Evaluating the Environmental Efficiency of Bioplastics using Advanced Life Cycle Assessment

George Bishop

Supervisor: Prof. Piet N. L. Lens
Co-supervisor: Dr. David Styles

A thesis submitted to the National University of Ireland Galway as fulfilment of the requirements for the degree of Doctor of Philosophy

School of Natural Sciences

November 2021

Declaration

I, George Bishop, declare that this thesis or any part thereof has not been, or is not currently being, submitted for any degree at the National University of Ireland, or any other University. I further declare that the work embodied is my own.

G M Bishop

Abstract

International bans on single-use petrochemical plastic products are accelerating the uptake of bioplastics to fill the gap in the market. Previous life cycle assessment (LCA) research exploring the environmental impacts from various bioplastic products has often been inadequate, leading to incomplete, biased, or misleading conclusions on their environmental sustainability. Thus, the primary aim of this thesis was to advance the understanding of the comparative environmental performance of bioplastic production, use, and disposal against conventional petrochemical plastic production, use, and disposal. This was achieved within this thesis via an extensive critical review of the published literature (**Chapter 2**), new analysis of plastic recycling chains (**Chapter 3**), and innovative consequential LCA of bioplastic value chains (**Chapters 4** and **5**). The research demonstrated the need for application of consequential LCA to facilitate a better understanding of the wider consequences of displacing petrochemical plastic with bioplastic. These LCAs further evaluated how possible environmental hotspots for bioplastic production could be reduced. Characterisation of petrochemical plastic recycling value chains was improved to enable more accurate benchmarking of bioplastics, where it was shown that a significant percentage of plastic from European recycling value chains is likely to end up as ocean debris. Overall, the thesis found that bioplastics can play a role in reducing global greenhouse gas emissions. However, simple substitution of petrochemical plastics with bioplastics will not drive environmental savings unless consumer behaviour and wider value chain logistics also change. Nevertheless, the uptake of bioplastics represents an important opportunity to design production pathways compatible with net-zero greenhouse gas emissions, and waste elimination in line with a fully circular bioeconomy.

From this thesis, five main focal points are identified to drive environmentally sustainable bioplastic expansion: i) acquisition of environmentally sound bioplastic feedstocks, considering the wider (potentially indirect) impacts. Bioplastics derived from lignocellulosic biomass or waste were shown to have great potential for environmental sustainability, avoiding the (potentially considerable) indirect land-use change burdens from purpose grown crops. ii) Development of circular and optimised value chains for the subsequent bioplastic production, for example, introducing decentralisation of production, utilising production residues, and ensuring that bioplastics can be treated via anaerobic digestion or directed to insect feed at its end-of-life. iii) Development of strategies to generate consumer behaviour change around bioplastics, especially for consumers to identify compostable bioplastics, and to place them within a dedicated food waste collection bin if appropriate after use. iv) Investment and implementation of regulations and incentives to aid the sustainable transition to bioplastics, based on the latest research, ensuring that the proceeding bioplastics don't just fill the market gap, but can also be restorative by nature. Such policies should support the preferred waste management hierarchy of compostable bioplastics, and safeguard against environmentally poor bioplastic feedstock acquisition. v) Increased research into the

potential environmental impacts of bioplastic production, use, and disposal, considering further consequential and dynamic LCAs and full life cycle sustainability assessments.

Overall, the guidance and understanding developed in this thesis will be a major asset to academic, industry, consumer, and policy stakeholders alike, enabling the rigorous assessment and design of environmentally sustainable bioplastic value chains, guiding stakeholders to bioplastics which can be more environmentally efficient than their petrochemical alternatives.

Acknowledgements

Firstly, I would like to express my sincere gratitude to my supervisor Prof. Piet Lens for giving me the opportunity to undertake this PhD in Galway. His motivation and immense knowledge were unwavering over the last three years, providing a source of continuous support and guidance. Thank you for giving me the freedom to choose the research that interested me. I am extremely grateful for that.

This PhD would not have been possible without the direction and encouragement of Dr. David Styles. His enthusiasm, vast intellect, and willingness to offer me so much of his time helped shape not only this PhD research, but also myself as a researcher. It is a privilege to have him as a mentor and I have benefitted enormously, both professionally and personally. I am truly grateful for his expertise, insights, and the many hours of discussion provided. Thank you, Dave.

I would also like to acknowledge Dr. Muriel Grenon, Prof. Charles Spillane, and Prof. Jamie Goggins for their contribution as members of my Graduate Research Committee.

To my family, thank you for helping me to get to where I am today and for your ceaseless love and support throughout the years.

Finally, I would like to thank Hannah for her endless love and encouragement. Thank you for your patience as I talked non-stop at you on our long walks as I attempted to solve the latest academic problem I was trying to overcome.

Contents

Table of figures

Table of tables

List of publications and chapter contributions

The work contained in this thesis consists of the following publications in international peer-reviewed journals:

Publication	Author contributions
Chapter 2: Bishop, G., Styles, D. & Lens, P.N.L., 2021. Environmental performance comparison of bioplastics and petrochemical plastics: A review of life cycle assessment (LCA) methodological decisions. *Resources, Conservation & Recycling*. **168**, 105451.	**GB**: Conceptualization, Methodology, Formal analysis, Writing - original draft. **DS**: Conceptualization, Validation, Writing - review & editing. **PNLL**: Writing - review & editing, Supervision, Funding acquisition.
Chapter 3: Bishop, G., Styles, D., Lens, P.N.L., 2020. Recycling of European plastic is a pathway for plastic debris in the ocean. *Environment International*. **142**, 105893.	**GB**: Conceptualization, Methodology, Formal analysis, Writing - original draft. **DS**: Conceptualization, Validation, Writing - review & editing. **PNLL**: Writing - review & editing, Supervision, Funding acquisition.
Chapter 4: Bishop, G., Styles, D., Lens, P.N.L., 2021. Environmental performance of bioplastic packaging on fresh food produce: a consequential life cycle assessment. *Journal of Cleaner Production*. **317**, 128377.	**GB**: Conceptualization, Methodology, Formal analysis, Writing - original draft. **DS**: Conceptualization, Validation, Writing - review & editing. **PNLL**: Writing - review & editing, Supervision, Funding acquisition.
Chapter 5: Bishop, G., Styles, D., Lens, P.N.L., in review. Land-use change and valorisation of side-streams determine the climate mitigation potential of bioplastics. *Resources, Conservation & Recycling*.	**GB**: Conceptualization, Methodology, Formal analysis, Writing - original draft. **DS**: Conceptualization, Validation, Writing - review & editing. **PNLL**: Writing - review & editing, Supervision, Funding acquisition.

Funding

This PhD thesis was supported by the Science Foundation Ireland Research Professorship, *Innovative Energy Technologies for Biofuels, Bioenergy and a Sustainable Irish Bioeconomy* (15/RP/2763).

Nomenclature

AD: anaerobic digestion
BAU: business-as-usual
BECCS: bioenergy with carbon capture and storage
Bio-PA: biopolyamide
Bio-PE: biopolyethylene
Bio-PET: biopolyethylene terephthalate
Bio-PP: biopolypropylene
C: carbon
CF: carbon fraction
CH_4: methane
CHP: combined heat and power
CO_2: carbon dioxide
CO_2 eq.: carbon dioxide equivalent
DOM: dead organic matter
dLUC: direct land-use change
DM: dry matter
EF: emission factor
EoL: end-of-life
EPD: Environmental Product Declaration
EU: European Union
GHG: greenhouse gas
GNI: gross national income
GWP: global warming potential
HDPE: high-density polyethylene
HI: high income
iLUC: indirect land-use change
IPCC: Intergovernmental Panel on Climate Change
ISO: International Standards Organisation
K_2O: potassium oxide
LCA: life cycle assessment
LCI: life cycle inventory
LCIA: life cycle impact assessment
LDPE: low-density polyethylene
LHV: lower heating value
LI: low income
LMI: low-middle income
LUC: land-use change
MRF: materials recovery facility
MSW: municipal solid waste
N: nitrogen
N_2O: dinitrogen monoxide
N_2O-N: dinitrogen monoxide nitrogen
NH_3: ammonia
NH_3-N: ammonia nitrogen
NH_4^+-N: ammonium nitrogen
NO_3^--N: nitrate nitrogen
P: phosphorus
P_2O_5: phosphorus pentoxide
PBAT: polybutylene adipate terephthalate
PBS: polybutylene succinate
PE: polyethylene
PEF: Product Environmental Footprint
PEFCRs: Product Environmental Footprint Category Rules
PET: polyethylene terephthalate
PHA: polyhydroxyalkanoates
PHB: polyhydroxybutyrate
PLA: polylactic acid
POPs: persistent organic pollutants
PP: polypropylene
PS: polystyrene
PVC: polyvinyl chloride
SOC: soil organic carbon
tkm: tonne kilometre
TPS: thermoplastic starch
TS: total solids
UK: United Kingdom
UMI: upper-middle income
USD: United States dollar

1. Introduction

1.1. Plastics in the circular economy

1.1.1. Plastics

Plastics are long chain polymer organic compounds, and can include other elements such as oxygen, nitrogen, sulphur, chlorine, fluorine, phosphorous, and silicon (Haslam et al., 1961). The raw materials for these traditional plastics are typically derived from non-renewable fossil resources such as oil and natural gas (Ren et al., 2009). Plastics are an important and ubiquitous material in the global economy and everyday life. The term "plastic" is derived from the Greek word "plastikos", meaning 'capable of being moulded into different shapes' (Shah et al., 2008). The mouldability of plastics, as well as the light, durable, and cheap properties of the materials, enable it to support many functional uses, contributing positively to global challenges including improving food security by improving the longevity of food (Barlow and Morgan, 2013). There are many different types of plastics, typically characterised into seven categories. These categories are defined by their resin identification code, where: 1 refers to polyethylene terephthalate (PET), 2 means high-density polyethylene (HDPE), 3 means polyvinyl chloride (PVC), 4 means low-density polyethylene (LDPE), 5 means polypropylene (PP), and 6 means polystyrene (PS). The classification number 7 refers to the packaging that is made from a type of plastic other than the previous six, or is a mixture of plastics. These plastics enhance almost every aspect of human life, including transportation, conservation, packaging, construction, medicine, human safety, and entertainment. In Europe, over 1.56 million people are directly employed by the plastic industry, and in 2019, the European plastic industry had a turnover of more than 350 billion euros (PlasticsEurope, 2020).

Plastic is often reported as a more environmentally efficient material when compared to alternative materials, such as glass and paper (e.g., Accorsi et al., 2015; Muthu et al., 2009). A report by Franklin Associates (2018) compared the environmental impacts of plastic packaging to steel, aluminium, glass, paper, textiles, wood, cork, and rubber packaging substitutes. They found significantly higher impacts for total energy demand, water consumption, solid waste generation, global warming, acidification, eutrophication, smog formation, and ozone depletion for the substitutes studied. The environmental efficiency of plastic is largely due to its low density and strength, which typically required less mass to perform equivalent functions over the substitutes. Although impacts per kg of plastic packaging can, in some cases, be higher than impacts per kg of substitute packaging, significantly more kg of substitute packaging is required to perform the same function (Franklin Associates, 2018).

1.1.2. Linear versus circular economy

The properties of plastic that make it so widely used also make its disposal challenging. The low cost of plastics mean that they are often discarded after a single use, especially for packaging and sheeting (Hopewell et al., 2009). However, due to the durability of the material, discarded plastics can persist in the environment for a very long time (Thompson et al., 2004). When mass production of plastic products began in the twentieth century, waste was not taken into account when designing products. The system was based on a linear model of take-make-dispose (Bocken et al., 2017). This model still dominates today, which incurs large quantities of waste. After a short first-use cycle, 95% of plastic packaging material value, or USD 80–120 billion annually, is lost to the economy (Ellen MacArthur Foundation, 2017), meaning that only 5% of material value is retained for subsequent use via recycling. Even then, the plastics that do get recycled are mostly recycled into lower-value applications that are not able to be recycled after use (Ellen MacArthur Foundation, 2017). The production of plastics is vast. Geyer et al. (2017) estimated that a cumulative total of 8300 million metric tonnes of virgin plastics had been produced by 2017, with nearly half of this produced within the previous 13 years. This large increase in plastic production correlates to increases in human population size and affluence, which have been increasing exponentially throughout the last century (Myers and Kent, 2003; Ogunola et al., 2018). With increased population growth and affluence has come increased demand for resources (Ellen MacArthur Foundation, 2013).

Owing to the negative environmental impacts arising from the linear economy model, the "circular economy" concept was developed. The circular economy model is a regenerative system which performs within ecological limits by reducing the need for resource extraction and abandoning the concept of waste (Geissdoerfer et al., 2017). The cradle-to-cradle concept aims to close material loops through a "cradle-to-cradle" approach, where the circular economy concept can be split into biological nutrients and technical nutrients (McDonough et al., 2003; McDonough and Braungart, 2002). Biological nutrients are non-toxic organic materials that at the end of their life can be safely returned in the biosphere (Korhonen et al., 2018), whereas technical nutrients are produced from inorganic or synthetic materials which can be cycled through the production system indefinitely, possibly degraded but without being transferred into waste (Mestre and Cooper, 2017). The circular economy concept involves products eco-designed for durability, disassembly, retrieval of technical nutrients, and refurbishment (Bocken et al., 2016). As well as the vast environmental benefits that could be achieved by closing loops in the circular economy (Geissdoerfer et al., 2017), it has been estimated that waste prevention, eco-design, reuse, and similar measures could bring net savings of 600 billion euros, or 8% of annual turnover, for businesses in the EU, whilst also reducing total annual greenhouse gas emissions by 2–4% (European Commission, 2014). Multiple European directives and strategies that affect plastic waste have been implemented (**Table 1.1**), which all either directly or indirectly stimulate a transition towards a more circular economy.

Table 1.1 – Overview of the main European Union policy involving plastics

Title of policy	Description relating to plastics
Packaging and packaging waste directive, Directive 94/62/EC (European Commission, 1994)	• The directive lays out the EU's rules on managing packaging and packaging waste • The directive requires member states to meet targets for the prevention, reuse, recovery, and recycling of packaging waste
Landfill directive, Directive 99/31/EC (European Commission, 1999)	• Regulates waste management of landfills in the European Union, including the banning of certain waste types • Reduction targets for biodegradable municipal waste sent to landfill
Waste framework directive, Directive 08/98/EC (European Commission, 2008)	• The directive requires Member States to apply the waste management hierarchy for waste management decisions • Further recycling and recovery targets introduced • The directive requires Member States to adopt waste management plans and waste prevention programmes
Plastic bags directive, Directive 2015/720 (European Parliment and the Council, 2015)	• The directive requires Member States to take measures to deal with the unsustainable consumption and use of lightweight plastic carrier bags, such as national reduction targets and/or economic instruments (e.g., fees, taxes) and marketing restrictions (bans), provided that the latter are proportionate and non-discriminatory
A European strategy for plastics in a circular economy (European Commission, 2018a)	• The strategy presents key commitments and a vision for a new circular plastic economy within the EU, and a call for action from European Parliament • Part of the EU's circular economy action plan • By 2030, all plastics packaging placed on the EU market will either reusable or can be recycled in a cost-effective manner
Directive on single-use plastics, Directive 2019/904 (European Parliment and the Council, 2019)	• The directive introduces a mix of measures tailored to common single-use products covered by the directive, including an EU-wide ban on single-use plastic products whenever alternatives are available
Pathway to a healthy planet for all EU action plan: 'Towards zero pollution for air, water and soil'. COM/2021/400 final. (European Commission, 2021)	• By 2030 the EU should reduce by 50% plastic litter at sea and by 30% micro-plastics released into the environment • By 2050 air, water, and soil pollution to be reduced to levels no longer considered harmful to health and natural ecosystems, that respect the boundaries with which our planet can cope, thereby creating a toxic-free environment
Mission Starfish 2030: Restore our ocean and waters (European Commission, 2020)	• The mission is to enable the restoration of the water cycle as a whole, via a set of ambitious, concrete, and measurable targets. • By 2030 all single-use plastics should be banned worldwide

1.1.3. Bioplastics

Alternatives to conventional fossil-based plastics exist in the form of bio-based plastics (bioplastics), i.e., plastics that are produced from renewable biomass sources. Bioplastics can retain the beneficial material characteristics of conventional petrochemical plastics whilst allowing for a transition towards a circular economy. Through a potential closed-loop system, the biogenic carbon taken up by the bio-based plastic feedstock can get released back into the atmosphere after use (preferable after several use cycles), e.g., by biodegradation or incineration, and can then once again be taken up by biomass sources (Spierling et al., 2018b).

Bioplastics are not a new family of plastics. In fact, one of the very first types of plastics created was a bioplastic called "parkesine", which was first produced and patented by Alexander Parkes in 1862 and was derived from a cellulose feedstock (Parkes, 1865). Bioplastics can be split into two categories, so called "drop-in" bioplastics, and "novel" bioplastics (Spierling et al., 2018a). Drop-in bioplastics may be produced from different raw materials but have the same chemical structure and properties as an existing fossil-based plastic on the market, and therefore can be managed in conventional recycling streams without adaptation (Spierling et al., 2018a). An example of a drop-in bioplastic is bio-PE (bio-based polyethylene). On the other hand, a novel bioplastic implies a new chemical structure and specific material properties (Spierling et al., 2018a), for example polylactic acid (PLA), a compostable bioplastic that is fully biodegradable in suitable environments. Bioplastics have been derived from many different feedstocks, including but not limited to: grass (Patterson et al., 2021), crops (Jimenez-Rosado et al., 2019), organic waste (Tsang et al., 2019), and algae (Prieto et al., 2017).

Although bioplastics can support the transition to a more circular economy by helping to reduce the environmental burdens of fossil resource extraction and potentially reduce the waste going to disposal, not all bioplastics are biodegradable, such as drop-in bioplastics, so some may still have the same issues and obstacles of plastic persistence as conventional petrochemical plastics if they are released into the environment.

1.2. Life cycle assessment

1.2.1. Life cycle assessment methodology

Life cycle assessment (LCA) is a method of quantifying the environmental impacts arising over the entire value chain of a product or service, from "cradle-to-grave", meaning that all the environmental aspects and impacts of product systems, from raw material acquisition to final disposal, are systematically assessed (**Figure 1.1**), using guidelines set by the International Standards Organisation (ISO) within the documents ISO 14040 and ISO 14044 (ISO, 2006a, 2006b).

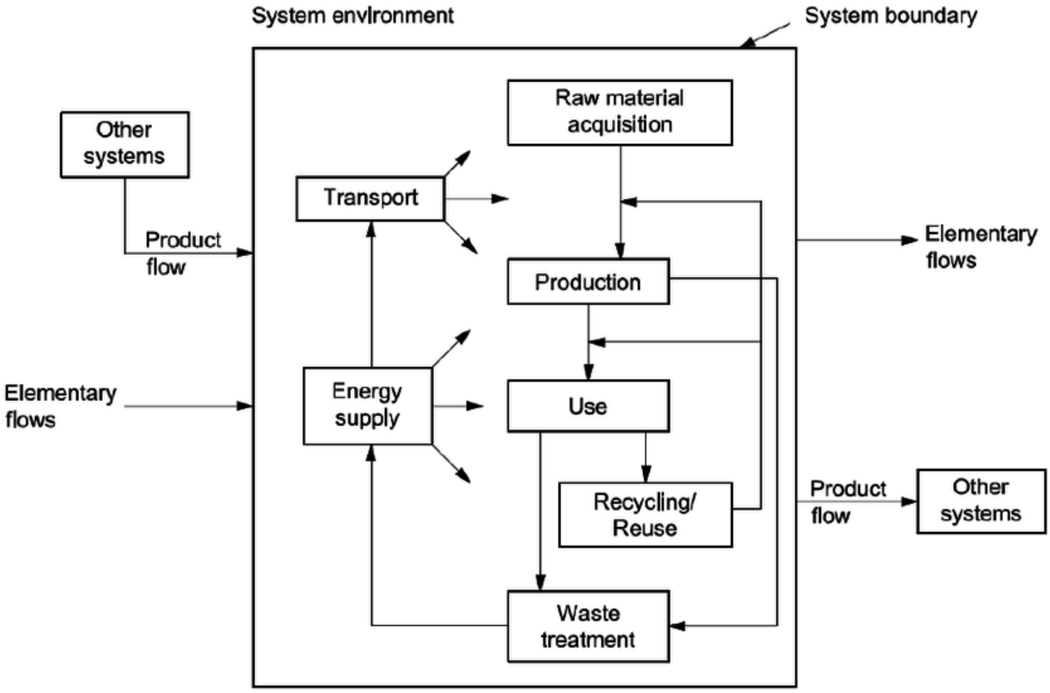

Figure 1.1 – Example of a product system for life cycle assessment (ISO, 2006a)

There are four main phases pertaining to a life cycle assessment (**Figure 1.2**) (ISO, 2006a, 2006b). The stages, as described by ISO (2006a, 2006b), are:

1. Defining the goal and scope. The goal of the LCA covers the intended application and the motives for carrying out the study. The scope includes multiple factors to be defined, such as: the product system to be studied, the functions of the product system and functional unit, reference flow, system boundary, impact categories selected, allocation procedures, data requirements, assumptions, and limitations.

2. Developing the life cycle inventory (LCI), where all significant inputs and outputs related to all relevant processes within the defined system boundaries are included. All raw material and energy requirements; emissions to the atmosphere, land, and water; resource use; and other releases over the life cycle of a product or process are quantified.

3. Life cycle impact assessment (LCIA). The environmental impacts based on the life cycle inventory data are quantified linking to specific impact categories (or "areas of environmental concern"). The LCIA includes i) selection of impact categories, category indicators, and characterisation models; ii) assignment of LCI results to the chosen impact categories (classification); and iii) calculation of category indicator results, where LCI results within each impact category are quantitatively transformed using characterisation factors (characterisation).

4. Interpreting results. The LCA results are identified, checked, evaluated, and presented.

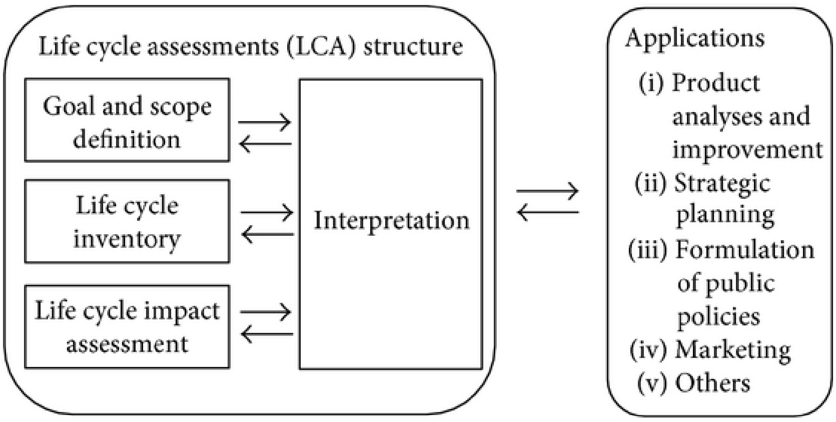

Figure 1.2 – Life cycle assessment phases (ISO, 2006a)

1.2.2. Use of LCA to assess environmental performance

LCA calculates the wider environmental impacts relating to many different impact categories, including climate change, eutrophication, acidification, ozone depletion, resource depletion, ecotoxicity, human toxicity, ionising radiation, and photochemical ozone formation (European Commission, 2018b), which provides a holistic picture of the environmental efficiency of a product (Rebitzer et al., 2004). A key part of the LCA methodology involves transparency relating to methodological choices and data sources (ISO, 2006b). This allows an LCA to be used in an unbiased way as a quantitative evidence base to inform policy decisions, or for marketing of products by businesses. Product systems can also be analysed for environmental hotspots (environmental inefficiencies) along the value chains, thus increasing the environmental and economic benefits of the product.

A consequential LCA is a system modelling approach in which activities are included in the product system to the extent that they are expected to *change* as a consequence of a change in demand for the functional unit (UNEP, 2011). The purpose of a consequential LCA is to describe how environmentally relevant flows will change in response to possible decisions (Finnveden et al., 2009), providing an estimate of how the production, use, and disposal of the study object affects the global environmental burdens (Ekvall, 2019). This implies that in such a system, the consequences are traced forward in time, which means that it is relevant to use data on marginal suppliers and substitution of displaced activities (Consequential-LCA, 2021).

1.3. Scope of this PhD thesis

1.3.1. Research motivation

Comprehensive and appropriately designed LCA studies are imperative to provide clear evidence on the comparative sustainability of bioplastics. However, it has been noted that previous LCA studies of bioplastics are often inadequate (Hottle et al., 2013; Pawelzik et al., 2013; Spierling et al., 2018a), leading to incomplete, biased, or misleading environmental footprints. The aspects of (bio)plastic LCAs which have previously been neglected, and thus the specific gaps that need filling, include i) the need for better understanding of the potential for closed petrochemical plastic recycling loops in a circular economy, ii) the need to consider the interaction between bioplastic end-of-life treatment options and consumer behaviour concerning waste separation, and iii) the need to critically explore the potential of different bioplastic feedstocks, considering the competing demands for land and potential impacts on biogenic carbon cycling directly and via indirect land-use change. It is important that comprehensive LCA evaluations of bioplastic sustainability are undertaken to ensure that genuine environmental savings are achieved, and that one set of major environmental impacts are not simply swapped with another set of impacts as we transition away from petrochemical towards bio-based plastics.

1.3.2. Research aims and objectives

The overarching aim of this doctoral thesis is to facilitate a greater understanding of the comparative environmental performance of bioplastic production, use, and disposal against conventional petrochemical-based plastic production, use, and disposal via cutting edge life cycle thinking and value chain analysis. This work will facilitate a better understanding of the consequences of displacing petrochemical plastic with bioplastic, as well as evaluating how possible environmental hotspots for bioplastic production can be mitigated. The specific objectives of this research are as follows:

1. To review the state-of-the-art regarding how previous bioplastic LCA studies have been modelled, identifying key gaps in studies, and therefore potential weaknesses in LCA results. Potential solutions to overcome key methodological gaps and to support more rigorous environmental assessments will be suggested (**Chapter 2**).

2. To characterise and quantify pathways of plastic waste release into the oceans according to specific plastic fractions from European recycling value chains, in order to better understand the efficiency and net environmental effects of plastic recycling – and potential as a circular economy solution.

Mass flows of this plastic leakage into the environment will be quantified for the first time (**Chapter 3**).

3. To investigate the environmental consequences of replacing petrochemical-based plastic food packaging with compostable bioplastic, within future-orientated scenarios. An advanced LCA model will be developed to holistically estimate the consequences from this forward-looking study (**Chapter 4**).

4. To quantify the environmental envelopes of bioplastic production from multiple feedstocks in relation to net-zero GHG targets, using a consequential LCA approach (**Chapter 5**).

1.4. Structure and layout

This thesis is composed of six chapters. Following this introductory chapter, the remainder of the thesis is structured into five additional chapters that are outlined below (**Figure 1.3**):

Chapter 2 (Bishop et al., 2021a) explores the growing collective of LCA literature that compares the environmental footprints of specific bioplastics against those of petrochemical plastics. Good practice examples facilitate identification of common gaps and weaknesses in LCA studies applied to benchmark bioplastics against petrochemical plastics.

Chapter 3 (Bishop et al., 2020) models and quantifies the end-of-life fates of polyethylene (PE) exported for recycling from Europe (EU-28, Norway and Switzerland). The end-of-life fates of the PE include recycled resins, landfilled PE, incinerated PE, and ocean debris.

Chapter 4 (Bishop et al., 2021b) rigorously assesses the environmental impact of displacing petrochemical plastic packaging of fresh fruit and vegetables with PLA, using advanced consequential LCA. Multiple future orientated scenarios are explored based on consumer behaviour decisions. LCA boundaries are expanded to include the end-of-life impacts of fruit and vegetable food waste within a UK context.

Chapter 5 (Bishop et al., in review) performs LCA to broadly screen potential compostable bioplastic feedstocks for greenhouse gas hotspots and compatibility with the objective of climate neutrality, considering both direct and potential indirect effects of their use for bioplastic production. The study calculates the greenhouse gas emissions balance of indicative value chains for four feedstocks from different origin: maize, lignocellulosic biomass from forestry, food waste digestate, and food waste.

Chapter 6 presents a summary and the conclusions of this PhD thesis. The recommendations and future perspectives on benchmarking the environmental impacts of bioplastic are discussed, based on the findings from the previous chapters.

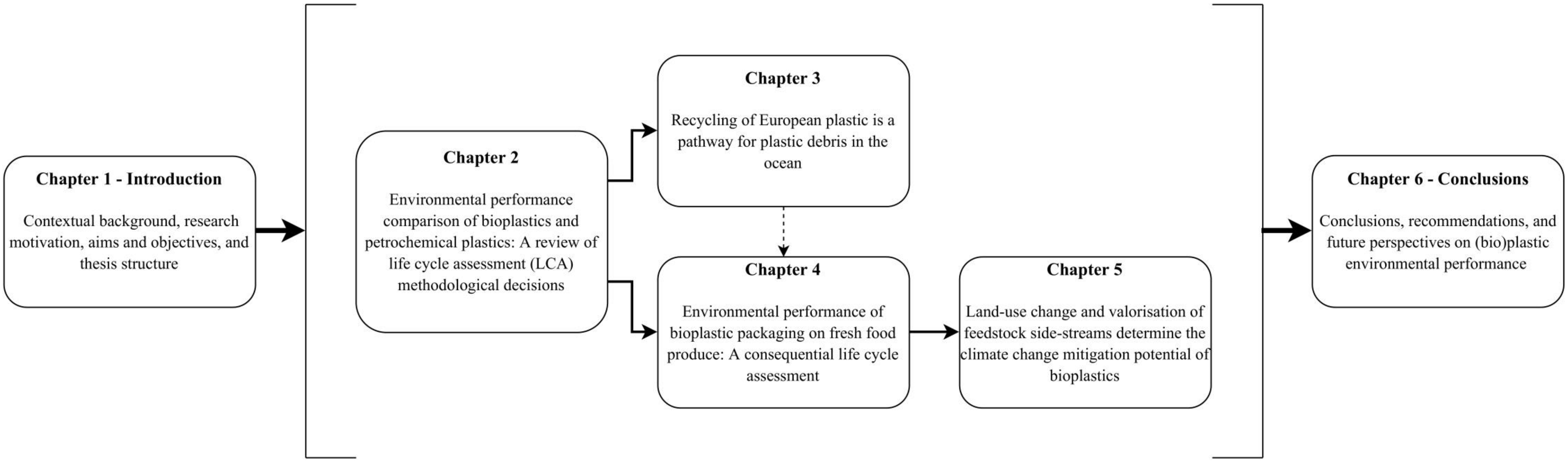

Figure 1.3 – Thesis structure and connection among the different chapters

1.5. References

Accorsi, R., Versari, L., Manzini, R., Accorsi, R., Versari, L., Manzini, R., 2015. Glass vs. plastic: life cycle assessment of extra-virgin olive oil bottles across global supply chains. Sustainability 7, 2818–2840. https://doi.org/10.3390/su7032818

Barlow, C.Y., Morgan, D.C., 2013. Polymer film packaging for food: An environmental assessment. Resour. Conserv. Recycl. 78, 74–80. https://doi.org/10.1016/J.RESCONREC.2013.07.003

Bishop, G., Styles, D., Lens, P.N.L., 2021a. Environmental performance comparison of bioplastics and petrochemical plastics: A review of life cycle assessment (LCA) methodological decisions. Resour. Conserv. Recycl. 168, 105451. https://doi.org/10.1016/j.resconrec.2021.105451

Bishop, G., Styles, D., Lens, P.N.L., 2021b. Environmental performance of bioplastic packaging on fresh food produce: A consequential life cycle assessment. J. Clean. Prod. 317, 128377. https://doi.org/10.1016/J.JCLEPRO.2021.128377

Bishop, G., Styles, D., Lens, P.N.L., 2020. Recycling of European plastic is a pathway for plastic debris in the ocean. Environ. Int. 142, 105893. https://doi.org/10.1016/j.envint.2020.105893

Bishop, G., Styles, D., Lens, P.N.L., in review. Land-use change and valorisation of feedstock side-streams determine the climate change mitigation potential of bioplastics. Resour. Conserv. Recycl.

Bocken, N.M.P., de Pauw, I., Bakker, C., van der Grinten, B., 2016. Product design and business model strategies for a circular economy. J. Ind. Prod. Eng. 33, 308–320. https://doi.org/10.1080/21681015.2016.1172124

Bocken, N.M.P., Olivetti, E.A., Cullen, J.M., Potting, J., Lifset, R., 2017. Taking the circularity to the next level a special issue on the circular economy. J. Ind. Ecol. 21, 476–482. https://doi.org/10.1111/jiec.12606

Consequential-LCA, 2021. Why and when? [WWW Document]. URL https://consequential-lca.org/clca/why-and-when/ (accessed 10.21.21).

Ekvall, T., 2019. Attributional and consequential life cycle assessment. Sustain. Assess. 21st century 395, 1–22. https://doi.org/10.5772/INTECHOPEN.89202

Ellen MacArthur Foundation, 2017. The new plastics economy: rethinking the future of plastics.

Ellen MacArthur Foundation, 2013. Towards the circular economy - Economic and business rationale for an accelerated transition. Isle of Wight.

European Commission, 2021. Pathway to a healthy planet for all EU action plan: "towards zero pollution for air, water and soil". COM/2021/400 final. Brussels.

European Commission, 2020. Mission Starfish: Restore our ocean and waters by 2030. Brussels.

European Commission, 2018a. A European strategy for plastics in a circular economy. COM/2018/028 final. Brussels.

European Commission, 2018b. PEFCR Guidance document, - Guidance for the development of Product

Environmental Footprint Category Rules (PEFCRs) 1–238.

European Commission, 2014. Towards a circular economy: A zero waste programme for Europe. Brussels.

European Commission, 2008. Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. Off. J. Eur. Union 51.

European Commission, 1999. Council Directive 99/31/EC of 26 April 1999 on the landfill of waste. Off. J. Eur. Communities L 182.

European Commission, 1994. European Parliament and Council Directive 94/62/EC of 20 December 1994 on packaging and packaging waste. Off. J. Eur. Union L 365, 10.

European Parliment and the Council, 2019. Directive (EU) 2019/904 of the European Parliament and of the Council of 5 June 2019 on the reduction of the impact of certain plastic products on the environment. Brussels.

European Parliment and the Council, 2015. Directive (EU) 2015/720 of the European Parliament and of the Council of 29 April 2015 amending directive 94/62/EC as regards reducing the consumption of lightweight plastic carrier bags. Brussels.

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S., 2009. Recent developments in life cycle assessment. J. Environ. Manage. 91, 1–21. https://doi.org/10.1016/J.JENVMAN.2009.06.018

Franklin Associates, 2018. Life cycle impacts of plastic packaging compared to substitutes in the United States and Canada.

Geissdoerfer, M., Savaget, P., Bocken, N.M.P., Hultink, E.J., 2017. The circular economy – A new sustainability paradigm? J. Clean. Prod. 143, 757–768. https://doi.org/10.1016/J.JCLEPRO.2016.12.048

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3, e1700782. https://doi.org/10.1126/sciadv.1700782

Haslam, J., Hamilton, J.B., Squirrell, D.C.M., 1961. The detection of "additional elements" in plastic materials the oxygen flask combustion method. Analyst 86.

Hopewell, J., Dvorak, R., Kosior, E., 2009. Plastics recycling: challenges and opportunities 364, 2115–2126. https://doi.org/10.1098/rstb.2008.0311

Hottle, T.A., Bilec, M.M., Landis, A.E., 2013. Sustainability assessments of bio-based polymers. Polym. Degrad. Stab. 98, 1898–1907. https://doi.org/10.1016/j.polymdegradstab.2013.06.016

ISO, 2006a. ISO 14040: Environmental management -- Life cycle assessment -- Principles and framework. Geneva.

ISO, 2006b. ISO 14044: Environmental management -- Life cycle assessment -- Requirements and guidelines. Geneva.

Jimenez-Rosado, M., Zarate-Ramirez, L.S., Romero, A., Bengoechea, C., Partal, P., Guerrero, A., 2019. Bioplastics based on wheat gluten processed by extrusion. J. Clean. Prod. 239. https://doi.org/10.1016/j.jclepro.2019.117994

Korhonen, J., Honkasalo, A., Seppälä, J., 2018. Circular economy: the concept and its limitations. Ecol. Econ. 143, 37–46. https://doi.org/10.1016/J.ECOLECON.2017.06.041

McDonough, W., Braungart, M., 2002. Cradle to cradle : remaking the way we make things. North Point Press.

McDonough, W., Braungart, M., Anastas, P.T., Zimmerman, J.B., 2003. Peer reviewed: applying the principles of green engineering to cradle-to-cradle design. Environ. Sci. Technol. 37, 434A-441A. https://doi.org/10.1021/es0326322

Mestre, A., Cooper, T., 2017. Circular product design. A multiple loops life cycle design approach for the circular economy. Des. J. 20:sup1, S1620–S1635. https://doi.org/10.1080/14606925.2017.1352686

Muthu, S., Li, Y., Hu, J.-Y., Mok, P.-Y., 2009. An exploratory comparative study on eco-impact of paper and plastic bags firefighter protective clothing view project. An exploratory comparative study on eco-impact of paper and plastic bags. J. Fiber Bioeng. Informatics Regul. Artic. 307 JFBI 1. https://doi.org/10.3993/jfbi03200909

Myers, N., Kent, J., 2003. New consumers: The influence of affluence on the environment. Proc. Natl. Acad. Sci. 100, 4963–4968. https://doi.org/10.1073/PNAS.0438061100

Ogunola, O.S., Onada, O.A., Falaye, A.E., 2018. Mitigation measures to avert the impacts of plastics and microplastics in the marine environment (a review). Environ. Sci. Pollut. Res. 25, 9293–9310. https://doi.org/10.1007/s11356-018-1499-z

Parkes, A., 1865. On the properties of parkesine, and its application to the arts and manufactures. J. Soc. Arts 14, 81–85.

Patterson, T., Massanet-Nicolau, J., Jones, R., Boldrin, A., Valentino, F., Dinsdale, R., Guwy, A., 2021. Utilizing grass for the biological production of polyhydroxyalkanoates (PHAs) via green biorefining: Material and energy flows. J. Ind. Ecol. 25, 802–815. https://doi.org/10.1111/JIEC.13071

Pawelzik, P., Carus, M., Hotchkiss, J., Narayan, R., Selke, S., Wellisch, M., Weiss, M., Wicke, B., Patel, M.K., 2013. Critical aspects in the life cycle assessment (LCA) of bio-based materials - Reviewing methodologies and deriving recommendations. Resour. Conserv. Recycl. 73, 211–228. https://doi.org/10.1016/j.resconrec.2013.02.006

PlasticsEurope, 2020. Plastics – the Facts 2020. Brussels, Belgium.

Prieto, C.V.G., Ramos, F.D., Estrada, V., Villar, M.A., Diaz, M.S., 2017. Optimization of an integrated algae-based biorefinery for the production of biodiesel, astaxanthin and PHB. Energy 139, 1159–1172. https://doi.org/10.1016/j.energy.2017.08.036

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment Part 1: Framework, goal and scope definition, inventory analysis, and applications. Environ. Int. 30, 701–720. https://doi.org/10.1016/j.envint.2003.11.005

Ren, T., Daniëls, B., Patel, M.K., Blok, K., 2009. Petrochemicals from oil, natural gas, coal and biomass: Production costs in 2030–2050. Resour. Conserv. Recycl. 53, 653–663. https://doi.org/10.1016/J.RESCONREC.2009.04.016

Shah, A.A., Hasan, F., Hameed, A., Ahmed, S., 2008. Biological degradation of plastics: A comprehensive review. Biotechnol. Adv. 26, 246–265. https://doi.org/10.1016/j.biotechadv.2007.12.005

Spierling, S., Knupffer, E., Behnsen, H., Mudersbach, M., Krieg, H., Springer, S., Albrecht, S., Herrmann, C., Endres, H.J., 2018a. Bio-based plastics - A review of environmental, social and economic impact assessments. J. Clean. Prod. 185, 476–491. https://doi.org/10.1016/j.jclepro.2018.03.014

Spierling, S., Rottger, C., Venkatachalam, V., Mudersbach, M., Herrmann, C., Endres, H.-J.J., Röttger, C., Venkatachalam, V., Mudersbach, M., Herrmann, C., Endres, H.-J.J., 2018b. Bio-based plastics - a building block for the circular economy? Procedia CIRP 69, 573–578. https://doi.org/10.1016/j.procir.2017.11.017

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., Russell, A.E., 2004. Lost at sea: where is all the plastic? Science (80-.). 304, 838–838. https://doi.org/10.1126/SCIENCE.1094559

Tsang, Y.F., Kumar, V., Samadar, P., Yang, Y., Lee, J., Ok, Y.S., Song, H., Kim, K.H., Kwon, E.E., Jeon, Y.J., 2019. Production of bioplastic through food waste valorization. Environ. Int. 127, 625–644. https://doi.org/10.1016/j.envint.2019.03.076

UNEP, 2011. Global guidance principles for life cycle assessment databases. A basis for greener processes and products, United Nations Environmental Programme. Paris/Pensacola: UNEP/SETAC Life Cycle Initiative.

2. Environmental performance comparison of bioplastics and petrochemical plastics: a review of life cycle assessment (LCA) methodological decisions

Abstract

There is currently a shift from petrochemical to bio-based plastics (bioplastics). The application of comprehensive and appropriately designed LCA studies are imperative to provide clear evidence on the comparative sustainability of bioplastics. This review explores the growing collective of LCA studies that compare the environmental footprints of specific bioplastics against those of petrochemical plastics. 44 relevant studies published between 2011 and 2020 were reviewed to explore important methodological choices regarding impact category selection, inventory completeness (e.g., inclusion of additives), boundary definition (e.g., inclusion of land-use change impacts), representation of biogenic carbon, choice of end-of-life scenarios, type of LCA, and the application of uncertainty analysis. Good practice examples facilitated identification of common gaps and weaknesses in LCA studies applied to benchmark bioplastics against petrochemical plastics. Many studies did not provide a holistic picture of the environmental impacts of bioplastic products, thereby potentially supporting misleading conclusions. For comprehensive evaluation of bioplastic sustainability, we recommend that LCA practitioners: embrace more detailed and transparent reporting of LCI data within plastic LCA studies; adopt a comprehensive impact assessment methodology pertaining to all priority environmental challenges; incorporate multiple plastic use cycles within functional unit definition and system boundaries where plastics can be recycled; include additives in life cycle inventories unless there is clear evidence that they contribute <1% to all impact categories; apply biogenic carbon storage credits only to long-term carbon sinks; account for (indirect) land-use change arising from feedstock cultivation; prospectively consider realistic scenarios of deployment and end-of-life, preferably within a consequential LCA framework.

2.1. Introduction

The environmental damage arising from the persistence of non-degradable plastic waste, typically produced by petrochemistry, has created an increasingly negative shift in public perception of petrochemical plastics (Rochman et al., 2016). To deal with the changing desires and concerns of the public, and to reduce environmental problems, European policy aims to reduce the quantities of single-use petrochemical plastic being used and produced (European Commission, 2018a, 2018b, 2008, 1994). Bio-based polymers (bioplastics) are being developed as a replacement material and a potential solution by retaining the beneficial material characteristics of petrochemical plastics whilst allowing for a transition towards a circular economy, reducing fossil resource extraction, and potentially reducing environmental burdens arising at end-of-life. The definition of "bioplastic" is generic, meaning that the term is often misleading. "Bioplastic" encompasses plastics which are durable and non-degradable (neat or partial blends) made from a biological source or plastics that are biodegradable (Soroudi and Jakubowicz, 2013). "Biodegradable bioplastics" can include biological-based biodegradable plastics, but also include biodegradable petrochemical plastics, such as polybutylene adipate terephthalate (PBAT) and polybutylene succinate (PBS) (Spierling et al., 2018a).

As most bio-based plastics are created as a potential replacement for petrochemical plastics, an accurate comparison of the environmental efficiency of these different plastics via life cycle assessment (LCA) is crucial. To be able to benchmark bio-based plastics against petrochemical plastics, the "full" life cycle of the different plastics should be represented, which can be complex owing to potentially long production-use-reuse/recycling value chains. Typical system boundaries for plastic value chains are represented in **Figure 2.1**. Failure to represent the complete system through boundary truncation or process simplification can result in studies misrepresenting the true comparative environmental efficiency of systems and products. The European Strategy for Plastics in a Circular Economy suggests that innovative materials and alternative feedstocks for plastic production should be developed and used where evidence clearly shows that they are more sustainable compared to petrochemical plastics (European Commission, 2018a). Therefore, comprehensive and appropriately designed LCA studies are imperative to provide clear evidence on the sustainability of bioplastics, and how they benchmark against conventional plastics.

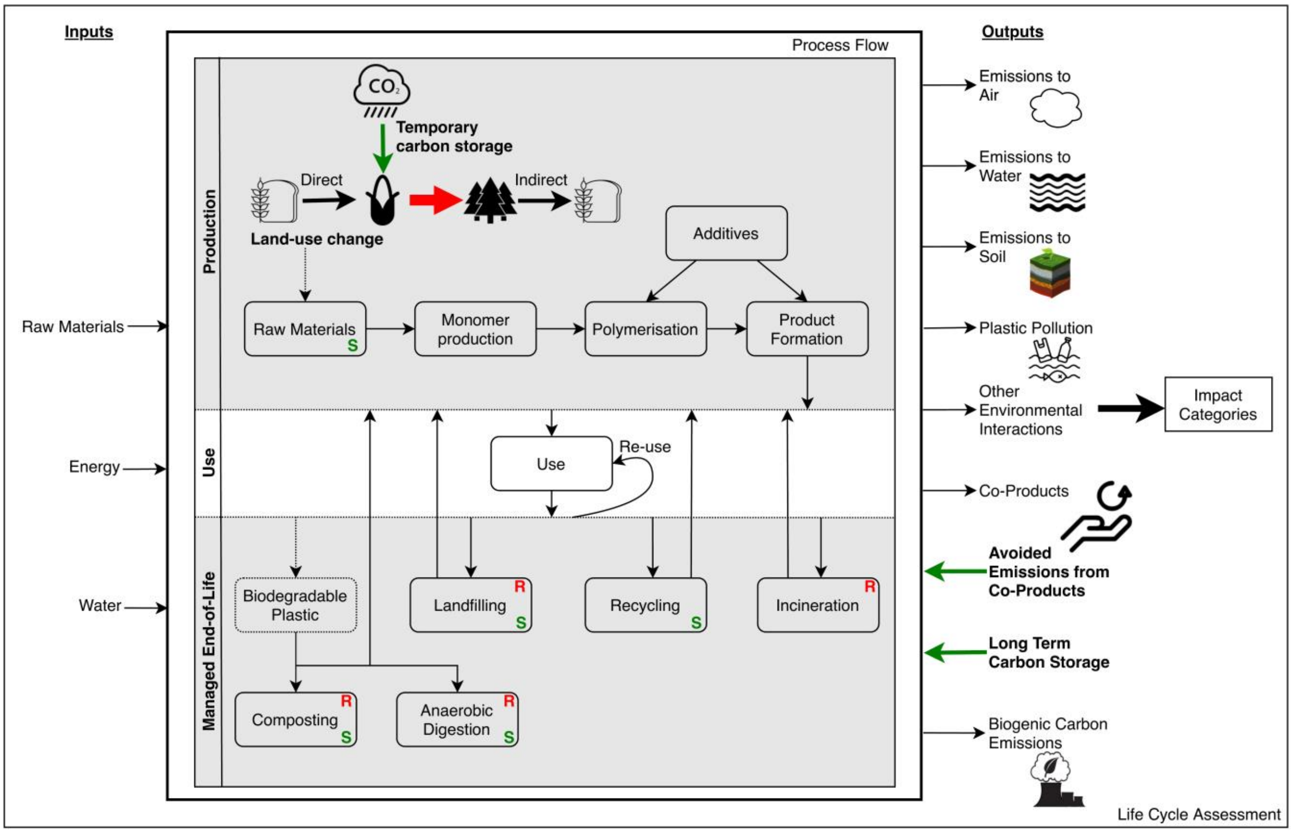

Figure 2.1 – A simplified schematic of a plastic value chain represented in LCA. The main processes, inputs, and outputs are displayed. Dashed lines represent flows specific to biodegradable plastics. The diagram also gives an indication of carbon flows throughout the system, with S representing carbon storage, and R representing the release of the carbon

Previous LCA reviews have studied the environmental performance of different end-of-life options for bioplastics (Spierling et al., 2018b), critical aspects of LCA methodology for bio-based materials (Pawelzik et al., 2013), and aspects of comparative environmental efficiency between bioplastics and petrochemical plastics (Hottle et al., 2013; Spierling et al., 2018a; Yates and Barlow, 2013). These studies have identified some methodological inconsistencies among studies, including: limited (or even biased) selection of impact categories resulting in incomplete footprints; differences in goal and scope definitions (including variations in choice of functional unit, system boundaries and allocation methods); selective evaluation of possible end-of-life options; and selective representation of indirect land-use change and accounting of biogenic carbon. This study expands upon previous reviews by exploring the growing collective of LCA studies that explicitly benchmark the environmental impacts of specific bioplastics against petrochemical plastics. This review critically analyses the methodological choices of how petrochemical and bio-based plastics are represented and environmentally benchmarked within studies. The aim of this study was to review a large segment of the literature to clarify the state of knowledge, identifying key gaps in studies and therefore potential weaknesses in LCA results hitherto. Potential solutions to overcome key methodological gaps and support more rigorous environmental assessments are suggested.

2.2. Material and methods

Life cycle assessment (LCA) is a method of quantifying the environmental impacts arising over the entire value chain of a product or service (ISO, 2006a, 2006b). Compared with more prevalent carbon footprinting, a full LCA calculates the wider environmental impacts in relation to multiple impact categories, providing a holistic picture of the environmental efficiency of a product (Rebitzer et al., 2004). A critical aspect of LCA studies is transparency on methodological choices and data sources that can strongly influence results. Transparent, non-biased LCA results provide a rigorous quantitative assessment of the environmental efficiency of products or systems, and constitute strong evidence to inform policy decisions (ISO, 2006b). Product systems can be analysed for improvements relating to environmental hotspots (points of comparatively high environmental impact) along value chains, allowing industry to recognise environmental and economic weaknesses within the product life cycle, as well as to assess the impacts of targeted mitigation strategies.

In this review, the focus was on studies which benchmarked, through LCA, the environmental efficiency of bioplastic against conventional petrochemical plastic. Web of Science and Scopus were used to search the literature, ensuring broad coverage of pertinent studies. The search included variations of the following keywords: *life cycle assessment*, *life cycle analysis*, *LCA* or *footprint* in connection with various combinations and variations of terms for bioplastics including: *bioplastic*, *bio-plastic*, *biobased plastic*, *bio-based plastic*, *biopolymer*, *bio-polymer*, *biobased polymer*, *bio-based polymer*, *renewable plastic*, *green plastic*, *sustainable plastic* and *biodegradable plastic*. The search also included various combinations of common bioplastic names and their associated acronyms: *thermoplastic starch* (*TPS*), *polylactic acid* (*PLA*), *polyhydroxyalkanoates* (*PHA*), *polyhydroxybutyrate* (*PHB*), *biopolyethylene* (*bio-PE*), *biopolypropylene* (*bio-PP*), *biopolyamide* (*bio-PA*), *biopolyethylene terephatalate* (*bio-PET*), *starch blend*, *polybutylene adipate terephthalate* (*PBAT*), *polybutylene succinate* (*PBS*). The literature search included studies which were published from January 2011 until November 2020. The time-related screening criterion was selected to reflect the state-of-art regarding LCA methodology, which is ever-changing over time. Only peer reviewed journal articles were considered within this review. From our search terms, all studies which actively performed LCA on at least one bioplastic were recorded. Further review of these recorded studies was completed to identify the papers which undertook an LCA of at least one bioplastic and at least one petrochemical plastic. From our approach, 44 LCA papers were found to be suitable and have been reviewed by this study (the list of reviewed studies can be found in **Appendix Table A2.1**).

2.3. Results and discussion

2.3.1. Impact categories

2.3.1.1. Impact categories covered

The average number of midpoint impact categories covered by the reviewed studies was eight (**Figure 2.2**), with a range between 1 and 18. The reviewed studies cover a wide range of impact categories. In total, 42 different midpoint impact categories were considered, with a total of 342 impact categories included over the 44 studies (**Table A2.2**). Many of the different impact categories cover similar environmental aspects but with different names and different methods, which makes it difficult to compare the results between studies. However, all 44 studies included (some variation of) global warming potential (GWP) within their impact categories, with five studies only evaluating this impact category. Apart from GWP, the most prevalent impact categories within the reviewed papers included (at least one variation of): acidification potential (29 studies); eutrophication potential (28 studies); resource depletion (26 studies); photochemical oxidant formation (23 studies); ozone depletion (20 studies); ecotoxicity (19 studies); human toxicity (17 studies); particulate matter formation (17 studies); energy (16 studies); land-use (14 studies); and water consumption (15 studies). The full break-down of midpoint impact categories evaluated by each study can be found in **Table A2.2**.

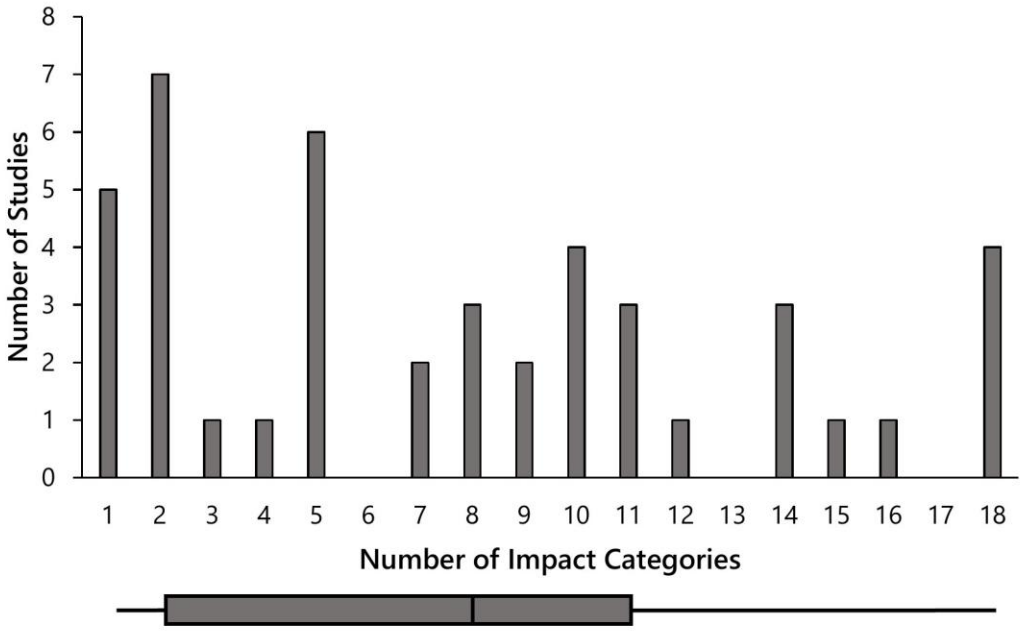

Figure 2.2 – The number of impact categories covered by each of the 44 studies reviewed. Below the graph is a box plot displaying the distribution of the number of impact categories in terms of the minimum, first quartile, median, third quartile, and maximum of the data

The number of impact categories each study used is closely related to the impact assessment method employed. The four studies that observed the highest number of impact categories utilised the ReCiPe Midpoint impact assessment methodology (Changwichan et al., 2018; Deng et al., 2013; Rodríguez et al., 2020; Vigil et al., 2020). However, other impact assessments used in the studies included CML 2001, Cumulative energy demand, Ecoindicator 99, ILCD, IMPACT 2002+, International EPD System, IPCC, and TRACI. Previous works have compared the technical differences, benefits, and limitations of the different impact methods (e.g., Bueno et al., 2016; Owsianiak et al., 2014; Renou et al., 2008). However, 11 studies were unclear or didn't include a standard impact assessment methodology within their paper, which typically reflects the inclusion of a limited suite of impact categories. Of the reviewed papers, nine studies also included endpoint impact categories (Alvarenga et al., 2013; Changwichan et al., 2018; Durkin et al., 2019; Gironi and Piemonte, 2011; Lorite et al., 2017; Piemonte, 2011; Saibuatrong et al., 2017; Tsiropoulos et al., 2015; Vigil et al., 2020). These categories included "resources", "human health", "ecosystem quality" and "climate change".

2.3.1.2. Pertinence of impact category selection to global environmental challenges

The planetary boundaries concept, developed by Rockström (2009) and Steffen et al. (2015), includes nine biogeophysical boundaries which define a "safe operating space for humanity" with respect to the Earth System. These boundaries have quantitative thresholds or limits, where transgression risks altering the planet's stable Holocene-like state, the only state known capable, with certainty, of supporting modern society (Steffen et al., 2015). The planetary boundaries represent thresholds of climate change, change in biosphere integrity (i.e., biodiversity loss and species extinction), stratospheric ozone depletion, ocean acidification, biogeochemical flows, land-system change, freshwater use, atmospheric aerosol loading, and the introduction of novel entities (e.g., such as radioactive materials, heavy metals and microplastics). The development by Steffen et al. (2015) suggested a two-level hierarchy of boundaries, in which climate change and biosphere integrity should be recognised as "core" planetary boundaries based on their "fundamental importance" for the Earth System. Steffen et al. (2015) identified that four of the boundaries have exceeded their proposed thresholds: climate change, biosphere integrity, biogeochemical flows, and land-system change, representing the main processes which need action to return to the safe operating space.

To map across the relevance of impact categories analysed by the reviewed studies to Earth System environmental priorities, this study paired each impact category to one planetary boundary with which it links most prominently (**Figure 2.3**). It was found that many studies only pertained to a single planetary boundary (**Figure 2.3a**). However, excluding those studies, the spread of the number of planetary boundaries covered per study had a relatively normal distribution, with a peak at six planetary boundaries being represented (**Figure 2.3a**). The two studies which were found to cover all nine planetary boundaries (Fieschi and Pretato, 2018; Maga et al., 2019) provide a more holistic comparison on the environmental efficiency of bioplastics and petrochemical plastics across primary biogeophysical (environmental) challenges. Both studies followed the Product Environmental Footprint (PEF) recommendations (described in **Chapter 2.3.2**) for impact categories. A count of the number of studies which observed each planetary boundary found that every study investigated the planetary boundary of climate change (**Figure 2.3b**). Although the other planetary boundaries were not so well covered, the exceeded planetary boundaries were typically explored except land-system change, which was only represented in approximately one quarter of the studies.

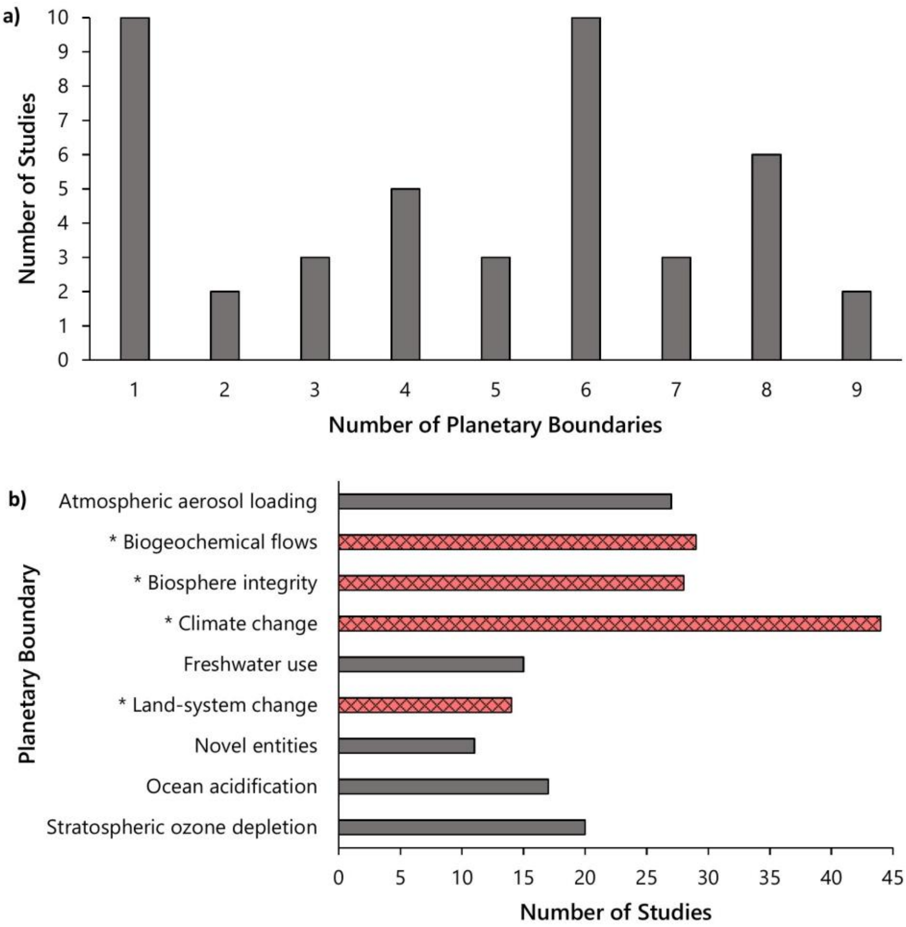

Figure 2.3 – a) The number of planetary boundaries covered by each of the 44 studies reviewed, with each study's observed impact categories linked to their most relevant planetary boundary, and b) The number of studies which observed each planetary boundary, with each study's observed impact categories linked to their most relevant planetary boundary. *red hashed columns represent the planetary boundaries which have exceeded the threshold of a safe operating space for humanity according to Steffen et al. (2015)

These results are only indicative as impact categories have links to multiple planetary boundaries. For example, land-use impact categories were paired with the land-system change planetary boundary, but are highly relevant to the biosphere integrity boundary (Kareiva et al., 2007; Sala et al., 2000). Nevertheless, the relationships presented highlight the relevance of impact categories used to evaluate bioplastics to critical environmental challenges. These reveal patchy coverage of the critical planetary boundaries which should be integral to environmental assessments. Future studies could focus more on environmental priorities by mapping impact category selection against global environmental priorities, such as those summarised in the planetary boundaries concept. The PEF impact assessment suite represents all planetary boundaries, and attempts to harmonise LCA application across Europe and globally (Fazio et al., 2018). Therefore, its use is strongly recommended when evaluating plastics. Other studies have attempted to bridge the gap between LCA and planetary boundaries (e.g., Ryberg et al., 2018, 2016; Sala et al., 2020; Sandin et al., 2015; Vanham et al., 2019).

2.3.1.3. Plastic pollution in LCA

The effects of plastic debris pollution (littering) into the environment are not included within any current LCA impact category (Sonnemann and Valdivia, 2017). Plastic pollution has wide-ranging and large potential impacts on ecosystem quality, human health, and climate change. Some plastics, such as polyethylene terephthalate, polystyrene, and polyvinylchloride have greater densities than seawater and thus sink more readily to the seabed (Haegerbaeumer et al., 2019). These plastics can cause smothering and/or mechanical damage to benthic organisms and vulnerable ecosystems such as coral reefs (Haegerbaeumer et al., 2019; Pawar et al., 2016). Plastic polymers are typically persistent and can survive for hundreds of years before being degraded (Thompson et al., 2004). Whilst the plastic remains in the environment, animals can become entangled within plastic fragments (Franco-Trecu et al., 2017; Gregory, 2009; Rodríguez et al., 2013) or ingest (Andrade et al., 2019; Clukey et al., 2017; Li et al., 2016; Poon et al., 2017) them, which can lead to suffocation, starvation, and death (Gregory, 2009; Koelmans et al., 2017; Wilcox et al., 2018). Entanglement and/or ingestion have been documented for at least 557 marine species (Kühn et al., 2015).

Even when plastic fragments have degraded, microplastics can still cause damage to larger organisms though potential biomagnification in trophic interactions (Saley et al., 2019), although more research on this topic is required (Law and Thompson, 2014). Microplastics have been found to affect soil ecosystems, negatively influencing factors such as germination success, shoot length, earthworm weight, soil structure, and pH (Boots et al., 2019). Additives within plastics have the potential to leach into the surrounding environment, resulting in toxicity to organisms, including potential carcinogenesis and endocrine disruption in humans (Cole et al., 2011; Talsness et al., 2009). The hydrophobic nature and large surface area-to-volume ratio of microplastics have also been found to concentrate persistent organic pollutants (POPs) from the surrounding environment, again potentially introducing toxins into the food chain (Li et al., 2016). Primary producers have been found to adsorb nanoplastic, which can hinder photosynthesis (Bhattacharya et al., 2010). A reduction in the photosynthesis rate can have additional effects of decreasing CO_2 uptake, encouraging climate change.

These plastic littering impacts are not only relevant for petrochemical plastics, but also occur with non-biodegradable bioplastics, and even potentially with plastics which are marketed as biodegradable bioplastics (Emadian et al., 2017; Parker, 2019; Straub et al., 2017). Such effects fit within the "novel entities" category of the planetary boundary concept (Villarrubia-Gómez et al., 2018).

Some LCA studies have started to consider littering within impact categories. Civancik-Uslu et al. (2019) developed and included the impact category: "Risk due to the abandonment of waste bags in marine environment" in their study. This impact category was created based on the combination of four parameters: 1) the quantity of bags required for the same function, based on number of bags used and the surface area; 2) the probability of the bags being released to the environment, based on the price of the bags; 3) the dispersion of the bags within the environment, based on the weight of the bag; and 4) the environmental persistence of the bag's material, based on biodegradability. The probability of the bags becoming litter is assumed to be directly proportional to the number of bags required, while it is indirectly proportional to the price, weight, and biodegradability. Although an interesting impact category to include, it has so far only been calculated as a comparative littering risk, and a lot more development is required to translate such a proxy into a mid-point impact category for LCA. Consequently, developing new impact assessment methods, or adapting existing ones, to represent potential environmental damage arising from plastic pollution should be a priority. Such development could have a large influence on conclusions drawn from LCA studies and is likely to have a significant bearing on the environmental sustainability credentials of biodegradable bioplastics used to substitute petrochemical plastics.

2.3.2. Additives

Of the 44 reviewed papers, only seven explicitly included additives within the life cycle inventories of plastics. Additives are chemical compounds added to plastic polymers during the production phase to enhance and determine the performance, functionality, appearance, and/or ageing properties of the final product (Hahladakis et al., 2018). The most commonly used types of additives in plastics are: plasticisers, flame retardants, antioxidants, acid scavengers, light and heat stabilisers, lubricants, pigments/colorants, antistatic agents, slip agents, fillers, and reinforcements (Hahladakis et al., 2018). Despite the considerable weight that additives can contribute to plastic (e.g., some plasticisers can account for up to 70% of the mass (%w/w) of the plastic material (Hahladakis et al., 2018; Hermabessiere et al., 2017)), and the relatively high impacts that additives can have on the overall environmental impact (Broeren et al., 2017), it is likely that additives are often not included within studies due to the quantities and specific substances used as additives being commercially-sensitive and widely neglected information. This represents an important gap that is typically ignored in LCA studies for final products that could lead to uncertain and misleading results (Broeren et al., 2016).

Where additives account for a small weight of the plastics, they may be consciously excluded (Hahladakis et al., 2018) based on materiality threshold cut-off criteria, which ISO standards allow based on either mass, energy, or environmental significance (ISO, 2006b). For example, PAS 2050 allows exclusions on the basis of materiality – i.e., if an item contributes <1% of the anticipated total GHG emissions associated with the product being assessed (BSI, 2011). However, Gallagher et al. (2015) discovered that adopting the suggested 1% materiality threshold led to a cumulative omission of between 2.6 and 7.5 % of the GWP burden for a micro-hydropower system, indicating failure to meet the threshold of accounting for at least 95% of the total system impacts (BSI, 2011). Similar results were found for the impact categories abiotic resource depletion, acidification potential, and fossil resource depletion. Therefore, future bioplastic LCA studies should explicitly include additives within the LCI unless there is clear evidence that they contribute less than 1% to mass, energy, or environmental burdens. Where multiple additives are used, it may be necessary to further reduce this materiality threshold to avoid neglecting cumulatively significant environmental burdens.

To reduce the methodological variability among LCA studies, there have been recent attempts to focus interpretation of the generic LCA framework provided in ISO 1400X series standards. The European PEF initiative (Fazio et al., 2018) aims to harmonise life cycle system boundary definitions, representation of common processes, and impact assessment categories, to support development of coherent environmental footprint databases that enable reliable benchmarking and communication of the environmental sustainability of products (Manfredi et al., 2012). The current PEF guidelines contain recommended impact category models (Fazio et al., 2018). Of the 44 studies, only one study followed the PEF guidelines (Fieschi and Pretato, 2018). No cut-off is allowed in the PEF methodology (Manfredi et al., 2012). However, the question has been raised of how this methodological requirement of PEF could practically be followed, as it is theoretically impossible to achieve (Finkbeiner, 2014a). Product Environmental Footprint Category Rules (PEFCRs) have been developed to improve the reproducibility of specific product LCA studies in relation to critical parameters (European Commission, 2018c). However, no PEFCR has yet been developed for bioplastics. It will be crucial that additives are included in any future PEFCRs. Another established system which develops PCRs is the Environmental Product Declarations (EPDs), which are also lack rules for bioplastics (Spierling et al., 2018a).

When included, typically only the impacts from the production of additives are modelled within the LCA studies, and the impacts related to the end-of-life are not. As previously mentioned, additives have the potential to leach into the surrounding environment (air, water, and soil) at the end-of-life, resulting in potential toxicity to organisms and ecosystems (Cole et al., 2011; Hermabessiere et al., 2017). The exclusion of end-of-life impacts of additives is an important area of research which needs future investigation to better understand the full potential life cycle impacts of additives, and at what concentration within plastics they are likely to exceed the 1% contribution threshold for specific impact categories.

2.3.3. Life cycle inventories

Inventory data sources for the studies were a mixture of primary data, literature, and LCI databases. For the petrochemical plastics, most of the production data was extracted from databases (41 studies), supplemented with literature (21 studies). Nine studies included some primary data, collated mostly via collaborations with companies, or measured primary data for specific novel products being compared. For the bioplastic production inventories, primary data was more prevalent, used in 20 studies, mostly collected from actors within bioplastic supply chains. However, data was still heavily augmented with extraction from LCI databases (36 studies) and from the literature (32 studies). One study was unclear on where its data was sourced from.

Further investigation of the data used by the studies is constrained owing to the generic nature and lack of detailed description of many data sources. To indicate data clarity and reproducibility, a subjective rating was applied to every reviewed paper based on data availability and data description within the studies. The analysis showed that 25% of the studies were "very clear and easily reproducible", 25% of the reviewed studies were "clear and moderately reproducible", 20% of the studies were "unclear but somewhat reproducible", and 30% of the studies were found to be "unclear and unreproducible". It is reasonable that some data may not be made fully available due to its confidential nature with respect to the industry providing the data. However, if LCA studies lack transparency and cannot be replicated, their validity is thrown into question. In line with general good practice for LCA studies, it is imperative that the LCIs for plastic LCA studies are: 1) easily understandable; 2) transparent; 3) complete; 4) clear; 5) reproducible.

2.3.4. Land-use and land-use change

2.3.4.1. Land-use change concepts

Land-use and land-use change are important aspects of the environmental impacts arising from bioplastics produced from crop feedstocks. Accounting for land-use change emissions can drastically alter the conclusions of LCA studies on the environmental impact of bioplastics, changing the rankings of bioplastics and conventional petrochemical plastics (Piemonte and Gironi, 2011). Despite bioplastic crop feedstocks being renewable production sources, they require land that could otherwise serve another function, such as provision of natural habitats or food production. Searchinger (2008) introduced the concept of "carbon cost": the accounting for the carbon storage and sequestration sacrificed by diverting land from existing uses. In LCA, emissions from two types of land-use change may be captured in the inventories for bio-based products: direct land-use change (dLUC), and indirect land-use change (iLUC). dLUC refers to a recent change in the use of land on which feedstocks for biobased products are produced, thus displacing prior land-use, e.g., the conversion of rainforests to sugarcane plantations. iLUC refers to the process where the production of biobased feedstocks displaces prior production, without incurring a direct change in land use (e.g., land remains cropland), but the displaced production causes land-use change elsewhere, potentially via a cascade effect. Thus, if demand for the displaced production remains, then that displaced production will cause subsequent land-use change in other locations around the world (Schmidt et al., 2015).

Land-use change can have considerable effects on the global carbon cycle, causing significant greenhouse gas emissions via disturbance of carbon stocks in soil and vegetation (Schulp et al., 2008). Different land-uses support different stocks of carbon in soils and vegetation, and different potential rates of carbon stock change (Schulp et al., 2008). Soil organic carbon (SOC) stocks under cropland are typically lower than SOC stocks under pasture or forest (Poeplau and Don, 2013). Consequently, conversion to cropland likely decreases SOC stocks. Similarly, forests accumulate large quantities of carbon in their biomass, so conversion to cropland (or grassland) will result in the loss of biomass carbon. Other impacts from land-use change can result from increased fertiliser use through intensification, thus increasing the release of nitrous oxide (N_2O), ammonia (NH_3), and nitrogen and phosphorus leachates, as well as fertiliser manufacturing burdens. It should be noted that, however, any displaced fertiliser use may also be represented by direct fertiliser use within life cycle inventories.

Despite potentially large impacts, accounting for iLUC is not mandatory in any LCA international standard (Finkbeiner, 2014b). This is partly because of the difficulty in establishing iLUC effects, as they are, as the name infers, indirect, and may ultimately arise at the end of a long cascade of consequences (Lambin and Meyfroidt, 2011). Also, iLUC is considered to be outside the direct scope of study when applying an attributional LCA approach. The inclusion of iLUC fits more with the consequential LCA framework as iLUC is usually a market-induced effect (see **Chapter 3.8**). Several studies have proposed how LCA studies can include iLUC, e.g., Schmidt et al. (2015) and Searchinger et al. (2018).

2.3.4.2. Land-use change covered by reviewed studies

Of the reviewed studies, 14 included some form of land-use impact category. The impact categories covered were: "land-use", which was measured as the carbon deficit (Belboom and Léonard, 2016; Fieschi and Pretato, 2018; Giovenzana et al., 2019), or as a relative contribution (Maga et al., 2019), or was included in the macro-category "Ecosystem Quality" (Gironi and Piemonte, 2011); "land occupation" (Changwichan et al., 2018; Horowitz et al., 2018; Lorite et al., 2017; Tsiropoulos et al., 2015); "agricultural land occupation" (Deng et al., 2013; Rodríguez et al., 2020; Vigil et al., 2020; Zhang et al., 2018); "urban land occupation" (Deng et al., 2013; Rodríguez et al., 2020; Vigil et al., 2020); "Natural land transformation" (Deng et al., 2013; Rodríguez et al., 2020; Vigil et al., 2020); or "biodiversity loss due to land-use" (Alvarenga et al., 2013).

Eight studies included emissions from dLUC within their methodology (Alvarenga et al., 2013; Hansen et al., 2015; Kikuchi et al., 2013; Leejarkpai et al., 2016; Liptow and Tillman, 2012; Razza et al., 2015; Suwanmanee et al., 2013b; Tsiropoulos et al., 2015). Many of the studies (Hansen et al., 2015; Leejarkpai et al., 2016; Suwanmanee et al., 2013b) directly modelled their dLUC emissions from IPCC equations (IPCC, 2006). Leejarkpai et al. (2016) calculated dLUC using primary data, where land-use change emissions were calculated from a combination of the change in soil carbon, the change of carbon stock in biomass, the non-CO_2 emissions from burning required for crop change, and the emissions from managed soils (i.e., fertiliser emissions), all measured in kg CO_2 eq. per hectare. In a similar method, an adapted IPCC Tier 1 approach was implemented by Hansen et al. (2015) to estimate the release of C from land transformation calculated as the carbon difference before and after cultivation of the (sugarcane) crop. Other studies utilised emission factors for specific land-uses (Razza et al., 2015; Tsiropoulos et al., 2015) or previously calculated results (Kikuchi et al., 2013; Liptow and Tillman, 2012). The type and method of land transformation varies per study. In the year of establishment, Leejarkpai et al. (2016) modelled that abandoned land was changed to cropland (corn). Kikuchi et al. (2013) estimated the land-use transformation from Cerrado wooded areas to sugarcane areas. Liptow and Tillman (2012) and Alvarenga et al. (2013) both modelled expansion of sugarcane plantations into pasture areas. Liptow and Tillman (2012) determined that, under the aforementioned situation, dLUC may have led to subsequent soil carbon accumulation following initial soil carbon release. Therefore, they assumed that net emissions upon dLUC were zero. However, their study also contained a second approach, wherein the dLUC was modelled from a 5% sugarcane expansion directly transformed from virgin areas of Brazil (Cerradao region), which resulted in a substantial release of GHG emissions. dLUC is associated with high emissions uncertainty (Razza et al., 2015), however, the likely bounds can be evaluated with sensitivity analysis. Razza et al. (2015) estimated dLUC emissions within sensitivity analysis, where "hypothetical" tapioca was produced in Thailand on land previously occupied by grassland (best-case) or rainforests (worst-case).

Only four studies included iLUC emissions within their methodologies (Alvarenga et al., 2013; Eerhart et al., 2012; Liptow and Tillman, 2012; Tsiropoulos et al., 2015). Tsiropoulos et al. (2015) used a range of emission factors taken from the literature, utilising 3 – 46 g CO_2 eq./$MJ_{ethanol}$ to produce 0.16 to 2.38 kg CO_2 eq./$kg_{bio\text{-}HDPE}$. Eerhart et al. (2012) included four scenarios of iLUC, which resulted in values of 0, 7, 14 and 30 g CO_2 eq./$MJ_{ethanol}$. Liptow and Tillman (2012) presented two possible extremes of iLUC emission factors taken from the literature, using a worst-case of 46 g CO_2 eq./$MJ_{ethanol}$ and a best-case scenario of 0 g CO_2 eq./$MJ_{ethanol}$. Alvarenga et al. (2013) considered that the pasture lands displaced by sugarcane (dLUC) would be diverted into areas with natural vegetation, which in this study was assumed to be the Amazon Forest, thus causing iLUC. The study contained three scenarios based on the area of iLUC attributable to sugarcane cultivation, which was developed from the early models that used a 1:1 ratio (Searchinger et al., 2008). The study uses the iLUC ratios of 1:1, 0.13:1, and 0:1, where a 1:1 ratio means that for every hectare of new crop cultivation, one hectare of new land would be indirectly cleared. The required 1.88 m^2 $year^{-1}$ of land occupation for 1 kg of bioethanol-based PVC resin (the functional unit) was divided over 20 years. iLUC was subsequently calculated as 0 – 1.34 kg CO_2 eq. per kg PVC resin.

2.3.5. Biogenic carbon accounting

2.3.5.1. Biogenic carbon accounting concepts

Bioplastics are typically produced, completely or partially, from a feedstock which has converted atmospheric CO_2 into carbon compounds via photosynthesis, termed as biogenic carbon in LCA studies. Representation of the climate forcing effect of temporarily storing this biogenic carbon out of the atmosphere has the potential to considerably alter the environmental rankings of bioplastics compared to petrochemical plastics. There are two main approaches in how biogenic carbon is modelled within LCA studies: 1) temporary carbon storage and 2) carbon neutrality (Pawelzik et al., 2013). Carbon is sequestered by the feedstock, thus storing the carbon within the bioplastic for a (potentially considerable) length of time. It is argued that carbon storage should be modelled because it delays radiative forcing, and thus decreases cumulative impacts (Levasseur et al., 2012). Carbon storage can also offset current anthropogenic carbon emissions (Pawelzik et al., 2013). Exclusion of carbon storage effects causes inaccuracies in LCA modelling of waste management because it omits potential long-term carbon sequestration (e.g., within landfills or compost-amended soils) that would decrease the impact on global warming (Christensen et al., 2009).

Climate neutrality is often assumed, where the carbon that is sequestered by the feedstock is released back into the environment in a closed loop with no net climate forcing effect. Some argue that biogenic carbon storage should be excluded from LCA, as the bioproduct (almost always) releases the stored carbon emissions in the future (Pawelzik et al., 2013). It has also been suggested that by temporarily reducing atmospheric CO_2 concentrations, temporary carbon storage can lower the CO_2 removal rates of other sinks, eventually leading to higher atmospheric concentrations and temperatures when the carbon is later released (Brandão et al., 2012). Further, the "carbon debt" concept (Fargione et al., 2008), which is prevalent in bioenergy and land-use change modelling, is also relevant to the end-of-life for bioplastic systems that originate from forestry feedstocks. If the initial emission of biogenic carbon from the bioplastic (e.g., during incineration) exceeds the emissions from a reference fossil system (which the bioproducts replace), it creates a carbon debt. The debt is paid back as the biomass re-grows and sequesters carbon from the atmosphere. With the continuous substitution of fossil fuels, bioplastics will over time repay the carbon debt. Nevertheless, biogenic CO_2 spends time in the atmosphere before being captured by biomass re-growth, which can possibly lead to climate change related impacts. This issue pertains more to forestry with long growth cycles (>40 years) rather than annual crop feedstocks where CO_2 uptake is within one growth season, and will become more important if lignocellulosic biomass-derived bioplastics gain traction (Brodin et al., 2017).

The calculated benefits from modelling biogenic carbon storage are especially sensitive to the time horizon over which the GWP is considered (see Levasseur et al., 2012). Recent groups and studies have developed different methodological decisions in dealing with such carbon emissions, often focusing on issues surround the time horizon. These have been covered concisely and analytically by Brandão et al. (2019). Some of the developed methodologies differ in the characterisation of climate change impacts of a given quantity of emissions. For example, GWP_{100} (ISO, 2018), which almost all LCA studies follow, is the time-integrated radiative forcing due to an emission, relative to the emission of an equal mass of CO_2; Global Temperature change Potential (GTP) (Shine et al., 2005) estimates the effect of greenhouse gas emissions on the average global temperature at a specified future time, relative to the temperature rise which the same mass of CO_2 would cause; and GWP_{bio} (Cherubini et al., 2011) utilises characterisation factors specific for CO_2 emissions from biomass, with the time profile of regrowth taken into account. Other methods relate to how to calculate the net balance of carbon emissions through time. Several differing methods combine cumulative radiative forcing and the benefits of temporary carbon storage, e.g., the Moura-Costa method (Moura Costa and Wilson, 2000), the Lashof method (Fearnside et al., 2000), the Müller-Wenk and Brandão method (Müller-Wenk and Brandão, 2010), and the Clift and Brandão method (Clift and Brandao, 2008). Time-averaged carbon stocks as a method for carbon accounting has also been introduced (Kirschbaum et al., 2001). The Climate Tipping Potential (CTP) (Lenton et al., 2008) is a planetary boundary style approach based on the notion of thresholds in the global climate system. The climate change threshold is quantified as a maximum temperature increase expressed as a corresponding atmospheric CO_2 concentration. The method then calculates the capacity of the atmosphere to absorb GHG emissions without exceeding the tipping point, and any emission is assessed against that remaining capacity (Brandão et al., 2019). Pawelzik et al. (2013) reviewed how different LCA guidelines recommend the accounting of biobased carbon storage: ADEME's methodology for bio-based materials (BIS, 2009); the European Commission's Lead Market Initiative (European Commission, 2009); GHG Protocol Initiative (GHGP, 2011); PAS 2050 (BSI, 2011); and the process/material carbon footprint (Narayan, 2011).

2.3.5.2. Biogenic carbon emissions covered by reviewed studies

Below, it is discussed how the different studies covered by this review have treated biogenic carbon cycling within bioplastic life cycles. Thirty of the reviewed studies explicitly referred to biogenic carbon modelling. As previously mentioned, many of these studies just treated the biogenic carbon as having a neutral GWP effect – i.e., assumed that very short term biogenic carbon storage in products and product end-of-life did not influence net climate forcing (Deng et al., 2013; Eerhart et al., 2012; Forte et al., 2016; Hottle et al., 2017; Kikuchi et al., 2013; Leejarkpai et al., 2016; Maga et al., 2019; Papong et al., 2014; Semba et al., 2018; Suwanmanee et al., 2013b; van der Harst et al., 2014; Zhang et al., 2018)[1]. It is likely that the studies that did not explicitly discuss biogenic carbon simply assumed no GWP effect from changes to biogenic carbon cycling within bioplastic value chains.

Some studies modelled biogenic carbon uptake but did account for end-of-life management (i.e., cradle-to-gate) or were not clear in how biogenic carbon was treated at the end-of-life, potentially generating misleading conclusions on GWP savings from bioplastic production. Changwichan et al. (2018) modelled CO_2 fixation via photosynthesis of 1834, 2199, and 2046 kg per kg resin for PLA, PHA, and PBS, respectively. However, this study did not fully explain how these very high values were derived, nor how these biogenic emissions were treated at the end-of-life. It is possible that the resin represented a small percentage of the plant biomass, and that the study related back to total biomass CO_2 sequestration. However, as an annual crop feedstock was used, almost all that carbon is likely to be released back to the atmosphere within a short time span (e.g., via animal feed or decomposition). Similarly, Unger et al. (2017) calculated the biogenic CO_2 uptake from corn feedstock but did not provide a clear description of how the biogenic carbon was treated at the end-of-life. Nikolic et al. (2015) also attributed credits to atmospheric carbon fixed by plants in the process of photosynthesis. The CO_2 absorbed by corn grain through the

[1] Hottle et al. (2017) mostly modelled carbon neutrality. However, under some conditions biopolymers did not degrade and the carbon was considered sequestered, while in other conditions biopolymers released methane which resulted in net positive GHG emissions from biogenic sources.

photosynthesis process (1.34 kg CO_2 kg^{-1} corn) was subtracted from the gross life cycle GHG emissions captured in this cradle-to-gate study. This method treats any difference between initial biogenic CO_2 fixation in plants and release of biogenic CO_2 during the time horizon of the study as carbon storage, and was applied in other cradle-to-gate studies reviewed here (e.g., Mahalle et al., 2014; and Tsiropoulos et al., 2015). However, the time horizon over which storage is accounted for may deviate from the 100-year time horizon pertinent to the GWP_{100} metric that is usually applied (IPCC, 2014). In the study by Liptow and Tillman (2012), neither biogenic CO_2 uptake during cultivation nor release during the bioplastic life cycle were explicitly accounted for. However, in some sensitivity analyses, waste treatment was changed from incineration to landfill, and carbon sequestration was considered in the latter (see **Figure 2.1**). Chen et al. (2016) applied carbon storage credits to bioplastic bottles on the basis that the carbon in the bio-PET bottles is potentially sequestered from the atmosphere long-term in a recyclable plastic product. Therefore, although the end-of-life stage was not included in the scope of the analysis, the authors found it reasonable to include sequestered biogenic carbon as a credit to bio-PET bottles. The credits ranged from 0.46 – 2.29 kg CO_2 eq. per kg bottle, depending on the scenario. These carbon sequestration credits were critical to the better environmental outcome for bioplastics compared with conventional plastics in that study.

Other studies implemented various approaches to represent biogenic carbon with varying levels of detail. Nguyen et al. (2020) included GHG credits for the biogenic carbon storage, taken as an average from previous studies. Patel et al. (2018) assumed 99% release of stored carbon for incineration and 95% release of stored carbon for biodegradation of films and industrial composting of trays. When presenting results for the production of the studied bioplastic, they considered both temporary storage of atmospheric carbon and long-term carbon storage in the compost (both treated as negative emissions). Saibuatrong et al. (2017) modelled photosynthetic uptake of CO_2 per kg of cassava and sugarcane (4.94×10^{-2} and 8.50×10^{-1} kg CO_2 per kg fresh matter, respectively). End-of-life sequestration from the compost provided a CO_2 emission offset of 2.33×10^{-3} kg per kg feedstock. In the study by Belboom and Léonard (2016), only the CO_2 converted into starch or sucrose was modelled. It was assumed that the remaining carbon compounds (e.g., proteins, cellulose, and lipids) would be degraded and emitted as CO_2 to the atmosphere within one year and were therefore treated as GWP neutral. The amount of CO_2 captured during growth was calculated using a sucrose content of 17% for sugar beet and a starch content of 62.5% for wheat. Values of captured CO_2 were based on stoichiometric equations of photosynthesis and were modelled as 18.27 t ha^{-1} for sugar beet and 7.9 t ha^{-1} for wheat. All emissions of CO_2 during the life cycle of the biobased polymer were considered, including fermentation and the end-of-life. In a sensitivity analysis, Hansen et al. (2015) developed material balances to consider all the possible forms of carbon uptake and emission over the bioplastic life cycle. Razza et al. (2015) modelled calculated biogenic carbon sequestration credits for landfill and composting disposal, leading to GWP credits. Suwanmanee et al. (2013a) modelled the amount of CO_2 absorbed per kilogram cassava and kilogram corn, and the carbon sequestered by compost. Durkin et al.

(2019) applied a stoichiometric carbon-counting approach to "track" the biogenic carbon flows over the life cycle. Guo and Murphy (2012) and Benavides et al. (2020) utilised a carbon counting approach to track carbon flows during the life cycle of the bioplastic products, including sequestration into the product and any downstream release of this carbon during the subsequent processing, product use and final disposal stages of the life cycle. These carbon counting approaches allow for high levels of clarity and detail.

Analysis of the reviewed studies shows that biogenic carbon storage can have a modest influence on the net GWP burden of bioplastics when accounted for rigorously. Studies that claim a large effect appear to have applied incomplete accounting to the biogenic carbon cycle. Best practise is to explicitly model biogenic carbon uptake, storage and release over the extended bioplastic life cycle (e.g., Benavides et al., 2020; Durkin et al., 2019; Guo and Murphy, 2012) – and indeed to account for fossil carbon over an extended life cycle for petrochemical plastics. Such accounting can be complex and requires careful representation of biological processes. If this is beyond the scope of an LCA study, then it is safer to exclude explicit biogenic carbon cycling effects such as longer-term sequestration, and instead treat biogenic carbon fluxes as GWP neutral overall (a conservative approach). The least accurate approach is to attribute a large (implying permanent) CO_2 sequestration potential to bioplastics based on initial carbon uptake in the feedstock crops, without considering the end-of-life fate of that carbon. This should be avoided because most of this carbon is likely to be re-released into the atmosphere within a relatively short timeframe – certainly within the 100-year time horizon implicit in the GWP_{100} metric.

2.3.6. End-of-life

The waste management treatments examined for plastic end-of-life and methodological decisions surrounding the modelling of treatment processes vary among studies. Variations include different methods of approaching potential co-products generated at the end-of-life (e.g., electricity generated from incineration), through allocation of co-products derived from wastes, which is known to have large effects on the environmental burdens allocated to main products (Durkin et al., 2019; Yates and Barlow, 2013). Simplification in how the end-of-life processes are represented in LCA can lead to flawed results due to the system not being fully represented, as well as treating the different waste management options as perfectly closed systems, e.g., 100% of waste sent for recycling with no further waste diversion. This is rarely the case, as recycling is seldom closed-loop (i.e., mechanical recycling transforms products back into their original product system), and a significant amount of material is rejected, where it is redirected to other waste management types – a percentage of which may even end up as ocean debris (Bishop et al., 2020). The waste management options of bioplastics often differ from petrochemical plastics. Whilst some bioplastics can enter the same waste streams as conventional plastics (e.g., bio-PET), many can't be recycled alongside petrochemical plastics (within the current infrastructure) due to fundamental differences in their composition. However, bioplastics often have the options of composting or anaerobic digestion which is unavailable for typical petrochemical plastics. In order to ensure a good comparison between the end-of-life of bioplastics and petrochemical plastics, modelling requires the development of transparent, and preferably differentiated, realistic scenarios. Below it is explored how the five main end-of-life waste treatment options of landfill, incineration, recycling, anaerobic digestion, and composting (**Figure 2.1**) are modelled within the 34 reviewed LCA studies which evaluated the life cycle of the plastics from cradle-to-grave.

2.3.6.1. End-of-life scenarios covered by reviewed studies

27 of the reviewed studies modelled incineration, 22 included landfill, 18 included composting, 13 studies explored recycling, and 5 modelled anaerobic digestion (**Table 2.1**). One study also included littering, as discussed in **Chapter 2.3.1.3** (Civancik-Uslu et al., 2019), and one included degradation on agricultural land as an end-of-life option (Patel et al., 2018).

The destinations of the end-of-life fates are modelled differently in almost all reviewed studies, making it difficult to compare between studies. Some of the LCAs explored scenarios of 100% waste directed to individual waste management options. Other studies modelled "hybrid" scenarios of waste treatment to better reflect the diverse flows of plastic waste, based either on current practises or future end-of-life fates. Several studies also included a mix of approaches, or modelled varying percentages of waste management contribution across multiple scenarios. The full breakdown of the end-of-life fate scenarios evaluated in the reviewed literature can be found in **Appendix 3.3** (**Chapter 2.6.3**)**.** Assuming 100% single fate provides an indicative technical potential for comparison across waste management options, but does not give realistic results. When the LCA objectives are to benchmark the environmental impacts of bioplastic against petrochemical plastic impacts, it is important that appropriate and realistic options are modelled, and that sensitivity of results to these choices is undertaken.

Table 2.1 – Waste management systems considered in the pertinent reviewed studies which covered end-of-life waste management. See **Table A2.1** for the full list of studies

Study			Anaerobic Digestion [1]	Composting [1]	Incineration	Landfill	Recycling
1	Benavides et al	2020		Y		Y	
2	Vigil et al.	2020		Y	Y	Y	Y
3	Rodriguez et al.	2020				Y	
4	Nguyen et al.	2020			Y	Y	Y
5	Blanc et al.	2019			Y	Y	
6	Civancik-Uslu et al.	2019			Y	Y	Y [2]
7	Durkin et al.	2019	Y				
9	Giovenzana et al.	2019			Y		
10	Maga et al.	2019			Y	Y	Y
11	Changwichan et al.	2018		Y	Y	Y	Y
12	Choi et al.	2018			Y	Y	Y [2]
13	Dilkes-Hoffman et al.	2018	Y	Y		Y	
14	Fieschi and Pretato	2018		Y	Y [2]	Y [2]	
15	Gabriel et al.	2018			Y		
16	Horowitz et al.	2018				Y	
17	Patel et al.	2018		Y	Y		
18	Semba et al.	2018			Y		
20	Hottle et al.	2017		Y	Y	Y	Y [2]
21	Lorite et al.	2017		Y	Y	Y	
22	Saibuatrong et al.	2017		Y	Y	Y	
24	Belboom and Leonard	2016			Y		
27	Leejarkpai et al.	2016		Y		Y	
30	Razza et al.	2015		Y	Y	Y	Y [2]
33	Papong et al.	2014		Y	Y	Y	Y [3]
34	Van der Harst et al.	2014	Y	Y	Y		Y
36	Deng et al.	2013		Y	Y		
37	Kikuchi et al.	2013			Y		
38	Suwanmanee et al.	2013a		Y	Y [2]	Y [1]	
40	Eerhart et al.	2012			Y		
41	Guo and Murphy	2012	Y	Y [4]	Y [2]	Y	Y [2]
42	Liptow and Tillman	2012			Y	Y	
43	Gironi and Piemonte	2011		Y	Y	Y	Y
44	Piemonte	2011	Y	Y	Y		Y
23 [5]	Unger et al.	2017					

1. Only included in bioplastic waste management. 2. Only included in petrochemical plastic waste management. 3. Chemical recycling for the bioplastic. 4. Home composting and two industrial composting systems. 5. Unclear in what waste management type was covered.

2.3.6.2. End-of-life allocations

Value chains for products are often long and contain stages with multiple output processes. Allocation is the process of "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (ISO, 2006b). So, when a multifunctional process is linked with a process that only provides one functional reference flow, an allocation procedure must be applied in order to partition environmental burdens among the co-products (materials and energy flows) ensuing from the multifunctional process. Allocation is one of the main methodological choices that can potentially cause large variation and disparities in LCA results (Durkin et al., 2019; Yates and Barlow, 2013).

ISO guidelines (ISO, 2006b) recommend the following hierarchy for decisions on allocation: 1) avoiding allocation. by disentangling the unit process that has been recorded as a multi-functional unit process and separating out into two or more mono-functional unit sub-processes, or through system expansion (i.e., expanding the product system to include additional functions related to the co-products); 2) partitioning the inputs and outputs of the system between its different products or functions in a way that reflects the underlying physical relationships between them (that reflects how the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system); 3) the inputs are allocated between the products and functions in a way that reflects other relationships between them, e.g., economic value, mass, volume, or energy. These guidelines can be interpreted to support various approaches to represent the multifunctionality of end-of-life in waste management, which can include: second life from recycling materials; compost from composting; biogas, electricity, heat, and digestate from anaerobic digestion; biochar, electricity, and heat from incineration; and energy from landfill gas.

A system expansion approach for recovery from waste management systems is most common in the reviewed studies, where system credits are calculated to represent displaced materials, e.g., electricity, heat, or fertiliser (Belboom and Léonard, 2016; Civancik-Uslu et al., 2019; Deng et al., 2013; Dilkes-Hoffman et al., 2018; Durkin et al., 2019; Gironi and Piemonte, 2011; Guo et al., 2013; Horowitz et al., 2018; Leejarkpai et al., 2016; Liptow and Tillmanm, 2012; Lorite et al., 2017; Maga et al., 2019; Nguyen et al., 2020; Papong et al., 2014; Patel et al., 2018; Piemonte, 2011; Razza et al., 2015; Saibuatrong et al., 2017; Suwanmanee et al., 2013a; van der Harst et al., 2014). Liptow and Tillman (2012) treated the energy recovered slightly differently in their attributional approach by allocating between both the disposal of LDPE and generation of electricity functions, using the prices as partitioning bases.

When recycling was not represented by expanding the boundaries to provide credits for displacing virgin materials, which is often used in closed-loop recycling (e.g., Maga et al., 2019; Nguyen et al., 2020), the cut-off approach was sometimes used (Civancik-Uslu et al., 2019; Maga et al., 2019), which assigns the recycling process to the next product cycle. Hottle et al. (2017) included a range of approaches to recycling in their study, which included 'substitution-with-equal-quality' (a one-for-one displacement of virgin materials via recycling), 'substitution-with-alternative-material' (bio-PET and bio-PE offset their fossil-based counterparts during recycling), and 'substitution-with-correction-factor' (to represent a 30% material loss during end-of-life processing). Van der Harst et al. (2014) applied credits for the recycled plastic, based on avoided production of virgin material, but corrected according to economic values of virgin and recycled materials as to include the loss of quality. For open-loop recycling modelling, Piemonte (2011) allocated 50% of the benefits and burdens deriving from the recycling process between the two products (first-life and second-life).

In several studies, it was unclear how allocation had occurred (e.g., Blanc et al., 2019; Changwichan et al., 2018; Choi et al., 2018; Fieschi and Pretato, 2018; Gabriel et al., 2018; Giovenzana et al., 2019; Semba et al., 2018; Vigil et al., 2020). Whilst allocation of end-of-life co-products may indeed have been used in these studies, a LCA should clearly state all fundamental methodological decisions, so as to allow clarity in results (ISO, 2006a). Allocation from upstream processes can also affect the results, especially of bioplastics, and is further discussed in **Chapter 2.3.8**. It is clear that 100% recycling to the same quality material (100% closed loop) is impossible, either for bioplastics or petrochemical plastics. End-of-life treatments should be realistic in their assumption to avoid biasing results. Where bioplastics and petrochemical plastics being compared undergo different end-of-life treatments, especially where recycling into new plastic products vary, the number of recycling loops considered within the system boundaries could have a profound effect on results (van der Harst et al., 2016). Many studies lack rigour and transparency in this regard.

2.3.7. Uncertainty Analysis

The wide range of activity data, system boundaries, modelling choices (e.g., allocation method) and end-of-life scenarios required for modelling the whole life cycle of plastics can result in large uncertainties. Uncertainty analysis is often performed to determine how uncertainties in data and assumptions progress in the calculations and how they affect the reliability of the results of the life cycle inventory analysis (ISO, 2006b). ISO guidelines state that the results of uncertainty analysis and data quality analysis should supplement checks within the evaluation phase of the interpretation stage. However, out of the 44 reviewed studies, although many studies included some form of sensitivity analysis, only seven included an uncertainty analysis (Alvarenga et al., 2013; Deng et al., 2013; Forte et al., 2016; Nguyen et al., 2020; Razza et al., 2015; Rodríguez et al., 2020; Vigil et al., 2020).

All these seven studies performed Monte Carlo analysis to propagate error ranges through model parameters. Nguyen et al. (2020) combined Monte Carlo analysis with non-parametric bootstrapping to mitigate error in the financial and environmental outcomes. This extra analysis was performed because it did not require any assumption in data distribution, or normality, and could also be used when a parametric formula for uncertainty was inapplicable. Often the uncertainty of the parameters was estimated with a pedigree matrix to generate standard deviations on the inputs and outputs within each unit process of the study before the Monte Carlo analysis was run (Alvarenga et al., 2013; Rodríguez et al., 2020; Vigil et al., 2020).

An overview of best practices of treating uncertainties in LCA can be found in Igos et al. (2019). ISO standards (ISO, 2006b) state that an analysis of results for uncertainty shall be conducted for studies intended to be used in comparative assertions disclosed to the public. As a high degree of uncertainty surrounds emerging technologies pertaining to bioplastic production, use, and end-of-life, uncertainty analysis should be included in LCA studies to generate robust comparisons and conclusions on the environmental sustainability of bioplastics and petrochemical plastics.

2.3.8. Attributional and consequential LCA modelling choices

There are two main modelling approaches for life cycle inventory analysis: attributional and consequential LCA modelling. Within an attributional LCA, the inputs and outputs are retrospectively attributed to the functional unit of a product system by linking and/or partitioning the unit processes of the system based on a normative approach (Sonnemann and Vigon, 2011). In attributional modelling, all relevant material and energy inputs are based on average supply data to quantify the environmental impacts of a specific system (Ekvall et al., 2016). Attributional modelling is the most common approach used in product system LCAs and the calculation of environmental "footprints". The allocation of co-products from end-of-life wastes was discussed in **Chapter 2.3.6.2**, however, further allocation can occur during the production phase, which is especially relevant for crop feedstock cultivation. Allocation of production phase co-products was partitioned through system expansion (e.g., Nguyen et al., 2020), economic allocation (e.g., Belboom and Leonard, 2016; Changwichan et al., 2018; Deng et al., 2013; Durkin et al., 2019; Eerhart et al., 2012; Forte et al., 2016; Guo and Murphy, 2012; Hansen et al., 2015; Mahalle et al., 2014; Patel et al., 2018; Razza et al., 2015; Rodriguez et al., 2020; Tsiropoulos et al., 2015; van der Harst et al., 2014), mass allocation (e.g., Belboom and Leonard, 2016; Chen et al., 2016; Deng et al., 2013; Durkin et al., 2019; Eerhart et al., 2012; Gironi and Piemonte, 2011; Hottle et al., 2017; Liptow and Tillman, 2012; Papong et al., 2014; Piemonte, 2011; Razza et al., 2015; Semba et al., 2018; Tsiropoulos et al., 2015), and energy allocation (e.g., Belboom and Leonard, 2016; Durkin et al., 2019; Giovenzana et al., 2019; Hansen et al., 2015).

On the other hand, consequential LCA models are prospective as they aim to model the consequences of future decisions. A consequential LCA is a system modelling approach in which activities are included in the product system(s) being evaluated only to the extent that they are expected to *change* as a consequence of a change in demand for the functional unit (LCA-2.0, 2015; Sonnemann and Vigon, 2011). Consequential modelling subsequently uses unconstrained (or marginal) suppliers in the product systems that can increase (or decrease) production if there is an increase (or decrease) in demand for a product or process, as well as for the products and processes which will be substituted in other systems (i.e., system expansion) due to additional production of co-products (Ekvall et al., 2016; Ekvall and Weidema, 2004). Further, Weidema et al. (2018) emphasised the relationship of consequential thinking with responsibility. They stated that the "literal meaning of responsibility implies a focus on consequences that can be meaningfully acted upon and changed". They went on to conclude that, "a consistent socially responsible decision-maker *must* always take responsibility for the activities in the consequential product life cycle". Therefore, consequential LCA is arguably the pertinent methodological approach to assess the decisions of replacing petrochemical plastic with bioplastic materials. So far, consequential studies of bioplastics are not common. Within our reviewed literature, only two studies included clear consequential modelling within the life cycle inventory analysis (Alvarenga et al., 2013; Liptow and Tillmanm, 2012). These studies included iLUC (as described in **Chapter 2.3.4**), avoided allocation through boundary expansion, considered the marginal suppliers to produce the feedstocks, and marginal technologies were modelled for processes such as electricity production. As the development and deployment of bioplastics accelerates, there is a need for consequential LCA studies that evaluate wider environmental outcomes linked to realistic deployment scenarios, representing, *inter-alia*, plausible end-of-life management linked with infrastructure and consumer behaviour. Challenges from consequential modelling may revolve around the difficulties of dissemination of the consequential LCA results to the public and policy makers, as models are often considerably more complex and uncertain than attributional LCA studies.

2.4. Key recommendations

The following criteria are recommended for comprehensive LCA evaluation of bioplastic sustainability, to ensure that genuine environmental savings are achieved, and that one set of major environmental impacts are not simply swapped with another set of impacts as we transition away from petrochemical and towards bio-based plastics:

- Adopt a comprehensive impact assessment methodology such as PEF, or at least select impact categories that capture priority environmental challenges (e.g., those identified in the Planetary Boundaries concept) in order to adequately represent environmental sustainability
- There is a need to identify how plastic littering effects can be integrated into existing impact categories or represented in a new impact category. Whilst there are many projects and researchers looking at the integration of plastic litter into LCA (e.g., MarILCA, 2020), full representation of all environmental impacts attributable to plastic debris within life cycle impact assessment indicators remains some way off. Just some of the many factors that need to be considered when developing a model for the environmental impacts (as discussed in FSLCI,

2020) include: the polymer type and therefore the persistence of the plastic; the size and shape of the plastic; the degradability of the plastic related to the environment the plastic resides in; the magnitude and type of chemical release into the environment, and the subsequent toxic effect; the risk of ingestion or entanglement; the redistribution between environmental compartments; and wide fate uncertainty. All of which will require modelling of regionalised fate and transport modelling, and better understanding of ecological interactions. In the interim, simple reporting on quantities of plastic likely to accumulate in the open environment (based on littering rates and biodegradability) could be included alongside impact category results in LCA studies (e.g., Civancik-Uslu et al., 2019)

- There should be more detailed and transparent reporting of LCI data within plastic LCA studies, and improved effort should be made in presenting the data of the LCA studies so that the models are: 1) easily understandable; 2) transparent; 3) complete; 4) clear; 5) reproducible
- Additives should be included in LCIs for plastic LCA studies unless there is clear evidence that they contribute <1% to <u>all</u> impact categories. A need for more studies specifically evaluating the environmental impacts of these additives was identified, especially surrounding end-of-life fate and impacts
- Land use is a critical aspect of bioplastic life cycles. Significant direct and indirect land use change impacts should be accounted for
- If accounting for temporary biogenic carbon storage, studies should do so carefully with explicit end-of-life accounting of carbon release. Otherwise, it is better to just adopt a simplified approach in which biogenic carbon cycling is treated as GWP neutral
- End-of-life management of all plastics should be based on plausible, representative options appropriate to the plastic type (and infrastructure available), with sensitivity analyses partitioning plastic differently across relevant fates
- Uncertainty and sensitivity analyses should reflect major assumptions related to the above critical issues. Uncertainty analysis is especially critical when creating scenarios due to the potential magnitude of uncertainty the occurs when dealing with the emerging technologies of bioplastics
- There is a need for application of consequential LCA to represent the environmental outcomes of widespread substitution of petrochemical plastics with bioplastics. Shifting to more forward-looking consequential LCAs will be critical to more accurately capture the likely effects of displacing petrochemical plastics with bioplastics via specific scenarios of deployment. Due to the uncertainty of future context and decisions, establishing a suit of consequential models (Yang and Heijungs, 2018) may represent the most reliable way to approximate a "true" result. The process of developing consequential LCAs elucidates linkages that may otherwise be missed, and thus important for informing decision making around bioplastic deployment

2.5. References

Alvarenga, R., Dewulf, J., De Meester, S., Wathelet, A., Villers, J., Thommeret, R., Hruska, Z., 2013. Life cycle assessment of bioethanol-based PVC: Part 2: Consequential approach. Biofuels, Bioprod. Biorefining 7, 396–405. https://doi.org/10.1002/bbb.1398

Andrade, M.C., Winemiller, K.O., Barbosa, P.S., Fortunati, A., Chelazzi, D., Cincinelli, A., Giarrizzo, T., 2019. First account of plastic pollution impacting freshwater fishes in the Amazon: Ingestion of plastic debris by piranhas and other serrasalmids with diverse feeding habits. Environ. Pollut. 244, 766–773. https://doi.org/10.1016/j.envpol.2018.10.088

Belboom, S., Leonard, A., 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 Bioenergy 85, 159–167. https://doi.org/10.1016/j.biombioe.2015.12.014

Belboom, S., Léonard, A., 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

Benavides, P.T., Lee, U., Zarè-Mehrjerdi, O., 2020. Life cycle greenhouse gas emissions and energy use of polylactic acid, bio-derived polyethylene, and fossil-derived polyethylene. J. Clean. Prod. 277, 124010. https://doi.org/10.1016/j.jclepro.2020.124010

Bhattacharya, P., Lin, S., Turner, J.P., Ke, P.C., 2010. Physical adsorption of charged plastic nanoparticles affects algal photosynthesis. J. Phys. Chem. C 114, 16556–16561. https://doi.org/10.1021/jp1054759

BIS, 2009. Study for a simplified LCA methodology adapted to bioproducts: Final Report. Study performed for ADEME.

Bishop, G., Styles, D., Lens, P.N.L., 2020. Recycling of European plastic is a pathway for plastic debris in the ocean. Environ. Int. 142, 105893. https://doi.org/10.1016/j.envint.2020.105893

Blanc, S., Massaglia, S., Brun, F., Peano, C., Mosso, A., Giuggioli, N.R., 2019. Use of Bio-Based Plastics in the Fruit Supply Chain: An Integrated Approach to Assess Environmental, Economic, and Social Sustainability. Sustainability 11. https://doi.org/10.3390/su11092475

Boots, B., Russell, C.W., Green, D.S., 2019. Effects of Microplastics in Soil Ecosystems: Above and Below Ground. Environ. Sci. Technol. 53, 11496–11506. https://doi.org/10.1021/acs.est.9b03304

Brandão, M., Kirschbaum, M.U.F., Cowie, A.L., Hjuler, S.V., 2019. Quantifying the climate change effects of bioenergy systems: Comparison of 15 impact assessment methods. GCB Bioenergy 11, 727–743. https://doi.org/10.1111/gcbb.12593

Brandão, M., Levasseur, A., Kirschbaum, M.U.F., Weidema, B.P., Cowie, A.L., Jørgensen, S.V., Hauschild, M.Z., Pennington, D.W., Chomkhamsri, K., 2012. Key issues and options in accounting for carbon sequestration and temporary storage in life cycle assessment and carbon footprinting. Int. J. Life Cycle Assess. 2012 181 18, 230–240. https://doi.org/10.1007/S11367-012-0451-6

Brodin, M., Vallejos, M., Opedal, M.T., Area, M.C., Chinga-Carrasco, G., 2017. Lignocellulosics as sustainable resources for production of bioplastics - A review. J. Clean. Prod. 162, 646–664. https://doi.org/10.1016/j.jclepro.2017.05.209

Broeren, M.L.M., Kuling, L., Worrell, E., Shen, L., 2017. Environmental impact assessment of six starch plastics focusing on wastewater-derived starch and additives. Resour. Conserv. Recycl. 127, 246–255. https://doi.org/10.1016/j.resconrec.2017.09.001

Broeren, M.L.M., Molenveld, K., van den Oever, M.J.A., Patel, M.K., Worrell, E., Shen, L., 2016. Early-stage sustainability assessment to assist with material selection: a case study for biobased printer panels. J. Clean. Prod. 135, 30–41. https://doi.org/10.1016/j.jclepro.2016.05.159

BSI, 2011. PAS 2050:2011 Specification for the assessment of the life cycle greenhouse gas emissions of goods and services.

Bueno, C., Hauschild, M.Z., Rossignolo, J.A., Ometto, A.R., Mendes, N.C., 2016. Sensitivity analysis of the use of Life Cycle Impact Assessment methods: A case study on building materials. J. Clean. Prod. 112, 2208–2220. https://doi.org/10.1016/j.jclepro.2015.10.006

Changwichan, K., Silalertruksa, T., Gheewala, S.H., 2018. Eco-Efficiency Assessment of Bioplastics Production Systems and End-of-Life Options. Sustainability 10. https://doi.org/10.3390/su10040952

Chen, L.Y., Pelton, R.E.O., Smith, T.M., 2016. Comparative life cycle assessment of fossil and bio-based polyethylene terephthalate (PET) bottles. J. Clean. Prod. 137, 667–676. https://doi.org/10.1016/j.jclepro.2016.07.094

Cherubini, F., Peters, G.P., Bernsten, T., Stromman, A.H., Hertwich, E., 2011. CO2 emissions from biomass combustion for bioenergy: atmospheric decay and contribution to global warming. GCB Bioenergy 3, 413–426. https://doi.org/10.1111/j.1757-1707.2011.01102.x

Choi, B., Yoo, S., Park, S.I., 2018. Carbon Footprint of Packaging Films Made from LDPE, PLA, and PLA/PBAT Blends in South Korea. Sustainability 10. https://doi.org/10.3390/su10072369

Christensen, T.H., Gentil, E., Boldrin, A., Larsen, A.W., Weidema, B.P., Hauschild, M., 2009. C balance, carbon dioxide emissions and global warming potentials. Waste Manag. Res. 27, 707–715. https://doi.org/10.1177/0734242X08096304

Civancik-Uslu, D., Puig, R., Hauschild, M., Fullana-i-Palmer, P., 2019. Life cycle assessment of carrier bags and development of a littering indicator. Sci. Total Environ. 685, 621–630. https://doi.org/10.1016/j.scitotenv.2019.05.372

Clift, R., Brandao, M., 2008. Carbon storage and timing of emissions-a note by Roland Clift and Miguel Brandao. Cent. Environ. Strateg. Work. Pap.

Clukey, K.E., Lepczyk, C.A., Balazs, G.H., Work, T.M., Lynch, J.M., 2017. Investigation of plastic debris ingestion by four species of sea turtles collected as bycatch in pelagic Pacific longline fisheries. Mar. Pollut. Bull. 120, 117–125. https://doi.org/10.1016/j.marpolbul.2017.04.064

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: A review. Mar. Pollut. Bull. 62, 2588–2597. https://doi.org/10.1016/J.MARPOLBUL.2011.09.025

Deng, Y.L., Achten, W.M.J., Van Acker, K., Duflou, J.R., 2013. Life cycle assessment of wheat gluten powder and derived packaging film. Biofuels Bioprod. Biorefining-Biofpr 7, 429–458. https://doi.org/10.1002/bbb.1406

Dilkes-Hoffman, L.S., Lane, J.L., Grant, T., Pratt, S., Lant, P.A., Laycock, B., 2018. Environmental impact of biodegradable food packaging when considering food waste. J. Clean. Prod. 180, 325–334.

https://doi.org/10.1016/j.jclepro.2018.01.169

Durkin, A., Taptygin, I., Kong, Q.Y., Resul, M., Rehman, A., Fernandez, A.M.L., Haryey, A.P., Shah, N., Guo, M., 2019. Scale-up and Sustainability Evaluation of Biopolymer Production from Citrus Waste Offering Carbon Capture and Utilisation Pathway. Chemistryopen 8, 668–688. https://doi.org/10.1002/open.201900015

Eerhart, A., Faaij, A.P.C., Patel, M.K., 2012. Replacing fossil based PET with biobased PEF; process analysis, energy and GHG balance. Energy Environ. Sci. 5, 6407–6422. https://doi.org/10.1039/c2ee02480b

Ekvall, T., Azapagic, A., Finnveden, G., Rydberg, T., Weidema, B.P., Zamagni, A., 2016. Attributional and consequential LCA in the ILCD handbook. Int. J. Life Cycle Assess. 21, 293–296. https://doi.org/10.1007/s11367-015-1026-0

Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. Int. J. Life Cycle Assess. 9, 161–171. https://doi.org/10.1007/BF02994190

Emadian, S.M., Onay, T.T., Demirel, B., 2017. Biodegradation of bioplastics in natural environments. Waste Manag. https://doi.org/10.1016/j.wasman.2016.10.006

European Commission, 2018a. A European Strategy for Plastics in a Circular Economy. COM/2018/028 final. Brussels.

European Commission, 2018b. Proposal for a DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on the reduction of the impact of certain plastic products on the environment. COM/2018/340 final. Bussels.

European Commission, 2018c. PEFCR Guidance document, - Guidance for the development of Product Environmental Footprint Category Rules (PEFCRs) 1–238.

European Commission, 2009. Taking bio-based from promise to market measures to promote the market introduction of innovative bio-based products. https://doi.org/10.2769/34881

European Commission, 2008. DIRECTIVE 2008/98/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 19 November 2008 on waste and repealing certain Directives. Off. J. Eur. Union 51.

European Commission, 1994. EUROPEAN PARLIAMENT AND COUNCIL DIRECTIVE 94/62/EC of 20 December 1994 on packaging and packaging waste. Off. J. Eur. Union L 365, 10.

Fargione, J., Hill, J., Tilman, D., Polasky, S., Hawthorne, P., 2008. Land clearing and the biofuel carbon debt. Science (80-.). 319, 1235–1238. https://doi.org/10.1126/science.1152747

Fazio, S., Biganzioli, F., De Laurentiis, V., Zampori, L., Sala, S., Diaconu, E., 2018. Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2, from ILCD to EF 3.0, EUR 29600 EN. Ispra. https://doi.org/10.2760/002447

Fearnside, P.M., Lashof, D.A., Moura-Costa, P., 2000. Accounting for time in mitigating global warming through land-use change and forestry. Mitig. Adapt. Strateg. Glob. Chang. https://doi.org/10.1023/A:1009625122628

Fieschi, M., Pretato, U., 2018. Role of compostable tableware in food service and waste management. A life cycle assessment study. Waste Manag. 73, 14–25. https://doi.org/10.1016/j.wasman.2017.11.036

Finkbeiner, M., 2014a. Product environmental footprint - Breakthrough or breakdown for policy implementation of life cycle assessment? Int. J. Life Cycle Assess. https://doi.org/10.1007/s11367-013-0678-x

Finkbeiner, M., 2014b. Indirect land use change - Help beyond the hype? Biomass and Bioenergy 62, 218–221. https://doi.org/10.1016/j.biombioe.2014.01.024

Forte, A., Zucaro, A., Basosi, R., Fierro, A., 2016. LCA of 1,4-Butanediol Produced via Direct Fermentation of Sugars from Wheat Straw Feedstock within a Territorial Biorefinery. Materials (Basel). 9. https://doi.org/10.3390/ma9070563

Franco-Trecu, V., Drago, M., Katz, H., Machín, E., Marín, Y., 2017. With the noose around the neck: Marine debris entangling otariid species. Environ. Pollut. 220, 985–989. https://doi.org/10.1016/j.envpol.2016.11.057

FSLCI, 2020. Linking the Life Cycle Inventory and Impact Assessment of Marine Litter and Plastic Emissions. Workshop Report.

Gabriel, C.A., Bortsie-Aryee, N.A., Apparicio-Farrell, N., Farrell, E., 2018. How supply chain choices affect the life cycle impacts of medical products. J. Clean. Prod. 182, 1095–1106. https://doi.org/10.1016/j.jclepro.2018.02.107

Gallagher, J., Styles, D., McNabola, A., Williams, A.P., 2015. Inventory compilation for renewable energy systems: the pitfalls of materiality thresholds and priority impact categories using hydropower case studies. Int. J. Life Cycle Assess. 20, 1701–1707. https://doi.org/10.1007/s11367-015-0976-6

GHGP, 2011. Product Life Cycle Accounting and Reporting Standard GHG Protocol Team.

Giovenzana, V., Casson, A., Beghi, R., Tugnolo, A., Grassi, S., Alamprese, C., Casiraghi, E., Farris, S., Fiorindo, I., Guidetti, R., 2019. Environmental benefits: Traditional vs innovative packaging for olive oil. Chem. Eng. Trans. 75, 193–198. https://doi.org/10.3303/CET1975033

Gironi, F., Piemonte, V., 2011. Life Cycle Assessment of Polylactic Acid and Polyethylene Terephthalate Bottles for Drinking Water. Environ. Prog. Sustain. Energy 30, 459–468. https://doi.org/10.1002/ep.10490

Gregory, M.R., 2009. Environmental implications of plastic debris in marine settings—entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Philos. Trans. R. Soc. B Biol. Sci. 364, 2013–2025. https://doi.org/10.1098/rstb.2008.0265

Guo, M., Murphy, R.J., 2012. Is There a Generic Environmental Advantage for Starch-PVOH Biopolymers Over Petrochemical Polymers? J. Polym. Environ. 20, 976–990. https://doi.org/10.1007/s10924-012-0489-3

Guo, M., Stuckey, D.C., Murphy, R.J., 2013. Is it possible to develop biopolymer production systems independent of fossil fuels? Case study in energy profiling of polyhydroxybutyrate-valerate (PHBV). Green Chem. 15, 706–717. https://doi.org/10.1039/c2gc36546d

Haegerbaeumer, A., Mueller, M.T., Fueser, H., Traunspurger, W., 2019. Impacts of micro- and nano-sized plastic particles on benthic invertebrates: A literature review and gap analysis. Front. Environ. Sci. https://doi.org/10.3389/fenvs.2019.00017

Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E., Purnell, P., 2018. An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. J. Hazard. Mater. 344, 179–199. https://doi.org/10.1016/J.JHAZMAT.2017.10.014

Hansen, A.P., da Silva, G.A., Kulay, L., 2015. Evaluation of the environmental performance of alternatives for polystyrene production in Brazil. Sci. Total Environ. 532, 655–668. https://doi.org/10.1016/j.scitotenv.2015.06.049

Hermabessiere, L., Dehaut, A., Paul-Pont, I., Lacroix, C., Jezequel, R., Soudant, P., Duflos, G., 2017. Occurrence and effects of plastic additives on marine environments and organisms: A review. Chemosphere. https://doi.org/10.1016/j.chemosphere.2017.05.096

Horowitz, N., Frago, J., Mu, D.Y., 2018. Life cycle assessment of bottled water: A case study of Green2O products. Waste Manag. 76, 734–743. https://doi.org/10.1016/j.wasman.2018.02.043

Hottle, T.A., Bilec, M.M., Landis, A.E., 2017. Biopolymer production and end of life comparisons using life cycle assessment. Resour. Conserv. Recycl. 122, 295–306. https://doi.org/10.1016/j.resconrec.2017.03.002

Hottle, T.A., Bilec, M.M., Landis, A.E., 2013. Sustainability assessments of bio-based polymers. Polym. Degrad. Stab. https://doi.org/10.1016/j.polymdegradstab.2013.06.016

Igos, E., Benetto, E., Meyer, R., Baustert, P., Othoniel, B., 2019. How to treat uncertainties in life cycle assessment studies? Int. J. Life Cycle Assess. 24, 794–807. https://doi.org/10.1007/s11367-018-1477-1

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

IPCC, 2006. Guidlines for National Greenhouse Gas Inventories. Volume 4. Agriculture, Forestry and Other Land Use [WWW Document]. URL https://www.ipcc-nggip.iges.or.jp/public/2006gl/vol4.html (accessed 4.30.20).

ISO, 2018. IISO 14067:2018 - Greenhouse gases — Carbon footprint of products — Requirements and guidelines for quantification. Switzerland.

ISO, 2006a. ISO 14040: Environmental management -- Life cycle assessment -- Principles and framework. Geneva.

ISO, 2006b. ISO 14044: Environmental management -- Life cycle assessment -- Requirements and guidelines. Geneva.

Kareiva, P., Watts, S., McDonald, R., Boucher, T., 2007. Domesticated nature: Shaping landscapes and ecosystems for human welfare. Science (80-.). https://doi.org/10.1126/science.1140170

Kikuchi, Y., Hirao, M., Narita, K., Sugiyama, E., Oliveira, S., Chapman, S., Arakaki, M.M., Cappra, C.M., 2013. Environmental Performance of Biomass-Derived Chemical Production: A Case Study on Sugarcane-Derived Polyethylene. J. Chem. Eng. Japan 46, 319–325. https://doi.org/10.1252/jcej.12we227

Kirschbaum, M.U.F., Schlamadinger, B., Cannell, M.G.R., Hamburg, S.P., Karjalainen, T., Kurz, W.A., Prisley, S., Schulze, E.D., Singh, T.P., 2001. A generalised approach of accounting for biospheric carbon stock changes under the Kyoto Protocol. Environ. Sci. Policy 4, 73–85. https://doi.org/10.1016/S1462-9011(01)00018-1

Koelmans, A.A., Besseling, E., Foekema, E., Kooi, M., Mintenig, S., Ossendorp, B.C., Redondo-Hasselerharm, P.E., Verschoor, A., Van Wezel, A.P., Scheffer, M., 2017. Risks of Plastic Debris: Unravelling Fact, Opinion, Perception, and Belief. Environ. Sci. Technol. 51, 11513–11519. https://doi.org/10.1021/acs.est.7b02219

Kühn, S., Bravo Rebolledo, E.L., Van Franeker, J.A., 2015. Deleterious effects of litter on marine life, in: Marine Anthropogenic Litter. Springer International Publishing, pp. 75–116. https://doi.org/10.1007/978-3-319-

16510-3_4

Lambin, E.F., Meyfroidt, P., 2011. Global land use change, economic globalization, and the looming land scarcity. Proc. Natl. Acad. Sci. U. S. A. https://doi.org/10.1073/pnas.1100480108

Law, K., Thompson, R., 2014. Microplastics in the seas. Science (80-.). 345, 144–145. https://doi.org/10.1126/science.1256304

LCA-2.0, 2015. Why and when? - Consequential LCA [WWW Document]. URL https://consequential-lca.org/ (accessed 7.28.20).

Leejarkpai, T., Mungcharoen, T., Suwanmanee, U., 2016. Comparative assessment of global warming impact and eco-efficiency of PS (polystyrene), PET (polyethylene terephthalate) and PLA (polylactic acid) boxes. J. Clean. Prod. 125, 95–107. https://doi.org/10.1016/j.jclepro.2016.03.029

Lenton, T.M., Held, H., Kriegler, E., Hall, J.W., Lucht, W., Rahmstorf, S., Schellnhuber, H.J., 2008. Tipping elements in the Earth's climate system. Proc. Natl. Acad. Sci. U. S. A. 105, 1786–1793. https://doi.org/10.1073/pnas.0705414105

Levasseur, A., Brandão, M., Lesage, P., Margni, M., Pennington, D., Clift, R., Samson, R., 2012. Valuing temporary carbon storage. Nat. Clim. Chang. https://doi.org/10.1038/nclimate1335

Li, W.C., Tse, H.F., Fok, L., 2016. Plastic waste in the marine environment: A review of sources, occurrence and effects. Sci. Total Environ. 566–567, 333–349. https://doi.org/10.1016/J.SCITOTENV.2016.05.084

Liptow, C., Tillmanm, A.M., 2012. A Comparative Life Cycle Assessment Study of Polyethylene Based on Sugarcane and Crude Oil. J. Ind. Ecol. 16, 420–435. https://doi.org/10.1111/j.1530-9290.2011.00405.x

Lorite, G.S., Rocha, J.M., Miilumaki, N., Saavalainen, P., Selkala, T., Morales-Cid, G., Goncalves, M.P., Pongracz, E., Rocha, C.M.R., Toth, G., 2017. Evaluation of physicochemical/microbial properties and life cycle assessment (LCA) of PLA-based nanocomposite active packaging. Lwt-Food Sci. Technol. 75, 305–315. https://doi.org/10.1016/j.lwt.2016.09.004

Maga, D., Hiebel, M., Aryan, V., 2019. A Comparative Life Cycle Assessment of Meat Trays Made of Various Packaging Materials. Sustainability 11. https://doi.org/10.3390/su11195324

Mahalle, L., Alemdar, A., Mihai, M., Legros, N., 2014. A cradle-to-gate life cycle assessment of wood fibre-reinforced polylactic acid (PLA) and polylactic acid/thermoplastic starch (PLA/TPS) biocomposites. Int. J. Life Cycle Assess. 19, 1305–1315. https://doi.org/10.1007/s11367-014-0731-4

Manfredi, S., Allacker, K., Chomkhamsri, K., Pelletier, N., Maia De Souza, D., 2012. Product Environmental Footprint (PEF) Guide. Ispra, Italy.

MarILCA, 2020. Marine Impacts In LCA [WWW Document]. URL https://marilca.org/

Moura Costa, P., Wilson, C., 2000. An equivalence factor between CO2 avoided emissions and sequestration - Description and application in forestry. Mitig. Adapt. Strateg. Glob. Chang. 5, 51–60. https://doi.org/10.1023/A:1009697625521

Müller-Wenk, R., Brandão, M., 2010. Climatic impact of land use in LCA-carbon transfers between vegetation/soil and air. Int. J. Life Cycle Assess. 15, 172–182. https://doi.org/10.1007/s11367-009-0144-y

Narayan, R., 2011. Carbon footprint of bioplastics using biocarbon content analysis and life-cycle assessment. Mrs Bull. 36, 716–721. https://doi.org/10.1557/mrs.2011.210

Nguyen, L.K., Na, S., Hsuan, Y.G., Spatari, S., 2020. Uncertainty in the life cycle greenhouse gas emissions and costs of HDPE pipe alternatives 154.

Nikolic, S., Kiss, F., Mladenovic, V., Bukurov, M., Stankovic, J., Nikoliae, S., Kiss, F., Mladenoviae, V., Bukurov, M., Stankoviae, J., 2015. Corn-based Polylactide vs. PET Bottles - Cradle-to-gate LCA and Implications. Mater. Plast. 52, 517–521.

Owsianiak, M., Laurent, A., Bjørn, A., Hauschild, M.Z., 2014. IMPACT 2002+, ReCiPe 2008 and ILCD's recommended practice for characterization modelling in life cycle impact assessment: A case study-based comparison. Int. J. Life Cycle Assess. 19, 1007–1021. https://doi.org/10.1007/s11367-014-0708-3

Papong, S., Malakul, P., Trungkavashirakun, R., Wenunun, P., Chom-in, T., Nithitanakul, M., Sarobol, E., 2014. Comparative assessment of the environmental profile of PLA and PET drinking water bottles from a life cycle perspective. J. Clean. Prod. 65, 539–550. https://doi.org/10.1016/j.jclepro.2013.09.030

Parker, L., 2019. Biodegradable shopping bags buried for three years didn't degrade [WWW Document]. Natl. Geogr. Mag. URL https://www.nationalgeographic.com/environment/2019/04/biodegradable-shopping-bags-buried-for-three-years-dont-degrade/ (accessed 6.2.20).

Patel, M.K., Bechu, A., Villegas, J.D., Bergez-Lacoste, M., Yeung, K., Murphy, R., Woods, J., Mwabonje, O.N., Ni, Y.Z., Patel, A.D., Gallagher, J., Bryant, D., 2018. Second-generation bio-based plastics are becoming a reality - Non-renewable energy and greenhouse gas (GHG) balance of succinic acid-based plastic end products made from lignocellulosic biomass. Biofuels Bioprod. Biorefining-Biofpr 12, 426–441. https://doi.org/10.1002/bbb.1849

Pawar, P., Shirgaonkar, S., Patil, R., 2016. Plastic marine debris: Sources, distribution and impacts on coastal and ocean biodiversity. PENCIL Publ. Biol. Sci. 3, 40–54.

Pawelzik, P., Carus, M., Hotchkiss, J., Narayan, R., Selke, S., Wellisch, M., Weiss, M., Wicke, B., Patel, M.K., 2013. Critical aspects in the life cycle assessment (LCA) of bio-based materials - Reviewing methodologies and deriving recommendations. Resour. Conserv. Recycl. 73, 211–228. https://doi.org/10.1016/j.resconrec.2013.02.006

Piemonte, V., 2011. Bioplastic Wastes: The Best Final Disposition for Energy Saving. J. Polym. Environ. 19, 988–994. https://doi.org/10.1007/s10924-011-0343-z

Piemonte, V., Gironi, F., 2011. Land-use change emissions: How green are the bioplastics? Environ. Prog. Sustain. Energy 30, 685–691. https://doi.org/10.1002/ep.10518

Poeplau, C., Don, A., 2013. Sensitivity of soil organic carbon stocks and fractions to different land-use changes across Europe. Geoderma 192, 189–201. https://doi.org/10.1016/j.geoderma.2012.08.003

Poon, F.E., Provencher, J.F., Mallory, M.L., Braune, B.M., Smith, P.A., 2017. Levels of ingested debris vary across species in Canadian Arctic seabirds. Mar. Pollut. Bull. 116, 517–520. https://doi.org/10.1016/j.marpolbul.2016.11.051

Razza, F., Degli Innocenti, F., Dobon, A., Aliaga, C., Sanchez, C., Hortal, M., 2015. Environmental profile of a bio-based and biodegradable foamed packaging prototype in comparison with the current benchmark. J. Clean. Prod. 102, 493–500. https://doi.org/10.1016/j.jclepro.2015.04.033

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Suh, S., Weidema, B.P., Pennington, D.W., 2004. Life cycle assessment Part 1: Framework, goal and scope definition, inventory analysis, and applications. Environ. Int. 30, 701–720. https://doi.org/10.1016/j.envint.2003.11.005

Renou, S., Thomas, J.S., Aoustin, E., Pons, M.N., 2008. Influence of impact assessment methods in wastewater treatment LCA. J. Clean. Prod. 16, 1098–1105. https://doi.org/10.1016/j.jclepro.2007.06.003

Rochman, C.M., Cook, A., Koelmans, A.A., 2016. Plastic debris and policy: Using current scientific understanding to invoke positive change. Environ. Toxicol. Chem. 35, 1617–1626. https://doi.org/10.1002/ETC.3408@10.1002/(ISSN)1552-8618.MICROPLASTICS

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., De Wit, C.A., Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature. https://doi.org/10.1038/461472a

Rodríguez, B., Bécares, J., Rodríguez, A., Arcos, J.M., 2013. Incidence of entanglements with marine debris by northern gannets (Morus bassanus) in the non-breeding grounds. Mar. Pollut. Bull. 75, 259–263. https://doi.org/10.1016/j.marpolbul.2013.07.003

Rodríguez, L.J., Fabbri, S., Orrego, C.E., Owsianiak, M., 2020. Comparative life cycle assessment of coffee jar lids made from biocomposites containing poly(lactic acid) and banana fiber. J. Environ. Manage. 266, 110493. https://doi.org/10.1016/j.jenvman.2020.110493

Ryberg, M.W., Owsianiak, M., Richardson, K., Hauschild, M.Z., 2018. Development of a life-cycle impact assessment methodology linked to the Planetary Boundaries framework. Ecol. Indic. 88, 250–262. https://doi.org/10.1016/j.ecolind.2017.12.065

Ryberg, M.W., Owsianiak, M., Richardson, K., Hauschild, M.Z., 2016. Challenges in implementing a Planetary Boundaries based Life-Cycle Impact Assessment methodology. J. Clean. Prod. 139, 450–459. https://doi.org/10.1016/j.jclepro.2016.08.074

Saibuatrong, W., Cheroennet, N., Suwanmanee, U., 2017. Life cycle assessment focusing on the waste management of conventional and bio-based garbage bags. J. Clean. Prod. 158, 319–334. https://doi.org/10.1016/j.jclepro.2017.05.006

Sala, O.E., Chapin, F.S., Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L.R., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. Science (80-.). https://doi.org/10.1126/science.287.5459.1770

Sala, S., Crenna, E., Secchi, M., Sanyé-Mengual, E., 2020. Environmental sustainability of European production and consumption assessed against planetary boundaries. J. Environ. Manage. 269, 110686. https://doi.org/10.1016/j.jenvman.2020.110686

Saley, A.M., Smart, A.C., Bezerra, M.F., Burnham, T.L.U., Capece, L.R., Lima, L.F.O., Carsh, A.C., Williams, S.L., Morgan, S.G., 2019. Microplastic accumulation and biomagnification in a coastal marine reserve situated in a sparsely populated area. Mar. Pollut. Bull. 146, 54–59. https://doi.org/10.1016/j.marpolbul.2019.05.065

Sandin, G., Peters, G.M., Svanstrom, M., 2015. Using the planetary boundaries framework for setting impact-reduction targets in LCA contexts. Int. J. Life Cycle Assess. 20, 1684–1700. https://doi.org/10.1007/s11367-015-0984-6

Schmidt, J.H., Weidema, B.P., Brandão, M., 2015. A framework for modelling indirect land use changes in Life Cycle Assessment. J. Clean. Prod. 99, 230–238. https://doi.org/10.1016/j.jclepro.2015.03.013

Schulp, C.J.E., Nabuurs, G.J., Verburg, P.H., 2008. Future carbon sequestration in Europe-Effects of land use change. Agric. Ecosyst. Environ. 127, 251–264. https://doi.org/10.1016/j.agee.2008.04.010

Searchinger, T., Heimlich, R., Houghton, R.A., Dong, F., Elobeid, A., Fabiosa, J., Tokgoz, S., Hayes, D., Yu, T.H., 2008. Use of U.S. croplands for biofuels increases greenhouse gases through emissions from land-use change. Science (80-.). 319, 1238–1240. https://doi.org/10.1126/science.1151861

Searchinger, T.D., Wirsenius, S., Beringer, T., Dumas, P., 2018. Assessing the efficiency of changes in land use for mitigating climate change. Nature 564, 249–253. https://doi.org/10.1038/s41586-018-0757-z

Semba, T., Sakai, Y., Sakanishi, T., Inaba, A., 2018. Greenhouse gas emissions of 100% bio-derived polyethylene terephthalate on its life cycle compared with petroleum-derived polyethylene terephthalate. J. Clean. Prod. 195, 932–938. https://doi.org/10.1016/j.jclepro.2018.05.069

Shine, K.P., Fuglestvedt, J.S., Hailemariam, K., Stuber, N., 2005. Alternatives to the Global Warming Potential for comparing climate impacts of emissions of greenhouse gases. Clim. Change 68, 281–302. https://doi.org/10.1007/s10584-005-1146-9

Sonnemann, G., Valdivia, S., 2017. Medellin Declaration on Marine Litter in Life Cycle Assessment and Management Facilitated by the Forum for Sustainability through Life Cycle Innovation (FSLCI) in close cooperation with La Red Iberoamericana de Ciclo de Vida (RICV) on Wednesday 14 of June. Int. J. Life Cycle Assess. 22, 1637–1639. https://doi.org/10.1007/s11367-017-1382-z

Sonnemann, G., Vigon, B., 2011. Global Guidance Principles for Life Cycle Assessment Databases, United Nations Environmental Programme. Paris/Pensacola: UNEP/SETAC Life Cycle Initiative.

Soroudi, A., Jakubowicz, I., 2013. Recycling of bioplastics, their blends and biocomposites: A review. Eur. Polym. J. 49, 2839–2858. https://doi.org/10.1016/J.EURPOLYMJ.2013.07.025

Spierling, S., Knupffer, E., Behnsen, H., Mudersbach, M., Krieg, H., Springer, S., Albrecht, S., Herrmann, C., Endres, H.J., Knüpffer, E., Behnsen, H., Mudersbach, M., Krieg, H., Springer, S., Albrecht, S., Herrmann, C., Endres, H.J., 2018a. Bio-based plastics - A review of environmental, social and economic impact assessments. J. Clean. Prod. 185, 476–491. https://doi.org/10.1016/j.jclepro.2018.03.014

Spierling, S., Röttger, C., Venkatachalam, V., Mudersbach, M., Herrmann, C., Endres, H.-J., 2018b. Bio-based Plastics - A Building Block for the Circular Economy? Procedia CIRP 69, 573–578. https://doi.org/10.1016/j.procir.2017.11.017

Steffen, W., Richardson, K., Rockstrom, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sorlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science (80-.). 347, 1259855–1259855. https://doi.org/10.1126/science.1259855

Straub, S., Hirsch, P.E., Burkhardt-Holm, P., 2017. Biodegradable and Petroleum-Based Microplastics Do Not Differ in Their Ingestion and Excretion but in Their Biological Effects in a Freshwater Invertebrate Gammarus fossarum. Int. J. Environ. Res. Public Health 14, 774. https://doi.org/10.3390/ijerph14070774

Suwanmanee, U., Leejarkpai, T., Mungcharoen, T., 2013a. Assessment the Environmental Impacts of Polylactic Acid/Starch and Polyethylene Terephthalate Boxes Using Life Cycle Assessment Methodology: Cradle to Waste Treatments. J. Biobased Mater. Bioenergy 7, 259–266. https://doi.org/10.1166/jbmb.2013.1328

Suwanmanee, U., Varabuntoonvit, V., Chaiwutthinan, P., Tajan, M., Mungcharoen, T., Leejarkpai, T., 2013b. Life cycle assessment of single use thermoform boxes made from polystyrene (PS), polylactic acid, (PLA), and PLA/starch: cradle to consumer gate. Int. J. Life Cycle Assess. 18, 401–417. https://doi.org/10.1007/s11367-012-0479-7

Talsness, C.E., Andrade, A.J.M., Kuriyama, S.N., Taylor, J.A., vom Saal, F.S., 2009. Components of plastic: experimental studies in animals and relevance for human health. Philos. Trans. R. Soc. B Biol. Sci. 364, 2079–2096. https://doi.org/10.1098/rstb.2008.0281

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., Russell, A.E., 2004. Lost at Sea: Where Is All the Plastic? Science (80-.). 304, 838–838. https://doi.org/10.1126/SCIENCE.1094559

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. J. Clean. Prod. 90, 114–127. https://doi.org/10.1016/j.jclepro.2014.11.071

Unger, S.R., Hottle, T.A., Hobbs, S.R., Thiel, C.L., Campion, N., Bilec, M.M., Landis, A.E., 2017. Do single-use medical devices containing biopolymers reduce the environmental impacts of surgical procedures compared with their plastic equivalents? J. Health Serv. Res. Policy 22, 218–225. https://doi.org/10.1177/1355819617705683

van der Harst, E., Potting, J., Kroeze, C., 2016. Comparison of different methods to include recycling in LCAs of aluminium cans and disposable polystyrene cups. Waste Manag. 48, 565–583. https://doi.org/10.1016/j.wasman.2015.09.027

van der Harst, E., Potting, J., Kroeze, C., 2014. Multiple data sets and modelling choices in a comparative LCA of disposable beverage cups. Sci. Total Environ. 494, 129–143. https://doi.org/10.1016/j.scitotenv.2014.06.084

Vanham, D., Leip, A., Galli, A., Kastner, T., Bruckner, M., Uwizeye, A., van Dijk, K., Ercin, E., Dalin, C., Brandão, M., Bastianoni, S., Fang, K., Leach, A., Chapagain, A., Van der Velde, M., Sala, S., Pant, R., Mancini, L., Monforti-Ferrario, F., Carmona-Garcia, G., Marques, A., Weiss, F., Hoekstra, A.Y., 2019. Environmental footprint family to address local to planetary sustainability and deliver on the SDGs. Sci. Total Environ. 693, 133642. https://doi.org/10.1016/j.scitotenv.2019.133642

Vigil, M., Pedrosa-Laza, M., Alvarez Cabal, J., Ortega-Fernández, F., 2020. Sustainability Analysis of Active Packaging for the Fresh Cut Vegetable Industry by Means of Attributional & Consequential Life Cycle Assessment. Sustainability 12, 7207. https://doi.org/10.3390/su12177207

Villarrubia-Gómez, P., Cornell, S.E., Fabres, J., 2018. Marine plastic pollution as a planetary boundary threat – The drifting piece in the sustainability puzzle. Mar. Policy 96, 213–220. https://doi.org/10.1016/j.marpol.2017.11.035

Weidema, B.P., Pizzol, M., Schmidt, J., Thoma, G., 2018. Attributional or consequential Life Cycle Assessment: A matter of social responsibility. J. Clean. Prod. 174, 305–314. https://doi.org/10.1016/j.jclepro.2017.10.340

Wilcox, C., Puckridge, M., Schuyler, Q.A., Townsend, K., Hardesty, B.D., 2018. A quantitative analysis linking sea turtle mortality and plastic debris ingestion. Sci. Rep. 8, 1–11. https://doi.org/10.1038/s41598-018-30038-z

Yang, Y., Heijungs, R., 2018. On the use of different models for consequential life cycle assessment. Int. J. Life Cycle Assess. 23, 751–758. https://doi.org/10.1007/s11367-017-1337-4

Yates, M.R., Barlow, C.Y., 2013. Life cycle assessments of biodegradable, commercial biopolymers - A critical review. Resour. Conserv. Recycl. https://doi.org/10.1016/j.resconrec.2013.06.010

Zhang, D.D., del Rio-Chanona, E.A., Shah, N., 2018. Life cycle assessments for biomass derived sustainable biopolymer & energy co-generation. Sustain. Prod. Consum. 15, 109–118. https://doi.org/10.1016/j.spc.2018.05.002

2.6. Appendices for Chapter 2

2.6.1. Appendix 2.1 – The 44 reviewed studies

Table A2.1 – The 44 LCA studies explored in the review, which explicitly benchmark the environmental impacts of specific bioplastics against petrochemical plastics

Study	*Author(s)*	*Title*	*Year*	*Studied bioplastic*	*Studied petrochemical plastic*
1	Benavides, P.T., Lee, U., Zarè-Mehrjerdi, O.	Life cycle greenhouse gas emissions and energy use of polylactic acid, bio-derived polyethylene, and fossil-derived polyethylene	2020	Bio-PE	HDPE, LDPE
2	Vigil, M.,Pedrosa-Laza, M.,Cabal, J.,Ortega-Fernandez, F.	Sustainability Analysis of Active Packaging for the Fresh Cut Vegetable Industry by Means of Attributional & Consequential Life Cycle Assessment	2020	PLA with reinforced ZnO nanoparticles	PP, PP reinforced with ZnO nanoparticles
3	Rodriguez, L., Fabbri, S.,Orrego, C., Owsianiak, M.	Comparative life cycle assessment of coffee jar lids made from biocomposites containing poly(lactic acid) and banana fiber	2020	Composite of banana fibres and PLA	Composite of banana fibres and HDPE
4	Nguyen, L K., Na, S., Hsuan, Y G., Spatari, S	Uncertainty in the life cycle greenhouse gas emissions and costs of HDPE pipe alternatives	2020	Bio-HDPE	Pristine HDPE, HDPE/PCR, HDPE/PCR/nanoclay
5	Blanc, S., Massaglia, S., Brun, F., Peano, C., Mosso, A., Giuggioli, N.R.	Use of Bio-Based Plastics in the Fruit Supply Chain: An Integrated Approach to Assess Environmental, Economic, and Social Sustainability	2019	Materbi (mulch)/PLA (packaging)	PE (mulch)/PET (packaging)
6	Civancik-Uslu, D., Puig, R., Hauschild, M., Fullana-i-Palmer, P.	Life cycle assessment of carrier bags and development of a littering indicator	2019	Mater-Bi bag	HDPE, LDPE, PP, (paper) bag
7	Durkin, A., Taptygin, I., Kong, Q.Y., Resul, M., Rehman, A., Fernandez, A.M.L., Haryey, A.P., Shah, N., Guo, M.	Scale-up and Sustainability Evaluation of Biopolymer Production from Citrus Waste Offering Carbon Capture and Utilisation Pathway	2019	PLC	PS
8	Fernández-Braña, Á., Feijoo-Costa, G., Dias-Ferreira, C.	Looking beyond the banning of lightweight bags: analysing the role of plastic (and fuel) impacts in waste collection at a Portuguese city	2019	TPS bio-bag	HDPE, recycled HDPE bag
9	Giovenzana, V., Casson, A., Beghi, R., Tugnolo, A., Grassi, S., Alamprese, C., Casiraghi, E., Farris, S., Fiorindo, I., Guidetti, R.	Environmental benefits: Traditional vs innovative packaging for olive oil	2019	Trad packaging (PE/ Al/ PU/ PET)	Bio Packaging (PLA/ Bio-PE/Al)
10	Maga, D., Hiebel, M., Aryan, V.	A Comparative Life Cycle Assessment of Meat Trays Made of Various Packaging Materials	2019	PLA	XPS CC, XPS OC, XPS-EVOH, PS-EVOH, rPET, rPET-PE, APET, PP
11	Changwichan, K., Silalertruksa, T., Gheewala, S.H.	Eco-Efficiency Assessment of Bioplastics Production Systems and End-of-Life Options	2018	PLA, PHA, PBS	PP

12	Choi, B., Yoo, S., Park, S.I.	Carbon Footprint of Packaging Films Made from LDPE, PLA, and PLA/PBAT Blends in South Korea	2018	PLA, PLA/PBAT	LDPE
13	Dilkes-Hoffman, L.S., Lane, J.L., Grant, T., Pratt, S., Lant, P.A., Laycock, B.	Environmental impact of biodegradable food packaging when considering food waste	2018	PHA/TPS	PP
14	Fieschi, M., Pretato, U.	Role of compostable tableware in food service and waste management. A life cycle assessment study	2018	B&C Tableware (PLA/ Mater-Bi)	Traditional tableware (GPPS, HIPS, PP)
15	Gabriel, C.A., Bortsie-Aryee, N.A., Apparicio-Farrell, N., Farrell, E.	How supply chain choices affect the life cycle impacts of medical products.	2018	bioplastic dish	HDPE dish
16	Horowitz, N., Frago, J., Mu, D.Y.	Life cycle assessment of bottled water: A case study of Green2O products	2018	PLA bottle	rPET, PET, ENSO
17	Patel, M.K., Bechu, A., Villegas, J.D., Bergez-Lacoste, M., Yeung, K., Murphy, R., Woods, J., Mwabonje, O.N., Ni, Y.Z., Patel, A.D., Gallagher, J., Bryant, D.	Second-generation bio-based plastics are becoming a reality - Non-renewable energy and greenhouse gas (GHG) balance of succinic acid-based plastic end products made from lignocellulosic biomass	2018	PBS	PP, PET, PE
18	Semba, T., Sakai, Y., Sakanishi, T., Inaba, A.	Greenhouse gas emissions of 100% bio-derived polyethylene terephthalate on its life cycle compared with petroleum-derived polyethylene terephthalate	2018	bio-PET (30, 100)	PET
19	Zhang, D.D., del Rio-Chanona, E.A., Shah, N.	Life cycle assessments for biomass derived sustainable biopolymer & energy co-generation	2018	PLC	PS
20	Hottle, T.A., Bilec, M.M., Landis, A.E.	Biopolymer production and end of life comparisons using life cycle assessment	2017	PLA, TPS, Bio-PE and Bio-PET	PET, HDPE, LDPE
21	Lorite, G.S., Rocha, J.M., Miilumaki, N., Saavalainen, P., Selkala, T., Morales-Cid, G., Goncalves, M.P., Pongracz, E., Rocha, C.M.R., Toth, G.	Evaluation of physicochemical/microbial properties and life cycle assessment (LCA) of PLA-based nanocomposite active packaging	2017	PLA	PET
22	Saibuatrong, W., Cheroennet, N., Suwanmanee, U.	Life cycle assessment focusing on the waste management of conventional and bio-based garbage bags	2017	Bio-PE, PBAT/Starch	PE
23	Unger, S.R., Hottle, T.A., Hobbs, S.R., Thiel, C.L., Campion, N., Bilec, M.M., Landis, A.E.	Do single-use medical devices containing biopolymers reduce the environmental impacts of surgical procedures compared with their plastic equivalents?	2017	PLA	LDPE, PP, Polyisoprene, Nitrile, Neoprene
24	Belboom, S., Léonard, A.	Does biobased polymer achieve better environmental impacts than fossil polymer? Comparison of fossil HDPE and biobased HDPE produced from sugar beet and wheat	2016	Bio-HDPE	HDPE
25	Chen, L.Y., Pelton, R.E.O., Smith, T.M.	Comparative life cycle assessment of fossil and bio-based polyethylene terephthalate (PET) bottles	2016	Bio-PET (30, 70, 100)	PET
26	Forte, A., Zucaro, A., Basosi, R., Fierro, A.	LCA of 1,4-Butanediol Produced via Direct Fermentation of Sugars from Wheat Straw Feedstock within a Territorial Biorefinery	2016	Bio-BDO	Fossil-BDO

27	Leejarkpai, T., Mungcharoen, T., Suwanmanee, U.	Comparative assessment of global warming impact and eco-efficiency of PS (polystyrene), PET (polyethylene terephthalate) and PLA (polylactic acid) boxes	2016	PLA (w/ LUC AND w/o LUC)	PS, PET
28	Hansen, A.P., da Silva, G.A., Kulay, L.	Evaluation of the environmental performance of alternatives for polystyrene production in Brazil	2015	BIO-PS	PS (HI, GP)
29	Nikolic, S., Kiss, F., Mladenovic, V., Bukurov, M., Stankovic, J., Nikoliae, S., Kiss, F., Mladenoviae, V., Bukurov, M., Stankoviae, J.	Corn-based Polylactide vs. PET Bottles - Cradle-to-gate LCA and Implications	2015	PLA	PET
30	Razza, F., Degli Innocenti, F., Dobon, A., Aliaga, C., Sanchez, C., Hortal, M.	Environmental profile of a bio-based and biodegradable foamed packaging prototype in comparison with the current benchmark	2015	Starch-based foam prototype	EPS
31	Tsiropoulos, I., Faaij, A.P.C., Lundquist, L., Schenker, U., Briois, J.F., Patel, M.K.	Life cycle impact assessment of bio-based plastics from sugarcane ethanol	2015	Bio-PET, Bio-HDPE	PET, HDPE
32	Mahalle, L., Alemdar, A., Mihai, M., Legros, N.	A cradle-to-gate life cycle assessment of wood fibre-reinforced polylactic acid (PLA) and polylactic acid/thermoplastic starch (PLA/TPS) biocomposites	2014	Woodfibre/PLA/TPS, Woodfibre/ PLA	PP
33	Papong, S., Malakul, P., Trungkavashirakun, R., Wenunun, P., Chom-in, T., Nithitanakul, M., Sarobol, E.	Comparative assessment of the environmental profile of PLA and PET drinking water bottles from a life cycle perspective	2014	PLA	PET
34	van der Harst, E., Potting, J., Kroeze, C.	Multiple data sets and modelling choices in a comparative LCA of disposable beverage cups	2014	PLA	PS, (Biopaper)
35	Alvarenga, R.A., Dewulf, J., De Meester, S., Wathelet, A., Villers, J., Thommeret, R., Hruska, Z.	Life cycle assessment of bioethanol-based PVC: Part 2: Consequential approach.	2013	Bioethanol-based- PVC	PVC
36	Deng, Y.L., Achten, W.M.J., Van Acker, K., Duflou, J.R.	Life cycle assessment of wheat gluten powder and derived packaging film	2013	Gluten Film, PLA	LDPE
37	Kikuchi, Y., Hirao, M., Narita, K., Sugiyama, E., Oliveira, S., Chapman, S., Arakaki, M.M., Cappra, C.M.	Environmental Performance of Biomass-Derived Chemical Production: A Case Study on Sugarcane-Derived Polyethylene	2013	BIO-PE	Fossil-PE
38	Suwanmanee, U., Leejarkpai, T., Mungcharoen, T.	Assessment the Environmental Impacts of Polylactic Acid/Starch and Polyethylene Terephthalate Boxes Using Life Cycle Assessment Methodology: Cradle to Waste Treatments	2013	PLA/Starch	PET
39	Suwanmanee, U., Varabuntoonvit, V., Chaiwutthinan, P., Tajan, M., Mungcharoen, T., Leejarkpai, T.	Life cycle assessment of single use thermoform boxes made from polystyrene (PS), polylactic acid, (PLA), and PLA/starch: cradle to consumer gate	2013	PLA, PLA/Starch	PS
40	Eerhart, A., Faaij, A.P.C., Patel, M.K.	Replacing fossil based PET with biobased PEF; process analysis, energy and GHG balance	2012	PEF	PET

41	Guo, M., Murphy, R.J.	Is There a Generic Environmental Advantage for Starch-PVOH Biopolymers Over Petrochemical Polymers?	2012	Starch-PVOH	PE/EPS
42	Liptow, C., Tillman, A.-M.	A Comparative Life Cycle Assessment Study of Polyethylene Based on Sugarcane and Crude Oil	2012	Bio-LDPE	LDPE
43	Gironi, F., Piemonte, V.	Life Cycle Assessment of Polylactic Acid and Polyethylene Terephthalate Bottles for Drinking Water	2011	PLA	PET
44	Piemonte, V.	Bioplastic Wastes: The Best Final Disposition for Energy Saving	2011	PLA, Mater-Bi	PE, PET

2.6.2. Appendix 3.2 – Impact categories covered by reviewed studies

Table A2.2 – Impact categories covered by the reviewed studies, grouped by similar LCA impact categories

Study	Climate Change/ GWP	Photochemical oxidant /ozone formation/ Smog	Ozone Depletion	Particulate matter formation / respiratory inorganics	Acidification potential	Terrestrial acidification	Aquatic acidification	Acidification/ Eutrophication	Eutrophication potential	Freshwater Eutrophication	Terrestrial Eutrophication	Marine Eutrophication	Aquatic eutrophication	Fossil depletion	Metal /mineral Depletion	Abiotic resource depletion	Mineral, fossil & ren resource depletion	Water consumption/ depletion	Freshwater Ecotoxity	Terrestrial Ecotoxity	Ecotoxicity	Marine Ecotoxicity	Aquatic ecotoxicity	Human Toxicity - Cancer effects	Human Toxicity - Non-Cancer effects	Human toxicity (undefined)	Ironizing radiation	Non-Renewable Energy	Cumulative energy demand/ Total energy	Primary energy consumption	Renewable Energy (biomass, other)	Respiratory organics	Respiratory effects	Land-use / land occupation	Agricultural land occupation	Urban land occupation	Natural land transformation	Other	SUM
1	1													1																									**2**
2	1	1	1	1		1				1		1		1	1			1	1	1		1				1	1								1	1	1		**18**
3	1	1	1	1		1				1		1		1	1			1	1	1		1				1	1								1	1	1		**18**
4	1																																						**1**
5	1																											1											**2**
6	1	1			1				1							1		1												1								1	**8**
7	1	1	1	1		1				1		1		1				1																					**9**
8	1	1		1		1				1				1	1				1	1																		1	**10**
9	1																1	1	1					1	1									1					**7**
10	1	1	1	1	1					1	1	1				1		1									1							1					**12**
11	1	1	1	1		1				1		1		1	1			1	1	1		1				1	1								1	1	1		**18**
12	1																																						**1**

13	1																	1																					2
14	1	1	1	1	1					1	1	1					1	1	1					1	1		1							1					15
15	1	1		1		1						1																											5
16	1		1	1		1	1						1		1					1			1	1	1			1				1		1					14
17	1																											1											2
18	1																																						1
19	1		1											1				1	1	1						1									1				8
20	1	1	1		1				1					1							1			1	1								1						10
21	1		1	1		1	1						1		1					1			1	1	1		1	1				1		1				1	16
22	1			1				1						1							1																		5
23	1	1	1		1				1												1			1	1				1				1						10
24	1	1	1	1	1					1	1			1	1		1																	1					11
25	1	1	1	1	1						1			1							1																		8
26	1	1	1	1		1				1		1		1				1																					9
27	1																																						1
28	1	1				1								1				1																					5
29	1																												1										2
30	1	1	1		1				1							1												1											7
31	1									1								1										1						1					5
32	1	1	1			1			1					1							1			1	1			1	1		2		1						14
33	1				1				1																	1			1										5
34	1	1	1		1				1							1			1	1		1				1			1										11
35	1	1	1	1	1					1		1				1		1			1					1	1							1				1	14
36	1	1	1	1		1				1		1		1	1			1	1	1		1				1	1								1	1	1		18
37	1																																						1
38	1				1				1																			1											4
39	1	1			1																																		3

40	1																										1												**2**
41	1	1	1		1				1							1			1	1		1				1													**10**
42	1	1			1				1																					1									**5**
43	1			1				1						1	1						1			1			1					1		1				1	**11**
44	1																												1										**2**
SUM	**44**	**23**	**20**	**17**	**16**	**11**	**2**	**2**	**10**	**12**	**4**	**10**	**2**	**16**	**9**	**6**	**3**	**15**	**10**	**10**	**7**	**6**	**2**	**8**	**7**	**9**	**9**	**9**	**6**	**2**	**2**	**3**	**3**	**9**	**5**	**4**	**4**	**5**	**342**

2.6.3. Appendix 3.3 – End-of-life scenarios covered by reviewed studies

The most common method of modelling end-of-life processes is to model different scenarios of 100% waste directed to individual waste management options in order to then compare across products and systems, or across individual management options (e.g., Belboom and Léonard, 2016; Benavides et al., 2020; Choi et al., 2018; Deng et al., 2013; Dilkes-Hoffman et al., 2018; Durkin et al., 2019; Eerhart et al., 2012; Gabriel et al., 2018; Giovenzana et al., 2019; Guo et al., 2013; Horowitz et al., 2018; Kikuchi et al., 2013; Liptow and Tillman, 2012; Nguyen et al., 2020; Patel et al., 2018; Piemonte, 2011; Rodriguez et al., 2020; Semba et al., 2018; Suwanmanee et al., 2013a; van der Harst et al., 2014).

Many other studies modelled "hybrid" scenarios of waste treatment to better reflect the diverse flows of plastic waste. Some studies attempted to model current practises within scenarios. For example, Blanc et al. (2019) modelled that 20% of all plastics were incinerated and the remaining 80% were disposed of in landfills. Civancik-Uslu et al. modelled current end-of-life scenarios depending on the current Spanish waste management for each plastic polymer type. Razza et al. (2015) modelled current waste management scenarios of 41.3% composting + 21.3 incineration + 37.4% landfill for bioplastic, and 0.5% recycling + 47.4% incineration + 52.1% landfill for petrochemical plastic, followed by sensitivity analysis of higher recycling rates both for petrochemical plastic (30% and 50%) and bioplastic (50% composting).

The approach of modelling waste management to include the typical practices and then compare with future scenarios (including 100% directed waste) was common in the consulted studies. In the study by Maga et al. (2019), two scenarios of end-of-life waste treatment were explored. Scenario 1 investigated the current end-of-life management situation in Europe, based on the polymer type, whereas a second suite of scenarios was based on the European strategy for plastics in a circular economy in 2030. Changwichan et al. (2018) modelled five different end-of-life scenarios: 75% landfill + 25% recycling, 70% landfilling + 30% composting, 100% composting, 100% recycling, and 100% incineration. Fieschi and Pretato (2018) modelled 55% landfill + 45% incineration (which was considered the European average) and 100% composting for the bioplastic. Lorite et al. (2017) created two scenarios for the end-of-life phase. The first modelled the current plastic packages treatment scenario in Europe (PET: 60% incinerated + 40% landfilled, PLA: 20% composted + 40% incinerated + 40% landfilled), and the second modelled the desired scenario for the future (PET: 100% incinerated, PLA: 100% composted). Vigil et al. (2020) modelled current MSW of PP as a reference system, but up to 100% composting for the ZnO-PLA scenario, and 50% incineration and 50% landfill for the ZnO-PP scenario.

Several studies also compared varying percentages of waste management contribution, without including the current waste management practices. Hottle et al. (2017) presented their waste management results as a continuous range of two end-of-life scenarios for each polymer: from 100% landfilled to 100% recycled, or from 100% landfilled to 100% composted. Saibuatrong et al. (2017) explored 6 scenarios: 100% scenarios of landfill (with and without energy recovery), incineration, and composting, as well as 50% landfill + 50% composting, and 50% incineration + 50% composting. Gironi and Piemonte (2011) modelled 100% of waste diverted to incineration, recycling, and landfilling for both PLA and PET bottles, with 100% composting also considered for the PLA. Further, the following hybrid scenarios for the end-of-life of the product were considered: PET: 70% recycling + 30% landfilling, 70% recycling + 30% incineration, 50% recycling + 30% incineration + 20% landfilling, and PLA: 50% recycling + 50% landfill, 50% recycling + 50% composting, and 40% recycling + 30% composting + 15% landfilling + 15% incineration. Leejarkpai et al. (2016) include all three scenarios mentioned, where they explored: 100% landfill, 96% landfill + 4% composting (based on current recycling scenarios in Thailand), 50% landfill + 50% composting, and 100% composting. Papong et al. (2014) covered the scenarios of PLA as 100% composting, incineration landfill (with and without energy recovery), and chemical recycling, but also 80% composting + 20% landfill, and 80% composting + 20% incineration.

2.6.4. Appendices references

Alvarenga, R.A., Dewulf, J., De Meester, S., Wathelet, A., Villers, J., Thommeret, R., Hruska, Z., 2013. Life cycle assessment of bioethanol-based PVC: Part 2: Consequential approach. Biofuels, Bioprod. Biorefining 7, 396–405. https://doi.org/10.1002/bbb.1398

Belboom, S., Léonard, A., 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

Benavides, P., Lee, U., Zare-Mehrjerdi, O., 2020 Life cycle greenhouse gas emissions and energy use of polylactic acid, bio-derived polyethylene, and fossil-derived polyethylene. J. Clean. Prod. 277, 124010. https://doi.org/10.1016/j.jclepro.2020.124010

Blanc, S., Massaglia, S., Brun, F., Peano, C., Mosso, A., Giuggioli, N.R., 2019. Use of Bio-Based Plastics in the Fruit Supply Chain: An Integrated Approach to Assess Environmental, Economic, and Social Sustainability. Sustainability 11. https://doi.org/10.3390/su11092475

Changwichan, K., Silalertruksa, T., Gheewala, S.H., 2018. Eco-Efficiency Assessment of Bioplastics Production Systems and End-of-Life Options. Sustainability 10. https://doi.org/10.3390/su10040952

Chen, L.Y., Pelton, R.E.O., Smith, T.M., 2016. Comparative life cycle assessment of fossil and bio-based polyethylene terephthalate (PET) bottles. J. Clean. Prod. 137, 667–676. https://doi.org/10.1016/j.jclepro.2016.07.094

Choi, B., Yoo, S., Park, S.I., 2018. Carbon Footprint of Packaging Films Made from LDPE, PLA, and PLA/PBAT Blends in South Korea. Sustainability 10. https://doi.org/10.3390/su10072369

Civancik-Uslu, D., Puig, R., Hauschild, M., Fullana-i-Palmer, P., 2019. Life cycle assessment of carrier bags and development of a littering indicator. Sci. Total Environ. 685, 621–630. https://doi.org/10.1016/j.scitotenv.2019.05.372

Deng, Y.L., Achten, W.M.J., Van Acker, K., Duflou, J.R., 2013. Life cycle assessment of wheat gluten powder and derived packaging film. Biofuels Bioprod. Biorefining-Biofpr 7, 429–458. https://doi.org/10.1002/bbb.1406

Dilkes-Hoffman, L.S., Lane, J.L., Grant, T., Pratt, S., Lant, P.A., Laycock, B., 2018. Environmental impact of biodegradable food packaging when considering food waste. J. Clean. Prod. 180, 325–334. https://doi.org/10.1016/j.jclepro.2018.01.169

Durkin, A., Taptygin, I., Kong, Q.Y., Resul, M., Rehman, A., Fernandez, A.M.L., Haryey, A.P., Shah, N., Guo, M., 2019. Scale-up and Sustainability Evaluation of Biopolymer Production from Citrus Waste Offering Carbon Capture and Utilisation Pathway. Chemistryopen 8, 668–688. https://doi.org/10.1002/open.201900015

Eerhart, A., Faaij, A.P.C., Patel, M.K., 2012. Replacing fossil based PET with biobased PEF; process analysis, energy and GHG balance. Energy Environ. Sci. 5, 6407–6422. https://doi.org/10.1039/c2ee02480b

Fernández-Braña, Á., Feijoo-Costa, G., Dias-Ferreira, C., 2019. Looking beyond the banning of lightweight bags: analysing the role of plastic (and fuel) impacts in waste collection at a Portuguese city. Environ. Sci. Pollut. Res. https://doi.org/10.1007/s11356-019-05938-w

Fieschi, M., Pretato, U., 2018. Role of compostable tableware in food service and waste management. A life cycle assessment study. Waste Manag. 73, 14–25. https://doi.org/10.1016/j.wasman.2017.11.036

Forte, A., Zucaro, A., Basosi, R., Fierro, A., 2016. LCA of 1,4-Butanediol Produced via Direct Fermentation of Sugars from Wheat Straw Feedstock within a Territorial Biorefinery. Materials (Basel). 9. https://doi.org/10.3390/ma9070563

Gabriel, C.A., Bortsie-Aryee, N.A., Apparicio-Farrell, N., Farrell, E., 2018. How supply chain choices affect the life cycle impacts of medical products. J. Clean. Prod. 182, 1095–1106. https://doi.org/10.1016/j.jclepro.2018.02.107

Giovenzana, V., Casson, A., Beghi, R., Tugnolo, A., Grassi, S., Alamprese, C., Casiraghi, E., Farris, S., Fiorindo, I., Guidetti, R., 2019. Environmental benefits: Traditional vs innovative packaging for olive oil. Chem. Eng. Trans. 75, 193–198. https://doi.org/10.3303/CET1975033

Gironi, F., Piemonte, V., 2011. Life Cycle Assessment of Polylactic Acid and Polyethylene Terephthalate Bottles for Drinking Water. Environ. Prog. Sustain. Energy 30, 459–468. https://doi.org/10.1002/ep.10490

Guo, M., Murphy, R.J., 2012. Is There a Generic Environmental Advantage for Starch-PVOH Biopolymers Over Petrochemical Polymers? J. Polym. Environ. 20, 976–990. https://doi.org/10.1007/s10924-012-0489-3

Hansen, A.P., da Silva, G.A., Kulay, L., 2015. Evaluation of the environmental performance of alternatives for polystyrene production in Brazil. Sci. Total Environ. 532, 655–668. https://doi.org/10.1016/j.scitotenv.2015.06.049

Horowitz, N., Frago, J., Mu, D.Y., 2018. Life cycle assessment of bottled water: A case study of Green2O products. Waste Manag. 76, 734–743. https://doi.org/10.1016/j.wasman.2018.02.043

Hottle, T.A., Bilec, M.M., Landis, A.E., 2017. Biopolymer production and end of life comparisons using life cycle assessment. Resour. Conserv. Recycl. 122, 295–306. https://doi.org/10.1016/j.resconrec.2017.03.002

Kikuchi, Y., Hirao, M., Narita, K., Sugiyama, E., Oliveira, S., Chapman, S., Arakaki, M.M., Cappra, C.M., 2013. Environmental Performance of Biomass-Derived Chemical Production: A Case Study on Sugarcane-Derived Polyethylene. J. Chem. Eng. Japan 46, 319–325. https://doi.org/10.1252/jcej.12we227

Leejarkpai, T., Mungcharoen, T., Suwanmanee, U., 2016. Comparative assessment of global warming impact and eco-efficiency of PS (polystyrene), PET (polyethylene terephthalate) and PLA (polylactic acid) boxes. J. Clean. Prod. 125, 95–107. https://doi.org/10.1016/j.jclepro.2016.03.029

Liptow, C., Tillman, A.-M., 2012. A Comparative Life Cycle Assessment Study of Polyethylene Based on Sugarcane and Crude Oil. J. Ind. Ecol. 16, 420–435. https://doi.org/10.1111/j.1530-9290.2011.00405.x

Lorite, G.S., Rocha, J.M., Miilumaki, N., Saavalainen, P., Selkala, T., Morales-Cid, G., Goncalves, M.P., Pongracz, E., Rocha, C.M.R., Toth, G., 2017. Evaluation of physicochemical/microbial properties and life cycle assessment (LCA) of PLA-based nanocomposite active packaging. Lwt-Food Sci. Technol. 75, 305–315. https://doi.org/10.1016/j.lwt.2016.09.004

Maga, D., Hiebel, M., Aryan, V., 2019. A Comparative Life Cycle Assessment of Meat Trays Made of Various Packaging Materials. Sustainability 11. https://doi.org/10.3390/su11195324

Mahalle, L., Alemdar, A., Mihai, M., Legros, N., 2014. A cradle-to-gate life cycle assessment of wood fibre-reinforced polylactic acid (PLA) and polylactic acid/thermoplastic starch (PLA/TPS) biocomposites. Int. J. Life Cycle Assess. 19, 1305–1315. https://doi.org/10.1007/s11367-014-0731-4

Nguyen, L.K., Na, S., Hsuan, Y.G., Spatari, S., 2020. Uncertainty in the life cycle greenhouse gas emissions and costs of HDPE pipe alternatives 154.

Nikolic, S., Kiss, F., Mladenovic, V., Bukurov, M., Stankovic, J., Nikoliae, S., Kiss, F., Mladenoviae, V., Bukurov, M., Stankoviae, J., 2015. Corn-based Polylactide vs. PET Bottles - Cradle-to-gate LCA and Implications. Mater. Plast. 52, 517–521.

Papong, S., Malakul, P., Trungkavashirakun, R., Wenunun, P., Chom-in, T., Nithitanakul, M., Sarobol, E., 2014. Comparative assessment of the environmental profile of PLA and PET drinking water bottles from a life cycle perspective. J. Clean. Prod. 65, 539–550. https://doi.org/10.1016/j.jclepro.2013.09.030

Patel, M.K., Bechu, A., Villegas, J.D., Bergez-Lacoste, M., Yeung, K., Murphy, R., Woods, J., Mwabonje, O.N., Ni, Y.Z., Patel, A.D., Gallagher, J., Bryant, D., 2018. Second-generation bio-based plastics are becoming a reality - Non-renewable energy and greenhouse gas (GHG) balance of succinic acid-based plastic end products made from lignocellulosic biomass. Biofuels Bioprod. Biorefining-Biofpr 12, 426–441. https://doi.org/10.1002/bbb.1849

Piemonte, V., 2011. Bioplastic Wastes: The Best Final Disposition for Energy Saving. J. Polym. Environ. 19, 988–994. https://doi.org/10.1007/s10924-011-0343-z

Razza, F., Degli Innocenti, F., Dobon, A., Aliaga, C., Sanchez, C., Hortal, M., 2015. Environmental profile of a bio-based and biodegradable foamed packaging prototype in comparison with the current benchmark. J. Clean. Prod. 102, 493–500. https://doi.org/10.1016/j.jclepro.2015.04.033

Rodríguez, L.J., Fabbri, S., Orrego, C.E., Owsianiak, M., 2020. Comparative life cycle assessment of coffee jar lids made from biocomposites containing poly(lactic acid) and banana fiber. J. Environ. Manage. 266, 110493. https://doi.org/10.1016/j.jenvman.2020.110493

Saibuatrong, W., Cheroennet, N., Suwanmanee, U., 2017. Life cycle assessment focusing on the waste management of conventional and bio-based garbage bags. J. Clean. Prod. 158, 319–334. https://doi.org/10.1016/j.jclepro.2017.05.006

Semba, T., Sakai, Y., Sakanishi, T., Inaba, A., 2018. Greenhouse gas emissions of 100% bio-derived polyethylene terephthalate on its life cycle compared with petroleum-derived polyethylene terephthalate. J. Clean. Prod. 195, 932–938. https://doi.org/10.1016/j.jclepro.2018.05.069

Suwanmanee, U., Leejarkpai, T., Mungcharoen, T., 2013. Assessment the Environmental Impacts of Polylactic Acid/Starch and Polyethylene Terephthalate Boxes Using Life Cycle Assessment Methodology: Cradle to Waste Treatments. J. Biobased Mater. Bioenergy 7, 259–266. https://doi.org/10.1166/jbmb.2013.1328

Suwanmanee, U., Varabuntoonvit, V., Chaiwutthinan, P., Tajan, M., Mungcharoen, T., Leejarkpai, T., 2013. Life cycle assessment of single use thermoform boxes made from polystyrene (PS), polylactic acid, (PLA), and PLA/starch: cradle to consumer gate. Int. J. Life Cycle Assess. 18, 401–417. https://doi.org/10.1007/s11367-012-0479-7

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. J. Clean. Prod. 90, 114–127. https://doi.org/10.1016/j.jclepro.2014.11.071

Unger, S.R., Hottle, T.A., Hobbs, S.R., Thiel, C.L., Campion, N., Bilec, M.M., Landis, A.E., 2017. Do single-use medical devices containing biopolymers reduce the environmental impacts of surgical procedures compared with their plastic equivalents? J. Health Serv. Res. Policy 22, 218–225. https://doi.org/10.1177/1355819617705683

van der Harst, E., Potting, J., Kroeze, C., 2014. Multiple data sets and modelling choices in a comparative LCA of disposable beverage cups. Sci. Total Environ. 494, 129–143. https://doi.org/10.1016/j.scitotenv.2014.06.084

Vigil, M., Pedrosa-Laza, M., Alvarez Cabal, J., Ortega-Fernández, F., 2020. Sustainability Analysis of Active Packaging for the Fresh Cut Vegetable Industry by Means of Attributional & Consequential Life Cycle Assessment. Sustainability 12, 7207. https://doi.org/10.3390/su12177207

Zhang, D.D., del Rio-Chanona, E.A., Shah, N., 2018. Life cycle assessments for biomass derived sustainable biopolymer & energy co-generation. Sustain. Prod. Consum. 15, 109–118. https://doi.org/10.1016/j.spc.2018.05.002

3. Recycling of European plastic is a pathway for plastic debris in the ocean

Abstract

Polyethylene (PE) is one of the most common types of plastic. Whilst an increasing share of post-consumer plastic waste from Europe is collected for recycling, 46% of separated PE waste is exported outside of the source country (including intra-EU trade). The fate of this exported European plastic is not well known. This study integrated data on PE waste flows in 2017 from UN Comtrade, an open repository providing detailed international trade data, with best available information on waste management in destination countries, to model the fate of PE exported for recycling from Europe (EU-28, Norway, and Switzerland) into: recycled high-density PE (HDPE) and low-density PE (LDPE) resins, "landfill", incineration, and ocean debris. Data uncertainty was reflected in three scenarios representing high, low, and average recovery efficiency factors in material recovery facilities and reprocessing facilities, and different ocean debris fate factors. The fates of exported PE were then linked back to the individual European countries of export. Our study estimated that 83,187 Mg (tonnes) (range: 32,115 Mg – 180,558 Mg), or 3% (1% − 7%) of exported European PE in 2017 ended up in the ocean, indicating an important and hitherto undocumented pathway of plastic debris entering the oceans. The countries with the greatest percentage of exported PE ending up as recycled HDPE or LDPE were Luxembourg and Switzerland (90% recycled for all scenarios), whilst the country with the lowest share of exported PE being recycled was the United Kingdom (59% – 80%, average 69% recycled). The results showed strong, significant positive relationships between the percentage of PE exported out of Europe and the percentage of exports which potentially end up as ocean debris. Export countries may not be the ultimate countries of origin owing to complex intra-EU trade in PE waste. Although somewhat uncertain, these mass flows provide pertinent new evidence on the efficacy and risks of current plastic waste management practices pertinent to emerging regulations around trade in plastic waste, and to the development of a more circular economy.

3.1. Introduction

Plastics are a versatile and ubiquitous material in the global economy. Plastic popularity can be attributed to the polymers being light, durable, mouldable, bio-inert, hydrophobic, and inexpensive to produce. Plastic products and packaging have thus contributed significantly to global economic development and environmental efficiency, e.g., reducing food waste by increasing the longevity of food (Barlow and Morgan, 2013). The global production of plastics is vast (Geyer et al., 2017). Polyethylene (PE) is one of the most common types of plastic. European plastic demand by polymer type of low-density polyethylene (LDPE) and high density polyethylene (HDPE) was 17.5% and 12.3%, respectively, of all plastics produced in 2016 (PlasticsEurope, 2018).

The huge scale of plastic production has led to large quantities of plastic waste posing an environmental challenge. Inevitably, large quantities of plastic waste end up in the oceans (Auta et al., 2017), a phenomenon emerging as a major threat to ocean ecosystems and food chains (Wilcox et al., 2015). Plastic pollution is widespread, being found in even the most remote marine environments (Chiba et al., 2018; Jiang, 2018). It is estimated that there are currently over 150 million tonnes of plastic waste in the ocean (Ellen MacArthur Foundation, 2017). Plastic polymers can survive for hundreds or even thousands of years depending on the type of plastic and the environment that the plastic ends up in (Thompson et al., 2004; Verma et al., 2016; Xanthos and Walker, 2017). Animals can become entangled within or ingest larger plastic fragments, which can lead to suffocation or starvation (Free et al., 2014; Lusher et al., 2015). However, plastic pollution goes beyond macro-plastic litter. Microplastics can contain additives, which have the potential to leach into the surrounding environment, resulting in toxicity to organisms, including carcinogenesis and endocrine disruption in humans (Cole et al., 2011; Talsness et al., 2009). The hydrophobic nature and large surface area-to-volume ratio of microplastics can concentrate persistent organic pollutants (POPs) from the surrounding environment (Hong et al., 2017; Li et al., 2016). As microplastics are similar in size to the small food sources of many primary consumers, they can be digested by a wide range of organisms (Wright et al., 2013). Thus, POPs can accumulate in different organisms, potentially undergoing biomagnification along the food chain (Law and Thompson, 2014).

Due to the large burdens placed on the environment from waste, and the scarcity of some finite resources disposed of in waste, the "circular economy" concept has emerged (McDonough and Braungart, 2002). Inspired by the workings of natural ecosystems, the model is a regenerative system which performs within ecological limits by reducing the need for resource extraction and abandoning the concept of "waste" (Geissdoerfer et al., 2017). The main idea of the circular economy is the principle of cradle-to-cradle, which aims to close both biological and technical material loops (McDonough et al., 2003; McDonough and Braungart, 2002). Multiple European directives and strategies pertaining to plastic waste have been implemented, facilitating the transition towards a more circular economy (European Commission, 2018a, 2018b, 2008, 1994). Plastic recycling is integral to the EU policy on the circular economy (Lazarevic et al., 2010). The challenge of managing increasing quantities of plastic waste diverted for recycling in Europe has been partially met through export of plastics destined for recycling to low-cost countries outside of Europe, until recently spearheaded by China. Collectively, China and Hong Kong have imported 72.4% of all exported plastic waste globally (Brooks et al., 2018). 46% of post-consumer plastic waste from Europe, collected for recycling, is exported (which includes exportation within Europe) (Wilson et al., 2015). Increasing attention on environmental challenges in some destination countries is changing the market for plastic waste. In 2017, China implemented new policy, banning plastic waste importation from 2018 onwards (Ministry of Ecology and Environment, 2017). This is likely to have major consequences for the fate of European plastic waste, but so far, no complete official trade data have been published to explore the implications.

Plastics are usually distributed between three fractions: plastics in use, post-consumer managed plastic waste and mismanaged plastic waste (Lebreton and Andrady, 2019). Managed plastic waste is usually disposed of by recycling, incineration or landfill, whereas mismanaged waste is waste that is either discarded into the environment or is inadequately disposed (which includes disposal in dumps or open, uncontrolled landfills) and may end up in the oceans (Jambeck et al., 2015). Mismanaged waste has a strong inverse correlation with GDP per capita (Lebreton and Andrady, 2019). This suggests that if waste is exported from a high GDP country to be recycled in a lower income country, there is significant potential for "leakage" into the environment. The rejected material from recycling could be a major pathway into the environment if the waste is inadequately disposed of in the exporting countries which predominantly use open and uncontrolled landfills. To the best of our knowledge, this potential pathway of plastic ocean debris has not been previously quantified. Whilst studies have incorporated or evaluated the exportation of waste into mass-flow or end-of-life studies of plastic recycling (Kawecki et al., 2018; Mutha et al., 2006; Sevigné-Itoiz et al., 2015; Van Eygen et al., 2017), ocean debris has not been considered as an end-of-life fate for the plastic waste. In part, this gap reflects a sparsity of data on waste management and recycling systems (Christensen et al., 2007). Often, recycling rates are calculated based on quantities sent for recycling, irrespective of the final fate of that separated waste (Lazarevic et al., 2010; Velis, 2014). A more accurate understanding on the fate of plastic waste is imperative to inform better management practice and policy making around the circular economy. For example, comparing the environmental sustainability of petrochemical plastic recycling in a circular economy against a shift towards biodegradable (bio)plastics requires accurate information on flows and fates of all plastic.

The goal of this study was to generate quantitative mass flows of PE waste exported for recycling from Europe to estimate the end-of-life fate of this plastic, including ending up as ocean debris. Realistic recycling flows were created from the most recent available data to act as a benchmark for future scenarios.

3.2. Materials and Methods

3.2.1. Overview of Recycling Flow

The typical European PE recycling chain (**Figure 3.1**) is modelled in this study.

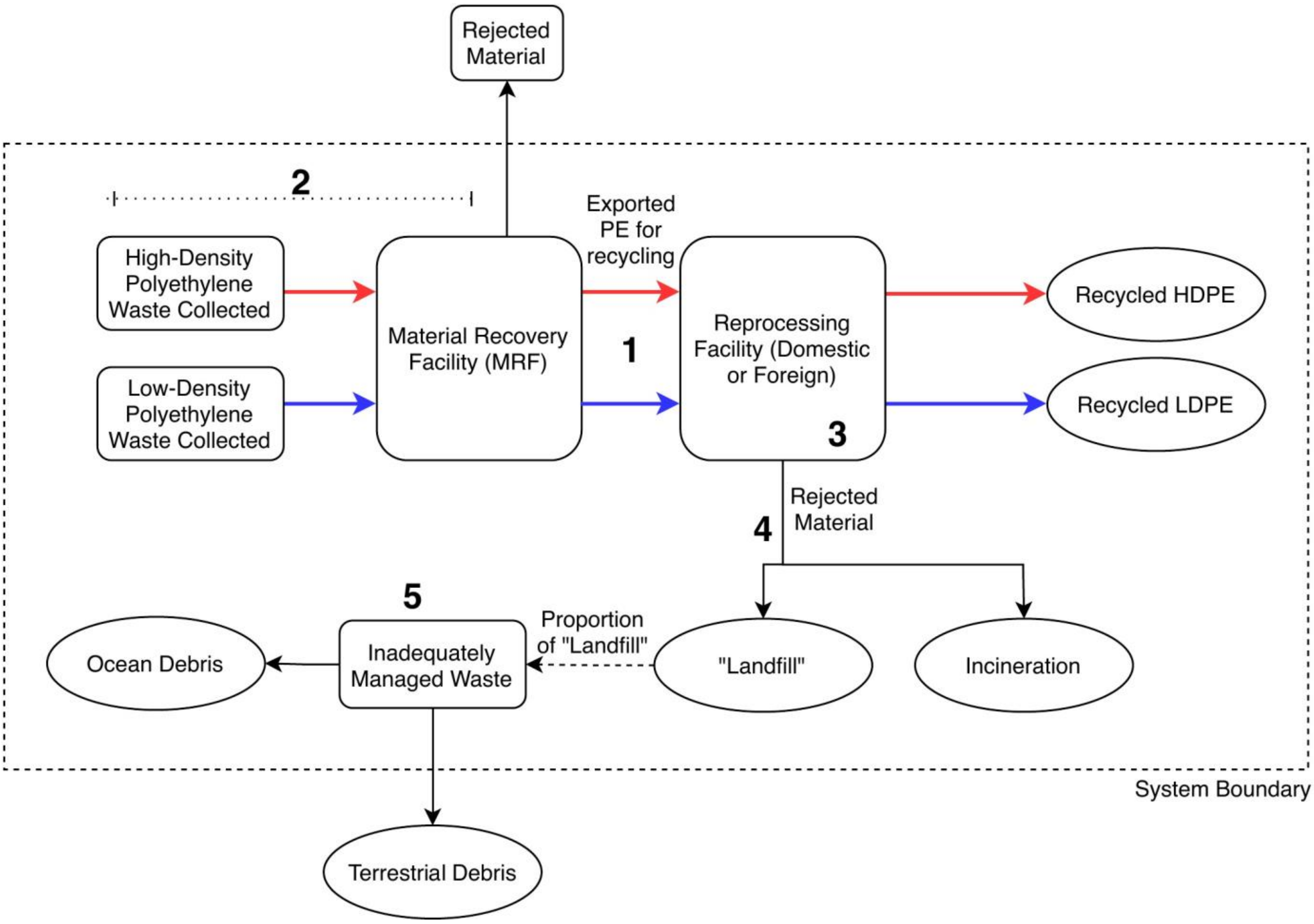

Figure 3.1 – The typical flow of polyethylene collected for recycling in Europe. Numbers **1-5** relate to the chronological sequence of analytical stages in the study, separated into subchapters in the Methodology description. Rejected material from European MRFs was not included in the final fate of the PE within this study. "Landfill" includes controlled landfills, dumps, and open, uncontrolled landfills. Terrestrial debris was not considered within this study, so is embedded within the "landfill" category. Rejected material reflects contaminated polyethylene which isn't recycled. Red lines represent the high-density polyethylene (HDPE) path. Blue lines represent the low-density polyethylene (LDPE) path. The dashed line to inadequately managed waste represents a proportion of "landfill" of which rejected material is inadequately disposed (**Chapter 3.2.7**)

Before describing each stage of analysis (represented by a number in **Figure 3.1**) in more detail in subsequent sections, we provide a brief elaboration of the recycling chain here. The initial step in the recycling chain involves the disposed waste being collected and sent to a materials recovery facility (MRF) (**Stage 2**, **Figure 3.1**), where materials are separated into desired plastic streams, typically by a single plastic type. The sorted waste is then baled and transported to a reprocessing facility domestically or abroad (**Stage 1**, **Figure 3.1**). In these reprocessing facilities, the plastic is recycled into resins where closed-loop recycling (the recycled material substituting virgin material to create an identical product type) or open-loop recycling (the recycled material mostly substituting other materials) occurs (Rigamonti et al., 2009) (**Stage 3**, **Figure 3.1**). The export of PE from the MRF to the reprocessing facilities is the first point of primary interest for this study, which focusses on quantification of the true fate of post-consumer PE exported as a commodity to be recycled.

3.2.2. Scenarios

There are a lack of official data from industry or government regarding the true fate of plastics exported outside of Europe for recycling. This study integrated best available evidence published across multiple sources. To prevent any potential forced assumptions, and to deal with the range of values obtained from the literature, three scenarios were designed to cover the range of values possible for key parameters at critical control points in the recycling value chain (**Figure 3.1**, **Table 3.1**).

Table 3.1 – The three scenarios used within this study, covering the three key parameters of control points based on the efficiency of recovery for the polyethylene recycling. MRF efficiency refers to the percentage of high-density and low-density polyethylene (HDPE and LDPE) which is sorted into single plastic streams (**Chapter 3.2.4**). Reprocessor efficiency refers to the levels of polyethylene which are recycled at the reprocessing facilities (**Chapter 3.2.5**). The reprocessing efficiencies are separated by economic classifications: low-middle income (LMI), upper-middle income (UMI), and high income (HI). Ocean debris refers to the percentage of inadequately managed waste entering the oceans (**Chapter 3.2.7**)

Scenarios	**MRF efficiency %**		**Reprocessor efficiency %**			**Ocean debris %**
	HDPE	LDPE	LMI	UMI	HI	
High recovery efficiency (HIGH)	98	95	70	80	90	15
Average recovery efficiency (AVG)	77.5	74.5	50	70	90	25
Low recovery efficiency (LOW)	57	54	30	60	90	40

The scenarios included with the study (**Table 3.1**) include: a high recovery efficiency scenario (HIGH), applying the highest MRF (**Chapter 3.2.4**) and reprocessing (**Chapter 3.2.5**) efficiencies and lowest rate of ocean debris transfer (**Chapter 3.2.7**); a low recovery efficiency scenario (LOW), applying the lowest MRF and reprocessing efficiencies and the highest ocean debris transfer factor; and the average recovery efficiency scenario (AVG), applying average efficiencies and ocean debris transfer factor. The AVG MRF efficiency was taken as an average of the LOW and HIGH scenario values. The values of the three key parameters (**Table 3.1**) are discussed in detail later.

3.2.3. Exported European PE trade flows

Stage 1 (**Figure 3.1**) involved identifying the quantities and destinations of the separated PE flows from the European MRFs to the reprocessing facilities, including European and non-European reprocessors. UN Comtrade (United Nations, 2019a), an open repository providing detailed annual international trade data for imports and exports of many commodities, was utilised as the principal data source for this study. UN Comtrade data for "Ethylene polymers; waste, parings and scrap" were compiled for each country in EU-28, Norway and Switzerland to map the flows of PE waste being exported from and within Europe (see **Chapter 3.2.9**). There were missing data in the 2018 dataset, so 2017 UN Comtrade data were used as the most recent complete data series to model the flows of PE (**Table S1, Appendix 3.1**). We discuss caveats around this dataset in **Chapter 3.2.9**. Within the mass flow (a closed mass balance that expresses the movement of material between stages), it was assumed that 63% of waste sent to Hong Kong was sent straight to mainland China, based on recently published data (Brooks et al., 2018).

Each individual export flow from each European country which exported greater than 0.010% of the total mass exported from Europe was recorded and used to quantify the flows. This cut-off accounted for 98.45% of the total waste flows. The individual flows of PE plastic waste smaller than 0.010% of the total mass exported, and the flows which did not report a specific location within the UN Comtrade dataset, were compiled as a category named "Other" location, equating to 1.55% of the total trade exported from the 30 European countries. Thus, results account for 100% of the reported traded waste. The full UN Comtrade dataset underpinning this study can be found in the supplementary material (**Table S1, Appendix 3.1**).

3.2.4. Breakdown of collected PE waste into ratios of exported HDPE and LDPE

The initial step in the recycling chain involves the collected waste being sent to a MRF. Polymer recycling rates are affected by ease of separation, quality of recycled material and market demand (Singh et al., 2017; Strangl et al., 2019). HDPE is widely accepted for recycling in Europe, but not all recycling schemes accept LDPE. However, the UN Comtrade data for the export of PE do not differentiate HDPE and LDPE, so assumptions had to be made to estimate the ratio of HDPE and LDPE being exported from MRFs (**Stage 2, Figure 3.1**). There is little information regarding the breakdown of materials or polymer types sent to MRFs for sorting (REPAK, 2018). However, recent MRF studies (Cimpan et al., 2015; Pressley et al., 2015) identified that MRF inputs of PE waste comprised approximately 34% LDPE and 66% HDPE (if assumed that film plastics are LDPE). Due to the differences between LDPE and HDPE separation efficiency at the MRF stage, we separated PE out into representative flows of LDPE and HDPE into the MRFs based on the aforementioned input split, and out of the MRFs based on specific MRF recovery efficiencies for LDPE and HDPE (described below). This allowed us to convert the UN Comtrade data for PE flows into LDPE and HDPE flows out of Europe, differentiated by scenario.

MRFs across Europe employ a range of technologies and are automated to different degrees. The quantities of rejected material in the MRF sorting stage can vary greatly, reflecting contamination with food waste or non-recyclable items and the heterogeneity of plastic products (Chilton et al., 2010; Lazarevic et al., 2010). Whilst plastic contamination could be removed within MRFs, it is usually more economically feasible for the waste to be diverted to another waste stream (Adrados et al., 2012). Published benchmarks and data on process efficiency for MRFs are rare (Mastellone et al., 2017). The efficiency of the MRFs depends especially on the technologies that the facilities have installed, the composition and the level of contamination of the incoming waste. The scenarios utilised a conservative range of values found in the literature (Eriksen et al., 2018; Pressley et al., 2015) to encompass a range of potential outcomes. The study by Pressley et al. (2015) uses data from Seattle (USA), but due to the limitation of data available, it was assumed that the high efficiency of the MRF would be similar with high-performing MRFs in Europe. Assuming from these studies that film plastics represented LDPE, for the HIGH scenario it was estimated that 95% LDPE sent to the MRF was recovered, and that 98% of the HDPE was recovered (**Table 3.1**) (Eriksen et al., 2018; Pressley et al., 2015). The LOW scenario employed MRF recovery efficiencies of 54% for LDPE and 57% for HDPE (**Table 3.1**) (Eriksen et al., 2018). The AVG scenario MRF efficiency was taken as an average of the HIGH and LOW MRF efficiencies, resulting in a recovery efficiency of 74.5% and 77.5% for LDPE and HDPE, respectively (**Table 3.1**).

3.2.5. PE reprocessing efficiency

Following sorting at a MRF, plastic is baled and transported to a reprocessor located in the same country or abroad. The fate of recycled plastic material in importing countries is not documented. A significant fraction of imported plastic destined for "recycling" ends up in landfill (Velis, 2014), however data on exact figures are lacking. At the reprocessing facility, the waste undergoes further processing before the plastic waste can be recycled (Barlow and Morgan, 2013). The levels of rejected waste at reprocessing facilities can vary, depending on the technology of the reprocessor, products that are to be made and the quality of material that the reprocessor receives (**Stage 3, Figure 3.1**). There is little information regarding the efficiency of individual recycling reprocessors, and various studies and media reports have referred or alluded to large reject rates in non-European reprocessing facilities (Beard, 2019; Retamal et al., 2019; Velis, 2014). Among the little hard data available, a 10% material loss of plastic from reprocessing facilities was reported by Merrild et al. (2012). Barlow and Morgan (2013) also noted that when the material is shredded, cleaned and extruded, a loss (reject rate) of 10% is typical, both in high-income (HI) countries (below).

It has been noted that there is a strong correlation between waste management efficiency and economic classification of countries (Kaza et al., 2018; Lebreton and Andrady, 2019). In this study, the efficiency of the reprocessing facilities, and thus the levels of PE rejects, was determined for each country importing PE waste by economic classification according to their gross national income (GNI) per capita, based on the World Bank Atlas method (The World Bank, 2019). Low-income (LI) economies are defined as those with a GNI per capita of $1,025 or less; lower middle-income (LMI) economies are those with a GNI per capita between $1,026 and $3,995; upper middle-income (UMI) economies are those with a GNI per capita between $3,996 and $12,375; and HI economies are those with a GNI per capita of $12,376 or more (Table S2) (The World Bank, 2019).

Due to large uncertainty, the wide range of reprocessing facility efficiencies from LOW to HIGH scenarios (**Table 3.1**) provides a sensitivity analysis covering the likely bounds of efficiency, where the lower-bound is likely to be the lowest prevailing efficiency that will be found in these countries, and the upper-bound is likely to be the highest prevailing efficiency. The efficiency of the reprocessing facilities for the HIGH scenario was assumed to be 90% for the HI countries, 80% for UMI countries and 70% for LMI countries. The efficiency of the reprocessing facilities for the LOW scenario was assumed to be 90% for the HI countries, 60% for UMI countries and 30% for LMI countries. The efficiency of the reprocessing facilities for the AVG scenario was assumed to be 90% for the HI countries, 70% for UMI countries and 50% for LMI countries (**Table 3.1**) (Barlow and Morgan, 2013; Beard, 2019; Merrild et al., 2012; Retamal et al., 2019; Velis, 2014). The study spans a broad range to cover the higher uncertainty and greater range of performance in LMI countries compared with the less range defined in HI countries as the HI countries are more tightly regulated.

3.2.6. Fate of rejected material

Rejected plastic waste is typically disposed of by incineration or landfilling (Huysman et al., 2017). The proportions of rejected PE (combined HDPE and LDPE) that were "landfilled" or incinerated from the reprocessing facilities were calculated (**Stage 4**, **Figure 3.1**). The specific fates of plastic reject waste from the repossessing facilities were estimated using country specific municipal solid waste (MSW) management practices taken from a 2018 report by the World Bank (Kaza et al., 2018). Within the report, waste treatment of MSW by country was calculated and reported from a range of sources. The fates of the entire MSW category included landfill, recycling, composting, anaerobic digestion, incineration and "unaccounted for" waste. As many countries did not differentiate between the types of landfill, whether controlled, uncontrolled, or open, the study treats all "landfill" together. Therefore, the "landfill" category is broad and in fact includes other fates, including debris loss to the oceans (**Chapter 3.2.7**) and terrestrial debris. Initially, the small quantities of "unaccounted for" waste were conservatively proportioned out by weight across all of the reported waste management types for that country (Kaza et al., 2018). In relation specifically to PE waste flows, composting and anaerobic digestion waste management types applied to aggregate MSW were then weightedly reproportioned to just "landfill", incineration, and recycling, the three managed waste treatments for which plastic waste is treated. From these waste management types, the recycling fate was then weightedly reproportioned to just "landfill" and incineration to identify the fate of rejected PE from each country's reprocessing facilities. Each country within our "Other" category was calculated in a similar way.

3.2.7. Ocean Debris

The percentage of "landfilled" PE rejects which ended up in the ocean was next calculated (**Stage 5**, **Figure 3.1**). Jambeck et al. (2015) estimated the percentage of plastic waste which is inadequately disposed of at country level (which includes disposal in dumps or open, uncontrolled landfills), a fraction of which enters the oceans. Jambeck et al. (2015) used a range of fixed conversion rates of mismanaged plastic waste to marine debris, to estimate the mass of plastic that entered the oceans (ocean debris percentage), proposing a best-case percentage of inadequately managed waste entering the oceans of 15%, an average of 25%, and a worst-case of 40%. These ocean debris percentages were calculated through the study of the percentage of waste collected by infrastructure in the San Francisco Bay watershed, and thus the residual uncollected percentage that is available to enter the ocean as marine debris (Jambeck et al., 2015). We assumed that the fate of the rejected material within the receiving countries was similar to the fate of the total plastic waste which Jambeck et al. (2015) reported.

Within this study, the percentage of inadequately managed waste (Jambeck et al., 2015) was multiplied by the ocean debris coefficient (**Table 3.1**) to calculate the final percentage of PE entering the ocean as debris for each destination country, and ultimately via flows to those countries, each export country, for each scenario. As it was assumed that the inadequately managed waste was included within the "landfill" proportion (**Stage 5**, **Figure 3.1**), the calculated quantities of PE debris entering the oceans were subtracted from the "landfill" quantities (from **Chapter 3.2.6**) to produce an updated "landfill" fate percentage. Thus, "landfill", incineration and ocean debris quantities in each recipient country equate to the quantity of PE rejects from reprocessing facilities. The fates for the countries in the "Other" category were individually calculated in a similar way, to get the three fates for each country for the three scenarios. A weighted average for the fate of "landfill", incineration and ocean debris within the "Other" category was undertaken for each scenario, to combine the countries into one result for each scenario. Full fates of rejected material for each receiving country scenario can be found in **Table S2, Appendix 3.1**.

3.2.8. Statistical Analysis

Two sets of PE mass flows were calculated: i) one for European (EU-28, Norway and Switzerland) aggregate export flows; ii) another for European-country-specific export flows. For i), aggregate quantities of exported European PE were calculated into recycled HDPE and LDPE resins, "landfill", incineration, or ocean debris (**Figure 3.2**; **Table 3.2**). For ii) over 2200 flows were constructed to follow the fate of every single flow reported by UN Comtrade, to relate the fate of the exported PE to each individual European country (**Figure 3.3**; **Table S3, Appendix 3.1**). The two sets of mass flows were created for each scenario using Microsoft Excel (2019), and converted into figures (**Figures 3.2**, **A3.1**, and **A3.2**) using open online software SankeyMATIC (Bogart, 2019). Ocean debris per head was calculated using 2017 population data (Eurostat, 2019).

Simple linear regression was performed to identify the relationship between the percentage of PE that was exported out of Europe and the percentage of PE exports resulting in ocean debris (**Chapter 3.3.2**). The data was run using Minitab® Statistical Software. Data were found to satisfy prerequisite assumptions for the simple linear regression model (Independence, Linearity, Normality and Equal Spreads). The study provides the fitted regression line, estimated standard deviation about the true regression line (S) and the R-Squared value for all the scenarios (**Figure 3.4**). The regression equation is provided for the AVG scenario.

3.2.9. The UN Comtrade data

The UN Comtrade database, maintained by the United Nations Statistics Division, is the official global database to which over 170 reporter countries provide their annual international trade statistics, detailed by commodity (or service) categories and partner countries. The data are therefore the most reliable repository of official international trade statistics. Nonetheless, there appear to be some discrepancies and gaps in the data, the quality of which ultimately depends on the completeness and accuracy of national reporting. We undertook data quality screening to ensure that the "Ethylene polymers; waste, parings and scrap" trade data used for the analyses were as consistent and accurate as possible. Firstly, to be sure that the 2017 data used for trade flows were not widely divergent from previous years, we compared 2017 export flows of PE from European countries with previous reported flows for the last 10 years. This identified potential anomalies for Slovenia in 2017 compared with earlier years, and missing data for Malta, so 2016 data were conservatively used in these cases. Secondly, we compared "export to" flows reported by countries of origin with "import from" flows reported by destination countries. It was immediately apparent that "import from" flows reported by many of the non-European developing countries were small or missing, consistent with other evidence that lower income countries do not adequately manage (and therefore do not report on) much of their waste (Jambeck et al., 2015; Kaza et al., 2018; Lebreton and Andrady, 2019). For this reason, we rely upon "export to" data reported by European countries of origin as a basis for our trade flows. A more complete data quality assessment was then performed matching "export to" with "import from" flows (of volumes equal to at least 0.01% of the total mass exported from Europe) for the pertinent countries assumed to implement more complete reporting, i.e., EU-28, Norway, and Switzerland (**Table S5, Appendix 3.1**). The results of this analysis show that there were some significant discrepancies for particular countries, but that in aggregate reported exports were within 2.1% of reported imports (**Table S5, Appendix 3.1**). This provides some confidence that the aggregate PE export flows from Europe underpinning core conclusions of this paper are reliable. In terms of discrepancies in specific country-to-country flows, such "bilateral trade asymmetries" are a well-acknowledged phenomenon that arise for the following reasons (United Nations, 2019b): i) the application of different criteria of partner attribution in import and export statistics; ii) the use of CIF-type values[2] in import statistics and FOB-type values in export statistics; iii) countries having different trade systems; iv) time lag between exports and imports; v) goods passing through third countries; vi) or goods being classified differently. Reason v) is likely to account for much of intra-European country-specific bilateral trade asymmetries in **Table S5** (**Appendix 3.1**), owing to extensive and complex trade and transit of PE within the European Economic Area. Whilst this is much less likely to be an issue for (long-distance) reported export flows outside of Europe, it does invoke some caution around the attribution of PE flows to specific countries of origin within - discussed in **Chapter 3.4.2**.

[2] Cost, Insurance, and Freight (CIF) and Free on Board (FOB) are international shipping agreements used in the transportation of goods. CIF-type values include the transaction value of the goods, the value of services performed to deliver goods to the border of the exporting country, and the value of the services performed to deliver the goods from the border of the exporting country to the border of the importing country. FOB-type values include the transaction value of the goods, and the value of services performed to deliver goods to the border of the exporting country (United Nations, 2010).

3.3. Results

3.3.1. Aggregate fate of PE exported from Europe

The analysis generated 345 PE export flows from 30 European countries to 50 major receiving countries in 2017, representing 98.4% of the total 2,487,351 Mg of PE exported out of the EU28, Norway and Switzerland (**Figure 3.2**). From this total, China received the largest share, at 31.0%, with the next highest receiving country being the Netherlands, receiving 8.9% of the exported PE. The mass flows highlight a considerable trade of PE outside of European countries, with 54.2% of exported PE received by countries not within EU28, Norway or Switzerland (**Figure 3.2**; **Table S2**, **Appendix 3.1**).

The fate of total European PE exported for recycling varied with each scenario (**Table 3.2**). The average recycle rate of the exported PE waste was determined to be 76% (range: 69% – 83%), with most of the PE waste that wasn't recycled – 59% (51% − 61%) – ending up in "landfill". The amount of PE exported for recycling estimated to enter the ocean was 83,187 Mg (range: 32,115 Mg – 180,558 Mg), accounting for 3% (1% − 7%) of the exported plastic. This represents an average of 14% (8% – 24%) of rejected PE entering the ocean. Mass flows for the AVG scenario can be visualized in **Figure 3.2**. The other scenarios are presented in the supplementary material (**Figures A3.1** and **A3.2**).

Table 3.2 – Mass and percentage breakdowns of the total mass of polyethylene exported from the EU-28, Norway, and Switzerland for recycling in 2017, across fates, for the three scenarios.

Fate	**Average recovery efficiency scenario**		**High recovery efficiency scenario**		**Low recovery efficiency scenario**	
	Quantity (Mg)	Percentage (%)	Quantity (Mg)	Percentage (%)	Quantity (Mg)	Percentage (%)
Recycled HDPE resin	1,268,689	51.0	1,379,093	55.4	1,160,002	46.6
Recycled LDPE resin	628,268	25.3	688,694	27.7	566,126	22.8
"Landfill"	347,049	14.0	257,538	10.4	390,259	15.7
Incineration	160,158	6.4	129,911	5.2	190,405	7.7
Ocean Debris	83,187	3.3	32,115	1.3	180,558	7.3

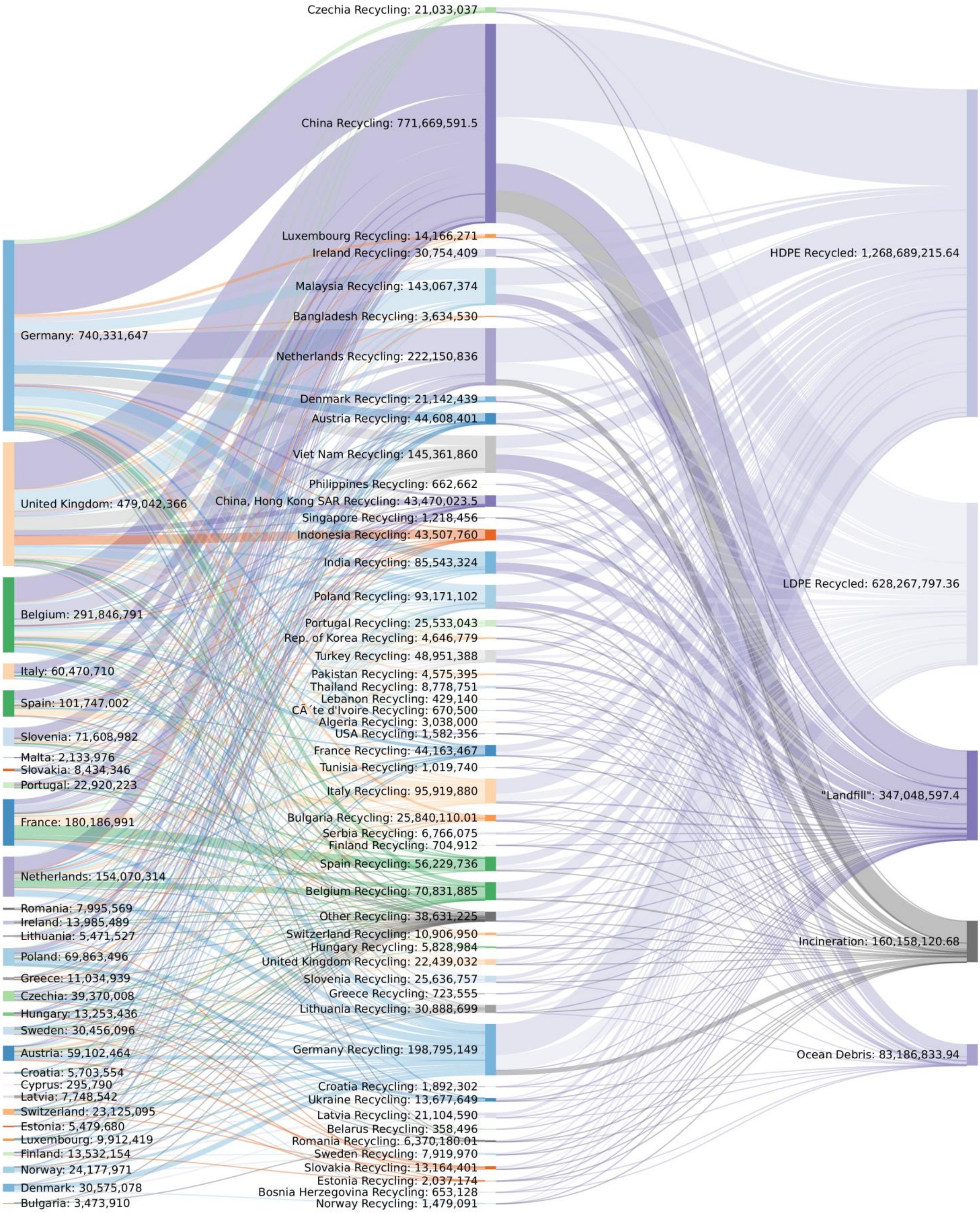

Figure 3.2 – The average recovery efficiency scenario (AVG) for the mass flows (kg) of polyethylene waste exported in 2017 from EU-28, Norway, and Switzerland, to different fates via different recipient countries. The thickness of the lines are relative to the mass flows. See **Figures A3.1** & **A3.2**, respectively, for the HIGH and LOW scenarios.

3.3.2. Fate of country-specific PE exports

The countries with the greatest percentage of exported PE ending up as recycled HDPE or LDPE were Luxembourg and Switzerland (90% recycled for all scenarios), whilst the country with the lowest share of exported PE being recycled was the United Kingdom (59% – 80%, average 69% recycled) (**Figure 3.3**). Consequently, the United Kingdom had the greatest percentage of exported PE estimated to end up as ocean debris, at 5% (2% − 11%). However, the largest absolute mass flow of PE to ocean debris originated from Germany, at 26,461 Mg (57,352 Mg − 10,246 Mg) (**Figure 3.5**). Germany also exported the greatest mass of PE that was recycled at 559,177 Mg (505,616 Mg − 612,738 Mg). The breakdown of AVG scenario fates is visualized in **Figure 3.3**. The full results can be found in the supplementary material (**Table S3, Appendix 3.1**).

The results demonstrate that the destination countries of the exported PE waste had a large effect on the end-of-life fate of the material. When countries exported the PE outside of the generally high-quality waste management systems of Europe to non-European countries with typically weaker waste management chains, the inadequately managed waste, and thus the PE potentially entering the oceans, increased. For example, Luxembourg and Switzerland only exported main flows (greater than 0.010% of total mass exported) to other European countries, whereas 85% of the main PE export flows from the United Kingdom were destined for non-European countries (**Tables S1** and **S2, Appendix 3.1**).

As previously mentioned, a large percentage of the exported PE waste is exported within Europe. A linear regression, which satisfied all assumptions, showed a significant positive relationship between the percentage of PE exported out of Europe and the percentage of exports which potentially enter the ocean ($p<0.001$) for each scenario (**Figure 3.4**). The regression equation for the AVG scenario of the percentage of exported PE debris in the ocean and the percentage of PE exported out of Europe was estimated to be the following equation (**Figure 3.4**):

Percentage of exported PE debris in ocean = 0.319 + (0.053 * Percentage PE exported out of Europe), R^2 = 90.4%, S = 0.51%

Thus, there is an estimated intercept at 0.319 (estimated standard error 0.138) and an estimated slope of 0.053 (estimated standard error 0.003).

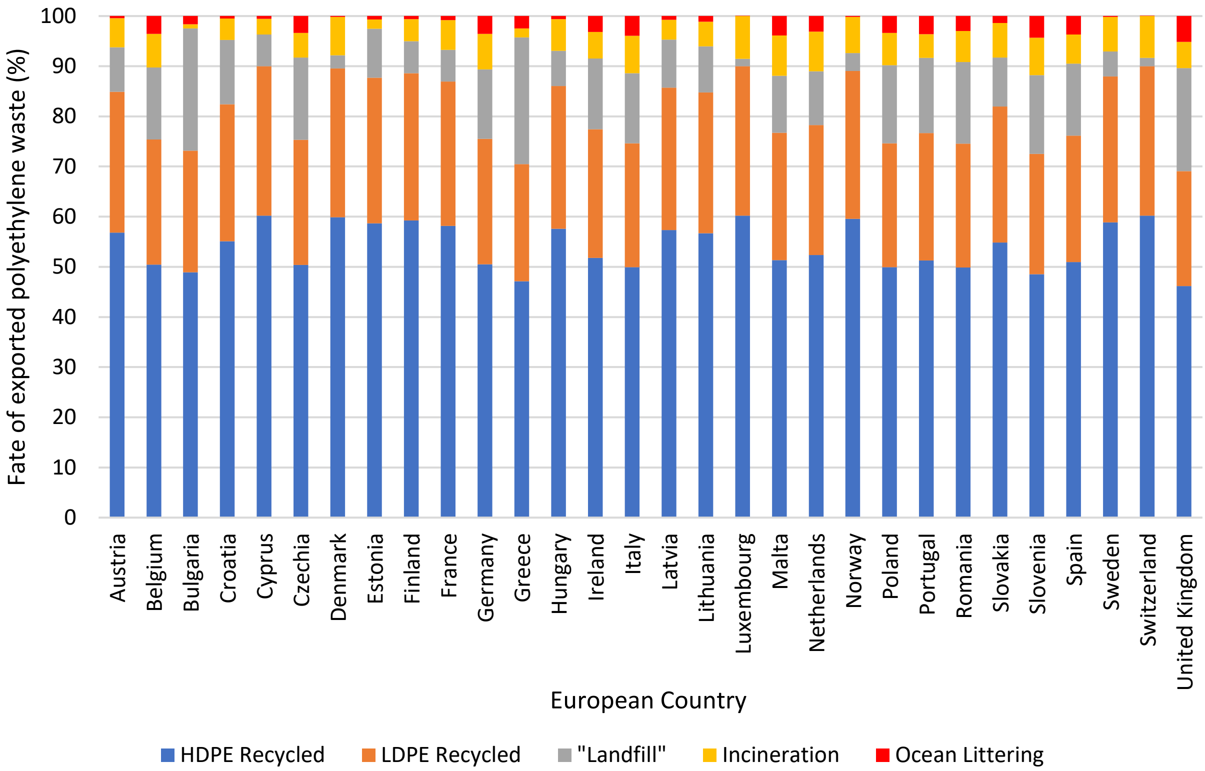

Figure 3.3 – Breakdown of end-of-life fates for polyethylene waste exported from EU-28, Norway, and Switzerland in 2017 for the average recovery efficiency scenario (AVG).

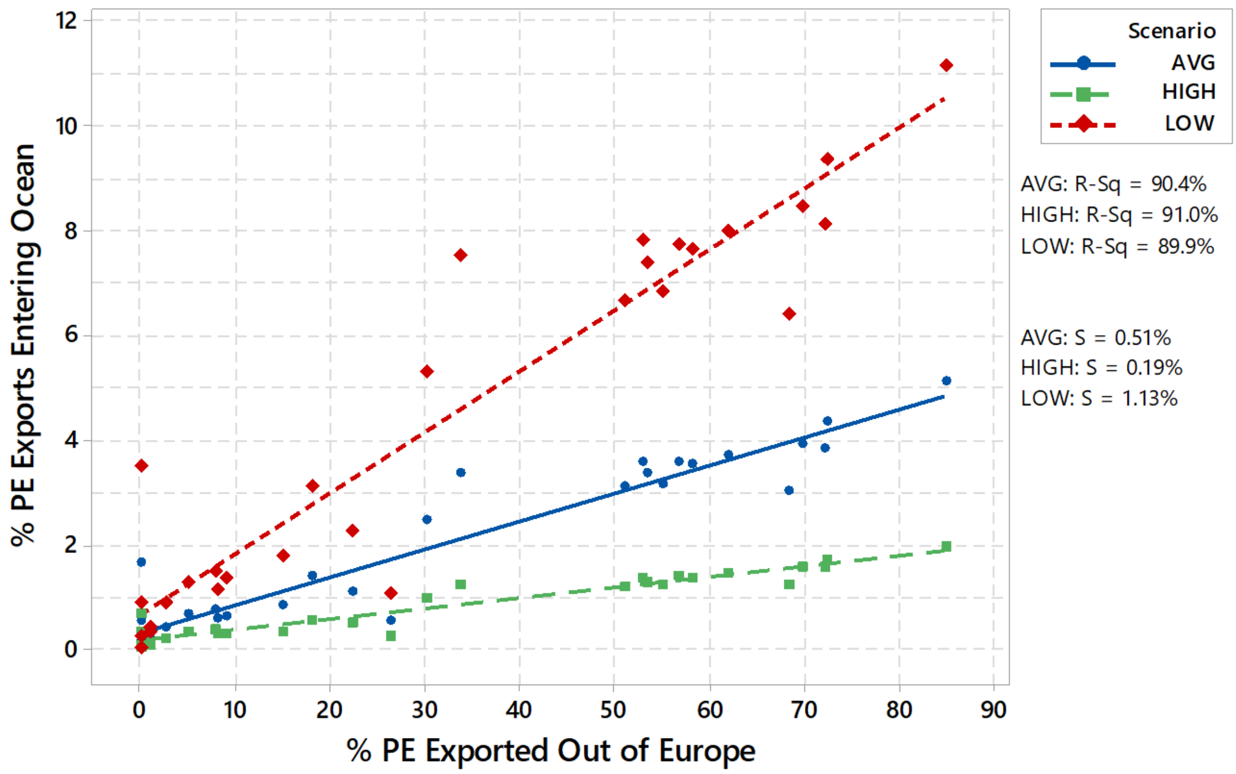

Figure 3.4 – Relationship between percentage of polyethylene waste exported out of Europe (EU-28, Norway, and Switzerland) to be recycled and the percentage of exported polyethylene debris ending up in the oceans for three scenarios based on recovery efficiency. Each point represents a European county. The figure gives the R-Squared value (R-Sq) and the standard error of the estimate (S) for each scenario

3.3.3. Contribution to total ocean debris

As mentioned in **Chapter 3.3.2**, the largest absolute mass flow of PE to ocean debris originated from Germany, which, from the AVG scenario, equated to 32% of total ocean debris. Other notable countries included United Kingdom with 29% of total ocean debris and Belgium with 12% of total ocean debris from the AVG scenario (**Figure 3.5**). The full results of the contribution to total ocean debris from the exportation of PE can be found in the supplementary material (**Table S4, Appendix 3.1**).

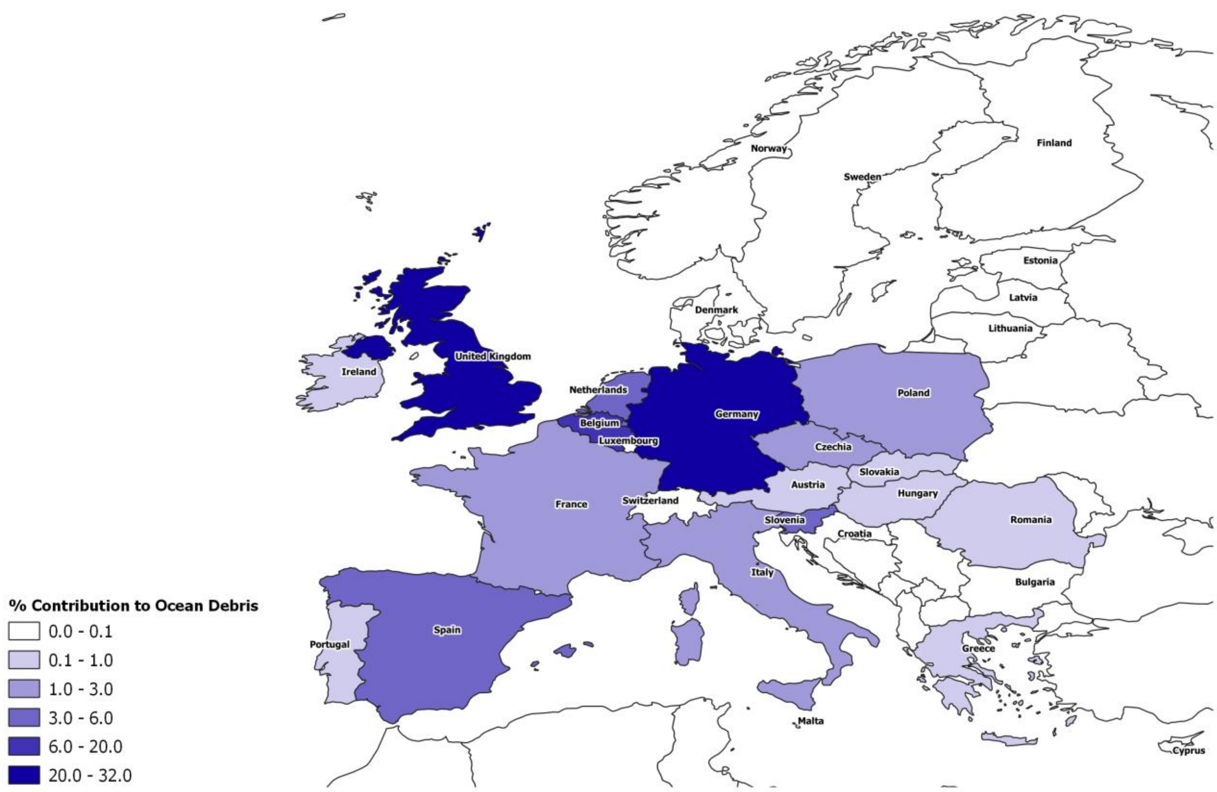

Figure 3.5 – Contribution to total ocean debris originating from the exportation of polyethylene for recycling originating from EU28, Norway, and Switzerland for the average recovery efficiency scenario (AVG). Full table of contributions to total ocean debris can be found in **Table S4, Appendix 3.1**.

3.3.4. Per capita PE exports

The results showed a large variation between countries in ocean debris per head, with Slovenia having considerable ocean debris per head at 1.5 kg.head^{-1}.year^{-1} (range: 0.6 – 3.2 kg.head^{-1}.year^{-1}) (**Figure 3.6**). Belgium also had large ocean debris per head reported at 0.9 kg.head^{-1}.year^{-1} (0.3 – 2.0 kg.head^{-1}.year^{-1}). Other notable large ocean debris per head countries include the United Kingdom, Germany, and the Netherlands. Luxembourg and Switzerland had the lowest ocean debris per head of 0.0 kg.head^{-1}.year^{-1}across all the scenarios. It is worth noting that these per capita values were calculated based on quantities of PE exported from each country, which may differ from PE waste generated in each country owing to through-trade.

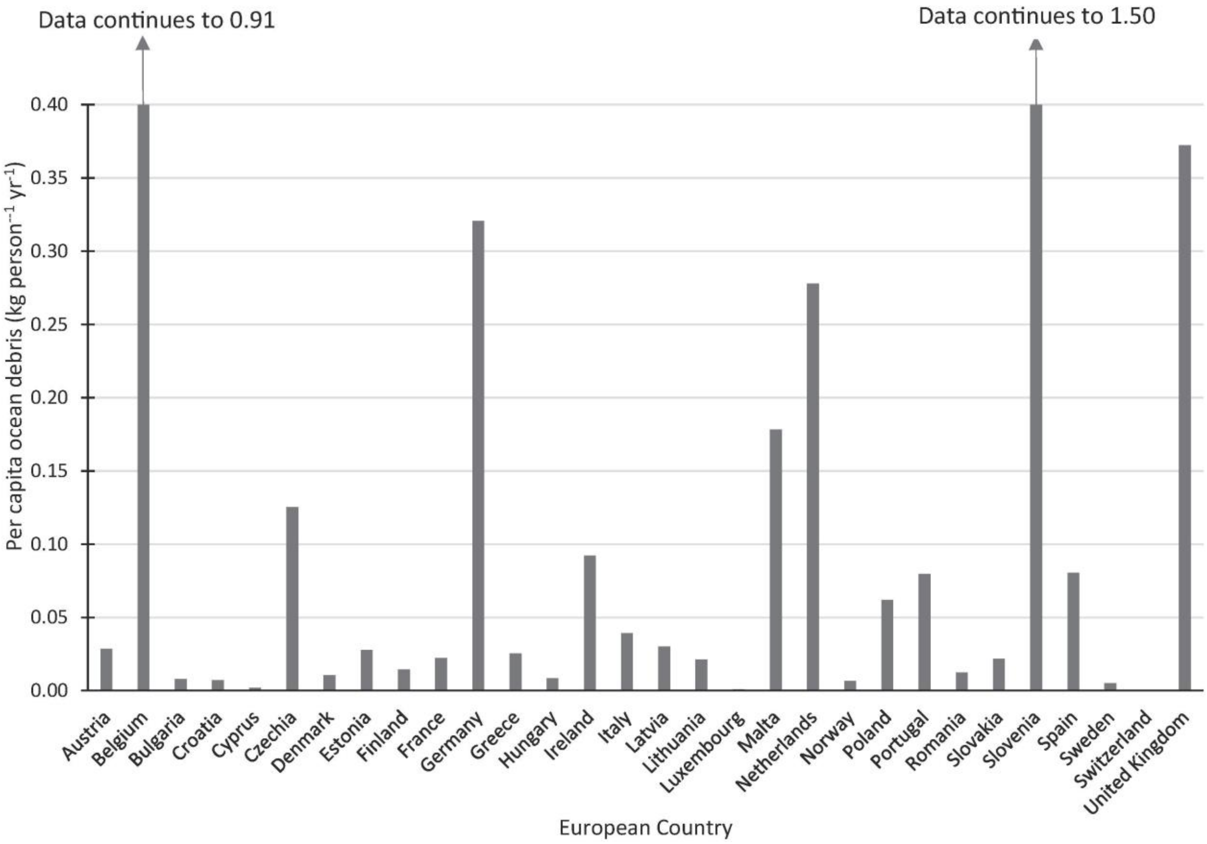

Figure 3.6 – Ocean debris per head of the population in 2017 for the polyethylene exported out of the original country for recycling for the average recovery efficiency scenario (AVG).

3.4. Discussion

3.4.1. Ocean debris

The aggregation of best available data in this study showed that a large percentage of PE exported for recycling (up to 31%) isn't actually recycled and that up to 24% of this rejected PE potentially enters the ocean. Jambeck et al. (2015) calculated a range of 4.8 – 12.7 million Mg of plastic waste entering the ocean per year. Based on these estimates and the estimates from this study, ocean debris from rejected European PE recycling accounts for 0.3 – 3.8% of the total debris entering the ocean. This hitherto undocumented flow of 83,187 Mg (tonnes) of PE entering the oceans in 2017 is therefore an important source of plastic loss into the ocean. If PE (which accounts for just 30% of European plastic (PlasticsEurope, 2018)) is representative of other types of plastic waste sent for recycling, then the total amount of plastic entering the oceans via exports for recycling will be considerably higher. This loss will have significant implications for marine life and ecosystems (Schneider et al., 2018; Xanthos and Walker, 2017). In addition to damaging global ocean ecosystems, the concentration of ocean debris around waste importing countries such as Bangladesh, Pakistan and Viet Nam may induce negative effects for fishing, tourism and potentially human health in those countries (Gall and Thompson, 2015; Li et al., 2016; Pawar et al., 2016).

The differences across scenarios for the quantities of ocean debris reflect the efficiency of the reprocessing facilities and the percentage of inadequately managed waste which is lost into the oceans. The level of economic development of countries receiving Europe's PE waste has a major influence on these factors (Kaza et al., 2018; Lebreton and Andrady, 2019), and thus plays a major role in determining the amount of ocean debris arising from exported PE waste. Waste management is expensive. In low-income countries, waste management can be the greatest single cost for many local governments, where it can equate to 20% of municipal budgets (Kaza et al., 2018). Comparatively, in middle-income countries, municipal solid waste management typically accounts for about 10% of municipal budgets, and for about 4% in high-income countries (Kaza et al., 2018). It is clear that many of the lower-income countries to which European plastic waste is sent have smaller budgets and lack infrastructure to deal with waste streams compared with the higher-income countries from which this waste originates. Essentially, export of plastic waste outside of Europe is a potentially major, unexplored pathway for ocean debris, as indicated by the strong relationship between the percentage of PE each country exported out of Europe and the percentage of waste estimated to be lost into the oceans. Realising a genuine circular economy in Europe (European Commission, 2018a) will require ending such pathways of international material and pollution "leakage".

3.4.2. Complex trade flows

The data highlight that there are long and complex flows of plastic waste trade within and outside of Europe. Large export quantities per capita from countries are likely to represent trade through those countries owing to the presence of large ports serving neighbouring countries. For example, according to UN Comtrade data (United Nations, 2019a), Slovenia, which has the highest ocean littering per head, imports the majority of its PE waste from its four surrounding countries (Italy, Austria, Hungary, and Croatia), where the majority is then exported outside of Europe (**Table S1, Appendix 3.1**). The other countries with high ocean debris per capita contain some of the world's biggest ports – Belgium, Germany, Netherlands, and the UK (**Figure 3.6**). Per capita ocean debris values in **Figure 3.6** may therefore represent transit of export flows originating from other countries, including those with very low values. This makes it difficult to relate the final fate of specific flows of plastic waste traded outside of Europe to specific countries of origin (rather than countries of export) of that waste – thus restricting conclusions on waste management practices per se within the countries directly exporting PE outside of Europe. Greater reporting at all stages of the recycling chain will be needed if countries (and ultimately municipalities or waste management companies) of origin are to be held accountable for the final fate of plastic waste streams in the future. Recent amendments to the Basel Convention, which will take effect in 2021 (Collins, 2019), have included plastic waste in a framework which will make the global trade of plastic waste more transparent, necessitating new monitoring, with more restrictions and controls of where the waste can be exported (United Nations, 2019c). To reduce the negative fates of exported plastic waste, European countries of origin should take greater responsibility for whom they export waste to. Individual countries should build upon the Basel Convention amendments, restricting the export of plastic waste from Europe to countries which fail to meet high efficiency waste management practices for the recycled material or by investing in the receiving countries which are importing the waste, to assist in the improvement of their waste processing efficiencies. From this study, the suggested areas which need greatest attention for improved waste reporting are revealed as: points of re-export (transfer) of waste within Europe; the breakdown of PE waste into constituent polymer types; and most importantly, the unknown efficiencies of reprocessing facilities and fates of residual waste streams (rejected plastic waste) in developing countries receiving the plastic waste. Improved data at these points in the plastic waste chain would improve the accuracy of the mass flow accounting required to underpin improved waste management practices and policy.

3.4.3. Environmental performance of plastic "recycling"

Life cycle assessment (LCA) has previously been used to assess the environmental impacts of HDPE and LDPE value chains (Belboom and Léonard, 2016; Bertolini et al., 2016; Günkaya and Banar, 2016; Liptow and Tillman, 2012; Sangwan and Bhakar, 2017), and as a tool to evaluate the environmental efficiency of alternative end-of-life options for plastics (Björklund and Finnveden, 2005; Hou et al., 2018; Lazarevic et al., 2010; Perugini et al., 2005). The common conclusion from these studies is that landfill is the least environmentally efficient destination for plastic waste, with recycling being the preferred option owing to lower environmental burdens a nd reduced resource depletion. However, if a significant fraction of material reported as recycled ultimately ends up as plastic debris in the oceans (or on land), as this study suggests, then the comparative environmental efficiency of average "recycling" outcomes could be re-ordered following updated LCA results. The accuracy of future LCA studies comparing waste management options could be greatly improved by reflecting the true (average) fates of waste collected for recycling, perhaps differentiated by recipient country type, within the "recycling" option. For example, unit processes in LCA could be adapted not just for country specific electricity sources as is currently the case in e.g., ecoinvent (Wernet et al., 2016), but for country-specific reprocessing rejection and subsequent fate coefficients. However, the environmental effects of plastic debris loss into the environment are not represented within current life cycle impact assessment methodology (Fazio et al., 2018). Therefore, even if ocean debris fates associated with non-European recycling were reflected within country-specific recycling unit processes, it may not be possible to fully represent the emerging environmental impacts linked with this loss pathway within state-of-the-art LCA studies (Jiang, 2018; Spierling et al., 2018). There is a need for the integration of plastic debris impacts into existing or new life cycle impact assessment categories.

As true recycling rates may differ from reported recycling rates, this will have implications for monitoring genuine progress towards recycling targets, with current reported recycling rates likely misleading. There are also implications for strategies to achieve the circular economy which Europe is striving towards (European Commission, 2018a). As plastic recycling is supposed to be closing the technical materials loop, potential leakage from the system shifts recycling away from the fundamental principles of the circular economy. A counter-intuitive implication of the findings from this study is that the efficiency of the European waste collection and MRF separation efficiency may be positively related to the rate of ocean debris arising from European plastic waste streams. Where European collection and separation efficiency of plastic waste fractions are high, greater quantities of plastic waste are likely to be exported out of Europe, thus increasing the rate of ocean debris. Therefore, improving efficiency of plastic separation domestically without paying greater attention to the final fate of exported plastic flows could be counter-productive from an environmental perspective.

As European municipalities adapt to China's recent ban on plastic waste import, early indications are that large quantities of plastic waste are being diverted to Southeast Asia (Qu et al., 2019). These lower-income countries are likely to have poorer waste management procedures and fewer resources to deal with the plastic imports compared with China. Therefore, the potential for plastic exports from Europe to end up as ocean debris is likely to increase. To deal with this uncertain future for waste trade, changes must occur in European accounting. Municipalities and MRF operators that export to these countries should be obliged to track the final fate of the exported waste in order to ensure that it is managed appropriately.

3.4.4. Study limitations

Due to the lack of data on plastic waste flows along the recycling value chain, especially in receiving countries, numerous assumptions and proxies needed to be applied in this study. Whilst UN Comtrade data provide a backbone to this analysis, MRF efficiency, reprocessing efficiency, the HDPE and LDPE composition of traded PE waste, and the fate of the rejected material, are critical aspects for which data availability are patchy. In order to reflect uncertainty around these aspects, we covered the bounded range of plausible outcomes across three scenarios, conservatively choosing outer-bound values found in the literature for each critical step in order to encompass the range of potential outcomes. The real situation in terms of the fate of exported PE waste should sit somewhere within these bounds. Therefore, we are confident that we provide new insight into the magnitude of European PE waste entering oceans via export for recycling, but more precise quantification of these flows requires further research to resolve some of the uncertainties and caveats discussed below. It should be noted that this study may represent a best-case scenario as the modelled assumption was that all exported PE waste undergoes reprocessing, and that ocean debris originates only from the rejects from reprocessing.

Total plastic pollution entering the environment will be larger than just the ocean debris reported in this study, and will include plastic debris on land, which has so far been less studied than plastic debris in oceans despite significant potential impacts (Boots et al., 2019). Terrestrial debris (within rivers and on land) is included within the "Landfill" category. Therefore, whilst not providing direct estimates of terrestrial debris arising from plastic recycling exports, the methodology applied in our study could be easily adapted to generate such estimates once terrestrial debris factors have been developed, analogous to ocean debris factors developed by Jambeck et al. (2015). Recent studies have highlighted that rivers are a major pathway for mismanaged plastic waste into the ocean (Emmerik and Schwarz, 2020; Schwarz et al., 2019; van Calcar and van Emmerik, 2019), as well as acting as large sinks of plastic pollution (Schwarz et al., 2019). These factors are affected by temporal changes, such as the seasons (Lebreton et al., 2017), but also spatial changes. Schmidt et al. (2017) found that the ocean debris flows via rivers are non-linear, with large and densely populated river catchments having considerably higher loss rates. Therefore, the generic rates of conversion of mismanaged plastic waste to marine debris (Jambeck et al. (2015) is a limitation of this study and could be refined in the future. However, such refinement would depend on the availability of more geographically explicit data on where, within receiving countries, plastic waste is sent. Nonetheless, considering the wide range of ocean debris conversion rates proposed by Jambeck et al. (2015), i.e., 15 –

40% of mismanaged waste, allowed us to calculate plausible bounds of PE waste exported for recycling that ends up in the ocean. The data are certainly sufficient to demonstrate that plastic waste export is an important pathway of plastic debris loss into the oceans from Europe, and to provide insight into the processes and control points that determine the rate of plastic leakage from exported recycling streams.

The UN Comtrade database is the largest, most reliable repository of official international trade statistics. However, there are some limitations with the data it contains owing to bilateral trade asymmetries. As previously mentioned, large intra-European trade of PE waste complicates the attribution of plastic debris volumes to specific source countries. However, the observed intra-European bilateral asymmetry has little effect on final ocean debris volumes which depend on practices in final destination countries outside of Europe. There is also large bilateral asymmetry between the exporting European countries and the importing countries outside of Europe, where in many cases the reported imports are much lower than reported exports to those countries. In our view, this is likely to be at last partly explained by under-reporting in these destination countries, consistent with the higher proportions of inadequately managed waste attributed to these countries (Jambeck et al., 2015). Thus, we base our core results on export data reported by European countries rather than import data reported by receiving countries, and conclude that there is an urgent need to tighten up on verifiable reporting of trade in waste plastic globally. We suggest that there is an onus on European countries leading the drive towards a circular economy to only allow external flows of materials where the fate can be verified, to avoid this pathway of source "leakage" that may be driving significant environmental damage.

3.5. Conclusion

This study quantified the fate of exported European polyethylene waste destined for recycling, combining reported trade data with estimations of reprocessing efficiencies and fates of residual waste streams in destination countries. We estimate 83,187 Mg (32,115 Mg – 180,558 Mg), or 3% (1% − 7%) of the exported European PE ended up in the ocean, suggesting that exported recycling has the potential to be an important pathway of plastic debris into the ocean that has so far not been accounted for. Given that such a large share of waste destined for recycling is exported, with poor downstream traceability, this study suggests that "true" recycling rates may deviate significantly from rates reported by municipalities where the waste originates. The 2017 mass flows presented here provide a baseline against which to evaluate changes in plastic waste management and policy, pertinent to the circular economy paradigm and emerging regulations around trade in plastic waste. Data are limited regarding convoluted pathways of trade in plastic waste, possible under-reporting of plastic waste flows and uncertain fates in destination countries. More effort is required to document these aspects of plastic waste management that have important implications for the circular economy. Life cycle assessment of plastic value chains, and associated management and policy decisions, should be modified to represent differential rates of plastic recycling related to final destinations of traded waste flows.

3.6. References

Adrados, A., de Marco, I., Caballero, B.M., López, A., Laresgoiti, M.F., Torres, A., 2012. Pyrolysis of plastic packaging waste: A comparison of plastic residuals from material recovery facilities with simulated plastic waste. Waste Manag. 32, 826–832. https://doi.org/10.1016/J.WASMAN.2011.06.016

Auta, H.S., Emenike, C.., Fauziah, S.., 2017. Distribution and importance of microplastics in the marine environment: A review of the sources, fate, effects, and potential solutions. Environ. Int. 102, 165–176. https://doi.org/10.1016/J.ENVINT.2017.02.013

Barlow, C.Y., Morgan, D.C., 2013. Polymer film packaging for food: An environmental assessment. Resour. Conserv. Recycl. 78, 74–80. https://doi.org/10.1016/J.RESCONREC.2013.07.003

Beard, T., 2019. War on Plastic with Hugh and Anita. BBC, England.

Belboom, S., Léonard, A., 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

Bertolini, M., Bottani, E., Vignali, G., Volpi, A., 2016. Comparative Life Cycle Assessment of Packaging Systems for Extended Shelf Life Milk. Pack. Technol. Sci. 29, 525–546. https://doi.org/10.1002/pts.2235

Björklund, A., Finnveden, G., 2005. Recycling revisited-life cycle comparisons of global warming impact and total energy use of waste management strategies. Resour. Conserv. Recycl. 44, 309–317. https://doi.org/10.1016/j.resconrec.2004.12.002

Bogart, S., 2019. SankeyMATIC [WWW Document]. URL http://sankeymatic.com/

Boots, B., Russell, C.W., Green, D.S., 2019. Effects of Microplastics in Soil Ecosystems: Above and Below Ground. Environ. Sci. Technol. 53, 11496–11506. https://doi.org/10.1021/acs.est.9b03304

Brooks, A.L., Wang, S., Jambeck, J.R., 2018. The Chinese import ban and its impact on global plastic waste trade. Sci. Adv. 4, 1–7.

Chiba, S., Saito, H., Fletcher, R., Yogi, T., Kayo, M., Miyagi, S., Ogido, M., Fujikura, K., 2018. Human footprint in the abyss: 30 year records of deep-sea plastic debris. Mar. Policy 96, 204–212. https://doi.org/10.1016/J.MARPOL.2018.03.022

Chilton, T., Burnley, S., Nesaratnam, S., 2010. A life cycle assessment of the closed-loop recycling and thermal recovery of post-consumer PET. Resour. Conserv. Recycl. 54, 1241–1249. https://doi.org/10.1016/J.RESCONREC.2010.04.002

Christensen, T.H., Bhander, G., Lindvall, H., Larsen, A.W., Fruergaard, T., Damgaard, A., Manfredi, S., Boldrin, A., Riber, C., Hauschild, M., 2007. Experience with the use of LCA-modelling (EASEWASTE) in waste management. Waste Manag. Res. 25, 257–262. https://doi.org/10.1177/0734242X07079184

Cimpan, C., Maul, A., Jansen, M., Pretz, T., Wenzel, H., 2015. Central sorting and recovery of MSW recyclable materials: A review of technological state-of-the-art, cases, practice and implications for materials recycling. J. Environ. Manage. 156, 181–199. https://doi.org/10.1016/J.JENVMAN.2015.03.025

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: A review. Mar. Pollut. Bull. 62, 2588–2597. https://doi.org/10.1016/J.MARPOLBUL.2011.09.025

Collins, C., 2019. The Global Environmental Recycling Crisis. What Options Exist for Plastic Waste? Washington, DC.

Ellen MacArthur Foundation, 2017. The New Plastics Economy: Rethinking the Future of Plastics & Catalysing Action.

Emmerik, T., Schwarz, A., 2020. Plastic debris in rivers. WIREs Water 7. https://doi.org/10.1002/wat2.1398

Eriksen, M., Damgaard, A., Boldrin, A., Astrup, T.F., 2018. Quality Assessment and Circularity Potential of Recovery Systems for Household Plastic Waste. J. Ind. Ecol. 23, 156–168. https://doi.org/10.1111/jiec.12822

European Commission, 2018a. A European Strategy for Plastics in a Circular Economy. COM/2018/028 final. Brussels.

European Commission, 2018b. Proposal for a DIRECTIVE OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL on the reduction of the impact of certain plastic products on the environment. COM/2018/340 final. Bussels.

European Commission, 2008. DIRECTIVE 2008/98/EC OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 19 November 2008 on waste and repealing certain Directives. Off. J. Eur. Union 51.

European Commission, 1994. EUROPEAN PARLIAMENT AND COUNCIL DIRECTIVE 94/62/EC of 20 December 1994 on packaging and packaging waste. Off. J. Eur. Union L 365, 10.

Eurostat, 2019. Population on 1 January by age and sex [WWW Document]. URL https://appsso.eurostat.ec.europa.eu/nui/show.do?dataset=demo_pjan&lang=en (accessed 9.18.19).

Fazio, S., Biganzioli, F., De Laurentiis, V., Zampori, L., Sala, S., Diaconu, E., 2018. Supporting information to the characterisation factors of recommended EF Life Cycle Impact Assessment methods, version 2, from ILCD to EF 3.0, EUR 29600 EN. Ispra. https://doi.org/10.2760/002447

Free, C.M., Jensen, O.P., Mason, S.A., Eriksen, M., Williamson, N.J., Boldgiv, B., 2014. High-levels of microplastic pollution in a large, remote, mountain lake. Mar. Pollut. Bull. 85, 156–163. https://doi.org/10.1016/J.MARPOLBUL.2014.06.001

Gall, S.C., Thompson, R.C., 2015. The impact of debris on marine life. Mar. Pollut. Bull. 92, 170–179. https://doi.org/10.1016/J.MARPOLBUL.2014.12.041

Geissdoerfer, M., Savaget, P., Bocken, N.M.P., Hultink, E.J., 2017. The Circular Economy – A new sustainability paradigm? J. Clean. Prod. 143, 757–768. https://doi.org/10.1016/J.JCLEPRO.2016.12.048

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3, e1700782. https://doi.org/10.1126/sciadv.1700782

Günkaya, Z., Banar, M., 2016. An environmental comparison of biocomposite film based on orange peel-derived pectin jelly-corn starch and LDPE film: LCA and biodegradability. Int. J. Life Cycle Assess. 21, 465–475. https://doi.org/10.1007/s11367-016-1042-8

Hong, S.H., Shim, W.J., Hong, L., 2017. Methods of analysing chemicals associated with microplastics: a review. Anal. Methods 9, 1361–1368. https://doi.org/10.1039/C6AY02971J

Hou, P., Xu, Y., Taiebat, M., Lastoskie, C., Miller, S.A., Xu, M., 2018. Life cycle assessment of end-of-life treatments for plastic film waste. J. Clean. Prod. 201, 1052–1060. https://doi.org/10.1016/J.JCLEPRO.2018.07.278

Huysman, S., De Schaepmeester, J., Ragaert, K., Dewulf, J., De Meester, S., 2017. Performance indicators for a circular economy: A case study on post-industrial plastic waste. Resour. Conserv. Recycl. https://doi.org/10.1016/j.resconrec.2017.01.013

Jambeck, J., Geyer, R., Wilcox, C., Siegler, T., Perryman, M., Andrady, A., Narayan, R., Law, K., 2015. Plastic waste inputs from land into the ocean. Sci. Adv. 347, 768–771. https://doi.org/10.1126/science.1260352

Jiang, J.-Q., 2018. Occurrence of microplastics and its pollution in the environment: A review. Sustain. Prod. Consum. 13, 16–23. https://doi.org/10.1016/J.SPC.2017.11.003

Kawecki, D., Scheeder, P.R.W., Nowack, B., 2018. Probabilistic Material Flow Analysis of Seven Commodity Plastics in Europe. Environ. Sci. Technol. 52, 9874–9888. https://doi.org/10.1021/acs.est.8b01513

Kaza, S., Yao, L., Bhada-Tata, P., Van Woerden, F., 2018. What a Waste 2.0 A Global Snapshot of Solid Waste Management to 2050. Washington, DC.

Law, K., Thompson, R., 2014. Microplastics in the seas. Science (80-.). 345, 144–145. https://doi.org/10.1126/science.1256304

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. Resour. Conserv. Recycl. 55, 246–259. https://doi.org/10.1016/J.RESCONREC.2010.09.014

Lebreton, L., Andrady, A., 2019. Future scenarios of global plastic waste generation and disposal. Palgrave Commun. 5, 1–11. https://doi.org/10.1057/s41599-018-0212-7

Lebreton, L.C.M., Van Der Zwet, J., Damsteeg, J.W., Slat, B., Andrady, A., Reisser, J., 2017. River plastic emissions to the world's oceans. Nat. Commun. 8, 1–10. https://doi.org/10.1038/ncomms15611

Li, W.C., Tse, H.F., Fok, L., 2016. Plastic waste in the marine environment: A review of sources, occurrence and effects. Sci. Total Environ. 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. J. Ind. Ecol. 16, 420–435. https://doi.org/10.1111/j.1530-9290.2011.00405.x

Lusher, A.L., Hernandez-Milian, G., O'Brien, J., Berrow, S., O'Connor, I., Officer, R., 2015. Microplastic and macroplastic ingestion by a deep diving, oceanic cetacean: The True's beaked whale Mesoplodon mirus. Environ. Pollut. 199, 185–191. https://doi.org/10.1016/J.ENVPOL.2015.01.023

Mastellone, M.L., Cremiato, R., Zaccariello, L., Lotito, R., 2017. Evaluation of performance indicators applied to a material recovery facility fed by mixed packaging waste. Waste Manag. 64, 3–11. https://doi.org/10.1016/J.WASMAN.2017.02.030

McDonough, W., Braungart, M., 2002. Cradle to cradle : remaking the way we make things. North Point Press.

McDonough, W., Braungart, M., Anastas, P.T., Zimmerman, J.B., 2003. Peer Reviewed: Applying the Principles of Green Engineering to Cradle-to-Cradle Design. Environ. Sci. Technol. 37, 434A-441A. https://doi.org/10.1021/es0326322

Merrild, H., Larsen, A.W., Christensen, T.H., 2012. Assessing recycling versus incineration of key materials in municipal waste: The importance of efficient energy recovery and transport distances. Waste Manag. 32, 1009–1018. https://doi.org/10.1016/J.WASMAN.2011.12.025

Microsoft Corporation, 2019. Excel.

Ministry of Ecology and Environment, 2017. Notice of the General Office of the State Council on Printing and Distributing the Implementation Plan for the Reform of the Import Management System for the Prohibition of Foreign Waste Entry [WWW Document]. URL http://zfs.mee.gov.cn/gz/bmhb/gwygf/201707/t20170728_418692.shtml (accessed 5.21.19).

Mutha, N.H., Patel, M., Premnath, V., 2006. Plastics materials flow analysis for India. Resour. Conserv. Recycl. 47, 222–244. https://doi.org/10.1016/J.RESCONREC.2005.09.003

Pawar, P., Shirgaonkar, S., Patil, R., 2016. Plastic marine debris: Sources, distribution and impacts on coastal and ocean biodiversity. PENCIL Publ. Biol. Sci. 3, 40–54.

Perugini, F., Mastellone, M.L., Arena, U., 2005. A life cycle assessment of mechanical and feedstock recycling options for management of plastic packaging wastes. Environ. Prog. 24, 137–154. https://doi.org/10.1002/ep.10078

PlasticsEurope, 2018. Annual Review 2017-2018. Plast. Annu. Rev. 2017-2018 15, 44.

Pressley, P.N., Levis, J.W., Damgaard, A., Barlaz, M.A., DeCarolis, J.F., 2015. Analysis of material recovery facilities for use in life-cycle assessment. Waste Manag. 35, 307–317. https://doi.org/10.1016/j.wasman.2014.09.012

Qu, S., Guo, Y., Ma, Z., Chen, W.-Q., Liu, J., Liu, G., Wang, Y., Xu, M., 2019. Implications of China's foreign waste ban on the global circular economy. Resour. Conserv. Recycl. 144, 252–255. https://doi.org/10.1016/J.RESCONREC.2019.01.004

REPAK, 2018. Plastic Packaging Recycling Strategy 2018-2030. Clondalkin, Ireland.

Retamal, M., Dominish, E., Thinh, L.X., Nguyen, A.T., Sharpe, S., 2019. Here's what happens to our plastic recycling when it goes offshore [WWW Document]. Conversat. URL http://theconversation.com/heres-what-happens-to-our-plastic-recycling-when-it-goes-offshore-110356 (accessed 9.18.19).

Rigamonti, L., Grosso, M., Sunseri, M.C., 2009. Influence of assumptions about selection and recycling efficiencies on the LCA of integrated waste management systems. Int. J. Life Cycle Assess. 14, 411–419. https://doi.org/10.1007/s11367-009-0095-3

Sangwan, K., Bhakar, V., 2017. Life cycle analysis of HDPE pipe manufacturing – a case study from an Indian industry. Procedia CIRP 61, 738 – 743.

Schmidt, C., Krauth, T., Wagner, S., 2017. Export of Plastic Debris by Rivers into the Sea. Environ. Sci. Technol. 51, 12246–12253. https://doi.org/10.1021/acs.est.7b02368

Schneider, F., Parsons, S., Clift, S., Stolte, A., McManus, M.C., 2018. Collected marine litter — A growing waste challenge. Mar. Pollut. Bull. 128, 162–174. https://doi.org/10.1016/J.MARPOLBUL.2018.01.011

Schwarz, A.E., Ligthart, T.N., Boukris, E., van Harmelen, T., 2019. Sources, transport, and accumulation of different types of plastic litter in aquatic environments: A review study. Mar. Pollut. Bull. 143, 92–100. https://doi.org/10.1016/j.marpolbul.2019.04.029

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

Singh, N., Hui, D., Singh, R., Ahuja, I.P.S., Feo, L., Fraternali, F., 2017. Recycling of plastic solid waste: A state of art review and future applications. Compos. Part B Eng. 115, 409–422. https://doi.org/10.1016/J.COMPOSITESB.2016.09.013

Spierling, S., Röttger, C., Venkatachalam, V., Mudersbach, M., Herrmann, C., Endres, H.-J., 2018. Bio-based Plastics - A Building Block for the Circular Economy? Procedia CIRP 69, 573–578. https://doi.org/10.1016/j.procir.2017.11.017

Strangl, M., Ortner, E., Buettner, A., 2019. Evaluation of the efficiency of odor removal from recycled HDPE using a modified recycling process. Resour. Conserv. Recycl. 146, 89–97. https://doi.org/10.1016/J.RESCONREC.2019.03.009

Talsness, C.E., Andrade, A.J.M., Kuriyama, S.N., Taylor, J.A., vom Saal, F.S., 2009. Components of plastic: experimental studies in animals and relevance for human health. Philos. Trans. R. Soc. B Biol. Sci. 364, 2079–2096. https://doi.org/10.1098/rstb.2008.0281

The World Bank, 2019. World Bank Country and Lending Groups - Country Classification [WWW Document]. URL https://datahelpdesk.worldbank.org/knowledgebase/articles/906519#High_income (accessed 7.9.19).

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., Russell, A.E., 2004. Lost at Sea: Where Is All the Plastic? Science (80-.). 304, 838–838. https://doi.org/10.1126/SCIENCE.1094559

United Nations, 2019a. UN Comtrade [WWW Document]. URL https://comtrade.un.org/ (accessed 6.27.19).

United Nations, 2019b. Bilateral asymmetries - International Trade Statistics [WWW Document]. URL https://unstats.un.org/unsd/tradekb/Knowledgebase/50657/Bilateral-asymmetries (accessed 3.26.20).

United Nations, 2019c. Basel Convention - Plastic waste, Marine Plastics Litter and Microplastics [WWW Document]. URL http://www.basel.int/implementation/plasticwastes/overview/tabid/6068/default.aspx (accessed 9.27.19).

United Nations, 2010. Trade valuation [WWW Document]. UN Trade Stat. URL https://unstats.un.org/unsd/tradekb/Knowledgebase/Trade-valuation (accessed 5.22.20).

van Calcar, C.J., van Emmerik, T.H.M., 2019. Abundance of plastic debris across European and Asian rivers. Environ. Res. Lett. 14, 124051. https://doi.org/10.1088/1748-9326/ab5468

Van Eygen, E., Feketitsch, J., Laner, D., Rechberger, H., Fellner, J., 2017. Comprehensive analysis and quantification of national plastic flows: The case of Austria. Resour. Conserv. Recycl. 117, 183–194. https://doi.org/10.1016/J.RESCONREC.2016.10.017

Velis, C., 2014. Global Recycling Markets: plastic waste - A story of one player- China. Vienna.

Verma, R., Vinoda, K.S., Papireddy, M., Gowda, A.N.S., 2016. Toxic Pollutants from Plastic Waste- A Review. Procedia Environ. Sci. 35, 701–708. https://doi.org/:10.1016/j.proenv.2016.07.069

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Wilcox, C., Van Sebille, E., Hardesty, B.D., 2015. Threat of plastic pollution to seabirds is global, pervasive, and increasing. PNAS 112, 11899–11904. https://doi.org/10.1073/pnas.1502108112

Wilson, D.C., Rodic, L., Modak, P., Soos, R., Rogero, A., Velis, C., Lyer, M., Simonett, O., 2015. Global Waste Management Outlook. https://doi.org/10.1177/0734242X15616055

Wright, S.L., Thompson, R.C., Galloway, T.S., 2013. The physical impacts of microplastics on marine organisms: A review. Environ. Pollut. 178, 483–492. https://doi.org/10.1016/J.ENVPOL.2013.02.031

Xanthos, D., Walker, T.R., 2017. International policies to reduce plastic marine pollution from single-use plastics (plastic bags and microbeads): A review. Mar. Pollut. Bull. 118, 17–26. https://doi.org/10.1016/J.MARPOLBUL.2017.02.048

3.7. Appendices for Chapter 3

3.7.1. Appendix 3.1 – Supplementary data file

Description: The attached excel file contains the key data utilised within the study, as referenced to within the **Chapter 3**.

Supplementary Table 1 (**S1**) contains the UN Comtrade data (United Nations, 2019a) showing the export destinations and quantities for EU28, Norway, and Switzerland for the waste, parings, and scraps of polymers of ethylene (HS code 391510) from 2017. Cells highlighted in blue represent the individual flows which exported greater than 0.010% of the total mass exported from Europe.

Supplementary Table 2 (**S2**) contains the compiled destination countries of the exported polyethylene from EU28, Norway and Switzerland. The table includes the quantity, percentage of exports which each destination country receives, economic classification and the fate of the rejected material in each of these countries for each scenario.

Supplementary Table 3 (**S3**) contains the end-of-life fates from each individual European country, including the exported PE by quantity, percentage, quantity per head for each fate (recycled HDPE and LDPE resins, "landfill", incineration, and ocean debris) for each scenario.

Supplementary Table 4 (**S4**) contains the contribution to total ocean debris originating from the exportation of polyethylene for recycling originating from EU28, Norway, and Switzerland.

Supplementary Table 5 (**S5**) quantifies the European bilateral asymmetry of intra-European trade of PE waste, by matching "export to" with "import from" flows (of volumes equal to at least 0.01% of the total mass exported from Europe, as modelled) for the pertinent countries assumed to implement more complete reporting, i.e., EU-28, Norway, and Switzerland.

Location: The data set has been uploaded to Zenodo data repository at Bishop (2021); https://doi.org/10.5281/zenodo.5798676

File name: Chapter_3_supplementary_material_GB.xls

3.7.2. Appendix 3.2 – Mass flows HIGH scenario

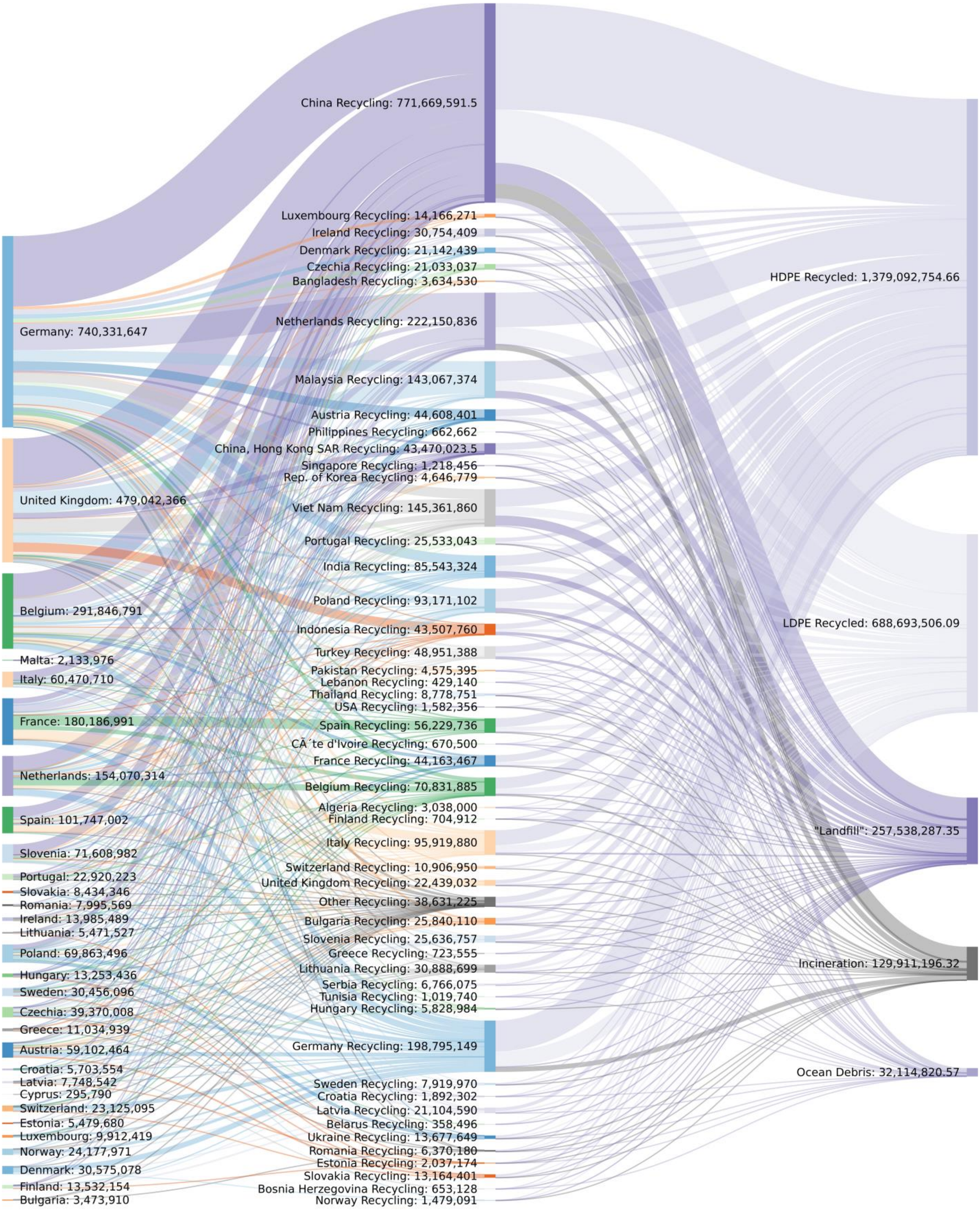

Figure A3.1 – Aggregate mass flows for the high recovery efficiency scenario (HIGH)

3.7.3. Appendix 3.3 – Mass flows LOW scenario

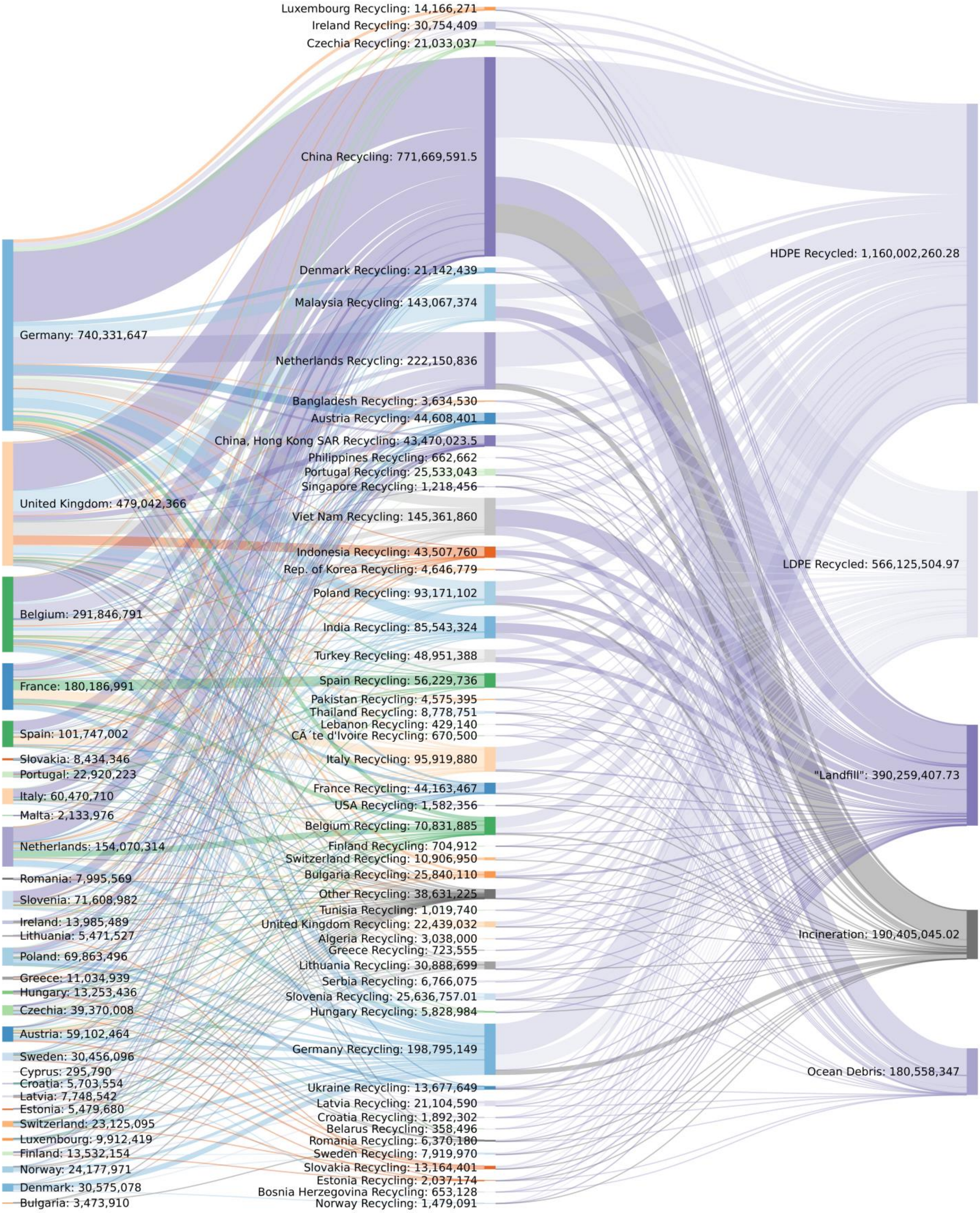

Figure A3.2 – Aggregate mass flows for the low recovery efficiency scenario (LOW)

4. Environmental performance of bioplastic packaging on fresh food produce: a consequential life cycle assessment

Abstract

Polylactic acid (PLA) is a compostable bio-based plastic that can be used for food packaging, potentially increasing separation of (packaged) food waste for targeted, more circular organic waste management via anaerobic digestion, industrial composting, or (in the future) insect protein meal feed production. Consequential life cycle assessment (LCA) was undertaken to rigorously assess the environmental impact of displacing petrochemical plastic packaging of fresh fruit and vegetables with PLA. Eight end-of-life scenarios of bioplastic packaging were evaluated against a business-as-usual petrochemical packaging scenario, expanding LCA boundaries to include end-of-life impacts of fruit and vegetable food waste within a UK context. PLA production has a higher impact compared with petrochemical plastic production across many impact categories, but diversion of PLA-packaged food waste to organic recycling can compensate for this, improving the overall environmental performance of bioplastic packaging scenarios. Future diversion of organic waste streams to insect feed (following regulatory change) would lead to the best environmental outcomes, followed by anaerobic digestion. Impact categories ameliorated in bioplastic scenarios include human health effects, climate change, freshwater eutrophication, ionising radiation, photochemical ozone formation, resource use energy carriers, and respiratory inorganics. On the other hand, petrochemical plastic scenarios generate smaller burdens for acidification, marine and terrestrial eutrophication, ozone depletion, and water scarcity. Sensitivity analyses indicate high improvement potential for bioplastic scenarios if the energy efficiency of PLA production can be increased, or if globalised production shifts to industrialised countries with cleaner energy mixes (that currently import most of their plastics). Whilst end-of-life management of the fruit and vegetable food waste has a considerable influence on environmental outcomes, plastic packaging represents a surprisingly large share of the dry matter material flow (about 25%) in fresh produce waste streams. Therefore, it is imperative that future LCA studies of food packaging account for both packaging and (diverted) food waste end-of-life flows.

4.1. Introduction

Global food production must increase by 60% from 2007 levels by 2050 in order to meet the demands of the growing world population (Alexandratos and Bruinsma, 2012). However, more than one third of the food produced for human consumption is wasted, with an estimated 1.3 billion tonnes of edible food waste disposed of globally each year (FAO, 2013, 2011). This wasted food requires almost 1.4 billion hectares of land to produce, which represents around 30 percent of the world's agricultural land area, with further impacts from water depletion and biodiversity loss (FAO, 2013). Thus, food waste[3] has substantial environmental and socioeconomic impacts and is one of the major challenges of the 21st century. In the UK, about 9.5 million tonnes of food is wasted post-farm-gate annually, of which 6.6 million tonnes is generated in the household (WRAP, 2020a). Over two-thirds of this (4.5 million tonnes) is food intended to be eaten, with a value of almost £14 billion (WRAP, 2020a). The remainder (2.1 million tonnes) consists of inedible portions (WRAP, 2020a). Fresh fruit and vegetables represent about 38% of all wasted food in the UK (WRAP, 2018). The UK is actively working towards the UN Sustainable Development Goal 12.3 food waste prevention target[4] and Courtauld 2025 targets[5].

Plastics are the major packaging material used for fresh foods. The global life cycle greenhouse gas (GHG) emissions of plastic use were estimated to be 1.7 Gt CO_2 eq. in 2015, projected to increase to 6.5 Gt CO_2 eq. by 2050 (Zheng and Suh, 2019). Moreover, persistent plastics are a well-known hazard to marine and terrestrial environments, but also to the health of animals and humans (Boots et al., 2019; Kühn et al., 2015; Li et al., 2016). The negative impacts of single-use petrochemical plastics have resulted in moves to phase them out throughout Europe (European Parliament, 2019).

A large share of household food waste is combined with the plastic packaging in which it is purchased. This food waste is difficult and expensive to separate, so the commingled waste often ends up in less preferred waste treatment, such as landfill or incineration (Bernstad et al., 2013). Thus, food waste incurs burdens across food and plastic value chains, representing a priority environmental "hotspot" that needs to be addressed in the shift towards a circular, low-carbon economy. Biodegradable bio-based polymers (biodegradable bioplastics) are being developed as a more sustainable replacement for petrochemical plastics (European Commission, 2018). Such plastics can retain the beneficial material

[3] The term "food waste" includes both wasted food and drink.
[4] Target 12.3 – By 2030, halve per capita global food waste at the retail and consumer levels and reduce food losses along production and supply chains, including post-harvest losses.
[5] Courtauld 2025 targets aims to prevent food waste and its associated greenhouse gas (GHG) emissions per person by 20% in the UK by 2025

characteristics (food preservation abilities) of petrochemical plastics whilst allowing for a transition towards a circular economy by reducing fossil resource extraction and lowering end-of-life burdens as a result of their biodegradable nature. The current bioplastic market accounts for less than 1% of the entire plastic packaging market, although bioplastic demand within the food packaging industry is rapidly growing (Zhao et al., 2020). Polylactic acid (PLA) is a commercially successful biodegradable bioplastic created from mainly starch feedstocks (Lim et al., 2008). PLA is a versatile material, being, *inter alia*, a thermoplastic, a gas barrier, UV resistant, biocompatible, elastic, rigid, and hydrophobic (Jabeen et al., 2015; Karan et al., 2019). PLA thus has the potential to replace many petrochemical plastics as a packaging material. However, before large-scale system changes are adopted, full environmental evaluations should be considered.

Life cycle assessment (LCA) is a method of quantifying the environmental impacts arising over the entire value chain of a product or service (ISO, 2006a, 2006b). Consequential LCA models are prospective as they aim to model the consequences of future decisions. Consequential LCA is a system modelling approach in which activities are included in the product system(s) being evaluated only to the extent that they are expected to change as a consequence of a change in demand for the functional unit (LCA-2.0, 2015; Sonnemann and Vigon, 2011). Consequential modelling therefore uses unconstrained (or marginal) suppliers in the product systems that can increase (or decrease) production if there is an increase (or decrease) in demand for a product or process, as well as for the products and processes which will be substituted in other systems (i.e., system expansion) due to additional production of co-products (Ekvall et al., 2016; Ekvall and Weidema, 2004). It is argued that consequential models are more appropriate than attributional models when evaluating changes to product systems (Weidema et al., 2018).

Recent reviews on LCA of bioplastics have found that LCA methodologies of bioplastic are often inadequate, through limited or even biased selection of impact categories, incomplete input data, inadequate representation of indirect land-use change and carbon storage, and lack of consequential modelling (Bishop et al., 2021; Pawelzik et al., 2013; Yates and Barlow, 2013). Due to the potential suitability of compostable bioplastics for organic recycling via anaerobic digestion (AD) or industrial composting, the use of a compostable bioplastic food packaging could make it easier to separate out organic waste in the household, wherefore the bioplastic could be disposed of with the food waste for targeted collection, thereby increasing collection and treatment efficiency. Previous studies have explored the consequential life cycle environmental impacts of food waste (Salemdeeb et al., 2017; Styles et al., 2020; Tonini et al., 2018), and a few studies have applied attributional LCA to assess bioplastic packaging and food waste together, e.g., homogeneous composting of bioplastic tableware and food waste (Fieschi and Pretato, 2018). However, a review by Kakadellis and Harris (2020) found a clear shortcoming of LCA studies of bioplastics that typically do not include food waste management (changes) within their system boundaries. Addressing this gap requires application of prospective LCA that considers the interaction between technology and consumer behaviour (Polizzi di Sorrentino et al., 2016).

The European Strategy for Plastics in a Circular Economy suggests that innovative materials and alternative feedstocks for plastic production should be developed and used where evidence clearly shows that they are more sustainable compared to petrochemical plastics (European Commission, 2018). There is thus an urgent need for more comprehensive and appropriately designed LCA studies to provide clear evidence on the sustainability of bioplastics, and how they benchmark against conventional plastics. There is a specific research gap in forward-looking, consequential LCA of bioplastic food packaging that accounts for potential diversion of food waste streams. The aim of this study is to fill that gap by comprehensively assessing the environmental consequences of replacing petrochemical plastic food packaging of fresh fruit and vegetables with PLA within future-orientated scenarios representing graduated levels of enhanced waste separation.

4.2. Methodology

4.2.1. Goal and scope

The goal of this consequential LCA was to rigorously evaluate the displacement of petrochemical food packaging used for fresh fruit and vegetables with a biodegradable bioplastic (PLA), in response to a future ban on single-use petrochemical plastics. In addition to PLA production, the possibility of greater levels of organic waste separation within the household, facilitated by the PLA packaging, was explored. Eight end-of-life scenarios of bioplastic packaging were evaluated against a baseline, business-as-usual (BAU) petrochemical packaging scenario (see **Chapter 4.2.2**).

The functional unit for this study was defined as **the management of 1 tonne of fresh fruit and vegetable food waste generated from UK households and associated food packaging of 51.12 kg**. Each tonne of fresh fruit and vegetable waste was comprised of 622.5 kg of vegetable waste, and 377.5 kg fruit waste, calculated from a disaggregated breakdown of the food waste from WRAP (2018) (described in **Chapter 4.2.3.1**). The mass of plastic packaging waste per tonne of fruit and vegetable food waste generated was calculated based on the study by Lebersorger and Schneider (2011). They estimated that food packaging represented 7% and 2% of product mass for vegetables and fruit, respectively, approximating to 51.12 kg per tonne of combined fruit and vegetable food waste in the UK.

Consequential modelling was applied with the geographic scope of the study being the UK. Therefore, all the foreground inventory data for marginal food waste composition, marginal technologies, and the legislative context were specific to UK conditions. Plastic production has a global market, so was not confined to the UK. Supplementary information is provided as an Excel workbook containing the input data and arithmetic manipulations involved in the life cycle inventory, full results, and uncertainty analyses. The calculation of LCA results and the uncertainty simulations were performed using openLCA 1.10.2 software (GreenDelta, 2021).

In the study there were a series of assumptions that were made for clarity and to aid the main goal of the LCA:

- It was assumed that the type of plastic packaging did not affect the quantities of household food waste generated.
- The process of product formation from plastic granulates was excluded from each scenario due to uncertainty of breakdown of the products, i.e., granulates into products such as films or trays. However, it was assumed that product formation burdens were the same per kg plastic across scenarios and across plastic types.
- Upstream food production burdens were excluded from the study, as not to distract from the studied results from the chapter. However, it was assumed that emissions were the same across scenarios and plastic types. For the food waste, just the end-of-life management was considered.
- The "use" phase of the household food and plastic packaging was excluded from the study (e.g., no emissions were modelled for culinary activities, or storage of the fruit and vegetables), due to large variations in households that can occur.
- Waste collection bags were excluded from the study, again due to considerable uncertainty.

The major processes accounted for in this study are shown in **Figure 4.1.**

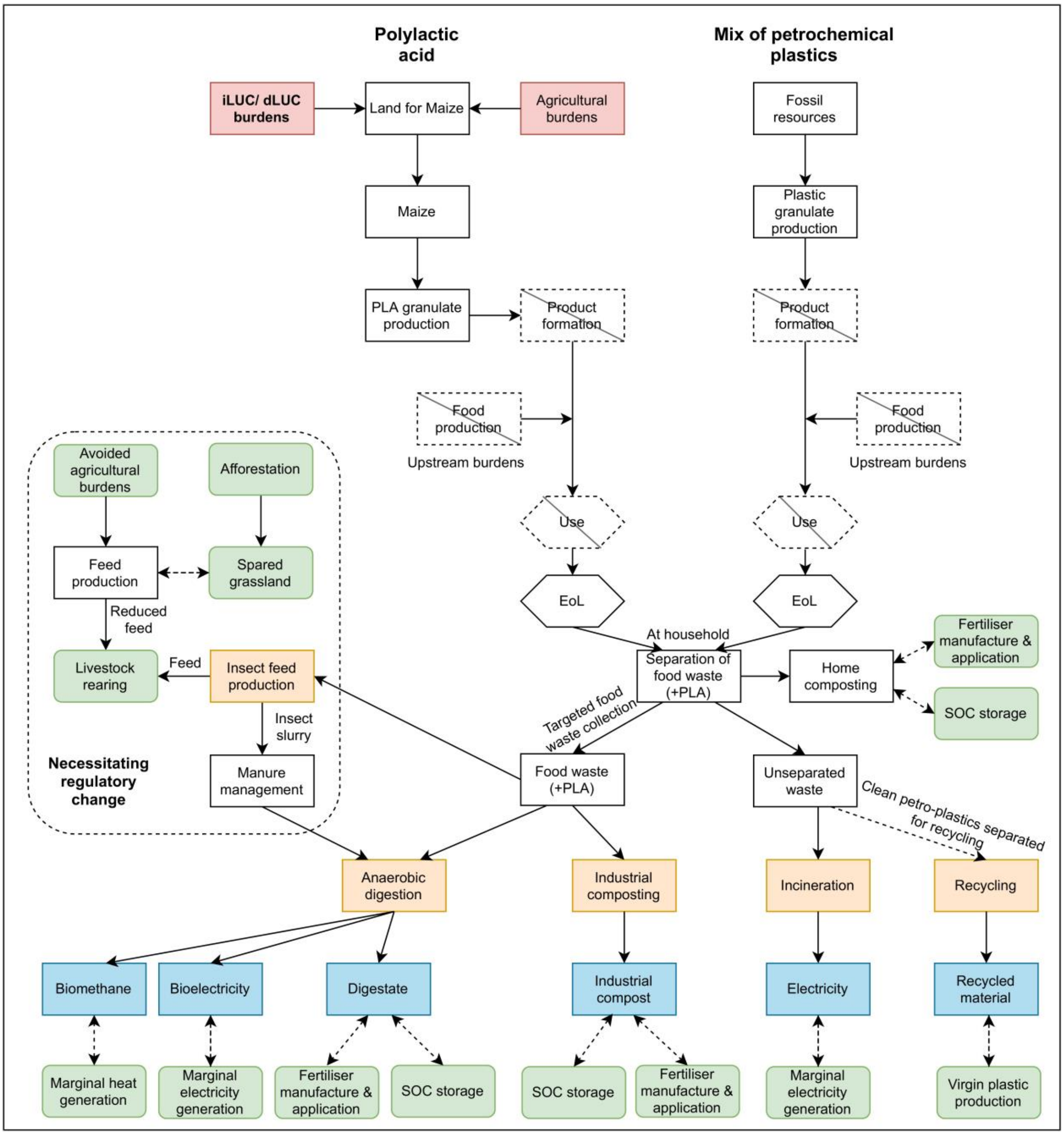

Figure 4.1 – The system boundary of the study. A schematic representation of the major processes modelled within this LCA study. iLUC: indirect land-use change; dLUC: direct land-use change; EoL: end-of-life; PLA: polylactic acid; SOC: soil organic carbon

4.2.2. Scenarios

This study incorporated eight bioplastic and food waste scenarios, and a business-as-usual (BAU) petrochemical plastic and food waste scenario. The scenarios evaluated in this study are described below and relate to **Figure 4.1**. Further details of the scenarios can be found in **Chapter 4.2.3**. To facilitate interpretation and benchmarking, in the first instance, scenario performance was calculated independently within the consequential LCA framework. Subsequent analyses consider full consequential *transitions* from the (avoided) petrochemical plastic BAU to each specific bioplastic scenario.

4.2.2.1. Petrochemical business-as-usual scenario (BAU)

This baseline scenario of the LCA study evaluated continuing usual practise. Food waste may be separated from the plastic packaging in households for weekly or biweekly targeted food waste collection, depending on the local municipality. In 2015, the proportion of total household food waste that was collected in targeted food waste collection schemes was 13.1% (WRAP, 2016). This value is increasing each year as more local authorities are increasing the coverage of targeted food waste collection (WRAP, 2016). This study therefore conservatively used a future BAU separation efficiency of 20% food waste collected for recycling. The end-of-life treatment of food waste was modelled as a mixture of home and industrial composting and AD for separated food waste, and incineration for non-separated food waste (**Table 4.1**). The destination of the targeted food waste collection was assumed to be a 50% split between AD and industrial composting (Tonini et al., 2018). Some food waste is home composted, which we assumed to be 15% of the generated household food waste, based on increasing levels of home composting (WRAP, 2020b). Landfilling is being phased out under EU regulation (European Commission, 2014; European Union, 1999), so, although it represents up to 48% of food waste treatment in parts of the UK (Salemdeeb et al., 2017), it is unlikely to represent a significant primary food waste disposal option in the future. Therefore, the remaining 65% of the fruit and vegetable food waste was incinerated following kerbside collection, representing a conservative and future-looking approach to the assessment of PLA impacts. The polypropylene (PP), high-density polyethylene (HDPE), low-density polyethylene (LDPE), and polyethylene terephthalate (PET) plastic packaging waste were modelled to be treated as 55% recycled, in line with proposed EU targets (European Commission, 2018). The low-density polyethylene (LDPE) plastic packaging was assumed to have a lower recycling rate of just 10%, double the current UK collection rate of plastic films (WRAP, 2019). The remaining plastic waste that is not recycled was incinerated.

Table 4.1 – Future business-as-usual scenario for the end-of-life fate of the food waste following separation at the household for targeted waste collection, given as percentages of total food waste generated in the UK

Scenario	Total targeted food waste collected	Industrial composting	Anaerobic digestion	Home composting	Incineration
	(% destinations of food waste generated in the household)				
Business-as-usual (BAU)	20	10	10	15	65

4.2.2.2. Bioplastic scenarios

The feedstock for the PLA in this study was maize. For the maize production, regional production processes were modelled (**Chapter 4.2.3.2.2**). As PLA can be organically recycled alongside the food waste, scenarios were evaluated relating to the levels of separation of the food waste for targeted food waste collection (i.e., in a food waste bin), including bioplastic packaging collection alongside the food waste (i.e., same organic recycling rate for PLA as for food waste). The scenarios included transportation of the waste from the household to different end-of-life treatments of incineration for non-separated organic waste, or industrial composting, AD, or feeding to insects to produce animal feed (**Chapter 4.2.3.4**) for separated organic waste. Home composting was assumed to occur for the food waste only, as PLA will not biodegrade within a reasonable time-frame in home compost (Su et al., 2019). For the fraction of food waste that was diverted for home compost, the associated PLA packaging was assumed to be separated in the household and placed in the targeted food waste collection bin.

The end-of-life fate proportions of waste were split across eight scenarios (**Table 4.2**) based on BAU, and then scenarios of increasing targeted waste collection from local authorities, culminating in total diversion to specific biowaste management options for comparative purposes. Increased targeted waste collection was considered to be diverted to anaerobic digestion due to it being the preferred waste treatment option for food waste in the UK (Styles et al., 2020).

Table 4.2 – Bioplastic scenarios for end-of-life fate of combined food waste and bioplastic packaging following separation at the household for targeted waste collection, given as percentages of total food and packaging waste generated

Scenario	Total targeted waste collected	Industrial composting	Anaerobic digestion	Home composting *	Incineration	Insect feed
		(% destinations of food waste generated in the household)				
Scenario 1 (S1)	**20**	10	10	15	65	0
Scenario 2 (S2)	**40**	10	30	15	45	0
Scenario 3 (S3)	**60**	10	50	15	25	0
Scenario 4 (S4)	**80**	10	70	15	5	0
Scenario Anaerobic Digestion (SAD)	**100**	0	100	0	0	0
Scenario Composting (SComp)	**100**	100	0	0	0	0
Scenario Incineration (SIncin)	**0**	0	0	0	100	0
Scenario Insect Feed (SIF)	**100**	0	0	0	0	100

*Home composting assumed for food waste only as PLA will not biodegrade within a reasonable timeframe in home composting. The separated PLA from the home composted food waste is assumed to be placed in targeted waste collection bin and split evenly between anaerobic digestion and industrial composting.

4.2.3. Inventory analysis

Within the life cycle inventory (LCI), the environmental aspects that were analysed included energy consumption, transportation, nutrient flows, water use, and chemical use. The full life cycle was modelled for the system (as described in **Figure 4.1**), to include production, construction, operation, and end-of-life impacts. Within this section, the calculations for the foreground data are described. Background data were sourced from the ecoinvent 3.5 consequential database (Wernet et al., 2016). Processes were chosen to refer to the UK reality. When unavailable, authors chose the closest available processes and changed the energy matrix to the UK matrix. A full breakdown of the LCI can be found in the **Supplementary Material, Appendix 4.1**.

4.2.3.1. Food waste composition

The composition of food waste strongly influences downstream waste management processes. **Table 4.3** shows the quantities of different fruit and vegetable waste modelled within this study. The disaggregated breakdown of UK household food waste was based on the latest available data (WRAP, 2018). These values were used to extrapolate the estimated food waste composition from households to the year 2030, by keeping the composition of food waste constant and uniformly adjusting to the anticipated total changes in food waste production identified from WRAP (2020b, 2018) and Styles et al. (2020). Thus, the total fresh fruit and vegetable food waste generated from households in 2030 was estimated to be 2,068,226 tonnes. See **supplementary material: tab 1** (**Appendix 4.1**) for a full methodological and nutritional breakdown of the modelled fruit and vegetable waste. The chemical, physical, and nutritional properties of the individual food products were taken from Tonini et al. (2018) to calculate weighted averages for the overall modelled food waste. The weighted average dry matter content of the fruit and vegetable waste was calculated to be 15.61%.

Table 4.3 – Composition of fruit and vegetable food waste from UK households, projected for the year 2030. Calculated from baseline data from WRAP (2018)

Food waste generated		**2030**	
Category	**Food Type**	**Food waste (tonnes)**	**Mass per tonne of food waste (tonnes)**
Vegetable	Potato (fresh)	592,591	0.287
Vegetable	Onion (fresh)	100,156	0.048
Vegetable	Carrot (fresh)	91,810	0.044
Vegetable	Lettuce (fresh)	56,755	0.027
Vegetable	Other root vegetables (fresh)	49,243	0.024
Vegetable	Cabbage (fresh)	48,409	0.023
Vegetable	Cucumber (fresh)	41,732	0.020
Vegetable	Tomato (fresh)	40,062	0.019
Vegetable	Cauliflower (fresh)	38,393	0.019
Vegetable	Broccoli (fresh)	34,220	0.017
Vegetable	Pepper (fresh)	29,212	0.014
Vegetable	Mixed vegetables (fresh)	25,874	0.013
Vegetable	Leafy salad (fresh)	20,866	0.010
Vegetable	Mushroom (fresh)	18,362	0.009
Vegetable	Leek (fresh)	17,527	0.008
Vegetable	Sweetcorn / corn on the cob (fresh)	13,354	0.006
Vegetable	Bean (all varieties) (fresh)	10,850	0.005

Vegetable	Spring onion (fresh)	9,181	0.004
Vegetable	All other fresh vegetables and salads	59,259	0.029
Fruit	Banana	267,083	0.129
Fruit	Melon	91,810	0.044
Fruit	Apple	83,464	0.040
Fruit	Orange	81,794	0.040
Fruit	Stone fruit	58,424	0.028
Fruit	Pineapple	56,755	0.027
Fruit	Other citrus	47,574	0.023
Fruit	Soft / berry fruit	42,566	0.021
Fruit	Pear	19,197	0.009
Fruit	All other fresh fruit	21,701	0.010
	Total	**2,068,226**	**1**

4.2.3.2. Marginal suppliers

The ability of suppliers to respond to a marginal increase in demand may be constrained by market failures, declining markets, regulations, redundant technologies, high financial cost of production, and/or shortage of raw materials and other necessary production factors (Weidema, 2003). A marginal supplier is identified as the most competitive with a steady increase or constant trend that is unaffected by such constraints (Weidema, 2003).

4.2.3.2.1. Petrochemical plastic packaging production

This study assumes the composition of the petrochemical plastic food packaging for the fruit and vegetables to be a mix of PP, HDPE, LDPE, and PET, the most common food packaging plastics. The split of plastic packaging into their individual polymers was based on the UK plastic packaging placed on the market (WRAP, 2019): 19% PP, 19% LDPE, 31% HDPE, and 31% PET. For plastic packaging production, global market data from ecoinvent 3.5 consequential were used (Wernet et al., 2016). The background data included all major processes of granulate production, from raw material extraction to delivery at plant, and is derived from the eco-profiles of the European plastics industry (Wernet et al., 2016).

4.2.3.2.2. PLA packaging production

The data for the production of PLA packaging were taken from ecoinvent 3.5 consequential (Wernet et al., 2016), based on data from the world's largest PLA plant, NatureWorks. Detailed maize production markets were modelled in the dataset, where the marginal maize is produced in multiple locations, as considered in ecoinvent. The main processes for the production of maize included: soil cultivation; transport of seeds, fertilisers, and pesticides to the field; sowing; fertilisation; irrigation; weed, pest, and pathogen control; harvest; transport from field to farm; and drying of grains. Direct land use change emissions were low in the ecoinvent datasets because most maize feedstock was modelled to be appropriated from existing agricultural land. Thus, in this study, potential indirect land-use change (iLUC) emissions from new maize PLA-feedstock cultivation were added to the LCI (**Chapter 4.2.3.3**). iLUC here refers to the process where the production of the maize feedstock displaces the prior crop onto other land in other locations around the world, inducing land-use change at the agricultural frontier (Schmidt et al., 2015). In this study, most biogenic carbon stored within the bioplastic was assumed to be released back to atmosphere in the short-term, so is treated as carbon neutral over its life cycle. However, a fraction of biogenic carbon returned to the soil in digestate and compost is assumed to remain out of the atmosphere long-term (13.2% and 11.3%, respectively) and is therefore considered to be "sequestered" within the model (**Figure 4.1**) (**Chapter 4.2.3.4**).

4.2.3.2.3. Energy production

The geographic market for energy is regional, so, the marginal electricity supply was modelled on the UK market using the method of calculating marginal mixes suggested by Schmidt et al. (2011). This method evaluates the change in the share of sources for energy production to the increasing market. The increasing market implies installation of more capacity, which is expected to be of modern technology, rather than old. Thus, the marginal electricity supply for the UK was based on extrapolation of electricity production sources with increasing shares of the market as reported by the IEA (2021). The different technologies were then modelled using ecoinvent 3.5 consequential (Wernet et al., 2016). The marginal mix of heat was calculated in a similar manner, with processes also modelled using ecoinvent 3.5 consequential (Wernet et al., 2016). The breakdown of the marginal energies can be found in the **supplementary material: tab 2** (**Appendix 4.1**).

4.2.3.3. Indirect land-use change

iLUC refers to the indirect consequence of land occupation, where the occupation of some production capacity needs to be compensated elsewhere. Following the consequential approach, iLUC was included in the impacts of PLA maize cultivation. The iLUC modelled in this study followed the deterministic model presented by Tonini et al. (2016). This framework uses a biophysical model that considers the global market for land as the marginal supplier of land. According to the model, additional demand for land, and thus additional production of crops, is supplied from two different sources: land expansion (i.e., deforestation) and intensification of land already in use. This model does not consider the social effects from reduced consumption induced by increases in prices since only long-term GHG impacts were considered, and thus the activity is constrained (Schmidt et al., 2015; Tonini et al., 2016). Observing global agricultural statistics over time, the proportion of sources for the change in the output of crop cultivation was 75% increased yields (intensification) and 25% expansion of the cultivated area (Tonini et al., 2016). Intensification is considered as 100% input driven, here modelled as increases in nitrogen (N), phosphorus (P), and potassium (K) fertilisers. The iLUC model considers the geographical location of expansion and affected biomes, the changed flows of carbon (C) and N as a result of expansion, the quantities of increased N, P, K fertiliser used for intensification, and the overall field emissions associated with the fertiliser application. For a detailed description of the model, the reader is referred to the original publication. The amount of arable land demanded by the maize for PLA production was identified as 0.4754 m^2a per kg PLA from ecoinvent 3.5 consequential (Wernet et al., 2016). Upstream emissions of the fertiliser production were also modelled from ecoinvent 3.5 consequential (Wernet et al., 2016). See **supplementary material: tab 8** (**Appendix 4.1**) for a full arithmetic breakdown of the iLUC inventory.

4.2.3.4. End-of-life treatments

For all the scenarios, regardless of the final treatment, waste collection from households was assumed to be performed with a municipal waste collection truck travelling 10 km (Tonini et al., 2018). Construction of waste treatment facilities was included in the life cycle inventories.

4.2.3.4.1. Industrial composting

Industrial composting was assumed to operate in an open-windrow system. **Table 4.4** summarises the framework methodology employed for industrial composting. Within this system, non-biogenic emissions arose from electricity consumption of 20 kWh per tonne organic waste, and from diesel machine operation, which was modelled from similar ecoinvent processes (Wernet et al., 2016). Decomposition emission factors (**Table 4.4**) were based on average emission factors from a literature review by Saer et al. (2013). Compost was assumed to be transported 20 km and applied on land by tractor, with spreading fuel consumption of 0.57 L diesel per tonne compost applied (Tonini et al., 2018). The direct and indirect N emissions from the application of the compost were estimated from the mass of N remaining in compost following decomposition (IPCC, 2006; Yoshida et al., 2016). Nutrients in compost were assumed to displace mineral fertilisers based on the N, P, and K content of the undigested food using chemical/nutritional properties of the individual food products from Tonini et al. (2018) and Kolstad et al. (2012) for the PLA packaging, weighted by proportions of UK household fruit and vegetable waste (see **Chapter 4.2.3.1**). P and K contents were then converted to phosphate (P_2O_5) and potassium oxide (K_2O) contents, based on relative molecular masses, to quantify mineral fertiliser substitution. The long-term fertiliser substitution for P and K was assumed to be 100%, whereas for N a substitution factor of 20% for compost was modelled (Tonini et al., 2018). Displaced embodied emissions in N, P, and K fertilisers were extracted from market data from ecoinvent 3.5 consequential (Wernet et al., 2016). Avoided field emissions from substituted fertilisers were also calculated. The long-term carbon sequestration was modelled as 11.3% of the C applied with the compost (Tonini et al., 2018). See **supplementary material: tab 4** (**Appendix 4.1**) for a full arithmetic breakdown of the inventory.

Table 4.4 – Methods applied to calculate activity data, emissions, and environmental burdens within the industrial composting scenarios

Process		**Method and data to calculate primary emissions and burdens**
Incurred processes	Decomposition	• kg CH_4 = feedstock input × 1.8297 kg/t feedstock (Saer et al., 2013). • kg N_2O = feedstock input × 0.075 kg/t feedstock (Saer et al., 2013). • kg NH_3 = feedstock input × 0.406 kg/t feedstock (Saer et al., 2013). • Burdens = electricity of 20 kWh/t organic waste (Takata et al., 2012), modelled from marginal electricity generation. • Burdens = machine operation, diesel, >= 74.57 kW, low load factor: 0.00035211 hour from ecoinvent 3.5 consequential.
	Compost transport	• Burdens = wet weight × 20 km × burdens per tkm for tractor-trailer from ecoinvent 3.5 consequential. • Burdens = wet weight × 0.57 L diesel/t × burdens of diesel burning from ecoinvent 3.5 consequential. • Assumes 1 kg digestate per 1 kg feedstock wet weight.
	Compost application	• kg NH_3-N and kg NO_3^--N = DM% × total N – N lost in decomposition (above) × EFs (1.6% NH_3-N, 21.8% NO_3^--N) (Yoshida et al., 2016). • kg N_2O-N = DM% × total N – N lost in composition × 1% + (NH_3-N (above) × 1%) + (NO_3-N (above) × 0.75%) (IPCC, 2006).
Avoided processes (credits)	Avoided fertiliser application	• Avoided NH_3-N = DM% × N contents – N lost in decomposition (above) × long-term substitution N fertiliser 20% (Tonini et al., 2018) × 1.7% (Misselbrook et al., 2012). • Avoided NO_3-N = DM% × N contents – N lost in decomposition × long-term substitution N fertiliser 20% (Tonini et al., 2018) × 10% (Duffy et al., 2013). • Avoided N_2O-N = DM% × N contents – storage NH_3-N loss × long-term substitution N fertiliser 20% (Tonini et al., 2018) × 1% + NH_3-N × 1% + NO_3^--N × 0.75% (IPCC, 2006). • Avoided P leach = DM% × P content × long-term substitution P fertiliser 100% (Tonini et al., 2018) × 1% (Styles et al., 2016).
	Avoided fertiliser manufacture	• Avoided burdens = DM% × nutrient contents – N lost in decomposition (above) × long-term fertiliser substitution factors (Tonini et al., 2018) × ecoinvent 3.5 burdens for N, P_2O_5, and K_2O fertilisers.
	C sequestration	• C sequestration = C not degraded (C degradation food waste 58%; PLA 65% (Boldrin et al., 2009; Hermann et al., 2011)) × 11.3% long-term C sequestration (Tonini et al., 2018)

tkm: tonne kilometre; DM%: dry matter percentage; PLA: polylactic acid; EFs: emission factors; CH_4: methane; N_2O: dinitrogen monoxide; NH_3: ammonia; NH_3-N: ammonia nitrogen; NO_3^--N: nitrate nitrogen; N_2O-N: dinitrogen monoxide nitrogen; N: nitrogen; P: phosphorus; C: carbon; P_2O_5: phosphorus pentoxide; K_2O: potassium oxide

4.2.3.4.2. Anaerobic digestion

Energy outputs and fugitive emissions from AD were calculated using the *LCAD* model framework described in Styles et al. (2016), summarised in **Table 4.5**. Here a large AD plant was modelled, where food waste and PLA were converted to biogas and digestate at an AD facility. It was assumed that 50% of AD biogas was burned in a combined heat and power (CHP) plant to produce electricity and heat. Electricity produced from the CHP plant, minus parasitic requirements within the plant, substitutes electricity from the UK market (**Chapter 4.2.3.2.3**). Heat from the CHP was used to heat the digester, with any remainder dumped. The other 50% of biogas was modelled to be upgraded into biomethane, which was then injected into the gas grid, substituting the market for heat (**Chapter 4.2.3.2.3**). These biogas uses are in line with future needs for dispatchable low-carbon heat and electricity to meet net zero GHG targets (CCC, 2019).

In the AD plant, the biomethane yield was calculated as 30.39 kg CH_4 per tonne food waste and 11.9 kg CH_4 for the associated flow of PLA (51.12 kg), based on the specific biochemical properties of these feedstocks (Kolstad et al., 2012; Tonini et al., 2018). Digestate was modelled to be stored in a sealed tank. Fugitive emissions of methane from the AD system were modelled as 1% from the digester and 1.5% from storage (Styles et al., 2016). NH_3 and CH_4 emissions from digestate storage were also modelled from Styles et al. (2016). Similarly to the industrial composting, the produced digestate was assumed to be transported 20 km and applied on land with tractors having fuel consumption of 0.57 L diesel per tonne digestate applied (Tonini et al., 2018). The direct and indirect N emissions from the application of the digestate through shallow soil injection were calculated. The long-term carbon sequestration equalled 13.2% of the C applied with the digestate (Tonini et al., 2018). As modelled in **Chapter 4.2.3.4.1**, digestate substitutes mineral fertilisers, mitigating the emissions incurred from the production and application of the N, P and K fertilisers.

Modelling of emissions mitigation arising from the use of the digestate was based on the N (excluding the N losses discussed), P, and K content of the undigested food. P and K contents were then converted to P_2O_5 and K_2O contents, based on relative molecular masses, to quantify mineral fertiliser replacement. The long-term fertiliser substitution for P and K was assumed to be 100%, whereas for N a substitution factor of 40% for the digestate was modelled (Tonini et al., 2018). Displaced embodied emissions in N, P, and K fertilisers were extracted from market data from ecoinvent 3.5 consequential (Wernet et al., 2016), and avoided emissions of fertiliser application were also calculated. For a full detailed description of the model, the reader is referred to Styles et al. (2016) and **supplementary material: tab 3** (**Appendix 4.1**) for a full arithmetic breakdown of the inventory.

Table 4.5 – Methods applied to calculate activity data, emissions, and environmental burdens within the anaerobic digestion scenarios

Process		**Method and data to calculate primary emissions and burdens**
Incurred processes	Digester leakage	• kg CH_4 = DM% × m^3 CH_4 yield × 0.67 kg/m^3 × 1% digester loss (Adams et al., 2015).
	Digestate storage	• kg CH_4 = DM% × m^3 CH_4 yield × 0.67 kg/m^3 × 1.5% (Styles et al., 2016). • kg NH_3-N = DM% × total N × % total N as NH_4^+-N (80%) × 2% (Misselbrook et al., 2012). • Indirect N_2O-N = NH_3-N × 1% (IPCC, 2006).
	Digestate transport	• Burdens = wet weight × 20 km × burdens per tkm for tractor-trailer from ecoinvent 3.5. • Burdens = wet weight × 0.57 L diesel t^{-1} × burdens of diesel burning from ecoinvent 3.5. • Assumes 1 kg digestate per 1 kg feedstock wet weight.
	Digestate application	• kg NH_3-N and kg NO_3^--N = DM% × total N × % total N as NH_4^+-N – storage NH_3-N loss × MANNER NPK EFs (7.5% NH_3-N, 15.5% NO_3^--N) (Nicholson et al., 2013). • kg N_2O-N = DM% × total N – storage NH_3-N loss × 1% + (NH_3-N (above) × 1%) + (NO_3^--N (above) × 0.75%) (IPCC, 2006). • kg P leached = DM% × P content × 1% (Styles et al., 2016).
	CHP combustion	• kg CH_4 = DM% × m^3 CH_4 yield × 0.67 kg/m^3 − 1% digester loss × 50% biomethane use × 0.5% CHP slip (Styles et al., 2016).
	Biogas upgrade	• kg CH_4 = DM% × m^3 CH_4 yield × 0.67 kg/m^3 − 1% digester loss × 50% biomethane use × 1.4% upgrade methane slip (Styles et al., 2016).
Avoided processes (credits)	Avoided fertiliser application	• Avoided NH_3-N = DM% × N contents – storage NH_3-N loss (above) × long-term substitution N fertiliser 40% (Tonini et al., 2018) × 1.7% (Misselbrook et al., 2012). • Avoided NO_3^--N = DM% × N contents – storage NH_3-N loss × long-term substitution N fertiliser 40% (Tonini et al., 2018) × 10% (Duffy et al., 2013). • Avoided N_2O-N = DM% × N contents – storage NH_3-N loss × long-term substitution N fertiliser 40% (Tonini et al., 2018) × 1% + NH_3-N × 1%+ NO_3^--N × 0.75% (IPCC, 2006). • Avoided P leach = DM% × P content × long-term substitution P fertiliser 100% (Tonini et al., 2018) × 1% (Styles et al., 2016).
	Avoided fertiliser manufacture	• Avoided burdens = DM% × nutrient contents – storage NH_3-N loss × long-term fertiliser substitution factors (Tonini et al., 2018) × ecoinvent 3.5 burdens for N, P_2O_5, and K_2O fertilisers.
	Avoided marginal electricity generation	• Avoided burdens = DM% × m^3 CH_4 yield × 0.67 kg/m^3 − 1% digester loss × 50% biomethane use − 0.5% CHP slip × 50 MJ/kg LHV × 1/3.6 MJ/kWh × 55% CHP electricity efficiency – 15.5% parasitic load × marginal electricity generation burdens generated in ecoinvent 3.5.
	Avoided margin heat generation	• Avoided burdens = DM% × m^3 CH_4 yield × 0.67 kg/m^3 − 1% digester loss × 50% biomethane use − 1.4% biomethane slip × 50 MJ/kg LHV × 1/3.6 MJ/kWh × 90% conversion efficiency of LHV in fuel to useful heat × marginal heat generation burdens generated in ecoinvent 3.5 consequential (Styles et al., 2016).
	C sequestration	• From Buswell's equation, 50% mol CO_2 and 50% mol CH_4 biogas composition • CH_4 moles lost in biogas = biomethane yield (kg) / 0.016 kg.mol^{-1}. • C sequestration = C in input – [(biomethane yield/0.016 × 0.012 kg.mol^{-1} CH_4-C) + (biomethane yield/0.016 × 0.012 kg.mol^{-1} CO_2-C)] × 13.2% long-term C sequestration (Tonini et al., 2018).

tkm: tonne kilometre; DM%: dry matter percentage; PLA: polylactic acid; EFs: emission factors; CH_4: methane; NH_3-N: ammonia nitrogen; NO_3^--N: nitrate nitrogen; N_2O-N: dinitrogen monoxide nitrogen; NH_4^+-N: ammonium nitrogen; N: nitrogen; P: phosphorus; C: carbon; P_2O_5: phosphorus pentoxide; K_2O: potassium oxide; CHP: combined heat and power; LHV: lower heating value

4.2.3.4.3. Incineration

The modelling of incineration was based on the methods described in Moult et al. (2018), utilising calculations of net energy released based on the gross energy and water content of the feedstocks. **Table 4.6** summarises the framework methodology employed for incineration. Net thermal energy from combustion was used to generate electricity on site, of which surplus was exported to the UK grid to avoid marginal generation (**Chapter 2.3.2.3**). Incineration energy conversion efficiency was modelled at 22%, and gross thermal energy outputs were reduced by 15.5% to account for parasitic heat loss to the walls of the incinerator (Moult et al., 2018). All excess heat was assumed to be reused within the process or dumped. Small quantities of residues and slag were diverted to inert landfill after incineration, though have the potential to be used as construction infill materials (Aubert et al., 2004). Emissions from the incineration process were adapted from ecoinvent 3.5 consequential (Wernet et al., 2016) to be relevant to the geographical location of the UK and to fit the modelled scenario. See **supplementary material: tab 5** (**Appendix 4.1**) for a full arithmetic breakdown of the net energy produced for each feedstock.

Table 4.6 – Methods applied to calculate activity data, emissions, and environmental burdens within the incineration scenarios

Process		**Method and data to calculate primary emissions and burdens**
Incurred processes	Burdens	• Input and outputs from the incineration process are calculated from adapted ecoinvent 3.5 consequential processes of: ○ Food Waste - treatment of biowaste, municipal incineration, CH ○ PLA - treatment of waste plastic, mixture, municipal incineration, CH ○ HDPE and LDPE - treatment of waste polyethylene, municipal incineration, CH ○ PET - treatment of waste polyethylene terephthalate, municipal incineration , CH ○ PP - treatment of waste polypropylene, municipal incineration , CH ○ Processes adapted to be relevant to the geographical location of the UK and to fit the modelled scenarios and calculated substitutions
Avoided processes (credits)	Avoided marginal electricity generation	• Mitigated emissions = net energy recovered (J/kg) (calculated below) × incinerator energy conversion efficiency (22%) × emissions intensity of grid electricity (taken from ecoinvent 3.5 consequential of modelled marginal mix) (Moult et al., 2018). • Net energy recovered = energy content in the food (Tonini et al., 2018) – (water content (Tonini et al., 2018) × energy required to heat a unit of water to boiling point and then to boil it (calculated below)) (Moult et al., 2018). • Energy required to heat a unit of water to boiling point and then to boil it = [(temperature of water (373 K) – starting temperature of the water (taken as 298 K)) × specific heat capacity of water (4.18 kJ/kg/K)] + latent heat of vaporisation of water (2257 kJ/kg) (Moult et al., 2018). • Gross thermal energy outputs were reduced by 15.5% to account for parasitic heat loss to the walls of the incinerator (Moult et al., 2018).

CH: Switzerland; PLA: polylactic acid; HDPE: high-density polyethylene; LDPE: low-density polyethylene; PET: polyethylene terephthalate; PP: polypropylene

4.2.3.4.4. Insect feed

Producing animal feed from food waste via insects is an ultimate circular use of waste represented in the most ambitious scenario (necessitating regulatory change). In this scenario, the separated waste was fed to insects which were converted into a protein meal to feed livestock. Land was spared due to a reduction in feed demand; this land could be diverted to other priority uses in line with GHG mitigation and circular economy objectives. The production of animal feed via insects was modelled based on an LCA study by van Zanten et al. (2015) which produced house fly larvae meal from a mixture of mainly food waste with some chicken manure. The methodology within this study follows that of Styles et al. (2020), whereby the LCA was simplified by considering that all dry matter feed was provided by the combined household food waste and PLA. 2.8 tonnes of dry matter (DM) food waste was required per tonne of DM larvae meal (Styles et al., 2020). Therefore, one tonne of fruit and vegetable food waste and 51.12 kg of PLA produced 0.074 tonnes of DM larvae meal. 378 kWh of electricity, and 183 kWh of heat were also required for the larvae meal feed production process. Energy was sourced from marginal sources mentioned in **Chapter 2.3.2.3**.

The original study estimated that 1 tonne of larvae meal replaced 0.5 tonnes of fishmeal and 0.5 tonnes of soybean meal on a DM basis (Van Zanten et al., 2015). The avoided upstream burdens and amount of land sparing associated with animal feed substitution were calculated based on ecoinvent 3.5 consequential burdens (Wernet et al., 2016). Reduced soybean demand spares arable land that we assume is afforested to help meet net zero GHG emission targets and potentially longer-term energy security and bioeconomy objectives depending on the use of harvested wood (Brodin et al., 2017). An average rate of C sequestration in soil and biomass following afforestation of 3600 kg C ha^{-1} yr^{-1} was assumed, based on average values for temperate forest regeneration (Searchinger et al., 2018). Finally, approximately 7.88 tonnes of insect manure are produced per tonne larvae meal (van Zanten et al., 2015). This insect manure was assumed to be diverted to AD. The larvae manure was assumed to have an N, P, and K content of 3.28, 0.76, and 0.98 % per tonne of DM, respectively. The DM of the larvae manure was modelled as 38%. The AD process was modelled as per **Chapter 4.2.3.4.2**. See **supplementary material: tab 6** (**Appendix 4.1**) for a full arithmetic breakdown of the inventory.

4.2.3.4.5. Home composting

Home composting produces insufficient heat to decompose PLA (Su et al., 2019). Hence, home composting was modelled using a similar method to industrial composting, applied only to food waste (i.e., excluding PLA), including the emissions from decomposition and application of compost, and emission mitigation via mineral fertiliser substitution and soil carbon sequestration. However, only 18% of the fertiliser substitution credit calculated for industrial composting was applied to home composting, reflecting the proportion of home compost that actually replaces fertiliser use (Andersen et al., 2012). See **supplementary material: tab 7** (**Appendix 4.1**) for a full arithmetic breakdown of the inventory.

4.2.3.4.6. Recycling

PP, HDPE, and PET packaging waste was modelled as 55% recycled, and LDPE packaging as 10% recycled (see **Chapter 2.2.1**). In this study, emissions relating to the sorting at a materials recovery facility and the recycling of the plastics to granulates were included using data from ecoinvent 3.5 consequential burdens (Wernet et al., 2016). The recycled plastics were modelled as substituting virgin granulates from their respective markets. In this study, the transportation from the materials recovery facility to the recycling facility was assumed to be 200 km via lorry and 3000 km via ocean transport (explored in uncertainty analyses), reflecting the global trade of plastic recycling (Bishop et al., 2020).

4.2.4. Impact assessment

4.2.4.1. Impact categories

The life cycle impact assessment (LCIA) used in this study was the Environmental Footprint (EF) 2.0 method. The EF LCIA is a European initiative to harmonise LCA, and includes a thorough collection of 16 midpoint impact categories, that aims to encompass the holistic impacts from the hitherto mentioned modelled system (Manfredi et al., 2012). All 16 impact categories were considered for this study.

4.2.4.2. Uncertainty analysis

Error propagation via Monte Carlo simulations was performed with 1,000 iterations to obtain estimates of result uncertainty associated with multiple model parameter uncertainties. Parameter uncertainties were based on background ecoinvent uncertainties where applicable, and a pedigree matrix for all generated foreground data (Ciroth et al., 2016). The pedigree matrix creates a score based on five aspects of data uncertainty (reliability, completeness, temporal correlation, geographical correlation, and further technological correlation), and applies a geometric standard deviation to the intermediate and elementary exchanges at the unit process level. The applied pedigree matrices scores can be found in **supplementary material: tab 9** (**Appendix 4.1**). For some uncertain parameters, triangular distribution was used which supplied a lower and upper value. These parameters included composting decomposition emission factors, recycling transportation distances, and collection distances for the food and plastic waste. The applied distributions can be found in **supplementary material: tab 9** (**Appendix 4.1**).

The statistical approach followed was similar to that of Pizzol (2019). Statistical analysis was performed to explore where the nine different scenarios were significantly different across all 16 impact categories. The null hypothesis tested assumed that the environmental impacts of the inventory of the different scenarios were equal. Since normality was rejected via Shapiro-Wilk tests, the differences between the environmental impacts of the scenarios within the same impact categories were statistically tested using nonparametric pairwise Mann-Whitney U tests. Significant differences were considered as $\alpha = 0.05$, with Bonferroni correction applied to avoid type 1 errors (false positives).

4.2.4.3. Sensitivity analyses

To explore the influence of critical factors for PLA production that could vary with policy and management decisions, two further sensitivity analyses were conducted. The PLA ecoinvent production data used within the study originated from 2007, and although the background data were extrapolated to the year of the 3.5 consequential database release (2018) within the dataset, with the uncertainty adjusted accordingly (Wernet et al., 2016), PLA is still a developing technology and it is highly probable that the efficiency of the PLA production has and will improve as production scales up and as technologies mature. Therefore, in the first sensitivity analysis, the energy burdens required to produce the PLA granulate were reduced by 50%. Secondly, a sensitivity scenario where the maize was produced and converted to PLA granulate within the UK was modelled, to assess the consequences of the location of the production of the PLA granulate in terms of maize cultivation and marginal energy mix.

4.2.4.4. Total magnitude of burdens for the UK

Footprints per functional unit were then extrapolated up to the 2,068,226 tonnes of total fruit and vegetable food waste estimated to be generated annually in the UK by 2030 (**Table 4.3**). For this quantity of food waste, 105,728 tonnes of plastic packaging waste are also generated. Within this analysis further consequential thinking was applied, wherein the BAU petrochemical plastic production and subsequent waste treatment was modelled to be avoided by deployment of bioplastic packaging for all fresh fruit and vegetables. As such, the calculated total magnitude of burdens for the UK modelled the avoided BAU scenario as a “credit” to the system for the multiple bioplastic scenarios. Within this framework, the environmental savings are denoted by any negative values, whilst positive values represent an overall burden increase (i.e., new impacts greater than any credits).

4.3. Results

4.3.1. LCA results from using PLA food packaging

The contribution analysis of the results from the management of 1 tonne of fresh fruit and vegetable food waste plus 51.12 kg of plastic packaging in different end-of-life waste management scenarios is presented in **Figure 4.2.** A full and further disaggregated breakdown of the results for each scenario and impact category can be found in **supplementary material: tab 10** (**Appendix 4.1**)**.** Overall, the results show variations between the different scenarios, with the SIF scenario being the most environmentally efficient treatment of the PLA, and the bioplastic scenarios improving in environmental efficiency when more food waste is diverted to AD. Plastic production and food end-of-life treatment dominate burdens across the majority of impact categories, and bioplastic scenarios only result in savings across half (eight) of the environmental impact categories assessed, compared with the petrochemical plastic BAU, if 100% of food waste can be diverted to AD (SAD). Only six of the 16 impact categories show lower footprints for bioplastic by scenario S4, compared to eleven out 16 for 100% diversion to insect larvae animal feed production (SIF). The impact categories for which bioplastic scenarios performed comparatively well included (with the level of food waste diversion required for environmental performance to exceed that of petrochemical plastic packaging): cancer human health effects (40%), climate change (100%, for AD and insect feed), freshwater eutrophication (60%), ionising radiation (20%), non-cancer human health effects (20%), photochemical ozone formation (80%), resource (energy carriers) use (60%), and respiratory inorganics (100%, for AD and insect feed). The impact categories for which only SIF achieved lower burdens included: freshwater ecotoxicity, land use, and resource (minerals and metals) use. There were five impact categories for which all the bioplastic scenarios had higher environmental impacts relative to the BAU scenario, including: terrestrial and freshwater acidification, marine eutrophication, terrestrial eutrophication, ozone depletion, and water scarcity. These are typically the impacts where feedstock (maize) cultivation made a particularly large contribution to bioplastic production impacts.

4.3.2. Uncertainty analysis results

Figure 4.3 shows the distribution of results for each scenario over 1,000 Monte Carlo simulations. Simulation results can be found in **supplementary material: tab 11** (**Appendix 4.1**). The initial visual impression is that uncertainties are substantial and may not allow clear differentiation across scenarios in terms of environmental performance. However, it was observed that 89% of all 576 pairwise comparisons undertaken showed statistically significant differences (Bonferroni-corrected p-values < 0.05), with the majority of impact categories exhibiting statistically significant differences across 97% of the pairwise comparisons among scenarios. Therefore, most of the differences across scenarios observed in

Figure 4.2 and **Figure 4.3** are statistically significant. A full breakdown of the statistical pairwise Mann-Whitney U results for each scenario and impact category can be found in **supplementary material: tab 12** (**Appendix 4.1**).

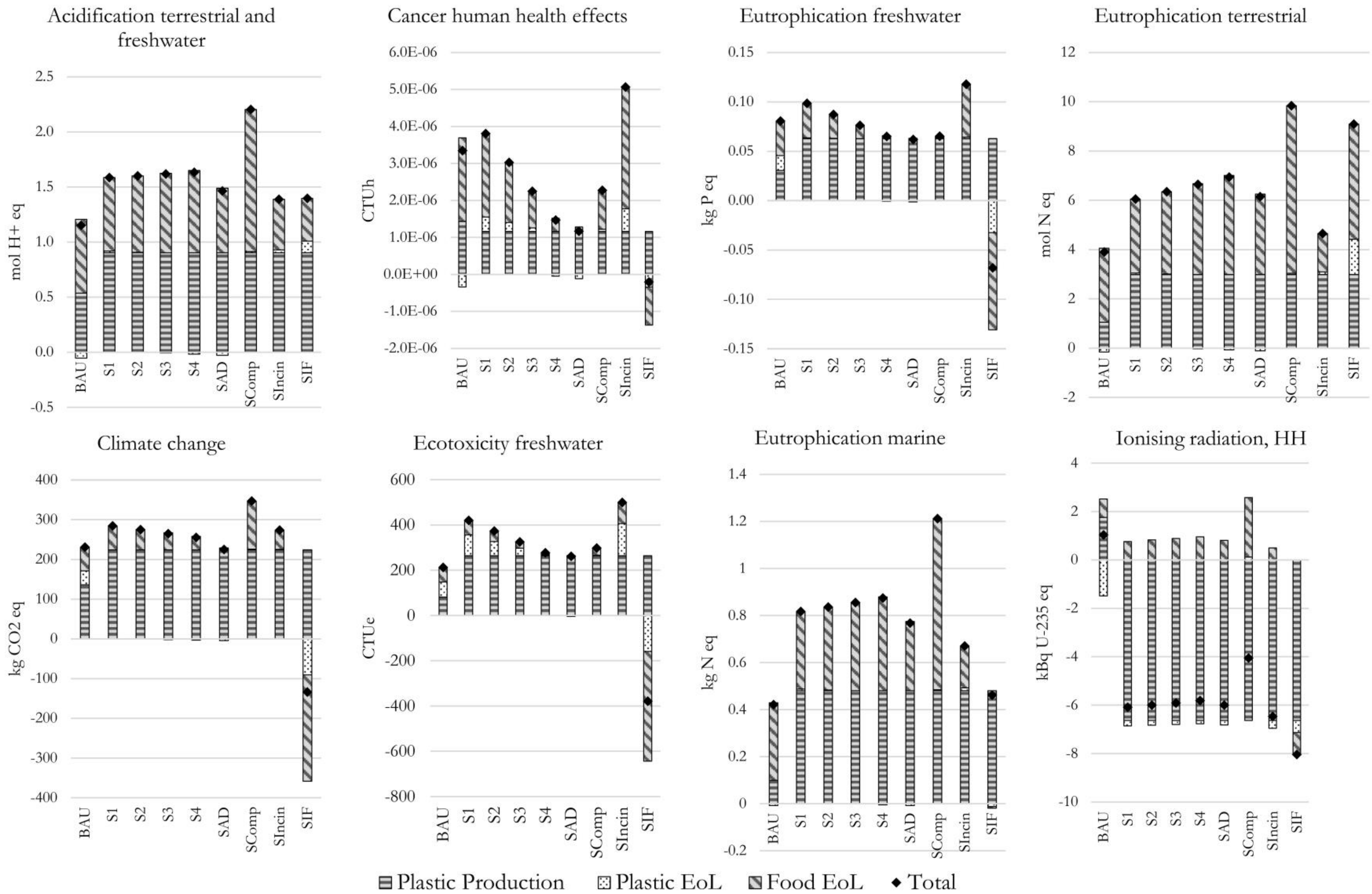

Figure 4.2a – Contribution analysis for the LCIA of the eight bioplastic and food waste scenarios, and the business-as-usual (BAU) petrochemical plastic and food waste scenario, across eight of the 16 impact categories assessed. Horizontal stripes represent burdens from plastic production, dotted bars represent burdens from plastic end-of-life, diagonal stripes represent burdens from food end-of-life. Black diamonds represent the total results for each scenario with each impact category. BAU: business-as-usual (20% separation); S1: scenario 1 (20% separation); S2: scenario 2 (40% separation); S3: scenario 3 (60% separation); S4: scenario 4 (80% separation); SAD: scenario anaerobic digestion (100% separation); SComp: scenario composting (100% separation); SIncin: scenario incineration (100% separation); SIF: scenario insect feed (100% separation)

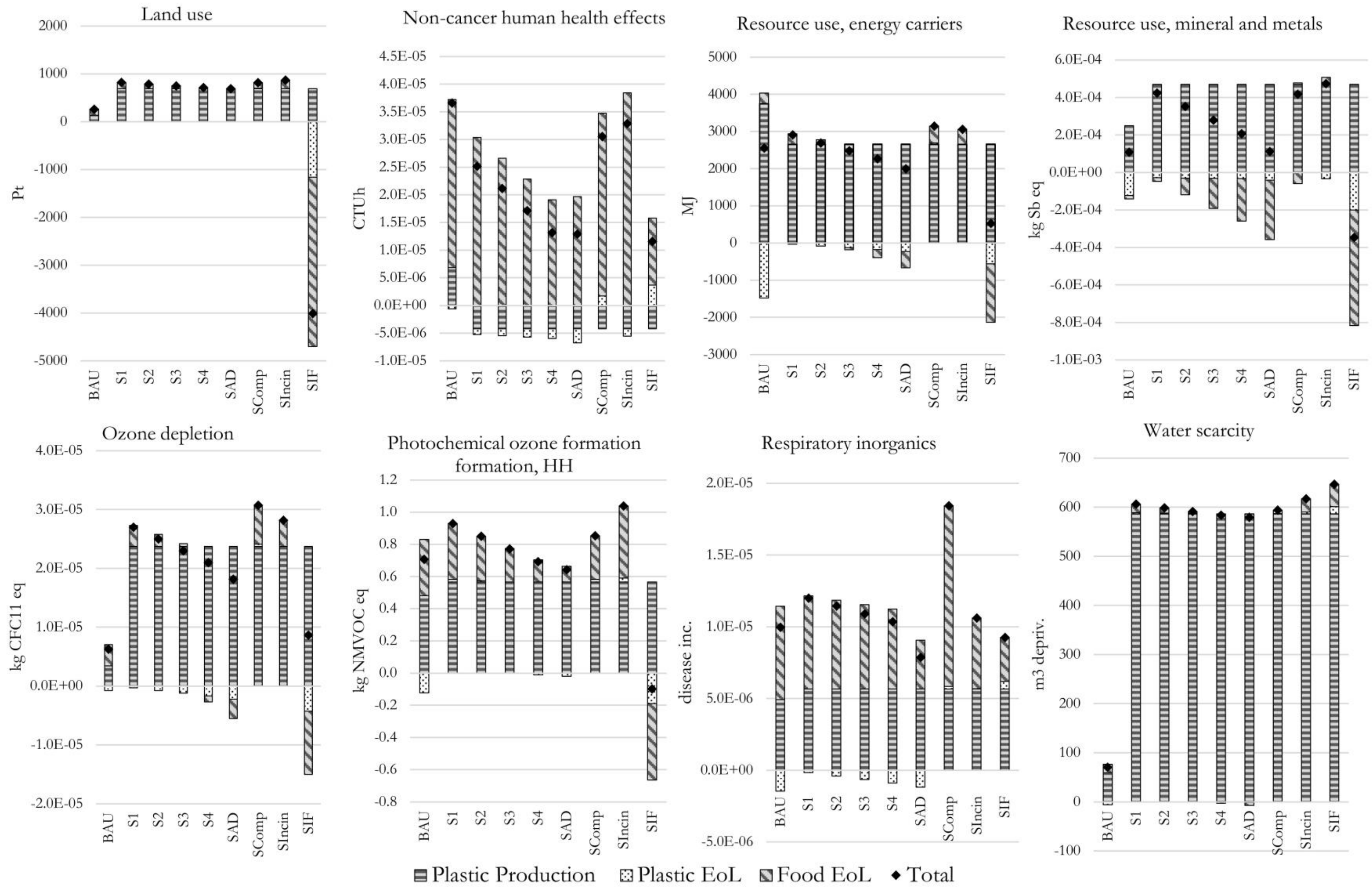

Figure 4.2b – Contribution analysis for the LCIA of the eight bioplastic and food waste scenarios, and the business-as-usual (BAU) petrochemical plastic and food waste scenario, across the remaining eight of 16 impact categories assessed. Horizontal stripes represent burdens from plastic production, dotted bars represent burdens from plastic end-of-life, diagonal stripes represent burdens from food end-of-life. Black diamonds represent the total results for each scenario with each impact category. BAU: business-as-usual (20% separation); S1: scenario 1 (20% separation); S2: scenario 2 (40% separation); S3: scenario 3 (60% separation); S4: scenario 4 (80% separation); SAD: scenario anaerobic digestion (100% separation); SComp: scenario composting (100% separation); SIncin: scenario incineration (100% separation); SIF: scenario insect feed (100% separation)

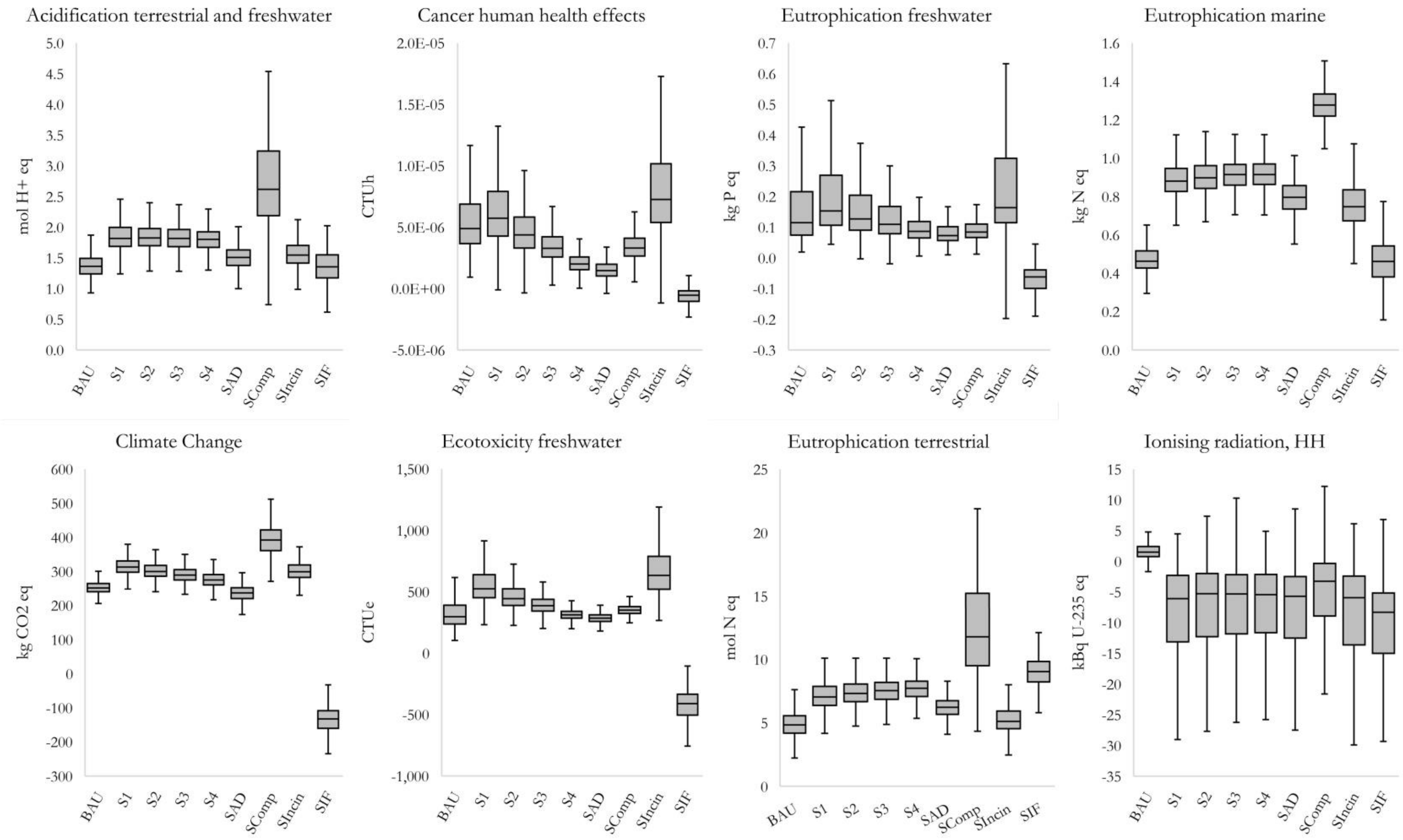

Figure 4.3a – Results of the Monte Carlo simulation for eight bioplastic and food waste scenarios, and the business-as-usual (BAU) petrochemical plastic and food waste scenario, across eight of the 16 impact categories assessed. For visualisation purposes, outliers have been excluded from the graphs, but were included in all statistical analyses. A full breakdown of the Monte Carlo results can be found in **supplementary material: tab 12.** BAU: business-as-usual (20% separation); S1: scenario 1 (20% separation); S2: scenario 2 (40% separation); S3: scenario 3 (60% separation); S4: scenario 4 (80% separation); SAD: scenario anaerobic digestion (100% separation); SComp: scenario composting (100% separation); SIncin: scenario incineration (100% separation); SIF: scenario insect feed (100% separation)

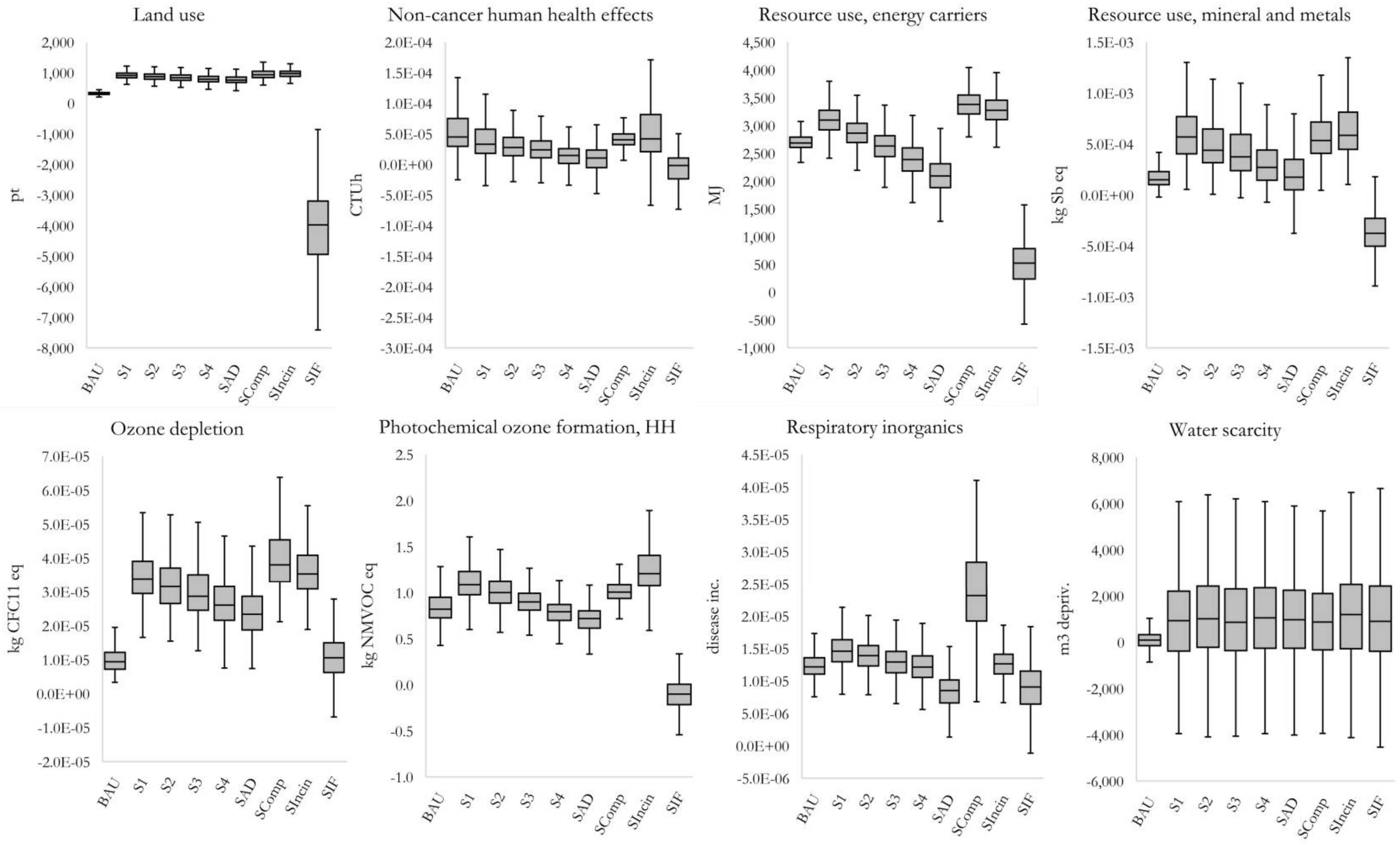

Figure 4.3b – Results of the Monte Carlo simulation for eight bioplastic and food waste scenarios, and the business-as-usual (BAU) petrochemical plastic and food waste scenario, across eight of the 16 impact categories assessed. For visualisation purposes, outliers have been excluded from the graphs, but were included in all statistical analyses. A full breakdown of the Monte Carlo results can be found in **supplementary material: tab 12.** BAU: business-as-usual (20% separation); S1: scenario 1 (20% separation); S2: scenario 2 (40% separation); S3: scenario 3 (60% separation); S4: scenario 4 (80% separation); SAD: scenario anaerobic digestion (100% separation); SComp: scenario composting (100% separation); SIncin: scenario incineration (100% separation); SIF: scenario insect feed (100% separation)

4.3.3. Sensitivity analyses results

The sensitivity analyses undertaken in this study found that there is huge potential to reduce the impacts of bioplastic production. Within the main model under default assumptions, many of the scenarios do not reduce environmental impact compared with the BAU scenario of conventional plastic packaging, as seen in **Figure 4.2** and **Figure 4.4**. However, both sensitivity analyses significantly improve the environmental ranking of the bioplastic scenarios against the petrochemical plastic BAU scenario, with bioplastic scenarios outperforming the petrochemical plastic packaging BAU scenario for the majority of impact categories (**Figure 4.4**). Even where rankings do not change, the absolute burdens of bioplastics were reduced, mitigating environmental trade-offs, as seen in **Figure 4.4** which displays the percentage difference of the bioplastic scenarios from the petrochemical scenario. For example, GWP burdens were only lower than the BAU scenario for two bioplastic scenarios in the baseline modelled LCA. However, both sensitivity analyses resulted in all of the bioplastic scenarios bar one having smaller GWP burdens than the petrochemical scenario. In fact, bioplastic scenario S1, which has the same separation efficiency as the BAU scenario (and therefore one of the least environmentally beneficial outcomes), resulted in 11% or 9% lower GHG emissions than BAU when energy use was reduced by 50% or global production switched to UK production, respectively. Differences from the BAU scenario GWP burden were as great as –192% and –190% for the SIF scenarios within the two sensitivity analyses (**Figure 4.4**). Full results from the sensitivity analysis can be found in **supplementary material: tab 13** (**Appendix 4.1**).

As Modelled		% Difference from BAU Scenario							
Impact category	**BAU**	**S1**	**S2**	**S3**	**S4**	**SAD**	**SComp**	**SIncin**	**SIF**
Acidification terrestrial and freshwater, mol H+ eq	1.15	38	39	40	42	27	91	20	21
Cancer human health effects, CTUh	3.35E-06	14	-10	-33	-56	-65	-32	51	-106
Climate change, kg CO2 eq	230.84	23	19	15	10	-3	50	18	-158
Ecotoxicity freshwater, CTUe	212.01	98	76	53	31	23	41	135	-279
Eutrophication freshwater, kg P eq	0.08	22	8	-6	-19	-23	-19	46	-184
Eutrophication marine, kg N eq	0.42	94	98	103	108	82	188	59	10
Eutrophication terrestrial, mol N eq	3.90	55	63	70	78	57	152	19	133
Ionising radiation, HH, kBq U-235 eq	1.03	-691	-682	-673	-664	-682	-493	-727	-880
Land use, Pt	260.03	215	201	188	174	165	214	233	-1644
Non-cancer human health effects, CTUh	3.66E-05	-31	-42	-53	-64	-65	-17	-10	-68
Ozone depletion, kg CFC11 eq	6.23E-06	333	301	269	237	191	393	352	39
Photochemical ozone formation, HH, kg NMVOC eq	0.71	31	20	9	-2	-9	21	47	-114
Resource use, energy carriers, MJ	2553.43	14	5	-3	-11	-22	23	20	-79
Resource use, mineral and metals, kg Sb eq	1.08E-04	293	226	159	92	4	288	340	-422
Respiratory inorganics, disease inc.	9.96E-06	20	15	9	4	-21	85	6	-7
Water scarcity, m3 depriv.	70.37	761	750	740	729	723	744	776	819

Sensitivity 1 - Improved energy efficiency		% Difference from BAU Scenario							
Impact category	**BAU**	**S1**	**S2**	**S3**	**S4**	**SAD**	**SComp**	**SIncin**	**SIF**
Acidification terrestrial and freshwater, mol H+ eq	1.15	28	29	30	31	17	81	10	11
Cancer human health effects, CTUh	3.35E-06	9	-14	-38	-61	-70	-37	46	-111
Climate change, kg CO2 eq	230.84	-11	-15	-20	-24	-37	16	-16	-192
Ecotoxicity freshwater, CTUe	212.01	91	69	46	24	16	34	129	-285
Eutrophication freshwater, kg P eq	0.08	19	5	-9	-23	-27	-22	43	-188
Eutrophication marine, kg N eq	0.42	87	92	96	101	75	181	53	3
Eutrophication terrestrial, mol N eq	3.90	48	55	63	71	50	145	12	125
Ionising radiation, HH, kBq U-235 eq	1.03	-363	-354	-345	-336	-354	-165	-399	-551
Land use, Pt	260.03	210	196	182	169	160	208	228	-1649
Non-cancer human health effects, CTUh	3.66E-05	-30	-41	-52	-63	-63	-15	-9	-67
Ozone depletion, kg CFC11 eq	6.23E-06	177	145	113	81	35	237	196	-117
Photochemical ozone formation, HH, kg NMVOC eq	0.71	16	5	-6	-17	-24	5	32	-129
Resource use, energy carriers, MJ	2553.43	-28	-37	-45	-53	-64	-19	-22	-122
Resource use, mineral and metals, kg Sb eq	1.08E-04	298	231	163	96	8	292	344	-418
Respiratory inorganics, disease inc.	9.96E-06	15	10	4	-1	-26	80	1	-12
Water scarcity, m3 depriv.	70.37	762	751	740	730	724	744	777	819

Sensitivity 2 - UK PLA production		% Difference from BAU Scenario							
Impact category	**BAU**	**S1**	**S2**	**S3**	**S4**	**SAD**	**SComp**	**SIncin**	**SIF**
Acidification terrestrial and freshwater, mol H+ eq	1.15	2	4	5	6	-8	56	-15	-14
Cancer human health effects, CTUh	3.35E-06	9	-14	-38	-61	-70	-37	46	-111
Climate change, kg CO2 eq	230.84	-9	-13	-17	-22	-35	18	-14	-190
Ecotoxicity freshwater, CTUe	212.01	85	63	40	18	10	28	123	-292
Eutrophication freshwater, kg P eq	0.08	-12	-26	-40	-54	-58	-53	12	-219
Eutrophication marine, kg N eq	0.42	80	84	89	93	68	173	45	-5
Eutrophication terrestrial, mol N eq	3.90	33	41	49	56	36	130	-2	111
Ionising radiation, HH, kBq U-235 eq	1.03	-4087	-4078	-4069	-4060	-4078	-3889	-4123	-4276
Land use, Pt	260.03	19	5	-9	-22	-31	17	37	-1840
Non-cancer human health effects, CTUh	3.66E-05	-34	-45	-56	-67	-68	-19	-13	-71
Ozone depletion, kg CFC11 eq	6.23E-06	102	70	38	5	-40	161	121	-192
Photochemical ozone formation, HH, kg NMVOC eq	0.71	4	-7	-18	-30	-37	-7	19	-141
Resource use, energy carriers, MJ	2553.43	-40	-49	-57	-65	-76	-31	-34	-134
Resource use, mineral and metals, kg Sb eq	1.08E-04	383	316	249	182	94	378	430	-333
Respiratory inorganics, disease inc.	9.96E-06	0	-6	-11	-17	-41	65	-14	-28
Water scarcity, m3 depriv.	70.37	751	740	729	719	713	733	766	808

Difference from BAU Scenario (%)	>200	100→200	0→100	-100→0	-100→-200	<-200

Figure 4.4 – Heat map of the eight different bioplastic scenarios showing the percentage difference of the multiple scenarios from the BAU baseline scenario. The figure contains the results as modelled, and the two sensitivity scenarios as described in **Chapter 4.2.4.3.** BAU: business-as-usual (20% separation); S1: scenario 1 (20% separation); S2: scenario 2 (40% separation); S3: scenario 3 (60% separation); S4: scenario 4 (80% separation); SAD: scenario anaerobic digestion (100% separation); SComp: scenario composting (100% separation); SIncin: scenario incineration (100% separation); SIF: scenario insect feed (100% separation)

4.3.4. *Total UK fruit and vegetable food waste and associated packaging*

Table 4.7 shows the magnitude of environmental impacts associated with the total fresh fruit and vegetable food waste estimated to be generated annually in the UK by 2030. Applying more consequential thinking to the approach provided a credit to the bioplastic scenarios from the displaced petrochemical BAU scenario. It is worth noting that, for the results presented in **Table 4.7**, environmental savings are denoted by any negative values, whilst positive values represent an overall burden increase. As such, converting to a PLA food packaging material appears to be able to considerably reduce the UK's overall emissions from packaging and food waste management when 100% diversion of organic waste to AD or insect feed is achieved. National GHG emissions savings are -12,791,668 and -754,742,657 kg CO_2 eq. for 100% waste diversion to AD and insect feed, respectively. For scenario S1, meanwhile, where the collection efficiency remains unchanged, net environmental savings are only seen in ionising radiation and non-cancer human health effects. Full results from the extrapolated total UK scenario, including results from the extrapolated sensitivity analysis, can be found in **supplementary material: tab 14** (**Appendix 4.1**).

Table 4.7 – Consequential LCA results for all UK fresh fruit and vegetable food waste (2,068,226 tonnes) and associated PLA packaging in 2030 transitioning from BAU (100% petrochemical plastic packaging) to the scenarios of 100% bioplastic packaging and enhanced food waste diversion to dedicated biowaste treatment. Environmental savings are denoted by negative values, whilst positive values represent burden increases following the transition

	Scenarios							
Impact category	**S1**	**S2**	**S3**	**S4**	**SAD**	**SComp**	**SIncin**	**SIF**
Acidification terrestrial and freshwater, mol H^+ eq	8.99E+05	9.30E+05	9.61E+05	9.93E+05	6.44E+05	2.17E+06	4.87E+05	5.08E+05
Cancer human health effects, CTUh	9.46E-01	-6.66E-01	-2.28E+00	-3.89E+00	-4.52E+00	-2.22E+00	3.54E+00	-7.36E+00
Climate change, kg CO_2 eq	1.10E+08	9.03E+07	7.02E+07	5.00E+07	-1.28E+07	2.40E+08	8.79E+07	-7.55E+08
Ecotoxicity freshwater, CTUe	4.30E+08	3.31E+08	2.33E+08	1.34E+08	1.01E+08	1.78E+08	5.94E+08	-1.22E+09
Eutrophication freshwater, kg P eq	3.71E+04	1.39E+04	-9.24E+03	-3.24E+04	-3.88E+04	-3.16E+04	7.70E+04	-3.07E+05
Eutrophication marine, kg N eq	8.18E+05	8.58E+05	8.98E+05	9.38E+05	7.16E+05	1.64E+06	5.16E+05	8.37E+04
Eutrophication terrestrial, mol N eq	4.44E+06	5.06E+06	5.68E+06	6.29E+06	4.63E+06	1.23E+07	1.55E+06	1.07E+07
Ionising radiation, HH, kBq U-235 eq	-1.47E+07	-1.45E+07	-1.43E+07	-1.42E+07	-1.45E+07	-1.05E+07	-1.55E+07	-1.88E+07
Land use, Pt	1.15E+09	1.08E+09	1.01E+09	9.36E+08	8.90E+08	1.15E+09	1.26E+09	-8.84E+09
Non-cancer human health effects, CTUh	-2.37E+01	-3.20E+01	-4.03E+01	-4.85E+01	-4.90E+01	-1.26E+01	-7.73E+00	-5.18E+01
Ozone depletion, kg CFC11 eq	4.29E+01	3.88E+01	3.46E+01	3.05E+01	2.47E+01	5.06E+01	4.54E+01	5.00E+00
Photochemical ozone formation, HH, kg NMVOC eq	4.60E+05	2.97E+05	1.33E+05	-3.07E+04	-1.33E+05	3.02E+05	6.86E+05	-1.67E+06
Resource use, energy carriers, MJ	7.29E+08	2.90E+08	-1.50E+08	-5.90E+08	-1.16E+09	1.22E+09	1.04E+09	-4.19E+09
Resource use, mineral and metals, kg Sb eq	6.54E+02	5.04E+02	3.55E+02	2.05E+02	9.15E+00	6.41E+02	7.57E+02	-9.41E+02

Respiratory inorganics, disease incidence	4.21E+00	3.08E+00	1.95E+00	8.18E-01	-4.33E+00	1.75E+01	1.33E+00	-1.45E+00
Water scarcity, m^3 deprived	1.11E+09	1.09E+09	1.08E+09	1.06E+09	1.05E+09	1.08E+09	1.13E+09	1.19E+09

S1: scenario 1 (20% separation); S2: scenario 2 (40% separation); S3: scenario 3 (60% separation); S4: scenario 4 (80% separation); SAD: scenario anaerobic digestion (100% separation); SComp: scenario composting (100% separation); SIncin: scenario incineration (100% separation); SIF: scenario insect feed (100% separation)

4.4. Discussion

4.4.1. Importance of packaging production on the environmental performance

The results show that a switch from petrochemical plastic to PLA bioplastic packaging for fresh fruit and vegetables within the UK, without any associated increased diversion of organic waste to biowaste treatment, could increase overall environmental burdens. Bioplastic production incurs high environmental costs for maize cultivation and processing energy consumption. Indirect land-use change was found to be a major PLA burden, but is not included in many bioplastic LCA studies (Bishop et al., 2021). The inclusion of iLUC incurs a penalty for the renewable feedstock that reflects the constrained availability of land and the risk of displacing existing agricultural production elsewhere. Conversely, whilst previous studies have attributed high biogenic carbon storage to bioplastics (Bishop et al., 2021), short-term biogenic carbon storage in plastics was disregarded in this study (although long-term soil carbon storage from residue application was included), providing a conservative and robust approach to assessing life cycle emissions from bioplastic use and end-of-life.

Although the analysis identified that bioplastic production and food waste end-of-life were the major environmental burden contributors, it was found that, despite plastic packaging only representing 5% fresh weight of fresh fruit and vegetable food waste, it represented 25% of the dry weight waste due to the high water content of fruit and vegetables. Plastic packaging therefore constitutes a major material flow within food waste streams that strongly influences the energy recovery potential. Food waste LCA studies involving different rates of separation should therefore also account for any diversion of plastic packaging waste.

The sensitivity analyses show that the energy efficiency and location of PLA production can have a large influence on the environmental footprint of bioplastic. Although somewhat crudely calculated, the sensitivity analyses show there is high potential for the performance of bioplastics to improve as processing technology matures and as energy systems decarbonise. Petrochemical plastic production has been refined over decades to approach technical potential for economic and environmental efficiency, whilst PLA is still an emerging material. The forward-looking LCA and sensitivity analyses applied in this study attempt to compensate for the different improvement potentials of these materials. This research suggests that, in order to realise environmental savings, bioplastics should be produced within industrialised countries where demand is initially greatest owing to more efficient feedstock cultivation and processing technologies, and cleaner energy supplies. This may require "on-shoring" of plastic production from low-cost developing and transitioning countries.

It is important to note that, whilst maize feedstock was modelled within this study due to it being the primary commercially used feedstock (Vink et al., 2007), many other feedstocks exist and are being developed for PLA production. These include feedstocks such as lignocellulosic material (Cubas-Cano et al., 2018; Danner et al., 2004; Singhvi and Gokhale, 2013), food waste (Kwan et al., 2018), and other crops (Morão and de Bie, 2019), among many other novel substrates for valorisation (Djukić-Vuković et al., 2019). These various feedstocks could significantly lower the environmental impact of bioplastic packaging, especially if production impacts can also be reduced (**Figure 4.2**). The results of this study therefore only reflect one production technology, and as such do not necessarily reflect the wider future potential of PLA packaging production. Future research should assess the potential future impacts from alternative production pathways, to identify the most environmentally efficient feedstocks and production techniques, and also the sustainable niche of bioplastics within a circular, net-zero carbon future (European Commission, 2019).

4.4.2. Importance of food waste diversion on the environmental performance

The analyses show that food waste end-of-life contributes a considerable share of environmental impact across all impact categories. Bioplastic LCA system boundaries are rarely expanded to include possible food waste diversion to biowaste treatment (Kakadellis and Harris, 2020), despite bioplastics potentially facilitating such food waste diversion that is shown in this study to be critical to the overall environmental efficiency of bioplastic use. If use of bioplastics means more food waste is diverted to AD and insect animal feed, because it is easier for consumers to separate out food waste without first de-packaging it, then that could leverage significant environmental savings.

Overall, the results show that supplying organic waste to insect feed was the most environmentally beneficial end-of-life option out of the modelled scenarios (**Figure 4.2**). For many impact categories, including global warming potential (carbon footprint), net credits are achieved when bioplastics increase diversion to this treatment. Although an extreme case, this scenario demonstrates that land-use and land-use change are important factors to consider when expanding LCA boundaries, and that more circular management of waste (as well as waste prevention) can spare land. Net carbon sequestration via afforestation of land no longer needed for feed cultivation illustrates the well-recognised benefits of afforestation for GHG mitigation (Duffy et al., 2020; Richards and Stokes, 2004) and (although not modelled) longer-term energy security and bioeconomy objectives depending on use of harvested wood (Brodin et al., 2017). Following insect feed, the next most environmentally efficient waste management option was diverting food waste and PLA to AD (**Figure 4.2**), in line with other waste management LCA studies (Moult et al., 2018; Salemdeeb et al., 2017). The industrial composting scenario showed relatively high emissions when compared to the other scenarios, placing composting lower down the food waste hierarchy than other recent studies (Moult et al., 2018; WRAP, 2020b). This was partly due to the high decomposition emission factors applied (Saer et al., 2013), though even after applying lower emission factors within the uncertainty analysis, the composting scenario remained less environmentally efficient than AD (**Figure 4.2**).

Bioplastics have the potential to reduce global warming potential (**Figure 4.4**), but the EF impact assessment method provides a more holistic view of PLA performance, indicating that PLA remains more environmentally damaging than conventional petrochemical plastics for several impact categories. The analysis therefore illustrates important trade-offs associated with a transition to PLA food packaging, including notable increases in terrestrial and freshwater acidification, terrestrial and marine eutrophication, ozone depletion, and water scarcity. These relate to critical planetary boundaries (Rockström et al., 2009; Steffen et al., 2015) and future sustainability challenges (Mekonnen and Hoekstra, 2016). Therefore, there is a particular need to identify feedstocks that can reduce these environmental hotspots in order to minimise the risk of burden shifting. Meanwhile, reducing the weight of plastic packaging used for fresh fruit and vegetables, without compromising preservation, and reducing food waste arising, should be priorities for the circular economy.

4.4.3. Limitations and future research

A major issue for biodegradable bioplastics is the uncertainty that still exists around the suitability of biodegradable bioplastics for AD and composting. It has been noted that industrial composting and AD facilities are reluctant to accept bioplastics (Kakadellis and Harris, 2020). This may be largely due to the challenge of distinguishing the biodegradable plastics from the non-biodegradable plastics during screening. Petrochemical plastic waste is a major contamination problem when it ends up within these systems, and can negatively affect the treatment processes, as well as contaminate the compost or digestate products (Bátori et al., 2018; Kale et al., 2007). Nevertheless, it has been suggested that optical sorting systems are capable of identifying and separating the bio-based and petrochemical plastics (Kakadellis and Harris, 2020). Another potential limitation of supplying the PLA to AD is that, whilst most compostable bioplastics are certified for specific environments in industrial composting conditions, the use of bioplastics within anaerobic digestion systems is relatively understudied. Although some biopolymers can degrade in AD within the hydraulic retention time (HRT) applied for a typical biogas plant treating the organic fraction of municipal solid waste, it has been found that PLA can take longer than the usual HRTs used in food waste biogas plants (Bátori et al., 2018; Narancic et al., 2018). Although this research highlights the benefits of developing biopolymers fit for AD end-of-life management, waste infrastructure and management practises may also need to adapt to the evolving composition of waste.

A barrier for bioplastic entry to the market is the current high cost of the plastic (Changwichan et al., 2018). For this future-looking study it was assumed that downwards price trends will continue, especially if production is to be increased, achieving economies of scale (Brizga et al., 2020). However, the economic costs over the PLA life cycle will have effects on the sustainability of the product. As such, life cycle cost analysis should be conducted in the future to explore the systematic economic evaluation of the bioplastic over its life cycle.

From our scenarios, the truly circular option of directing the waste to insect feed would necessitate regulatory change. Research would have to be conducted to ensure no significant disease-transmission risks within and across species arises from such recycling. Further, research would have to be undertaken to confirm the assumption that PLA is suitable to be digested by the insect larvae. In the meantime, this scenario demonstrates that innovative approaches to waste management, and the development of disruptive waste treatment technologies, may be required to realise the potential environmental benefits of bioplastics (Salemdeeb et al., 2017).

The effects of plastic debris pollution (littering) into the environment are not included within any current LCA impact category (Bishop et al., 2021). Plastic pollution has wide-ranging and large potential impacts on ecosystem quality, and human health. Consequently, developing new impact assessment methods, or adapting existing ones, to represent potential environmental damage arising from plastic pollution should be a priority for future research. Such development could have a large influence on conclusions drawn from LCA studies and is likely to have a significant bearing on the environmental sustainability credentials of biodegradable bioplastics used to substitute petrochemical plastics.

The error propagation performed via Monte Carlo simulation provides an idea of the uncertainties due to the data used in the model (Pizzol, 2019). The use of error propagation on LCA inventories has some limitations, the foremost one being that any covariance between parameters is not considered, and this can lead to under/overestimation of the output variance (Groen and Heijungs, 2017). A further major limitation with the uncertainty analysis is that the uncertainty estimates were calculated using the pedigree-matrix approach (Ciroth et al., 2016), which is a semi-quantitative method and has arguably a lower accuracy compared to using primary data to estimate uncertainty. Nevertheless, the uncertainties analyses showed, with strong confidence, statistical significance of differences across scenarios (**Figure 4.3**).

The regional scope of this study was the UK, and although the UK is somewhat representative of other European countries with respect to more circular waste management objectives (European Commission, 2020), the model developed in this study highlights that variations in waste composition and marginal energy supplies strongly influence the comparative environmental performance of bioplastic. Future work should therefore explore different locations of deployment. Future work should also explore the growing collection of novel feedstocks for PLA production, which have the potential to improve overall environmental performance.

4.5. Conclusion

This study demonstrated that quantification of possible consequences for food waste separation is critical to understand the net environmental performance of a shift towards use of bioplastics for food packaging. The results show that PLA production can have a high impact compared with petrochemical plastic production across many impact categories, but diversion of PLA and food waste to be organically recycled, via AD, or potentially insect feed in the future, can compensate for this, improving the overall environmental performance of the bioplastic packaging. The impact categories for which bioplastic scenarios performed comparatively well against petrochemical plastic use included: human health effects, climate change, freshwater eutrophication, ionising radiation, photochemical ozone formation, resource use (energy carriers), and respiratory inorganics. On the other hand, bioplastic plastic scenarios generated larger burdens for terrestrial and freshwater acidification, terrestrial and marine eutrophication, ozone depletion, and water scarcity. However, often these environmental savings require 100% diversion of waste, which is, in reality, unrealistic. The immediate focus for the UK should focus on increasing the targeted food waste rate. Nevertheless, sensitivity analyses indicated high improvement potential for bioplastics through improved energy efficiency in PLA production, and/or a shift of bioplastic production to industrialised, plastic-importing countries such as the UK (production "on-shoring"). Changes in fruit and vegetable food waste end-of-life management had a large influence on environmental outcomes. Conversely, the plastic packaging of fresh fruit and vegetables represents a large share of dry matter material flow in waste streams and therefore has a strong influence on environmental outcomes for biowaste treatment. It is important that future LCA studies of food packaging account for both packaging and (diverted) food waste end-of-life flows.

4.6. References

Adams, P.W.R., Mezzullo, W.G., McManus, M.C., 2015. Biomass sustainability criteria: Greenhouse gas accounting issues for biogas and biomethane facilities. Energy Policy 87, 95–109. https://doi.org/10.1016/j.enpol.2015.08.031

Alexandratos, N., Bruinsma, J., 2012. World Agriculture Towards 2030/2015: The 2012 Revision. ESA Work. Pap. No. 12-03. https://doi.org/10.1016/S0264-8377(03)00047-4

Andersen, J.K., Boldrin, A., Christensen, T.H., Scheutz, C., 2012. Home composting as an alternative treatment option for organic household waste in Denmark: An environmental assessment using life cycle assessment-modelling. Waste Manag. 32, 31–40. https://doi.org/10.1016/j.wasman.2011.09.014

Aubert, J.E., Husson, B., Vaquier, A., 2004. Use of municipal solid waste incineration fly ash in concrete. Cem. Concr. Res. 34, 957–963. https://doi.org/10.1016/j.cemconres.2003.11.002

Bátori, V., Åkesson, D., Zamani, A., Taherzadeh, M.J., Sárvári Horváth, I., 2018. Anaerobic degradation of bioplastics: A review. Waste Manag. 80, 406–413. https://doi.org/10.1016/j.wasman.2018.09.040

Bernstad, A., Malmquist, L., Truedsson, C., la Cour Jansen, J., 2013. Need for improvements in physical pretreatment of source-separated household food waste. Waste Manag. 33, 746–754. https://doi.org/10.1016/j.wasman.2012.06.012

Bishop, G., Styles, D., Lens, P.N.L., 2021. Environmental performance comparison of bioplastics and petrochemical plastics: A review of life cycle assessment (LCA) methodological decisions. Resour. Conserv. Recycl. 168, 105451. https://doi.org/10.1016/j.resconrec.2021.105451

Bishop, G., Styles, D., Lens, P.N.L., 2020. Recycling of European plastic is a pathway for plastic debris in the ocean. Environ. Int. 142, 105893. https://doi.org/10.1016/j.envint.2020.105893

Boldrin, A., Andersen, J.K., Møller, J., Christensen, T.H., Favoino, E., 2009. Composting and compost utilization: Accounting of greenhouse gases and global warming contributions. Waste Manag. Res. https://doi.org/10.1177/0734242X09345275

Boots, B., Russell, C.W., Green, D.S., 2019. Effects of Microplastics in Soil Ecosystems: Above and Below Ground. Environ. Sci. Technol. 53, 11496–11506. https://doi.org/10.1021/acs.est.9b03304

Brizga, J., Hubacek, K., Feng, K., 2020. The Unintended Side Effects of Bioplastics: Carbon, Land, and Water Footprints. One Earth. https://doi.org/10.1016/j.oneear.2020.06.016

Brodin, M., Vallejos, M., Opedal, M.T., Area, M.C., Chinga-Carrasco, G., 2017. Lignocellulosics as sustainable resources for production of bioplastics – A review. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2017.05.209

CCC, 2019. Net Zero: The UK's contribution to stopping global warming, Committe on Climate Change. London.

Changwichan, K., Silalertruksa, T., Gheewala, S.H., 2018. Eco-Efficiency Assessment of Bioplastics Production Systems and End-of-Life Options. Sustainability 10. https://doi.org/10.3390/su10040952

Ciroth, A., Muller, S., Weidema, B., Lesage, P., 2016. Empirically based uncertainty factors for the pedigree matrix in ecoinvent. Int. J. Life Cycle Assess. 21, 1338–1348. https://doi.org/10.1007/s11367-013-0670-5

Cubas-Cano, E., González-Fernández, C., Ballesteros, M., Tomás-Pejó, E., 2018. Biotechnological advances in lactic acid production by lactic acid bacteria: lignocellulose as novel substrate. Biofuels, Bioprod. Biorefining 12, 290–303. https://doi.org/10.1002/bbb.1852

Danner, Herbert, Neureiter, M., Danner, H, Madzingaidzo, L., Miyafuji, H., Thomasser, C., Bvochora, J., Bamusi, S., Braun, R., 2004. Lignocellulose Feedstocks for the Production of Lactic Acid. Chem. Biochem. Eng. Q. 18.

Djukić-Vuković, A., Mladenović, D., Ivanović, J., Pejin, J., Mojović, L., 2019. Towards sustainability of lactic acid and poly-lactic acid polymers production. Renew. Sustain. Energy Rev. 108, 238–252. https://doi.org/10.1016/j.rser.2019.03.050

Duffy, C., O'Donoghue, C., Ryan, M., Styles, D., Spillane, C., 2020. Afforestation: Replacing livestock emissions with carbon sequestration. J. Environ. Manage. 264, 110523. https://doi.org/10.1016/j.jenvman.2020.110523

Duffy, P., Hanley, E., Hyde, B., O'Brien, P., Ponzi, J., Cotter, E., Black, K., 2013. Greenhouse gas emissions 1990 – 2011 reported to the United Nations Framework Convention on Climate Change. Dublin.

Ekvall, T., Azapagic, A., Finnveden, G., Rydberg, T., Weidema, B.P., Zamagni, A., 2016. Attributional and consequential LCA in the ILCD handbook. Int. J. Life Cycle Assess. 21, 293–296. https://doi.org/10.1007/s11367-015-1026-0

Ekvall, T., Weidema, B.P., 2004. System boundaries and input data in consequential life cycle inventory analysis. Int. J. Life Cycle Assess. 9, 161–171. https://doi.org/10.1007/BF02994190

European Commission, 2020. A New Circular Economy Action Plan. COM(2020) 98 final. Brussels.

European Commission, 2019. The European Green Deal. COM/2019/640 final. Brussels.

European Commission, 2018. A European Strategy for Plastics in a Circular Economy. COM/2018/028 final. Brussels.

European Commission, 2014. Towards a circular economy: A zero waste programme for Europe. Brussels.

European Parliament, 2019. Parliament seals ban on throwaway plastics by 2021 [WWW Document]. URL https://www.europarl.europa.eu/news/en/press-room/20190321IPR32111/parliament-seals-ban-on-throwaway-plastics-by-2021 (accessed 2.22.21).

European Union, 1999. Council directive 1999/31/EC on the landfill of waste. Off. J. Eur. Communities L182/1.

FAO, 2013. Food wastage footprint - Impacts on natural resouces. Summary Report., Food and Agriculture Organization of the United Nations.

FAO, 2011. Food loss and food waste - Extent, causes and prevention, Food and Agricultural Organization of the United Nations. Rome. https://doi.org/10.4337/9781788975391

Fieschi, M., Pretato, U., 2018. Role of compostable tableware in food service and waste management. A life cycle assessment study. Waste Manag. 73, 14–25. https://doi.org/10.1016/j.wasman.2017.11.036

GreenDelta, 2021. openLCA 1.10.2 [WWW Document]. URL https://www.openlca.org/ (accessed 1.13.21).

Groen, E.A., Heijungs, R., 2017. Ignoring correlation in uncertainty and sensitivity analysis in life cycle assessment: what is the risk? Environ. Impact Assess. Rev. 62, 98–109. https://doi.org/10.1016/j.eiar.2016.10.006

Hermann, B.G., Debeer, L., De Wilde, B., Blok, K., Patel, M.K., 2011. To compost or not to compost: Carbon and energy footprints of biodegradable materials' waste treatment. Polym. Degrad. Stab. 96, 1159–1171. https://doi.org/10.1016/j.polymdegradstab.2010.12.026

IEA, 2021. IEA – International Energy Agency [WWW Document]. URL https://www.iea.org/ (accessed 1.11.21).

IPCC, 2006. IPCC Guildlines for National Greenhouse Gas Inventories. Cambridge, UK.

ISO, 2006a. ISO 14040: Environmental management -- Life cycle assessment -- Principles and framework. Geneva.

ISO, 2006b. ISO 14044: Environmental management -- Life cycle assessment -- Requirements and guidelines. Geneva.

Jabeen, N., Majid, I., Nayik, G.A., 2015. Bioplastics and food packaging: A review. Cogent Food Agric. 1. https://doi.org/10.1080/23311932.2015.1117749

Kakadellis, S., Harris, Z.M., 2020. Don't scrap the waste: The need for broader system boundaries in bioplastic food packaging life-cycle assessment – A critical review. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2020.122831

Kale, G., Kijchavengkul, T., Auras, R., Rubino, M., Selke, S.E., Singh, S.P., 2007. Compostability of bioplastic packaging materials: An overview. Macromol. Biosci. https://doi.org/10.1002/mabi.200600168

Karan, H., Funk, C., Grabert, M., Oey, M., Hankamer, B., 2019. Green Bioplastics as Part of a Circular Bioeconomy. Trends Plant Sci. https://doi.org/10.1016/j.tplants.2018.11.010

Kolstad, J.J., Vink, E.T.H., De Wilde, B., Debeer, L., 2012. Assessment of anaerobic degradation of Ingeo™ polylactides under accelerated landfill conditions. https://doi.org/10.1016/j.polymdegradstab.2012.04.003

Kühn, S., Bravo Rebolledo, E.L., Van Franeker, J.A., 2015. Deleterious effects of litter on marine life, in: Marine Anthropogenic Litter. Springer International Publishing, pp. 75–116. https://doi.org/10.1007/978-3-319-16510-3_4

Kwan, T.H., Hu, Y., Lin, C.S.K., 2018. Techno-economic analysis of a food waste valorisation process for lactic acid, lactide and poly(lactic acid) production. J. Clean. Prod. 181, 72–87. https://doi.org/10.1016/j.jclepro.2018.01.179

LCA-2.0, 2015. Why and when? - Consequential LCA [WWW Document]. URL https://consequential-lca.org/ (accessed 7.28.20).

Lebersorger, S., Schneider, F., 2011. Discussion on the methodology for determining food waste in household waste composition studies. Waste Manag. 31, 1924–1933. https://doi.org/10.1016/j.wasman.2011.05.023

Li, W.C., Tse, H.F., Fok, L., 2016. Plastic waste in the marine environment: A review of sources, occurrence and effects. Sci. Total Environ. 566–567, 333–349. https://doi.org/10.1016/J.SCITOTENV.2016.05.084

Lim, L.T., Auras, R., Rubino, M., 2008. Processing technologies for poly(lactic acid). Prog. Polym. Sci. 33, 820–852. https://doi.org/10.1016/j.progpolymsci.2008.05.004

Manfredi, S., Allacker, K., Chomkhamsri, K., Pelletier, N., Maia De Souza, D., 2012. Product Environmental Footprint (PEF) Guide. Ispra, Italy.

Mekonnen, M.M., Hoekstra, A.Y., 2016. Four billion people facing severe water scarcity. Sci. Adv. 2, e1500323. https://doi.org/10.1126/sciadv.1500323

Misselbrook, T.H., Gilhespy, S.L., Cardenas, L.M., B.J., C., Williams, J., Dragosits, U., 2012. Inventory of Ammonia Emissions from UK Agriculture 2012. London.

Morão, A., de Bie, F., 2019. Life Cycle Impact Assessment of Polylactic Acid (PLA) Produced from Sugarcane in Thailand. J. Polym. Environ. 27, 2523–2539. https://doi.org/10.1007/s10924-019-01525-9

Moult, J.A., Allan, S.R., Hewitt, C.N., Berners-Lee, M., 2018. Greenhouse gas emissions of food waste disposal options for UK retailers. Food Policy 77, 50–58. https://doi.org/10.1016/j.foodpol.2018.04.003

Narancic, T., Verstichel, S., Reddy Chaganti, S., Morales-Gamez, L., Kenny, S.T., De Wilde, B., Babu Padamati, R., O'Connor, K.E., 2018. Biodegradable Plastic Blends Create New Possibilities for End-of-Life Management of Plastics but They Are Not a Panacea for Plastic Pollution. Environ. Sci. Technol. 52, 10441–10452. https://doi.org/10.1021/acs.est.8b02963

Nicholson, F.A., Bhogal, A., Chadwick, D., Gill, E., Gooday, R.D., Lord, E., Misselbrook, T., Rollett, A.J., Sagoo, E., Smith, K.A., Thorman, R.E., Williams, J.R., Chambers, B.J., 2013. An enhanced software tool to support better use of manure nutrients: MANNER- *NPK*. Soil Use Manag. 29, 473–484. https://doi.org/10.1111/sum.12078

Pawelzik, P., Carus, M., Hotchkiss, J., Narayan, R., Selke, S., Wellisch, M., Weiss, M., Wicke, B., Patel, M.K., 2013. Critical aspects in the life cycle assessment (LCA) of bio-based materials - Reviewing methodologies and deriving recommendations. Resour. Conserv. Recycl. 73, 211–228. https://doi.org/10.1016/j.resconrec.2013.02.006

Pizzol, M., 2019. Deterministic and stochastic carbon footprint of intermodal ferry and truck freight transport across Scandinavian routes. J. Clean. Prod. 224, 626–636. https://doi.org/10.1016/j.jclepro.2019.03.270

Polizzi di Sorrentino, E., Woelbert, E., Sala, S., 2016. Consumers and their behavior: state of the art in behavioral science supporting use phase modeling in LCA and ecodesign. Int. J. Life Cycle Assess. 21, 237–251. https://doi.org/10.1007/s11367-015-1016-2

Richards, K.R., Stokes, C., 2004. A review of forest carbon sequestration cost studies: A dozen years of research. Clim. Change. https://doi.org/10.1023/B:CLIM.0000018503.10080.89

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., De Wit, C.A., Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature. https://doi.org/10.1038/461472a

Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: Environmental impact hotspots. J. Clean. Prod. 52, 234–244. https://doi.org/10.1016/j.jclepro.2013.03.022

Salemdeeb, R., zu Ermgassen, E.K.H.J., Kim, M.H., Balmford, A., Al-Tabbaa, A., 2017. Environmental and health impacts of using food waste as animal feed: a comparative analysis of food waste management options. J. Clean. Prod. 140, 871–880. https://doi.org/10.1016/j.jclepro.2016.05.049

Schmidt, J.H., Thrane, M., Merciai, S., Dalgaard, R., 2011. Inventory of country specific electricity in LCA-consequential and attributional scenarios Methodology report v2. Aalborg.

Schmidt, J.H., Weidema, B.P., Brandão, M., 2015. A framework for modelling indirect land use changes in Life Cycle Assessment. J. Clean. Prod. 99, 230–238. https://doi.org/10.1016/j.jclepro.2015.03.013

Searchinger, T.D., Wirsenius, S., Beringer, T., Dumas, P., 2018. Assessing the efficiency of changes in land use for mitigating climate change. Nature 564, 249–253. https://doi.org/10.1038/s41586-018-0757-z

Singhvi, M., Gokhale, D., 2013. Biomass to biodegradable polymer (PLA). RSC Adv. 3, 13558–13568. https://doi.org/10.1039/c3ra41592a

Sonnemann, G., Vigon, B., 2011. Global Guidance Principles for Life Cycle Assessment Databases, United Nations Environmental Programme. Paris/Pensacola: UNEP/SETAC Life Cycle Initiative.

Steffen, W., Richardson, K., Rockstrom, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sorlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science (80-.). 347, 1259855–1259855. https://doi.org/10.1126/science.1259855

Styles, D., Dominguez, E.M., Chadwick, D., 2016. Environmental balance of the UK biogas sector: An evaluation by consequential life cycle assessment. Sci. Total Environ. 560–561, 241–253. https://doi.org/10.1016/j.scitotenv.2016.03.236

Styles, D., Yesufu, J., Williams, P., Bowman, M., Luyckx, K., 2020. Identifying the Sustainable Niche for Anaerobic Digestion in a Low Carbon Future. Feedback Global, London.

Su, S., Kopitzky, R., Tolga, S., Kabasci, S., 2019. Polylactide (PLA) and Its Blends with Poly(butylene succinate) (PBS): A Brief Review. Polymers (Basel). 11, 1193. https://doi.org/10.3390/polym11071193

Takata, M., Fukushima, K., Kino-Kimata, N., Nagao, N., Niwa, C., Toda, T., 2012. The effects of recycling loops in food waste management in Japan: Based on the environmental and economic evaluation of food recycling. Sci. Total Environ. 432, 309–317. https://doi.org/10.1016/j.scitotenv.2012.05.049

Tonini, D., Albizzati, P.F., Astrup, T.F., 2018. Environmental impacts of food waste: Learnings and challenges from a case study on UK. Waste Manag. 76, 744–766. https://doi.org/10.1016/j.wasman.2018.03.032

Tonini, D., Hamelin, L., Astrup, T.F., 2016. Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes. GCB Bioenergy 8, 690–706. https://doi.org/10.1111/gcbb.12290

Van Zanten, H.H.E., Mollenhorst, H., Oonincx, D.G.A.B., Bikker, P., Meerburg, B.G., De Boer, I.J.M., 2015. From environmental nuisance to environmental opportunity: Housefly larvae convert waste to livestock feed. J. Clean. Prod. 102, 362–369. https://doi.org/10.1016/j.jclepro.2015.04.106

Vink, E.T.H., Glassner, D.A., Kolstad, J.J., Wooley, R.J., O'Connor, R.P., 2007. The eco-profiles for current and near-future NatureWorks® polylactide (PLA) production. Ind. Biotechnol. 3, 58–81.

https://doi.org/10.1089/ind.2007.3.058

Weidema, B., 2003. Market information in life cycle assessment. Copenhagen.

Weidema, B.P., Pizzol, M., Schmidt, J., Thoma, G., 2018. Attributional or consequential Life Cycle Assessment: A matter of social responsibility. J. Clean. Prod. 174, 305–314. https://doi.org/10.1016/j.jclepro.2017.10.340

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. Int. J. Life Cycle Assess. 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

WRAP, 2020a. UK progress against Courtauld 2025 targets and UN Sustainable Development Goal 12.3 3.

WRAP, 2020b. Food Surplus and Waste in the UK – key facts. Waste Resour. Action Program.

WRAP, 2019. WRAP Plastics market situation report 2019.

WRAP, 2018. Household food waste: restated data for 2007-2015.

WRAP, 2016. Synthesis of Food Waste Compositional Data 2014 & 2015.

Yates, M.R., Barlow, C.Y., 2013. Life cycle assessments of biodegradable, commercial biopolymers - A critical review. Resour. Conserv. Recycl. https://doi.org/10.1016/j.resconrec.2013.06.010

Yoshida, H., Nielsen, M.P., Scheutz, C., Jensen, L.S., Bruun, S., Christensen, T.H., 2016. Long-Term Emission Factors for Land Application of Treated Organic Municipal Waste. Environ. Model. Assess. 21, 111–124. https://doi.org/10.1007/s10666-015-9471-5

Zhao, X., Cornish, K., Vodovotz, Y., 2020. Narrowing the Gap for Bioplastic Use in Food Packaging: An Update. Environ. Sci. Technol. 54, 4712–4732. https://doi.org/10.1021/acs.est.9b03755

Zheng, J., Suh, S., 2019. Strategies to reduce the global carbon footprint of plastics. Nat. Clim. Chang. 9, 374–378. https://doi.org/10.1038/s41558-019-0459-z

4.7. Appendices for Chapter 4

4.7.1. Appendix 4.1 – Supplementary data file

Description: The attached excel file contains the key data and calculations utilised within the study, as referenced to within **Chapter 4**.

This Excel workbook contains all the input data and arithmetical manipulations used to calculate the life cycle inventory emissions resulting from the end-of-life managements (anaerobic digestion, industrial composting, incineration, home composting, and insect feed) and indirect land-use change of the fruit and vegetables food waste and associated packaging. The full results, uncertainty calculations, and sensitivity analyses are also included in this workbook

Location: The data set has been uploaded to Zenodo data repository at Bishop (2021); https://doi.org/10.5281/zenodo.5798676

File name: Chapter_4_supplementary_material_GB.xls

5. Land-use change and valorisation of feedstock side-streams determine the climate mitigation potential of bioplastics

Abstract

Globally, governments have increased their commitment to mitigate greenhouse gas (GHG) emissions. At the same time, the compostable bioplastic market is growing rapidly as many single-use petrochemical plastics are being banned internationally. A prospective consequential life cycle assessment approach was conducted to quantify the environmental envelopes of compostable bioplastic production for the value chain to operate within the bounds of climate neutrality. Four different feedstocks of i) lignocellulosic biomass from forestry, ii) maize biomass, iii) food waste digestate, and iv) food waste were evaluated as indicative feedstocks for potential bioplastic production. Upstream and end-of-life emissions for these feedstocks equated to GHG balances of -16.3 – +23.5, 0.3 – 1.0, 1.0 – 4.8, and -0.1 – +0.4 kg CO_2 eq. per kg bioplastic, respectively. The scenarios demonstrated that indirect land-use change could have a considerable negative impact on the environmental performance of maize-based plastic, but a positive impact, via terrestrial carbon sequestration, for lignocellulosic-derived plastic (unless increased feedstock demand drives deforestation). Appropriate use of residues and sidestreams is critical to the environmental performance of bioplastics. Efficient utilisation of residues may require decentralisation of bioplastic production and implementation of biorefinery and circular economy concepts.

5.1. Introduction

Global life cycle greenhouse gas (GHG) emissions of plastic use were estimated to be 1.7 Gt CO_2-equivalent (CO_2 eq.) in 2015, projected to increase to 6.5 Gt CO_2 eq. by 2050 under the current trajectory of increasing plastic production (Zheng and Suh, 2019). Moreover, persistent plastics are a well-known hazard to terrestrial, freshwater, and marine environments, negatively impacting the ecosystems when the plastic debris enters those environments (Barboza et al., 2018; Boots et al., 2019; Peng et al., 2020; Strungaru et al., 2019). The pervasive environmental impacts arising from single-use petrochemical plastics have resulted in moves to phase them out of production internationally (Xanthos and Walker, 2017). These bans have led to a shift in consumption towards bio-based polymers (bioplastics), which are rapidly increasing their share within the plastic market (Zhao et al., 2020). Bioplastics, and especially compostable bioplastics, are being developed as a more environmentally sustainable replacement for petrochemical plastics (European Commission, 2018). Such plastics are typically made from renewable, bio-based feedstocks and can retain the beneficial material characteristics of petrochemical plastics whilst allowing for a transition towards a circular economy by reducing fossil resource extraction and lowering end-of-life burdens as a result of their compostable nature (Bishop et al., 2021a). This displacement of petrochemical plastics by bioplastics may also have the benefit of potentially reducing the global carbon footprint of plastics (Bishop et al., 2021a).

Globally, governments have increased their commitment to mitigate GHG emissions. The 21st annual session of the Conference of the Parties to the 1992 United Nations Framework Convention on Climate Change, commonly known as COP 21, saw 195 countries sign the negotiated Paris Agreement, a global agreement on the reduction of climate change (UNFCCC, 2015). The goal of this legally binding international treaty is to limit the increase in the global average temperature to "well below 2°C above pre-industrial levels", whilst "pursuing efforts to limit the temperature increase to 1.5°C above pre-industrial levels" by 2100 (UNFCCC, 2015). The agreement also calls for a long-term goal of countries to achieve net-zero GHG emissions in the second half of the 21st century (UNFCCC, 2015). Net-zero carbon, sometimes referred to as "carbon neutral", is the concept of achieving an overall balance between anthropogenic GHG emissions released, and GHG emissions removed from the atmosphere via land-based sinks and technologies. A special report by the Intergovernmental Panel on Climate Change (IPCC) (2018) stated that in order to meet the Paris Agreement's temperature goals within a 1.5°C threshold, net CO_2 emissions need to be reduced by about 45% from 2010 levels by 2030, reaching net-zero CO_2 emissions by 2050 (IPCC, 2018).

First-generation feedstocks of maize and sugarcane dominate the market of commercial bioplastic feedstocks (Brizga et al., 2020). Large inputs are required for production of these bioplastic feedstocks, such as fertiliser and land-use, which can negate the sought environmental efficiency of these bio-based materials (Bishop et al., 2021a). The land-use for bioplastics was recently reported to be approximately 0.7 million hectares in 2020 (0.015% of global land area), increasing to 0.020% of global land area in 2025 under the projected market growth (European Bioplastics, 2020). A complete diversion of 250 million tonnes of plastic produced annually to bioplastics would require as much as 5% of all arable land (Reddy et al., 2013), potentially undermining the carbon benefits of bioplastics (Zheng and Suh, 2019). The use of second-generation feedstocks such as lignocellulosic or waste biomass, can, however, alleviate the pressure of cropland expansion and associated GHG emissions from land-use change (Piemonte and Gironi, 2011). If the bioplastic production emissions are greater than the potential credits earned through avoided petrochemical plastic production, avoided processes at end-of-life, and/or long-term sequestered carbon, then the potential environmental benefits provided by bioplastics will not be realised (Bishop et al., 2021a). Therefore, it is important to identify the sustainable niche of bioplastics, evaluating how bioplastics can be produced within a net-zero carbon future by calculating the production burden constraints required for environmental neutrality from different bioplastic feedstocks.

Recent reviews (Bishop et al., 2021b; Pawelzik et al., 2013; Spierling et al., 2018; Yates and Barlow, 2013) found that application of life cycle assessment (LCA) (see **Chapter 5.2.1**) to bioplastic innovations has hitherto been patchy and often incomplete, leading to potentially misleading conclusions. Further, very few studies have been found to explore the wider environmental impacts of bioplastics via consequential LCA, to include indirect impacts and emissions brought about via economic signals (Bishop et al., 2021b). Due to how new the technologies are, and due to the many different types of potential compostable bioplastics available, data on commercial bioplastic production is scarce, and is certain to change in the future as production increases (Zhao et al., 2020). However, it is important to consider that different bioplastic feedstocks may have embodied burdens from upstream acquisition (i.e., production or diversion) and end-of-life management before the feedstock even reaches bioplastic production, thus affecting the "allowable" emissions from bioplastic production for the system to operate with net-zero carbon targets.

This study fills a specific research gap in forward-looking, consequential LCA studies, quantifying the environmental envelopes of bioplastic production from multiple feedstocks in relation to net-zero GHG targets. Four different feedstocks of i) lignocellulosic biomass from forestry, ii) maize biomass, iii) food waste digestate, and iv) food waste are evaluated as indicative feedstocks for potential bioplastic production pathways.

5.2. Methodology

5.2.1. Goal and scope

The goal of this life cycle assessment (LCA) was to broadly screen potential compostable bioplastic feedstocks for GHG hotspots and compatibility with climate neutral targets, considering both direct and potential indirect effects of their use for bioplastic production. The study calculated the carbon footprints of indicative value chains for four feedstocks from different origin: maize, lignocellulosic biomass from forestry, food waste digestate, and food waste. A European context was considered for all foreground inventory data for marginal technologies, and for legislative drivers and constraints. A full breakdown of the life cycle inventory and results from this study can be found in the **Supplementary Material** (**Chapter 5.7**). The functional unit of this study was defined as the management of one kilogram of compostable bioplastic material produced. The holistic environmental impacts, including direct and indirect effects, arising from the cradle-to-grave life cycle of the compostable bioplastics were modelled within the present study, with system boundaries represented in **Figures 5.1 – 5.5**.

LCA is a method of quantifying the environmental impacts arising over the entire value chain of a product or service from "cradle-to-grave" (ISO, 2006a, 2006b). To account for the global net effects of bioplastic production arising from factors such as indirect land-use change (iLUC) and avoided emissions due to substituted processes, a consequential LCA approach was applied to this study. Consequential LCA models are prospective as they aim to model the consequences of future decisions. A consequential LCA expands system boundaries to account for marginal effects of system modifications induced via economic signals throughout the wider economy. Therefore, activities are included in the product system being evaluated only to the extent that they are expected to change as a consequence of a change in demand for the functional unit (Weidema, 2001).

The primary objective of this study was to evaluate prospective GHG mitigation efficacy of different bioplastic feedstocks. Accordingly, life cycle impact assessment focussed on contributions to climate change, represented within the study as global warming potential (GWP) over 100 years. GWP is the radiative forcing of a substance over a given time horizon, relative to the heat that would be absorbed by the same mass of CO_2 (IPCC, 2013). The GWP metrics employed within the study utilise IPCC (2013) values, with slightly adjusted methane (CH_4) values based on Mũnoz and Schmidt (2016). CH_4 burdens are separated into biogenic and fossil emissions, with burdens including the result of an additional indirect effect of oxidation of CH_4 to CO_2 during CH_4 decay, and an additional biogenic correction applied to CH_4 from biogenic sources (Muñoz and Schmidt, 2016). The environmental burden characterisation factors of the GHG investigated are presented in **Table 5.1**.

Table 5.1 – Environmental burden characterisation factors (per kg) applied to emissions attributable to global warming potential over 100 years (GWP_{100}) (IPCC, 2013; Muñoz and Schmidt, 2016).

Greenhouse Gas	GWP_{100}
CO_2	1
Biogenic CH_4	27.75
Fossil CH_4	30.50
N_2O	265

CO_2: carbon dioxide; CH_4: methane; N_2O: dinitrogen monoxide.

5.2.2. Indicative feedstock value-chains and inventory analyses

Four different materials were explored as potential bioplastic feedstocks. In this section, the indicative value chains evaluated by LCA modelling are described. Detailed life cycle inventories are developed for each scenario in the **Supplementary Material** (**Chapter 5.7**). For each scenario transportation to the processing facilities was considered to be a total of 130km, 240km, and 270km by truck, rail, and barge, respectively, for each feedstock (Nessi et al., 2020), related to anticipated economies of scale required for processing into bioplastics (Ni et al., 2021).

5.2.2.1. Lignocellulosic biomass feedstock

Lignocellulosic derived bioplastics are an emerging novel material (Reshmy et al., 2021). Increased demand for lignocellulosic biomass feedstock for bioplastic production could displace existing lower value uses of harvested wood, such as bioenergy generation, but could also induce afforestation – providing longer-term benefits in terms of climate mitigation, ecosystem enhancement, wider bioeconomy initiatives and energy security (Forster et al., 2021). Studies show that 6.7 – 31.2 kg of lignocellulosic biomass is required to produce one kg of bioplastic (Al-Battashi et al., 2019; Kim et al., 2020; Nieder-Heitmann et al., 2019). Thus, for this study, 20 kg of biomass feedstock was conservatively modelled to be utilised for the production of 1 kg bioplastic.

For the lignocellulosic biomass feedstock, the value chain modelled evaluated the valorisation of low-value wood from the sawmill process of harvested Sitka spruce YC18 (*Picea sitchensis*) (**Figure 5.1**). This value chain considered that the low-value by-products of bark and waste chips/sawdust from sawmilling, that are typically directed outside of the sawmill to bioenergy generation plants, are instead diverted to become bioplastic feedstock – representing about 13.6% of the total end harvest. Within the system, modelled using flows detailed in Forster et al. (2021), 81% of the harvest is destined for the sawmill (**Figure 5.2**). The saw logs are then debarked, and it is assumed that all bark is sent to bioenergy generation. A typical sawmill results in 50% sawn timber to chips/sawdust from the pallet logs, and a 60:40 split of sawn timber to chips/sawdust for green logs. From this chips/sawdust fraction, 40% is typically utilised as biomass for bioenergy. Finally, 40% of this bioenergy fraction is the quantity of material that is exported for bioenergy (as the other 60% of this biomass is used in the drying process of the sawn wood) (Forster et al., 2021). A mass flow for the biomass that is typically sent to bioenergy, and the desired feedstock for this study, is provided in **Figure 5.2**.

By valorising the low-value wood for bioplastic production, the previous use of this wood is avoided – in this case, bioenergy generation. As such, system boundaries were expanded to include the avoided bioenergy emissions, as well as the indirect emissions of increased marginal electricity generation required to compensate for the reduced bioenergy generation.

For this value chain, it was assumed that diverting the low-value wood for bioplastic production could increase the demand for biomass. This in turn could drive increased forest profitability, and therefore increase forest planting. It was assumed that these market signals result in an indirect land-use change (iLUC) of unmanaged and low productivity grassland to forestland, as there would be increased incentive for landowners to convert their land to forest. To calculate this iLUC, the 20 kg lignocellulosic feedstock required was divided by the total output over 50-hectare years, estimated at 617 tonnes per ha (Forster et al., 2021). This provided a fraction of the absolute harvest from one hectare that was taken out over 50 years. For this study, just the first 20 years of iLUC were considered. After this time, additional carbon storage from commercial forestry becomes uncertain due to future harvesting rates and wood uses outside the scope of this study (Forster et al., 2021). Therefore, the annual land-use change emissions were multiplied by 20 years to capture a relevant fraction of the transformation, as described in **Chapter 5.2.4.1**. Although the reduction in available grassland could result in the indirect intensification of livestock production, there is considerable scope for producing livestock on a smaller area without increasing overall GHG emissions (Styles et al., 2018b), and this effect has been excluded from the system boundary.

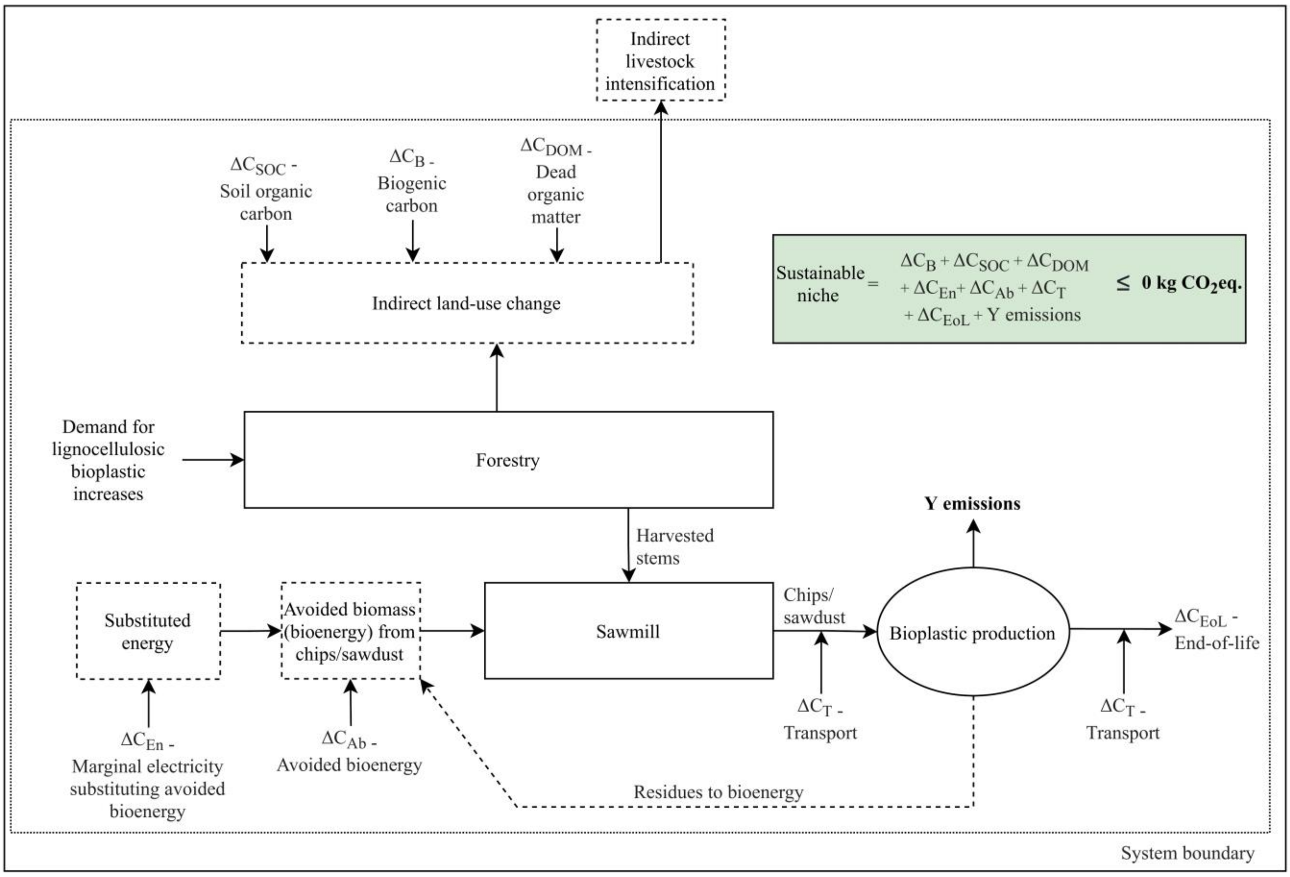

Figure 5.1 – Schematic representation of the major processes modelled for lignocellulosic-based bioplastic obtained from low-value wood from sawmilling. ΔC: change in GHG emissions from the different processes; Y: bioplastic production.

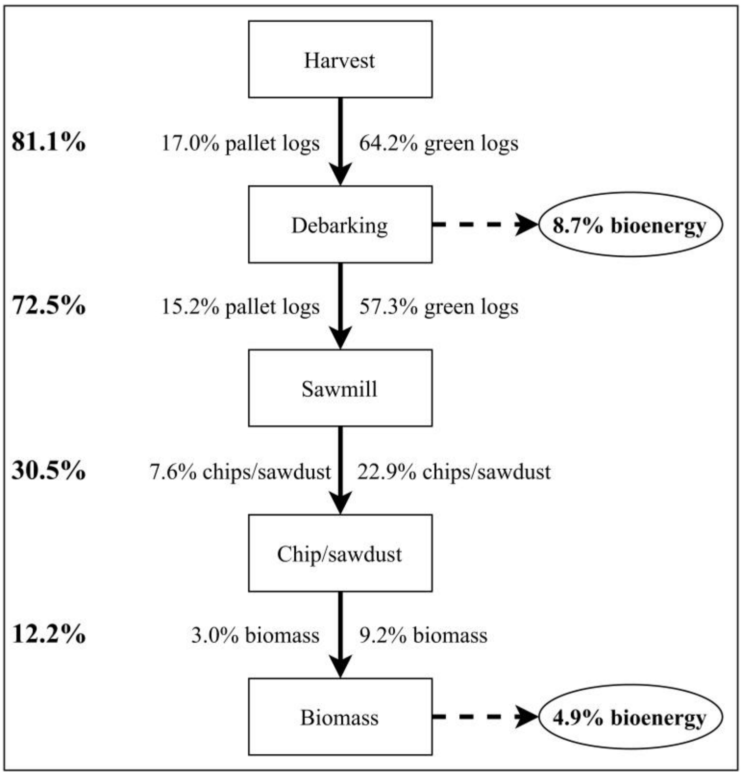

Figure 5.2 – Mass flow of the biomass typically sent to bioenergy generation plants from the sawmilling process for Sitka spruce. The percentages displayed represent the proportion of the total end harvest. The figure does not include percentages of harvest directed to other wood product types. Due to rounding, some totals may not correspond with the sum of the separate figures. Complete, detailed mass flows can be found in Forster et al. (2021). Note that green logs are the main logs used for sawn wood (mainly construction timber), whilst pallet logs are lower quality logs of sufficient diameter to make pallets.

5.2.2.2. Maize biomass feedstock

Maize is an established bioplastic feedstock used to produce commercial polylactic acid (PLA) bioplastic at a relatively large-scale (Vink et al., 2007). In this value chain, the environmental impacts from an increased demand of maize-based bioplastic were explored (**Figure 5.3**).

For this study, 2.47 kg of maize biomass was required to produce one kg of bioplastic (Vink and Davies, 2015), requiring 1.7 m^2 of land (Wernet et al., 2016). For the maize feedstock, direct land-use change burdens were negligible (see **Chapter 5.2.4.1**) as the maize was considered to be grown on existing arable land. However, as arable land is constrained, food and feed production are displaced, causing iLUC via arable land expansion (e.g., deforestation) and intensification of existing arable land (see **Chapter 2.4.2**). Fertiliser was applied to crops, with upstream burdens and soil application emissions considered (see **Chapter 2.4.3**). Harvesting, fertiliser, and transport burdens were extracted from ecoinvent 3.7 (Wernet et al., 2016).

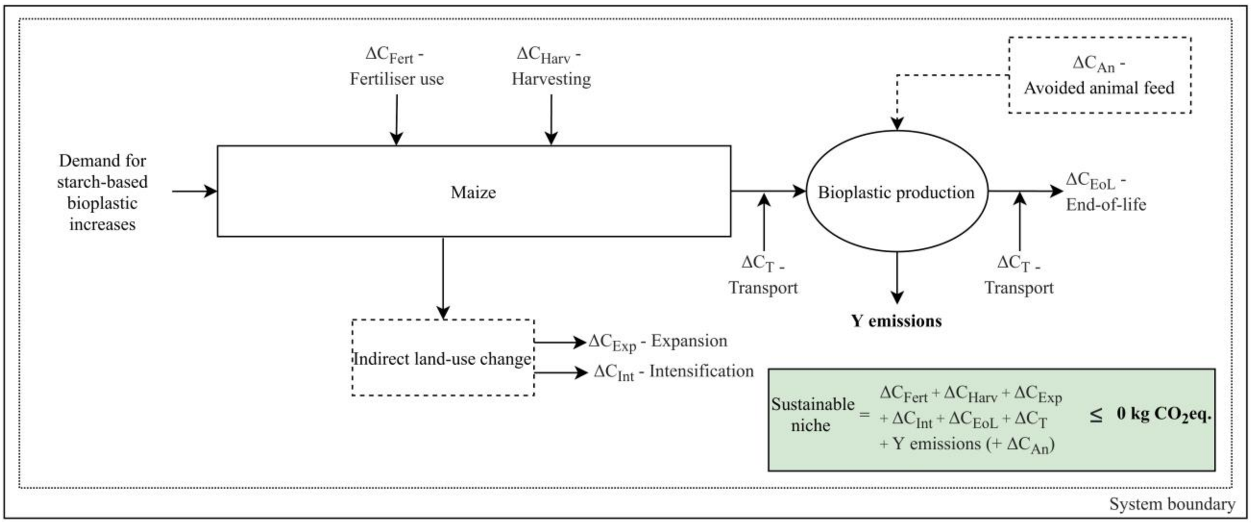

Figure 5.3 – Schematic representation of the major processes modelled for maize-based bioplastic. ΔC: change in GHG emissions from the different processes; Y: bioplastic production.

5.2.2.3. Food waste digestate feedstock

Food waste digestate was the third potential bioplastic feedstock evaluated within this study (**Figure 5.4**). Anaerobic digestion (AD) is the process of organic matter decomposition by microbes in the absence of oxygen. Organic matter is converted into biogas (comprised of CO_2, CH_4, and other trace gases) and a wet digestate that contains residual organic matter and nutrients. Following the capture of the biogas, it is usually combusted as a fuel or purified into a clean CH_4 biofuel for transport or injection into the natural gas grid, thus providing a source of renewable bioenergy (Styles et al., 2016). Meanwhile, the digestate is typically used as biofertiliser. This value chain explored the use of food waste digestate as a feedstock for bioplastic production.

Early digestate valorisation studies demonstrate PHA/PHB bioplastic yields of 4.6 – 12.3 g/L (Altun, 2019; Eshtaya et al., 2013; Passanha et al., 2013). Thus, for this study, 100 L of food waste digestate was assumed to be required for one kg of bioplastic production. The food waste digestate properties were assumed to be 4.9 % total solids (TS), 36.2 total C (%TS), 4.15 total N (%TS), 0.93 P_2O_5 (%TS), and 2.33 K_2O (%TS) (Peng and Pivato, 2019), with a density of 1.1 kg/L (Seruga et al., 2020).

In this value chain, the use of food waste digestate as a bioplastic feedstock displaced its use as biofertiliser. As such, the impacts from subsequent fertiliser production were evaluated for the consequential increased synthetic fertiliser use (**Chapter 5.2.4.3**). It was assumed that, based on the digestate composition, 0.26 kg of long-term C storage was avoided by the digestate application, assuming that 13.2 % of the digestate C was sequestered long-term (Tonini et al., 2018). However, some C was nevertheless sequestered long-term in the soil because of the end-of-life treatments (see **Chapter 5.2.4.6**).

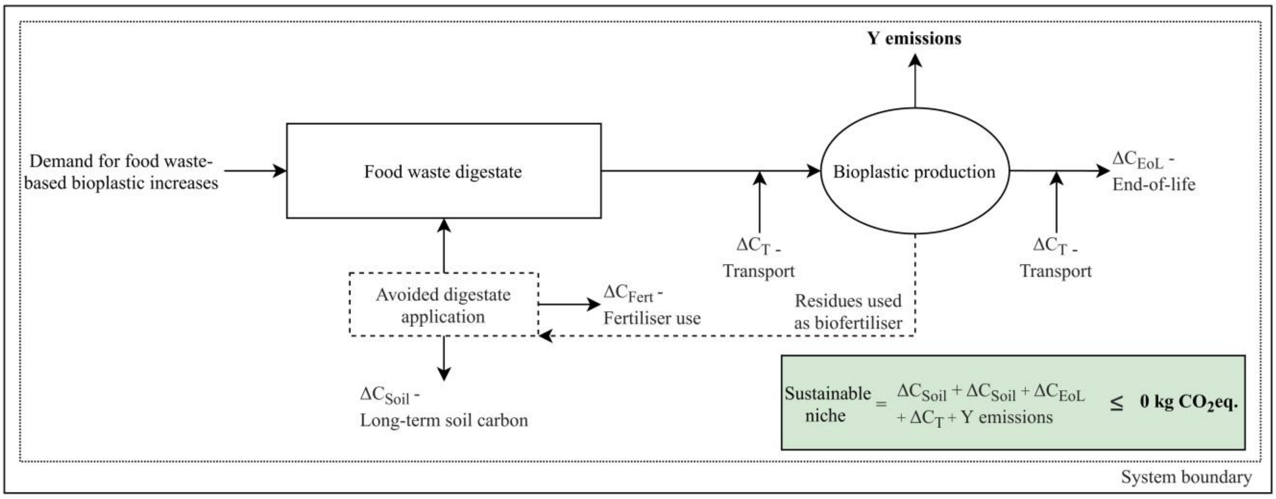

Figure 5.4 – Schematic representation of the major processes modelled for bioplastic production from the food waste digestate feedstock. ΔC: change in GHG emissions from the different processes; Y: bioplastic production.

5.2.2.4. Food waste feedstock

Food waste is a promising bioplastic feedstock (Tsang et al., 2019) and was evaluated as the fourth potential feedstock within the study. For this study, it was assumed that 12.5 kg of food waste was required per kg of bioplastic (Albizzati et al., 2021). In this value chain, it was assumed that an increase in demand for bioplastic would lead to an increased demand for food waste diverted to bioplastic. As such there would be incentivised separate food waste collection to capture the uncollected fraction of food waste, thus reducing the treatment of this waste in the non-separated form. This avoided end-of-life treatment was assumed to be incineration with energy recovery, in line with future European technologies (European Commission, 2014) (**Figure 5.5**).

The modelling of incineration was based on the methods described in Moult et al. (2018), using calculations of net energy released based on the gross energy and water content of the food waste. The incineration process modelled that net thermal energy from combustion was used to generate electricity on site, of which surplus was exported to the grid, avoiding marginal generation. The incineration energy conversion efficiency was modelled at 22%, and gross thermal energy outputs were reduced by 15.5% to account for parasitic heat loss (Moult et al., 2018). All excess heat was assumed to be reused within the process or dumped. Small quantities of residues and slag were diverted to landfill after incineration. In this value chain, as the incineration was avoided, the avoided electricity production was compensated via marginal electricity generation. The burdens from the incineration process were also avoided, modelled from ecoinvent 3.7 (see **Chapter 5.2.4.5**).

The food waste composition was modelled from Albizzati et al. (2021), with the water and energy content information of the individual foods taken from Tonini et al. (2018). The weighted average water and energy content of the feedstock was modelled to be 62.9% and 19.6 MJ/kg total solids, respectively.

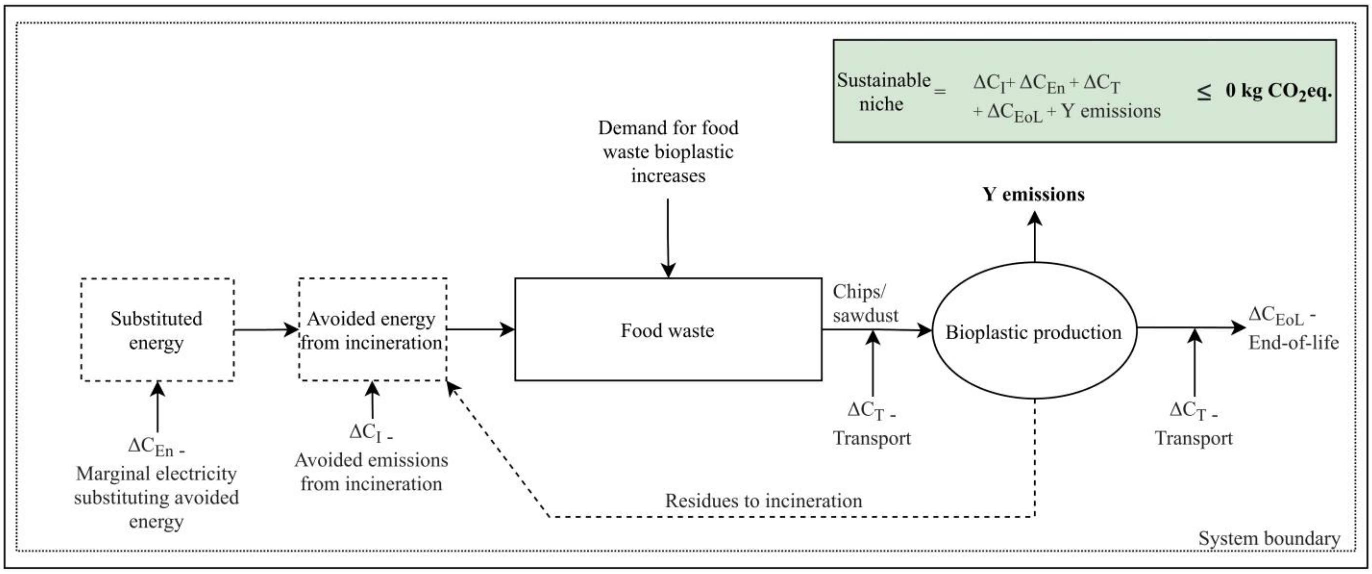

Figure 5.5 – Schematic representation of the major processes modelled for bioplastic production from the food waste feedstock. ΔC: change in GHG emissions from the different processes; Y: bioplastic production.

5.2.3. Scenario overview

To deal with several key areas of uncertainty within the system model, multiple potential scenarios were run to explore some "what if?" scenarios around the emissions resulting from the four feedstocks. Due to the prospective nature of the study, one scenario was assumed to be not more likely than any other.

As such, the value chains described above were modelled as **Scenario 1**. This section describes how different value chain flow and process combinations (scenarios) are considered to cover a range of plausible outcomes arising from the use of the four bioplastic feedstocks.

Although production processing burdens were excluded from the study, **Scenario 2** estimated the magnitude of effect that circularity in production can have on the environmental impact. All four feedstocks, especially the digestate and the lignocellulosic feedstocks, currently require a high mass of feedstock per kg of bioplastic produced. Thus, this analysis explored the net effects incurred from the production of the bioplastic if surplus residues can be returned to the original (marginal) feedstock use, i.e., biofertiliser, bioenergy, animal feed, or incineration for the digestate, lignocellulosic biomass, maize, and food waste feedstocks, respectively. This analysis therefore modelled that only 1 kg of lignocellulosic biomass was diverted from energy generation, reducing compensatory energy generation burdens and iLUC credits, due to 19 kg of the residual forestry feedstock ultimately being returned to bioenergy. Similarly, for the digestate scenario, estimates were made with the assumption that all the N, P, or K (not present in bioplastic) contained within the food digestate could still be applied to land as biofertiliser. Further, it was assumed that only 0.5 kg of C was prevented from being applied to the land, based on the molecular mass of PLA [$C_3H_4O_2)_n$] as an example compostable bioplastic. The maize residues were considered to substitute animal feed also produced from maize. As such, avoided emissions for the maize biomass included fertiliser application and production, harvesting, and iLUC burdens on a mass flow basis. The food waste residues were modelled to be returned to incineration, reducing the compensatory energy substitution (similarly to the lignocellulosic scenario). Full transportation of raw materials to the processing centres was considered.

Scenario 3 explored what if the production of the bioplastic was decentralised, resulting in less distance for the feedstock to travel to the production location. The new transport distance was modelled to be 100km via truck, only.

Scenario 4 evaluated the consequences for environmental performance where the wood chips/sawdust fraction was assumed to be diverted from an industrial furnace for heat production (rather than for electricity production as in **Scenario 1**) prior to being processed into bioplastic. A 300 kW furnace running off this wood was therefore modelled to be avoided, with the inventory considered as being representative of boilers with nominal capacities between 70 – 500 kW (Wernet et al., 2016). Three alternative fuel scenarios for the substitution of this displaced heat production were generated. These included natural gas and oil heating within industrial boilers, modelled using a condensing, modulating <100 kW and a condensing, non-modulating 10 kW boiler, respectively (Wernet et al., 2016). A third scenario within this sensitivity involved the substitution of an industrial furnace. Using the cement industry as an example, coal was the fuel modelled to substitute the avoided heat from the wood in this scenario, due to the high quantities used in production (Fierro et al., 2020). Heat from an average heat and power co-generation hard coal power plant was modelled from ecoinvent 3.7 (Wernet et al., 2016).

Scenario 5 estimated the GWP arising from circular production of the lignocellulosic biomass feedstock, when residues were returned back to bioenergy for heating, as modelled in **Scenario 2** and **4**.

Scenario 6 evaluated the consequence to the environmental performance of the food waste digestate feedstock when the digestate was dried on-site before being transported to bioplastic production. For this scenario it was assumed that the critical compounds required to produce the bioplastic could be obtained from the solid fraction of the digestate. A screw press was considered to be used to separate digestate fractions with minimum electricity and no additive inputs were required (Bamelis et al., 2015). The separation efficiency from the digestate into the solid fraction was assumed to be 33%, 13%, and 28% for solids, total N, and total P_2O_5, respectively (Tambone et al., 2017). The drying of the solid fraction was modelled to be supplemented with excess heat production from combustion of biogas in a combined heat and power generator, producing a solid faction that was assumed to have a moisture content of 20% (Lyons et al., 2021). The liquid fraction was applied to the soils, avoiding synthetic fertiliser application but incurring various emissions (see **Chapter 5.2.4.3**). From the above mass flows, the total quantity of digestate feedstock requiring transport and processing was reduced down to 2.2 kg per 1 kg bioplastic.

Scenario 7 investigated the uncertainty arising from the end-of-life fate of the bioplastic. Thus, for the four feedstocks, the impacts from 100% diversion to anaerobic digestion, industrial composting, and incineration were estimated. As before, this analysis was to explore the context of the potential end-of-life emissions, as the same net changes can be expected for each bioplastic feedstock scenario.

Final GWP balance results for each scenario were compared against a reference of typical production and end-of-life emissions per kg petrochemical plastic, extracted from Bishop et al. (2021a) where a detailed methodology of how the emissions were calculated can be found. Further, the environmental impacts from each scenario were estimated without considering iLUC emissions, where the demand for the lignocellulosic and maize biomass was assumed to not result in additional land-use change.

5.2.4. Environmental balance calculations

The equations required to fulfil the calculations within **Figures 5.1 – 5.5** are described within this section. See **Supplementary material** (**Chapter 5.7**) for a full arithmetic breakdown of each scenario and sensitivity analyses.

5.2.4.1. Land-use change from forestry

For the emissions arising from indirect land-use change from the forestry scenario, IPCC (2019a, 2006) Tier 1 methodology was followed. As such, annual changes in carbon stocks from land-use change (ΔC_{LU}) were calculated using **Equation 1** (IPCC, 2006) on the basis of annual changes in carbon stocks in above and below ground biomass (ΔC_{B}), dead organic matter (including dead wood and litter) (ΔC_{DOM}), and soil organic carbon (ΔC_{SOC}). For the forestry, the iLUC was considered over the first 20 years following conversion from grassland (see **Chapter 5.2.2.1**).

Equation (1):

$$\Delta C_{LU} = \Delta C_{B} + \Delta C_{DOM} + \Delta C_{SOC}$$

5.2.4.1.1. Biomass carbon stocks

In the LCA modelling of the primary feedstock scenarios, the IPCC (2006) Tier 1 assumptions were followed for estimating the changes in carbon stocks in above- and below-ground biomass. As such, for the maize feedstock, it was assumed that annual biomass production in a single year is equal to biomass losses from harvest and decomposition.

To estimate the annual biomass carbon stock changes from grassland to forestry conversion for the lignocellulosic feedstock scenario (ΔC_{B}), the annual gain in carbon was calculated. Tier 1 employs a default assumption that there is no change in initial biomass carbon stocks due to conversion. ΔC_{G} is estimated from **Equation 2** (IPCC, 2006), calculated from the area of land converted to forest land (A) of 0.00032 ha (see **Chapter 5.2.2.1**), the average annual above-ground biomass growth (G_{W}) of 4.0 tonnes dry matter ha^{-1} yr^{-1} (IPCC, 2006), the ratio of 0.340 below-ground biomass to above-ground biomass (R) (IPCC, 2019a), and the carbon fraction (CF) of dry matter of 0.51 (IPCC, 2006). Parameters are dependent on the ecological zone (i) and climate domain (j).

Equation (2):

$$\Delta C_{B} = \sum_{i,j} (A_{i,j} \bullet G_{W} \bullet (1 + R) \bullet CF_{i,j})$$

No thinning was expected to be undertaken during the first 20 years; therefore, no biogenic losses were modelled. Harvesting at circa 40-50 year intervals removes most of the aboveground biomass, so long-term average terrestrial aboveground biomass gain will be significantly less than maximum standing biomass before harvest. However, over a longer-time horizon, there is considerable potential for much of the removed wood to be sequestered long-term in harvested wood products or by bioenergy generation with carbon capture and storage (BECCS) (Forster et al., 2021). Therefore, considering the first 20 years of LUC following conversion to forest represents a conservative approach to estimating additional biogenic C storage linked with afforestation.

5.2.4.1.2. Dead organic matter

Dead organic matter (DOM) comprises dead wood and litter (**Equation 3**; IPCC, 2006), where the annual change in carbon stocks in DOM (ΔC_{DOM}) is determined by the change in carbon stocks in dead wood (ΔC_{DW}) and the change in carbon stocks in litter (ΔC_{LT}).

Equation (3):

$$\Delta C_{DOM} = \Delta C_{DW} + \Delta C_{LT}$$

The IPCC (2006) Tier 1 assumption is that carbon stocks in litter and dead wood pools for the maize land-use is zero. For the land converted to forest land, the Tier 1 assumption is that dead wood and litter pools increase linearly from zero to the default values for the climate region over a period of T years. Estimates of the average annual change of dead organic matter stocks were calculated separately for dead wood and litter (**Equation 4**; IPCC, 2006), where ΔC_{DOM} was determined by the difference in dead wood/litter stock under the new land-use category (C_n) (22.1 and 66.3 tonnes C ha^{-1} for dead wood and litter, respectively (IPCC, 2019a)) and dead wood/litter stock under the old land-use category (C_o) over the time period of the transition from old to new land-use category (T_{on}) (Tier 1 default of 20 years). The default Tier 1 assumption is that non-forest dead organic matter carbon stocks are zero and that the period of transition is 20 years. The average annual change was multiplied by the area undergoing conversion from old to new land-use category (A_{on}).

Equation (4):

$$\Delta C_{DOM} = \frac{(C_n - C_o) \bullet A_{on}}{T_{on}}$$

5.2.4.1.3. Soil organic carbon

The change in SOC stocks for the conversion of grassland to forestland was estimated for mineral soils with **Equation 5** (IPCC, 2006). Annual rates of stock changes in mineral soils ($\Delta C_{Mineral}$) were calculated as the difference in stocks (over time) divided by the time dependence (D) of the stock change factors. For Tier 1, the initial (pre-conversion) soil organic C stock ($SOC_{(0\text{-}T)}$) and C stock in the last year of the inventory time period (SOC_0) were determined from the common set of reference soil organic C stocks (SOC_{REF}) (see **Supplementary material** (**Chapter 5.7**)) and default stock change factors (F_{LU}, F_{MG}, F_I, for land-use systems, management regime, and input of organic matter, respectively) and land area of the stratum (A) being estimated, as appropriate for describing land use and management both pre- and post-conversion. The parameters were dependent on the climate zones (c), the soil types (s), and the set of management systems that are present (i).

Equation (5):

$$\Delta C_{Mineral} = \frac{\left(SOC_0 - SOC_{(0-T)}\right)}{D}$$

Where:

$$SOC = \sum_{c,s,i} \left(SOC_{REF_{c,s,i}} \bullet F_{LU_{c,s,i}} \bullet F_{MG_{c,s,i}} \bullet F_{I_{c,s,i}} \bullet A_{c,s,i}\right)$$

However, under Tier 1 values, managed forest and grassland are assumed to have the same soil C stock as the reference condition, i.e., all stock change factors are equal to 1 (IPCC, 2006)**.** Therefore, net annual SOC changes equal 0 tonnes C yr^{-1}. Due to net SOC equalling 0, no N_2O emissions arising from mineralised N resulting from the loss of SOC stocks in mineral soils through land-use change were modelled (IPCC, 2006).

5.2.4.2. Indirect land-use change of arable land

The iLUC impacts from the maize cultivation modelled in this study followed the deterministic model presented by Tonini et al. (2016). According to the biophysical model, additional demand for land is supplied from land expansion and intensification of land already in use. Observing global agricultural statistics over time, iLUC consists of 75% increased yields (intensification) and 25% expansion of the cultivated area (Tonini et al., 2016). The iLUC model considered the geographical location of expansion and affected biomes, and thus the changed flows of C and N from changes in biomass C as a result of expansion, plus the quantities of increased N, P, and K fertiliser used for intensification in terms of increased fertiliser manufacture and post-application field emissions. For a detailed description of the model, the reader is referred to the original publication of Tonini et al. (2016). The amount of arable land demanded by the maize for the bioplastic production was identified as 0.00017 ha per kg bioplastic from ecoinvent 3.7 consequential (Wernet et al., 2016). Upstream emissions of the fertiliser production were also modelled from ecoinvent 3.7 consequential (Wernet et al., 2016). As maize is an annual crop, iLUC emissions were considered over one year.

5.2.4.3. Fertiliser emissions

Direct and indirect N_2O emissions from the application of fertiliser for maize production were estimated using **Equation 6** (IPCC, 2006). The direct emissions were calculated from the annual amount of synthetic fertiliser N applied to soils (F_{SN}) (0.0049 kg N/kg maize (Wernet et al., 2016)), multiplied by the emission factor for N_2O emissions from the N fertiliser (EF_1) (0.010 kg N_2O-N/kg N (IPCC, 2019a)). The indirect N_2O emissions from N fertiliser leaching were calculated by multiplying the F_{SN} by the fraction of N added to the soil that is lost through leaching and runoff ($FRAC_{Leach-(H)}$) (0.24 kg N/kg N applied (IPCC, 2019a)), as well as the emission factor for N_2O emissions from the N present in leaching and runoff (EF_2) (0.011 kg N_2O-N/kg N leaching (IPCC, 2019a)). Similarly, the N_2O emissions from the volatilisation and subsequent redeposition of N were estimated via the F_{SN} multiplied by the fraction of synthetic fertiliser N that volatilises as NH_3 and NO_x ($FRAC_{GASF}$) (0.11 kg NH_3-N + NO_x-N/kg N applied (IPCC, 2019a)) and the emission factor for N_2O emissions from atmospheric deposited N (EF_3) (0.010 kg N-N_2O/kg NH_3-N + NO_x-N volatilised (IPCC, 2019a)).

Equation (6):

$$N_2O - N\ incurred = \left[(F_{SN} \bullet EF_1) + \left(F_{SN} \bullet FRAC_{LEACH-(H)} \bullet EF_2\right) + (F_{SN} \bullet FRAC_{GASF} \bullet EF_3)\right]$$

The digestate scenario saw credits from the avoided direct and indirect N_2O emissions from avoided digestate application following **Equation 7** (IPCC, 2006). The emission factors where the same as **Equation 6**, and 0.22 kg N of organic fertiliser (F_{ON}) was modelled to be prevented from being added to the soil, based on the chemical properties of the digestate (described in **Chapter 5.2.2.3**). The volatilisation from the organic fertiliser deposited ($Frac_{GASM}$) was modelled as 0.20 kg NH_3-N + NO_x-N/kg N applied (IPCC, 2019a).

Equation (7):

$$N_2O - N\ avoided = \left[(F_{ON} \bullet EF_1) + \left(F_{ON} \bullet FRAC_{LEACH-(H)} \bullet EF_2\right) + (F_{ON} \bullet FRAC_{GASM} \bullet EF_3)\right]$$

However, due to the avoided digestate acting as an organic fertiliser, 0.089 kg N synthetic fertiliser was added to the soil, due to digestate substituting N fertiliser 40% in the long-term (Tonini et al., 2018). This was modelled from **Equation 6**. Therefore, the total emission credits for the digestate scenario were calculated as the difference between **Equation 6** and **7**. The upstream burdens of N, P, and K fertiliser production were calculated using ecoinvent 3.7 (Wernet et al., 2016).

5.2.4.4. Avoided bioenergy from forestry

The avoided energy emissions (M_I) were modelled using **Equation 8** utilising calculations of net energy released (N.E.R.) based on the lower heating value (LHV) of the feedstock of 12.4 GJ/tonne, with a moisture content of 30% (Forest Research, 2021), an energy conversion efficiency of 35% (Wernet et al., 2016), and process emissions for the bioenergy of 0.043 kg CO_2 eq./kWh produced (bio_{proc}) (Wernet et al., 2016). All excess heat was assumed to be reused within the process or dumped.

Equation (8):

$$M_I = N.E.R. \bullet \eta_{el} \bullet bio_{proc}$$

From the above calculation, it was estimated that 24.3 kWh of electricity was avoided from bioenergy for the reference flow of 20 kg biomass per kg plastic. This demand for electricity was assumed to be met by an increase in marginal electricity generation. The marginal electricity supply for the European market was modelled using the method of calculating marginal mixes suggested by Schmidt et al. (2011), where the marginal electricity supply for Europe was based on extrapolation of electricity production sources with increasing shares of the market as reported by the IEA (2021). The emissions from the different technologies were modelled using ecoinvent 3.7 (Wernet et al., 2016).

5.2.4.5. Avoided energy from incineration of food waste

The modelling of avoided incineration of the food waste (see **Chapter 5.2.2.4**) was based on the methods described in Moult et al. (2018) using **Equations 9–11**:

Equation (9):

$$M_I = N.E.R. \times \eta_{el} \times Grid_{el}$$

Equation (10):

$$N.E.R. = E_C - (W_C \times W_B)$$

Equation (11):

$$W_B = [(T_1 - T_2) \times W_{Sp.H.C}] + W_{L.H.C}$$

Where M_I was based on the N.E.R., the incinerator energy conversion efficiency (η_{el}) and the emissions intensity of grid electricity ($Grid_{el}$). A value of 22% was used for η_{el} (Moult et al., 2018). Grid emissions were calculated from the market of European electricity (**Chapter 5.2.4.4**). N.E.R was calculated using **Equation 10**, where E_C is the energy content in the food, W_C is the water content, and W_B is the energy required to heat a unit of water to boiling point and then to boil it. W_B was calculated using **Equation 3**, where T_1 is the boiling temperature of water (373 K), T_2 is the starting temperature of the water (taken as 298 K), $W_{Sp.H.C}$ is the specific heat capacity of water (4.18 kJ/kg/K) and $W_{L.H.C}$ is the latent heat of vaporisation of water (2257 kJ/kg). The weighted average E_C and W_C of the food waste amounted to 19.6 MJ/kg total solids and 62.9%, respectively (Albizzati et al., 2021; Tonini et al., 2018). Gross thermal energy outputs were reduced by 15.5% to account for parasitic heat loss (Moult et al., 2018).

5.2.4.6. End-of-life treatment

It is important to include the end-of-life treatment when considering the potential environmental impacts of a product, so that the burdens related to the disposal and fate of the product are placed with the producer, rather than the consumer, a key principle within the circular economy (Maitre-Ekern, 2021). The end-of-life treatment from the bioplastic material followed the methodology of Bishop et al. (2021a), whereby the potential marginal technologies for waste management of the compostable bioplastic were modelled from anaerobic digestion (20%), industrial composting (20%), and incineration with energy recovery (60%), using estimated scenarios of organic material end-of-life fate. The proportions of these waste management destinations were explored within **Scenario 7**. Landfilling is being phased out under EU regulation (European Commission, 2014; European Union, 1999), so it was not considered as a marginal technology for this prospective, consequential study. Where chemical/biophysical properties of the bioplastic mattered in the end-of-life modelling, the compostable bioplastic PLA was used as a reference material.

In this study, most biogenic carbon stored within the compostable bioplastic was assumed to be released back to the atmosphere in the short-term and was thus treated as carbon neutral over its life cycle, i.e., CO_2 emissions from the different end-of-life fates as well as short-term biogenic carbon storage upstream were not included with the GWP accounting. Though, biogenic carbon uptake was considered for the lignocellulosic material iLUC because the carbon was stored long-term over 20 years, with no end-of-life considered for the afforested material in the present model. However, a fraction of the biogenic carbon was returned to the soil via the digestate and the compost, and was assumed to remain out of the atmosphere long-term (100 years) and was thus considered to be sequestered within the model. The end-of-life burdens are modelled similarly for plastic produced from each feedstock in this study. However, for context, it was still relevant to be included within the system boundaries.

For anaerobic digestion, the emissions and energy outputs were calculated using the *LCAD* model framework described in Styles et al. (2016). A large anaerobic digestion plant was modelled, where the bioplastic was converted to biogas and digestate. For this scenario, 50% of biogas was burned in a combined heat and power (CHP) unit to produce electricity and heat. Electricity produced from the CHP unit, minus parasitic requirements within the unit, substituted electricity from the European market mix. Heat from the CHP was used to heat the digester, with any remainder dumped. The other 50% of biogas was modelled to be upgraded to biomethane, which was then injected into the gas grid, substituting natural gas. These biogas uses are in line with future needs for dispatchable low-carbon heat and electricity to meet net-zero GHG targets (CCC, 2019). In the anaerobic digestion plant, the biomethane yield was calculated as 0.23 kg CH_4 per kg bioplastic sent to anaerobic digestion, based on the specific biochemical properties of the plastic (Kolstad et al., 2012). Fugitive emissions of methane from the system were modelled as 1% from the digester and 1.5% from digestate storage, as well as 0.5% from the CHP, and 1.4% from the upgraded methane (Styles et al., 2016). The produced digestate was assumed to be transported 20 km and applied on land with tractors having a fuel consumption of 0.57 L diesel per tonne digestate applied (Tonini et al., 2018). The long-term carbon sequestration equalled 13.2% of the C applied with the digestate (Tonini et al., 2018).

Industrial composting was assumed to operate in an open-windrow system. Within this system, process emissions arose from electricity consumption of 20 kWh per tonne waste (Takata et al., 2012) and from diesel machine operation, which was modelled from similar ecoinvent processes (Wernet et al., 2016). A methane decomposition emission factor of 1.83 kg CH_4 per tonne feedstock was used (Saer et al., 2013). Compost was assumed to be transported 20 km and applied on land by tractor, with a fuel consumption of 0.57 L diesel per tonne compost applied (Tonini et al., 2018). The long-term carbon sequestration was modelled as 11.3% of the C applied with the compost (Tonini et al., 2018).

The emissions arising from incineration with energy recovery followed a similar methodology to **Chapter 5.2.4.5** that was used to calculate the energy emissions, with an energy conversion efficiency of

22% (Moult et al., 2018), with feedstock properties relevant to the bioplastic of a lower heating value of 19.5 MJ/kg (Lokesh et al., 2019) and process emissions for the incineration of 0.053 kg CO_2 eq./kg plastic. Gross thermal energy outputs were also reduced by 15.5% to account for parasitic heat loss to the walls of the incinerator (Moult et al., 2018).

5.2.5. Sensitivity analyses

Interpretation of results, the final phase of an LCA study, should account for critical uncertainties and sensitivities. Accordingly, to explore the influence of critical factors from the above scenarios, three sensitivity analyses were run. The first sensitivity analysis explored the effect on **Scenario 1** if the demand for extra lignocellulosic material resulted in deforestation, rather than initiating afforestation. As such, no positive iLUC or diversion of bioenergy was considered, as these processes are no longer relevant. This sensitivity further treated all CO_2 and CH_4 release throughout the scenario (including end-of-life) as a "fossil" release (rather than biogenic) as the released carbon is no longer being taken up by the next cycle of forest growth.

The second sensitivity analysis explored the effect on **Scenario 1** if the afforestation associated with increased demand for lignocellulosic biomass occurred within a boreal zone, rather than a temperate zone, as it is currently modelled. This included factors such as ratio of below-ground biomass to above-ground biomass, carbon fraction of dry matter, average annual above-ground biomass growth, and dead wood/litter stock under the forest. New factors used IPCC (2019a) values (See **Chapter 5.2.4** and **Supplementary material** (**Chapter 5.7**)).

The third sensitivity scenario explored for **Scenarios 1** and **2** the effect of location on the marginal electricity for the substitution of energy for the lignocellulosic biomass and food waste scenarios, rather than the European market mix (see **Chapter 5.2.4.4**). The countries UK, Norway, and Germany were explored, with marginal electricity mixes calculated using the method of calculating marginal mixes suggested by Schmidt et al. (2011). This method evaluates the change in the share of sources for energy production to the market. The increasing market implies installation of more capacity, which is expected to be of modern technology, rather than old. Thus, the marginal electricity supplies were based on extrapolation of electricity production sources with increasing shares of the market as reported by the IEA (2021) (see **Supplementary material** (**Chapter 5.7**)).

5.2.6. Uncertainty analyses

Error propagation via Monte Carlo simulations was performed with 1,000 iterations per scenario to obtain estimates of result uncertainty. Parameter uncertainties were based on a pedigree matrix for the generated foreground data (Ciroth et al., 2016). The pedigree matrix creates a score based on five aspects of data uncertainty, i.e., reliability, completeness, temporal correlation, geographical correlation, and further technological correlation. The matrix applies a geometric standard deviation to the intermediate and elementary exchanges at the unit process level. The applied pedigree matrices scores can be found in the S**upplementary material** (**Chapter 5.7**).

5.3. Results

5.3.1. Potential indirect land-use effects

The results for **Scenario 1** show that the lignocellulosic biomass, maize biomass, food waste digestate and food waste feedstocks have an "embodied" burden of -15.8, 1.0, 4.8, and 0.2 kg CO_2 eq./kg bioplastic, respectively, arising from upstream and end-of-life emissions (**Figure 5.6**). Thereby suggesting that the lignocellulosic biomass alone is compatible with climate neutrality within this scenario. As seen in the contribution analysis of the expanded life cycle of one kilogram of compostable bioplastic material (**Figure 5.6**), emission credits from afforestation due to iLUC dominated the lignocellulosic environmental performance, whereas the iLUC emissions from the maize biomass production were the greatest contributor to the environmental impact for that feedstock. Within the scenarios, if the iLUC emissions were not considered, the embodied burdens from the maize biomass became less than the burdens attributed to the lignocellulosic biomass feedstock. In **Scenario 1**, if iLUC burdens were not considered, the embodied burdens of the lignocellulosic biomass from forestry became positive, with burdens totalling 1.2 kg CO_2 eq./kg bioplastic, compared to the maize biomass which had a burden of only 0.3 kg CO_2 eq./kg bioplastic.

5.3.2. Valorisation of sidestreams

The utilisation of residues for the original destination processes for the respective raw feedstock or to animal feed (**Scenario 2**) had a significant effect on the "embodied" burdens of the feedstocks. The embodied burdens of the food waste digestate were reduced 15% from **Scenario 1** levels to 4.0 kg CO_2 eq./kg bioplastic, improving the environmental performance of the material. Maize biomass burdens were also reduced by 59% to 0.4 kg CO_2 eq./kg bioplastic (0.2 kg CO_2 eq./kg bioplastic without iLUC). However, conversely, the environmental mitigation potential of the lignocellulosic biomass was reduced to

-0.2 kg CO_2 eq./kg bioplastic, a 99% reduction in GWP credit (or 0.7 kg CO_2 eq./kg bioplastic without iLUC). Food waste also saw a deterioration in its overall environmental performance, increasing GWP burdens by 105% to 0.4 kg CO_2 eq./kg bioplastic (**Scenario 2**, **Figure 5.6**).

5.3.3. Decentralisation

Decentralisation of bioplastic production (**Scenario 3**) had large effects on environmental burdens for feedstocks required in large volumes to make one kg of bioplastic. For example, bioplastic from food waste digestate, which required 100 L of feedstock per kg bioplastic produced, experienced an environmental impact reduction of 60% to 1.9 kg CO_2 eq./kg bioplastic when the distance to production was reduced to 100 km. An overall GWP reduction of 3% was also seen for the lignocellulosic biomass scenario, with revised embodied burdens equalling -16.3 kg CO_2 eq./kg bioplastic (0.7 kg CO_2 eq./kg bioplastic without iLUC), whilst a reduction of 7% was observed for the maize biomass scenario, with total embodied burdens of 0.9 kg CO_2 eq./kg bioplastic (0.2 kg CO_2 eq./kg bioplastic without iLUC). Food waste also saw a large GWP reduction of 171%, where the feedstock yielded a net environmental credit of -0.13 kg CO_2 eq./kg bioplastic (**Scenario 3**, **Figure 5.6**).

5.3.4. Bioenergy displacement

When the lignocellulosic biomass was considered to be diverted from generation of bioheat, rather than bioelectricity, the type of substituted (fossil-based) heating fuel had a large impact on the net environmental performance of the feedstock (**Scenario 4**, **Figure 5.6**). Whilst there were some avoided emissions from avoided wood-heat production, at -0.7 kg CO_2 eq. per kg bioplastic, the substituted process emissions from heat production from natural gas, heating oil, and coal were estimated to be 14.1, 17.5, and 40.6 kg CO_2 eq., respectively. Therefore, the overall global warming potential results for **Scenario 4** were estimated to be -3.0, 0.4, and 23.5 kg CO_2 eq./kg lignocellulosic bioplastic for the natural gas, oil, and coal fuel scenarios, respectively (or 14.1, 17.4, and 40.5 kg CO_2 eq./kg bioplastic without iLUC, respectively).

Scenario 5, where circularity of residues within the lignocellulosic biomass bioplastic production was considered and when the bioheat was displaced, led to emissions of 0.5, 0.7, and 1.8 kg CO_2 eq./kg bioplastic for the scenarios involving avoided substitution of natural gas, oil, and coal fuel, respectively (or 1.3, 1.5, and 2.7 kg CO_2 eq./kg bioplastic without iLUC, respectively) (**Scenario 5**, **Figure 5.6**).

5.3.5. Digestate drying

When the food waste digestate was dried within the anaerobic digestion plant (**Scenario 6**), a large GWP reduction from **Scenario 1** arose within the feedstock transportation process, which observed a reduction from 3.8 to 0.08 kg CO_2 eq./kg bioplastic. Overall, the GWP of the digestate feedstock was reduced 80% from **Scenario 1** down to 1.0 kg CO_2 eq./kg bioplastic (**Scenario 6**, **Figure 5.6**).

5.3.6. End-of-life impacts

The results from **Scenario 7** suggest that the future end-of-life fate of the plastics has little impact on the embodied environmental burdens of the different feedstocks investigated, compared with other environmental hotspots. However, it was found that diverting 100% of the bioplastic waste to anaerobic digestion had the lowest net GWP burden, and diverting waste to 100% incineration with energy recovery resulted in the largest net GWP burden. The net GWP burden for the lignocellulosic biomass value chain (excluding processing but including end-of-life) was -16.0, -15.8, and -15.7 kg CO_2 eq./ kg bioplastic for the anaerobic digestion, composting, and incineration scenarios, respectively. For maize bioplastic value chains, these burdens were estimated to be 0.8, 1.0, and 1.0 kg CO_2 eq. for the anaerobic digestion, composting, and incineration scenarios, respectively. Similarly, for digestate feedstock value chains these burdens were calculated as 4.6, 4.8, and 4.8 kg CO_2 eq./ kg bioplastic for the anaerobic digestion, composting, and incineration scenarios, respectively. Finally, the emissions for the anaerobic digestion, composting, and incineration scenarios of the food waste were -0.003, 0.22, and 0.24 kg CO_2 eq./ kg bioplastic, respectively.

5.3.7. Carbon neutrality

The scenarios which had net GWP credits for the feedstock upstream and end-of-life emissions included the lignocellulosic biomass from **Scenarios 1**, **2**, **3**, and **4** (from diverted natural gas), when iLUC was considered for the system and **Scenario 3** for the food waste feedstock. No other feedstocks or scenarios modelled achieved net zero GHG emissions (**Figure 5.6**).

However, all scenarios except food waste digestate from **Scenario 1** and **2**, and lignocellulosic biomass from **Scenario 4** (except when avoided heat substituted with natural gas and heating oil included iLUC), resulted in pre-production and end-of-life burdens that were lower than the petrochemical plastic production and end-of-life emissions (**Figure 5.6**). A full and further disaggregated breakdown of the results for each scenario can be found in the **Supplementary Material** (**Chapter 5.7**).

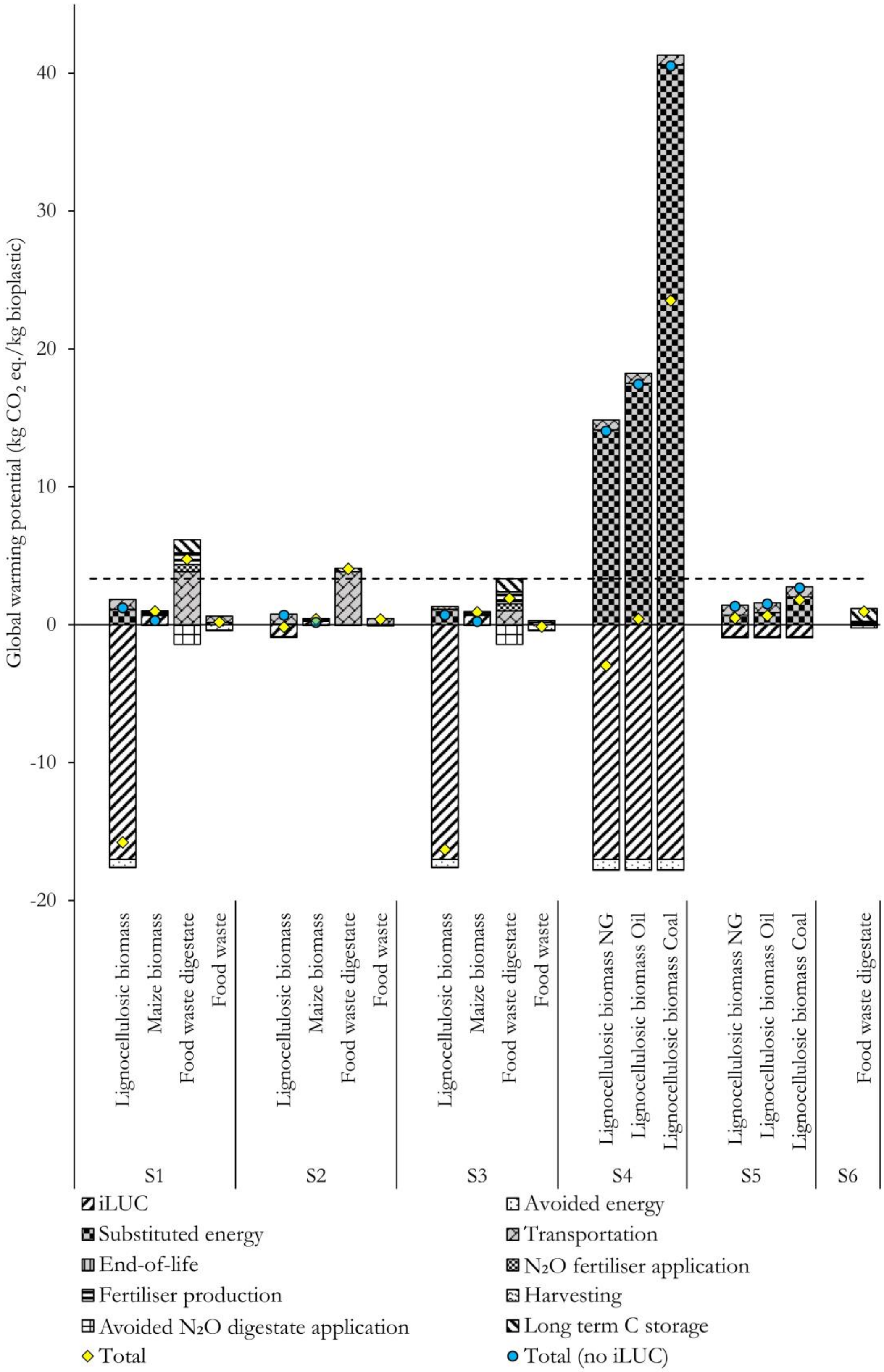

Figure 5.6 – Contribution analysis for the management of the four bioplastic feedstocks investigated (lignocellulosic biomass, maize biomass, food waste digestate, and food waste) and subsequent scenarios, relating to each scenario's global warming potential results. Six scenarios (S) are compared. S1: initial modelled systems; S2: utilising residues for circular production; S3: decentralised production; S4: heat from biomass avoided, with natural gas (NG), heating oil, and coal substituting the process; S5: utilising resides and heat from biomass avoided with NG, heating oil, and coal substituting the biomass; S6: dried digestate used as a feedstock. A seventh scenario was undertaken to explore 100% diversion of bioplastic waste to different treatments, but differences in results were insignificant and the same across all feedstocks. Yellow diamonds represent the total environmental impact for each scenario. Blue circles represent total results with no indirect land-use change. The dotted line represents typical emissions for petrochemical plastic (Bishop et al., 2021a). C: carbon; N_2O: dinitrogen monoxide.

5.3.8. Uncertainty analyses

Figure 5.7 shows the distribution of results for each scenario over 1,000 Monte Carlo simulations. Full simulation results can be found in the **Supplementary Material** (**Chapter 5.7**). **Figure 5.7** shows that the GWP results per scenario are highly variable and contain considerable uncertainty. Although some large uncertainty suggests the potential for overlap between scenarios, there remains considerable differences between the scenarios. As such, the uncertainty analysis results further corroborate the management decisions explored within the scenarios to reduce environmental emissions for the bioplastic feedstocks (see **Chapter 5.4**).

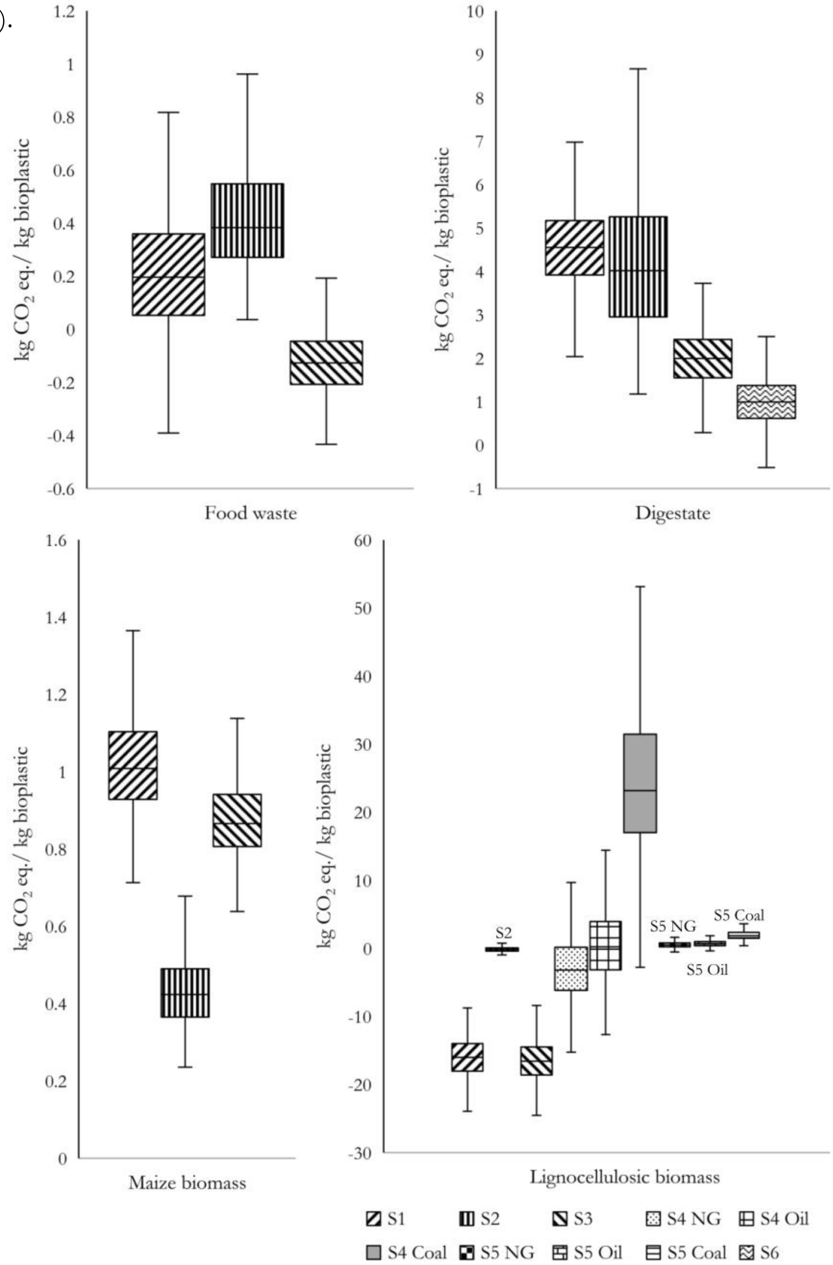

Figure 5.7 – Results of the Monte Carlo simulation for the six scenarios for the 4 studied feedstocks. For visualisation purposes, outliers have been excluded from the graphs. A full breakdown of the Monte Carlo results can be found in supplementary material. Six scenarios (S) are compared, as detailed in the legend of **Figure 5.6**. The box and whisker plots show the minimum, first quartile, median, third quartile, and maximum value for each scenario.

5.3.9. Sensitivity analyses

The sensitivity analyses undertaken in this study demonstrated how the results may change from different practices and locations (**Table 5.2**). Full sensitivity results can be found in the **Supplementary Material** (**Chapter 5.7**). As one would expect, when an increased demand for lignocellulosic material resulted in deforestation rather than afforestation, environmental emissions increased significantly from a net credit to 26.8 kg CO_2 eq./kg bioplastic.

Location had a large effect on the results (**Table 5.2**). The sequestration benefits of afforestation were significantly reduced by 51%, to -7.7 kg CO_2 eq./kg bioplastic, when the afforestation was assumed to occur in a boreal forest compared to a temperate forest (**Table 5.2**).

The sensitivity analyses showed that location effects on the marginal energy mix can also have a large impact on the feedstocks which are diverted from energy generation, reflecting the composition of the national marginal energy mixes (**Table 5.2**). For lignocellulosic biomass feedstock **Scenario 1** and **2**, as well as food waste **Scenario 1**, the lower the emissions per kWh from the country specific energy mixes, the better the environmental performance of the feedstock. However, for the food waste **Scenario 2**, the reduction in end-of-life credits from energy substitution from incineration and anaerobic digestion (due to the "cleaner" energy mixes) resulted in a reduction of the environmental performance of the feedstock.

Table 5.2 – Percentage difference between the sensitivity analyses results and the results from the original **Scenario 1** and **2** (defined in **Chapter 5.2.3**). Negative percentages represent a reduced GWP, whilst a positive percentage represents an increased environmental footprint.

	% Difference from original scenario results	
Sensitivity	**Scenario 1**	**Scenario 2**
Sensitivity 1 – Deforestation (instead of afforestation)	270 *	**
Sensitivity 2 – Boreal forest (instead of temperate forest)	51	**
Sensitivity 3 – UK electricity, lignocellulosic biomass	-1	-2
Sensitivity 3 – UK electricity, food waste	-9	1
Sensitivity 3 – Norway electricity, lignocellulosic biomass	-5	-9
Sensitivity 3 – Norway electricity, food waste	-48	4
Sensitivity 3 – Germany electricity, lignocellulosic biomass	5	10
Sensitivity 3 – Germany electricity, food waste	53	-4

* The scenario turns from a net environmental credit to a net environmental debit
** Sensitivity 1 and 2 were run for **Scenario 1**

5.4. Discussion

5.4.1. Importance of land-use change

This study demonstrates that not all feedstocks for bioplastic production are environmentally equal, with land-use change impacts, especially iLUC, having a profound influence on the climate mitigation potential of compostable bioplastic value chains derived from different feedstocks. When considering the entire expanded life cycle of four potential bioplastic feedstocks, it was found that the embodied environmental burdens of the feedstocks, before the material reaches bioplastic production, can vary dramatically (**Figure 5.6**), mostly due to emissions incurred or removed via iLUC.

Environmental credits from the lignocellulosic feedstock are derived mainly from the carbon mitigation potential from afforestation due to iLUC. The iLUC modelled in this study was considered over 20 years, which was a conservative approach due to uncertain future harvesting frequencies and fates (e.g., harvested wood products or bioenergy) outside the scope of this paper. Therefore, iLUC results from the forestry scenario are expected to be much larger over the full 50 years of forest growth and beyond when considering the carbon sequestered in harvested wood products or when including carbon capture and storage (BECCS) in bioenergy generation (Forster et al., 2021). Interestingly, when the volume of diverted feedstock from the total harvest was reduced due to circularity of production (**Scenario 2**), and thus the area of iLUC was diminished, the environmental benefits were lower due to reduced afforestation. However, the reduced iLUC credits arising from more circular production were also associated with less bioenergy displacement, which in turn mitigated the high emissions from coal, oil, and natural gas derived heat substitution (**Figure 5.6**). Positive land-use change arising from afforestation for bio-based production/bioenergy will be critical to achieving international climate targets (Englund et al., 2020; Forster et al., 2021). The findings of this study support to the positive land-use change required to achieve global climate stabilisation (IPCC, 2019b). The sensitivity analysis (**Table 5.2**) demonstrated the significance of the impacts that could be attributed to deforestation linked with feedstock acquisition – again highlighting the importance of achieving carbon sequestration through afforestation via positive land-use change, and the risks associated with sourcing bioplastic feedstock from non-sustainably-managed forests.

Conversely, the inclusion of iLUC incurred a penalty for the maize feedstock that reflects the constrained availability of cropland and the risk of displacing existing agricultural production elsewhere (Schmidt et al., 2015). These iLUC burdens dominate the life cycle impacts of primary feedstocks, but are rarely included within the life cycle inventory of bioplastic studies (Bishop et al., 2021b). Whilst accounting for iLUC presents many uncertainties, excluding the emission penalty from the extra demand for land may lead to misleading conclusions around the environmental performance of the feedstock (Bishop et al., 2021b; Searchinger et al., 2018). However, even when iLUC emissions were considered for the maize feedstock, in this study the emissions remained lower than those of the petrochemical plastic production and end-of-life across all scenarios investigated, thereby indicating the potential climate mitigation benefits of pursuing maize-based bioplastic.

5.4.2. Valorisation of biomass side-streams

This study demonstrated that reducing overall emissions from bioplastic production is possible and will require careful consideration as to how residues and sidestreams are treated within the production process. Sending back these residues, depending on their integrity and toxicity after the extraction of critical compounds for bioplastic production, to the process from which they were originally substituted or to use them for animal feed, reduces the net environmental impact (**Figure 5.6**). Large decreases in GWP were seen for the maize biomass, food waste, and digestate biomass scenarios when the residues were utilised (**Figure 5.6**). Valorisation of the residues within the lignocellulosic biomass value chain also resulted in decreased environmental impacts when the bioenergy avoided was from heat, but environmental performance of the material decreased when the avoided bioenergy was from electricity, as described in **Chapter 5.4.1**.

Other potential pathways for side-streams could include valorisation end-of-life treatments, such as anaerobic digestion, or diversion to extract additional high-value chemicals and products via a biorefinery approach (Octave and Thomas, 2009). If a biorefinery approach is not possible, side stream materials could be used to generate energy for production processes. A large share of emissions from production arises from energy generation for feedstock drying and process requirements (Bishop et al., 2021a). Using the bio-based materials from the bioplastic production would both displace the requirements for conventional energy sources and reduce the amount of waste material. If directed to anaerobic digestion, processing the digestate into multiple higher value compounds (Stiles et al., 2018; Styles et al., 2018a), as well as circular economy thinking, will be fundamental to further reduce emissions of the bioplastic production.

5.4.3. Further major hotspots from bioplastic feedstocks

Although the scenarios often placed the food waste digestate as the feedstock with the highest environmental impact, and greater environmental impacts than conventional petrochemical plastic (**Figure 5.6**), the scenarios explored for such an uncertain novel feedstock showed that there was huge potential for improvement of environmental performance. The main benefit of this secondary feedstock over the maize feedstock is that it has no competition with land use, and as such no direct LUC or iLUC burdens exist. However, the digestate did require transport and processing of large volumes of feedstock to produce 1 kg of bioplastic and transportation burdens dominated the environmental impact of the feedstock. When these burdens where reduced due to the drying of material or by reducing the transportation distances, the digestate GWP results were decreased dramatically (**Figure 5.6**), corresponding with other studies which highlight that utilisation of digestate should occur close to source (Styles et al., 2018a). As such, the "default" transport distances applied here are likely pessimistic for the food waste digestate, which is mostly produced at large centralised food waste digestion plants. Therefore, a best-case scenario for the food waste digestate may involve a decentralised approach (**Scenario 3**) adapted to include zero transportation. Environmental improvements were also seen for the other feedstocks when the transportation distances were decreased. In fact, the food waste feedstock became a net environmental credit for this scenario. Therefore, logistical optimisation around feedstock availability and plant economies of scales will be critical to reducing emissions.

Figure 5.6 shows that the upstream burdens of the bioplastic feedstock production and associated diversion of material flows have a greater impact on the overall environmental performance than the end-of-life emissions. An important reason for this was that the (future) electricity market mix utilised for this study was derived from mostly renewable sources (IEA, 2021; Schmidt et al., 2011) and thus energy recovery from the end-of-life did not substitute a high-emitting energy mix, providing little credit to the system. Similarly, this also meant that there was a low environmental cost of displacing the electricity from the lignocellulosic biomass and food waste (**Scenario 1**). This was visible in the sensitivity analysis when different marginal electricity mixes were explored (**Table 5.2**). When the lignocellulosic biomass scenario was run to model the bioenergy being substituted by fossil heat, large environmental penalties were applied to the system. Environmental outcomes are thus likely to be negative when the lignocellulosic material for bioplastic production competes with bioheat production in large scale boilers and industrial furnaces. This demonstrates the necessity of carefully considering any displaced uses of feedstocks when assessing the sustainability of bioplastic production. Specifically, new evidence presented here suggests that the use of lignocellulosic biomass for bioplastic production should only be considered where demand for bioheat (e.g., to decarbonise industrial processes such as cement production) can be met.

5.4.4. Limitations and future research

The feedstock scenarios modelled were some of the most likely situations and configurations that could arise for each of the four main feedstock types. Although this study tried to encompass a wide variety of potential outcomes from what could happen, it is beyond the scope of any study to cover the full range of value chain configurations. Nonetheless, the scenarios spanned the outer limits of key factors influencing the environmental performance of bioplastic value chains, bounding the potential outcomes of this study within a realistic range and elucidating the main drivers of environmental outcomes of bioplastic production.

A limitation in the lignocellulosic iLUC modelling was the use of a biophysical link between wood demand and afforestation, rather than an economic link related to market signals (Ekvall, 2019). Another limitation is the assumption that an extra demand for lignocellulosic material results in afforestation. In practice, it would also be possible that the demand for woody feedstock could drive additional extraction from existing forests, e.g., through increased thinnings and thus lower C storage. The expected outcome of this could be similar to the deforestation sensitivity (**Table 5.2**).

How the life cycle boundaries are drawn is crucial when designing an LCA study. For the digestate value chain, the boundaries for the food waste digestate started with the digestate available from existing anaerobic digestion plants. However, if food waste was diverted from other waste treatment to become bioplastic via digestate, counterfactual waste management credits will exist within the new framework, improving the environmental performance of the digestate material as a bioplastic precursor.

Although only four feedstocks were explored within this study, these differing feedstocks are analogous to other potential materials. The maize biomass will follow similar burdens to other primary crop feedstocks, the lignocellulosic biomass modelled from the sawmill will be similar to other low-value wood valorisation options, and food waste digestate will be similar to other digestate feedstocks. Future work can also evaluate additional diverse potential feedstocks for compostable bioplastic production, such as grass (Patterson et al., 2021), crops (Jimenez-Rosado et al., 2019), organic waste (Tsang et al., 2019), and algal biomass (Prieto et al., 2017).

Although used as a reference value within **Figure 5.6**, the results from the bioplastic feedstock scenarios did not include any credits for avoided petrochemical production. At the point when bioplastic production becomes large enough that it displaces petrochemical plastic production, full consequential transitions from the avoided petrochemical plastic to the specific bioplastics should be modelled as well. This may come about through increasing bioplastic production in response to high demand, and/or near cessation of petrochemical plastic production due to regulatory bans on its use (Xanthos and Walker, 2017).

Future work should consider alternative uses for the residue biomass, especially exploring the use of a biorefinery concept for valorisation of waste residues. The potential type of land-use change for the afforestation (rather than only low productivity grassland) could also be explored in future studies. Decentralisation of production may affect bioplastic production efficiency and ultimately the environmental performance of the bioplastic. Economies of scale, capital costs, and reduced transportation costs should all be considered and weighed when planning for decentralisation (Lauven et al., 2018). These potential impacts should be explored in future studies.

In a time when achieving net-zero GHG emissions over the coming 30 years has become the key target for tackling climate change (UNFCCC, 2015), it is important to make sure that the (bio)plastics industry is engaged with this transition. The values presented give an indication of the production emission constraints in which future bioplastic production should operate in order to at least achieve carbon neutrality, where emission removals equivalent to net emissions are required to achieve net-zero. Feedstocks which have a positive environmental burden entering production will need to be produced with negative production emissions. However, as bioplastic production is a relatively new technology, there is a unique opportunity to design the production pathways where net-zero GHG emissions can be "inset" within these value chains, where land-use change emissions can be decreased (or become net removals), where residues can be utilised, and where hotspots can be reduced. More specifically, the lignocellulosic biomass was found the most environmentally efficient feedstock of the feedstocks investigated when the demand was sourced from sustainably managed forests (i.e., deforestation was avoided) and was linked with increased afforestation. Food waste and digestate feedstocks were found to be most suitable when processing can be undertaken close to existing waste or anaerobic digestion facilities to minimise transport requirements (perhaps linked with biorefinery concepts in the future). Finally, maize feedstock emissions may be reduced if the displacement of food and feed cropping can be avoided, e.g., by crop rotation optimisation and coupling with diet change to reduce demand for animal feed production.

5.5. Conclusion

This study demonstrated that not all compostable bioplastic feedstocks are environmentally equal, and that feedstocks can come with highly variable upstream burdens. The global warming potential burdens from upstream emissions and end-of-life were found to range between -16.3 to +23.5 kg CO_2 eq./kg bioplastic across four potential feedstocks, namely lignocellulosic biomass, maize biomass, food waste digestate, and food waste. The inclusion of indirect land-use change had a considerable negative impact on the embodied burdens of maize-based plastic, but a positive impact, via terrestrial carbon sequestration within new commercial forest plantations, for lignocellulosic-based plastic (unless lignocellulosic feedstock demand drives deforestation, leading to the highest emission burdens per kg bioplastic). Appropriate use of residues and side-streams is critical to the environmental performance of bioplastics and should be actively considered when planning the production phase of the bioplastic. Efficient utilisation of residues may require decentralisation of bioplastic production, thus reducing transportation burdens. Overall, there is high potential to achieve net-zero GHG emissions over the whole life cycle of compostable bioplastic, but careful consideration is required so that the environmental impact of petrochemical plastic is not replaced with bioplastics which are equally (or even more) contributing to GHG emissions.

5.6. References

Al-Battashi, H., Annamalai, N., Al-Kindi, S., Nair, A.S., Al-Bahry, S., Verma, J.P., Sivakumar, N., 2019. Production of bioplastic (poly-3-hydroxybutyrate) using waste paper as a feedstock: Optimization of enzymatic hydrolysis and fermentation employing Burkholderia sacchari. Journal of Cleaner Production 214, 236–247. https://doi.org/10.1016/j.jclepro.2018.12.239

Albizzati, P.F., Tonini, D., Astrup, T.F., 2021. High-value products from food waste: An environmental and socio-economic assessment. Science of The Total Environment 755, 142466. https://doi.org/10.1016/J.SCITOTENV.2020.142466

Altun, M., 2019. Polyhydroxyalkanoate production using waste vegetable oil and filtered digestate liquor of chicken manure. Preparative Biochemistry and Biotechnology 49, 493–500. https://doi.org/10.1080/10826068.2019.1587626

Bamelis, L., Blancke, S., Camargo-Valero, M.A., De Clercq, L., Haumont, A., De Keulenaere, B., Delvigne, F., Meers, E., Michels, E., Ramirez-Sosa, D.R., Ross, A.B., Smeets, H., Tarayre, C., Williams, P.T., 2015. BIOREFINE Recycling inorganic chemicals from digestate derivatives Techniques for nutrient recovery from digestate derivatives.

Barboza, L.G.A., Cózar, A., Gimenez, B.C.G., Barros, T.L., Kershaw, P.J., Guilhermino, L., 2018. Macroplastics pollution in the marine environment, in: World Seas: An Environmental Evaluation Volume III: Ecological Issues and Environmental Impacts. Elsevier, pp. 305–328. https://doi.org/10.1016/B978-0-12-805052-1.00019-X

Bishop, G., Styles, D., Lens, P.N.L., 2021a. Environmental performance of bioplastic packaging on fresh food produce: A consequential life cycle assessment. Journal of Cleaner Production 317, 128377. https://doi.org/10.1016/J.JCLEPRO.2021.128377

Bishop, G., Styles, D., Lens, P.N.L., 2021b. Environmental performance comparison of bioplastics and petrochemical plastics: A review of life cycle assessment (LCA) methodological decisions. Resources, Conservation and Recycling 168, 105451. https://doi.org/10.1016/j.resconrec.2021.105451

Boots, B., Russell, C.W., Green, D.S., 2019. Effects of Microplastics in Soil Ecosystems: Above and Below Ground. Environmental Science & Technology 53, 11496–11506. https://doi.org/10.1021/acs.est.9b03304

Brizga, J., Hubacek, K., Feng, K., 2020. The Unintended Side Effects of Bioplastics: Carbon, Land, and Water Footprints. One Earth. https://doi.org/10.1016/j.oneear.2020.06.016

CCC, 2019. Net Zero: The UK's contribution to stopping global warming, Committe on Climate Change. London.

Ciroth, A., Muller, S., Weidema, B., Lesage, P., 2016. Empirically based uncertainty factors for the pedigree matrix in ecoinvent. International Journal of Life Cycle Assessment 21, 1338–1348. https://doi.org/10.1007/s11367-013-0670-5

Ekvall, T., 2019. Attributional and consequential life cycle assessment. Sustainability Assessment at the 21st century 395, 1–22. https://doi.org/10.5772/INTECHOPEN.89202

Englund, O., Börjesson, P., Berndes, G., Scarlat, N., Dallemand, J.F., Grizzetti, B., Dimitriou, I., Mola-Yudego, B., Fahl, F., 2020. Beneficial land use change: Strategic expansion of new biomass plantations can reduce environmental impacts from EU agriculture. Global Environmental Change 60, 101990. https://doi.org/10.1016/J.GLOENVCHA.2019.101990

Eshtaya, M.K., Rahman, N.A., Hassan, M.A., 2013. Bioconversion of restaurant waste into Polyhydroxybutyrate (PHB) by recombinant E. coli through anaerobic digestion. International Journal of Environment and Waste Management 11, 27–37. https://doi.org/10.1504/IJEWM.2013.050521

European Bioplastics, 2020. Bioplastics Market Data [WWW Document]. URL https://www.european-bioplastics.org/market/ (accessed 4.7.21).

European Commission, 2018. A European strategy for plastics in a circular economy. COM/2018/028 final. Brussels.

European Commission, 2014. Towards a circular economy: A zero waste programme for Europe. Brussels.

European Union, 1999. Council directive 1999/31/EC on the landfill of waste. Official Journal of the European Communities L182/1.

Fierro, J.J., Escudero-Atehortua, A., Nieto-Londoño, C., Giraldo, M., Jouhara, H., Wrobel, L.C., 2020. Evaluation of waste heat recovery technologies for the cement industry. International Journal of Thermofluids 7–8, 100040. https://doi.org/10.1016/J.IJFT.2020.100040

Forest Research, 2021. Typical calorific values of fuels [WWW Document]. URL https://www.forestresearch.gov.uk/tools-and-resources/fthr/biomass-energy-resources/reference-biomass/facts-figures/typical-calorific-values-of-fuels/ (accessed 7.15.21).

Forster, E.J., Healey, J.R., Dymond, C., Styles, D., 2021. Commercial afforestation can deliver effective climate change mitigation under multiple decarbonisation pathways. Nature Communications 2021 12:1 12, 1–12. https://doi.org/10.1038/s41467-021-24084-x

IEA, 2021. IEA – International Energy Agency [WWW Document]. URL https://www.iea.org/ (accessed 6.11.21).

IPCC, 2019a. 2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Cambridge, UK.

IPCC, 2019b. Climate Change and Land: an IPCC special report on climate change, desertification, land degradation, sustainable land management, food security, and greenhouse gas fluxes in terrestrial ecosystems. Cambridge.

IPCC, 2018. Global Warming of 1.5°C. An IPCC Special Report on the impacts of global warming of 1.5°C above pre-industrial levels and related global greenhouse gas emission pathways, in the context of strengthening the global response to the threat of climate change,. In Press.

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

IPCC, 2006. IPCC Guildlines for National Greenhouse Gas Inventories. Cambridge, UK.

ISO, 2006a. ISO 14040: Environmental management -- Life cycle assessment -- Principles and framework. Geneva.

ISO, 2006b. ISO 14044: Environmental management -- Life cycle assessment -- Requirements and guidelines. Geneva.

Jimenez-Rosado, M., Zarate-Ramirez, L.S., Romero, A., Bengoechea, C., Partal, P., Guerrero, A., 2019. Bioplastics based on wheat gluten processed by extrusion. Journal of Cleaner Production 239. https://doi.org/10.1016/j.jclepro.2019.117994

Kim, H., Lee, S., Ahn, Y., Lee, J., Won, W., 2020. Sustainable Production of Bioplastics from Lignocellulosic Biomass: Technoeconomic Analysis and Life-Cycle Assessment. ACS Sustainable Chemistry and Engineering 8, 12419–12429. https://doi.org/10.1021/acssuschemeng.0c02872

Kolstad, J.J., Vink, E.T.H., De Wilde, B., Debeer, L., 2012. Assessment of anaerobic degradation of Ingeo™ polylactides under accelerated landfill conditions. https://doi.org/10.1016/j.polymdegradstab.2012.04.003

Lauven, L.P., Karschin, I., Geldermann, J., 2018. Simultaneously optimizing the capacity and configuration of biorefineries. Computers & Industrial Engineering 124, 12–23. https://doi.org/10.1016/J.CIE.2018.07.014

Lokesh, K., Matharu, A., Clark, J., Rossi, V., Bengoa, X., Briassoulis, D., Hiskakis, M., Pikasi, A., Salim, I., Wojnowska-Baryła, I., 2019. Assessing Sustainability of Managed End-of-life Options for Bio-based Products in a Circular Economy. York.

Lyons, G.A., Cathcart, A., Frost, J.P., Wills, M., Johnston, C., Ramsey, R., Smyth, B., 2021. Review of Two Mechanical Separation Technologies for the Sustainable Management of Agricultural Phosphorus in Nutrient-Vulnerable Zones. Agronomy 2021, Vol. 11, Page 836 11, 836. https://doi.org/10.3390/AGRONOMY11050836

Maitre-Ekern, E., 2021. Re-thinking producer responsibility for a sustainable circular economy from extended producer responsibility to pre-market producer responsibility. Journal of Cleaner Production 286, 125454. https://doi.org/10.1016/J.JCLEPRO.2020.125454

Moult, J.A., Allan, S.R., Hewitt, C.N., Berners-Lee, M., 2018. Greenhouse gas emissions of food waste disposal options for UK retailers. Food Policy 77, 50–58. https://doi.org/10.1016/j.foodpol.2018.04.003

Muñoz, I., Schmidt, J.H., 2016. Methane oxidation, biogenic carbon, and the IPCC's emission metrics. Proposal for a consistent greenhouse-gas accounting. International Journal of Life Cycle Assessment 21, 1069–1075. https://doi.org/10.1007/s11367-016-1091-z

Nessi, S., Sinkko, T., Bulgheroni, C., Konti, A., Tonini, D., Pant, R., 2020. Comparative life cycle assessment (LCA) of alternative feedstock for plastics production- Part I. European Commission, Ispra.

Ni, Y., Richter, G., Mwabonje, O., Qi, A., Patel, M., Woods, J., 2021. Novel integrated agricultural land management approach provides sustainable biomass feedstocks for bioplastics and supports the UK's 'net-zero' target. Environmental Research Letters 16, 014023. https://doi.org/10.1088/1748-9326/abcf79

Nieder-Heitmann, M., Haigh, K., Görgens, J.F., 2019. Process design and economic evaluation of integrated, multi-product biorefineries for the co-production of bio-energy, succinic acid, and polyhydroxybutyrate (PHB) from sugarcane bagasse and trash lignocelluloses. Biofuels, Bioproducts and Biorefining 13, 599–617. https://doi.org/10.1002/bbb.1972

Octave, S., Thomas, D., 2009. Biorefinery: Toward an industrial metabolism. Biochimie 91, 659–664. https://doi.org/10.1016/J.BIOCHI.2009.03.015

Passanha, P., Esteves, S.R., Kedia, G., Dinsdale, R.M., Guwy, A.J., 2013. Increasing polyhydroxyalkanoate (PHA) yields from Cupriavidus necator by using filtered digestate liquors. Bioresource Technology 147, 345–352. https://doi.org/10.1016/j.biortech.2013.08.050

Patterson, T., Massanet-Nicolau, J., Jones, R., Boldrin, A., Valentino, F., Dinsdale, R., Guwy, A., 2021. Utilizing grass for the biological production of polyhydroxyalkanoates (PHAs) via green biorefining: Material and energy flows. Journal of Industrial Ecology 25, 802–815. https://doi.org/10.1111/JIEC.13071

Pawelzik, P., Carus, M., Hotchkiss, J., Narayan, R., Selke, S., Wellisch, M., Weiss, M., Wicke, B., Patel, M.K., 2013. Critical aspects in the life cycle assessment (LCA) of bio-based materials - Reviewing methodologies and deriving recommendations. Resources Conservation and Recycling 73, 211–228. https://doi.org/10.1016/j.resconrec.2013.02.006

Peng, L., Fu, D., Qi, H., Lan, C.Q., Yu, H., Ge, C., 2020. Micro- and nano-plastics in marine environment: Source, distribution and threats — A review. Science of the Total Environment 698, 134254. https://doi.org/10.1016/j.scitotenv.2019.134254

Peng, W., Pivato, A., 2019. Sustainable Management of Digestate from the Organic Fraction of Municipal Solid Waste and Food Waste Under the Concepts of Back to Earth Alternatives and Circular Economy. Waste and Biomass Valorization. https://doi.org/10.1007/s12649-017-0071-2

Piemonte, V., Gironi, F., 2011. Land-use change emissions: How green are the bioplastics? Environmental Progress & Sustainable Energy 30, 685–691. https://doi.org/10.1002/ep.10518

Prieto, C.V.G., Ramos, F.D., Estrada, V., Villar, M.A., Diaz, M.S., 2017. Optimization of an integrated algae-based biorefinery for the production of biodiesel, astaxanthin and PHB. Energy 139, 1159–1172. https://doi.org/10.1016/j.energy.2017.08.036

Reddy, R.L., Reddy, V.S., Gupta, G.A., 2013. Study of Bio-plastics As Green & Sustainable Alternative to Plastics. International Journal of Emerging Technology and Advanced Engineering 3, 82–89.

Reshmy, R., Thomas, D., Philip, E., Paul, S.A., Madhavan, A., Sindhu, R., Sirohi, R., Varjani, S., Pugazhendhi, A., Pandey, A., Binod, P., 2021. Bioplastic production from renewable lignocellulosic feedstocks: a review. Reviews in Environmental Science and Bio/Technology 2021 20:1 20, 167–187. https://doi.org/10.1007/S11157-021-09565-1

Saer, A., Lansing, S., Davitt, N.H., Graves, R.E., 2013. Life cycle assessment of a food waste composting system: Environmental impact hotspots. Journal of Cleaner Production 52, 234–244. https://doi.org/10.1016/j.jclepro.2013.03.022

Schmidt, J.H., Thrane, M., Merciai, S., Dalgaard, R., 2011. Inventory of country specific electricity in LCA-consequential and attributional scenarios Methodology report v2. Aalborg.

Schmidt, J.H., Weidema, B.P., Brandão, M., 2015. A framework for modelling indirect land use changes in Life Cycle Assessment. Journal of Cleaner Production 99, 230–238. https://doi.org/10.1016/j.jclepro.2015.03.013

Searchinger, T.D., Wirsenius, S., Beringer, T., Dumas, P., 2018. Assessing the efficiency of changes in land use for mitigating climate change. Nature 564, 249–253. https://doi.org/10.1038/s41586-018-0757-z

Seruga, P., Krzywonos, M., Wilk, M., 2020. Treatment of By-Products Generated from Anaerobic Digestion of Municipal Solid Waste. Waste and Biomass Valorization 11, 4933–4940. https://doi.org/10.1007/s12649-019-00831-6

Spierling, S., Knupffer, E., Behnsen, H., Mudersbach, M., Krieg, H., Springer, S., Albrecht, S., Herrmann, C., Endres, H.J., 2018. Bio-based plastics - A review of environmental, social and economic impact assessments. Journal of Cleaner Production 185, 476–491. https://doi.org/10.1016/j.jclepro.2018.03.014

Stiles, W.A.V., Styles, D., Chapman, S.P., Esteves, S., Bywater, A., Melville, L., Silkina, A., Lupatsch, I., Fuentes Grünewald, C., Lovitt, R., Chaloner, T., Bull, A., Morris, C., Llewellyn, C.A., 2018. Using microalgae in the circular economy to valorise anaerobic digestate: challenges and opportunities. Bioresource Technology 267, 732–742. https://doi.org/10.1016/J.BIORTECH.2018.07.100

Strungaru, S.A., Jijie, R., Nicoara, M., Plavan, G., Faggio, C., 2019. Micro- (nano) plastics in freshwater ecosystems: Abundance, toxicological impact and quantification methodology. TrAC - Trends in Analytical Chemistry. https://doi.org/10.1016/j.trac.2018.10.025

Styles, D., Adams, P., Thelin, G., Vaneeckhaute, C., Chadwick, D., Withers, P.J.A., 2018a. Life Cycle Assessment of Biofertilizer Production and Use Compared with Conventional Liquid Digestate Management. Environmental Science & Technology 52, 7468–7476. https://doi.org/10.1021/ACS.EST.8B01619

Styles, D., Dominguez, E.M., Chadwick, D., 2016. Environmental balance of the UK biogas sector: An evaluation by consequential life cycle assessment. Science of the Total Environment 560–561, 241–253. https://doi.org/10.1016/j.scitotenv.2016.03.236

Styles, D., Gonzalez-Mejia, A., Moorby, J., Foskolos, A., Gibbons, J., Gonzalez-Mejia, A., Moorby, J., Foskolos, A., Gibbons, J., 2018b. Climate mitigation by dairy intensification depends on intensive use of spared grassland. Global Change Biology 24, 681–693. https://doi.org/10.1111/gcb.13868

Takata, M., Fukushima, K., Kino-Kimata, N., Nagao, N., Niwa, C., Toda, T., 2012. The effects of recycling loops in food waste management in Japan: Based on the environmental and economic evaluation of food recycling. Science of the Total Environment 432, 309–317. https://doi.org/10.1016/j.scitotenv.2012.05.049

Tambone, F., Orzi, V., D'Imporzano, G., Adani, F., 2017. Solid and liquid fractionation of digestate: Mass balance, chemical characterization, and agronomic and environmental value. Bioresource Technology 243, 1251–1256. https://doi.org/10.1016/J.BIORTECH.2017.07.130

Tonini, D., Albizzati, P.F., Astrup, T.F., 2018. Environmental impacts of food waste: Learnings and challenges from a case study on UK. Waste Management 76, 744–766. https://doi.org/10.1016/j.wasman.2018.03.032

Tonini, D., Hamelin, L., Astrup, T.F., 2016. Environmental implications of the use of agro-industrial residues for biorefineries: application of a deterministic model for indirect land-use changes. GCB Bioenergy 8, 690–706. https://doi.org/10.1111/gcbb.12290

Tsang, Y.F., Kumar, V., Samadar, P., Yang, Y., Lee, J., Ok, Y.S., Song, H., Kim, K.H., Kwon, E.E., Jeon, Y.J., 2019. Production of bioplastic through food waste valorization. Environment International 127, 625–644. https://doi.org/10.1016/j.envint.2019.03.076

UNFCCC, 2015. Adoption of the Paris agreement. Report No. FCCC/CP/2015/L.9/Rev.1. Paris.

Vink, E.T.H., Davies, S., 2015. Life Cycle Inventory and Impact Assessment Data for 2014 Ingeo® Polylactide Production. Industrial Biotechnology 11, 167–180. https://doi.org/10.1089/ind.2015.0003

Vink, E.T.H.H., Glassner, D.A., Kolstad, J.J., Wooley, R.J., O'Connor, R.P., 2007. The eco-profiles for current and near-future NatureWorks® polylactide (PLA) production. Industrial Biotechnology 3, 58–81. https://doi.org/10.1089/ind.2007.3.058

Weidema, B., 2001. Avoiding Co-Product Allocation in Life-Cycle Assessment. Journal of Industrial Ecology 4, 11–33. https://doi.org/10.1162/108819800300106366

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., Weidema, B., 2016. The ecoinvent database version 3 (part I): overview and methodology. International Journal of Life Cycle Assessment 21, 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

Xanthos, D., Walker, T.R., 2017. International policies to reduce plastic marine pollution from single-use plastics (plastic bags and microbeads): A review. Marine Pollution Bulletin 118, 17–26. https://doi.org/10.1016/J.MARPOLBUL.2017.02.048

Yates, M.R., Barlow, C.Y., 2013. Life cycle assessments of biodegradable, commercial biopolymers-A critical review. Resources Conservation and Recycling 78, 54–66. https://doi.org/10.1016/j.resconrec.2013.06.010

Zhao, X., Cornish, K., Vodovotz, Y., 2020. Narrowing the Gap for Bioplastic Use in Food Packaging: An Update. Environmental science & technology 54, 4712–4732. https://doi.org/10.1021/acs.est.9b03755

Zheng, J.J., Suh, S., 2019. Strategies to reduce the global carbon footprint of plastics. Nature Climate Change 9, 374-+. https://doi.org/10.1038/s41558-019-0459-z

5.7. Appendices for Chapter 5

5.7.1. Appendix 5.1 – Supplementary data file

Description: The attached excel file contains the key data and calculations utilised for the four scenarios within the study, as referenced to within **Chapter 5**. Full results from the scenarios are also provided within the workbook.

Location: The data set has been uploaded to Zenodo data repository at Bishop (2021); https://doi.org/10.5281/zenodo.5798676

File name: Chapter_5_supplementary_material_GB.xls

6. Summary and conclusions

The primary aim of this thesis was to advance the understanding of the comparative environmental performance of bioplastic production, use, and disposal against conventional petrochemical plastic production, use, and disposal. Another major aim was to facilitate a better understanding of the wider consequences of displacing petrochemical plastic with bioplastic, as well as evaluating how possible environmental hotspots for bioplastic production can be mitigated. This was achieved within the research of the previous chapters via an extensive critical review of the published literature, advanced value chain analysis, and innovative life cycle assessment (LCA). The purpose of this chapter is to summarise the key findings of the research undertaken in the context of the research objectives outlined in **Chapter 1.3**, relating to the main stakeholders. The key recommendations for each stakeholder are synthesised in **Table 6.1**. Furthermore, this chapter outlines the limitations of the thesis and makes recommendations for areas of future research.

6.1. Summary of research findings

6.1.1. Environmental performance comparison of bioplastics and petrochemical plastics: a review of LCA methodological decisions

6.1.1.1. Key findings pertaining to academic stakeholders

In **Chapter 2**, the state-of-the-art regarding how bioplastic LCA studies have previously been modelled was reviewed to identify methodological weaknesses (and strengths) of how the environmental performance of bioplastics has been benchmarked against petrochemical plastics. The literature review focussed on methodological aspects such as the range of impact categories covered and their pertinence to global environmental challenges, how plastic pollution is modelled in LCA (if at all), the comprehensiveness and replicability of each study's life cycle inventory (especially regarding the inclusion of additives), how land-use change emissions and biogenic carbon sequestration impacts are quantified within studies (if at all), end-of-life treatment modelling, uncertainty analysis, and the type of LCA models applied (i.e., attributional vs consequential). A significant outcome of the review was a set of nine distinct recommendations derived from the learnings of the study, providing guidance for comprehensive, comparative LCA evaluation of (bio)plastic environmental performance (Bishop et al., 2021a).

This research made a number of contributions to the state of the knowledge in regard to the comparative environmental performance of bioplastics. The critical analysis of the current literature drew focus to the largest research gaps to inform subsequent chapters of this thesis, as well as recognising how LCA models should be improved for future research (including for the succeeding thesis chapters). Best-practice recommendations were presented to overcome these methodological shortcomings and promote more accurate environmental assessment of bioplastics. This review facilitates the application of comprehensive and appropriately designed LCA studies, ensuring the most accurate possible environmental impacts associated with bioplastics via LCA.

6.1.1.2. Key findings pertaining to industry stakeholders

As seen in **Chapter 2**, failure to represent the complete system, through boundary truncation or process simplification, can result in studies misrepresenting the true comparative environmental efficiency of systems and products. Therefore, by providing guidelines towards a more comprehensive assessment of bioplastics, this review should support bioplastic producers to assess the comparative environmental efficiency of their bioplastic products more reliably against petrochemical plastic alternatives. Recommendations proposed in this chapter provide producers with clear guidance on how to avoid "greenwashing", or the perception of it, when promoting their products – a common issue with bioplastic marketing, whether intentional or not (Nandakumar et al., 2021).

6.1.1.3. Key findings pertaining to policy stakeholders

Transparent, non-biased LCA results provide a rigorous quantitative assessment of the environmental efficiency of products or systems, and constitute strong evidence to inform policy decisions (ISO, 2006). By providing best practise recommendations for methodological decisions, more rigorous environmental assessments of (bio)plastics can be undertaken, improving the accuracy of the LCAs for the attestation of the environmental performance of bioplastic materials, informing appropriate bioplastic policies. The proposed recommendations provide a benchmark against which policy makers can assess and validate LCA studies, helping to screen out less robust evidence from decision making processes.

6.1.2. Recycling of European plastic is a pathway for plastic debris in the ocean

6.1.2.1. Key findings pertaining to academic stakeholders

Within **Chapter 2** it was noted that realistic scenarios of end-of-life management of plastics should be considered within LCA studies (Bishop et al., 2021a). Data on the quantities of European plastic exported for recycling, estimated to be as much as 46% of post-consumer plastic waste collected for recycling (Wilson et al., 2015), had not previously been used to assess the true end-of-life fate of European plastics. As such, for **Chapter 3**, pathways of European polyethylene (PE) plastic waste to final end-of-life fates, including release into the ocean, were characterised and quantified in order to better understand the efficiency and net environmental effects of plastic recycling (Bishop et al., 2020). The results from **Chapter 3** estimated that in the year 2017, between 32,115 – 180,558 tonnes of exported PE ended up in the ocean from EU28, Norway, and Switzerland.

Chapter 3 added to the literature by exploring and quantifying, for the first time, this hitherto unidentified pathway of plastic leakage into oceans from plastic waste originating in Europe. It was shown that a significant percentage of PE exported from Europe is likely to end up as ocean litter, with considerable variation from specific countries of origin (**Figure 3.3**). The chapter provided reliable new insight into the previously undocumented flows of plastic into the ocean.

Comparing the environmental sustainability of petrochemical plastic recycling in a circular economy against a shift towards bioplastics requires accurate information on flows and fates of all plastic. Therefore, the accuracy of future LCA studies comparing waste management options could be improved by reflecting the true fates of waste collected for recycling. However, as noted in **Chapter 2** (Bishop et al., 2021a), the environmental effects of plastic debris loss into the environment are not currently represented within existing life cycle impact assessment methodology. Therefore, even if ocean debris fates associated with broader European recycling were reflected within country-specific recycling unit processes, it may not be possible to fully represent the emerging environmental impacts linked with this loss pathway within state-of-the-art LCA studies (see **Chapter 6.3.1**). However, the mass flows generated enable destination-country-specific distances and electricity burdens to be considered for the transport and processing of "recycled" plastic, which can be incorporated into existing LCA methodology (and databases) to improve accuracy.

6.1.2.2. Key findings pertaining to industry stakeholders

The results from **Chapter 3** provided new evidence on the efficacy and risks of current plastic waste management practices pertinent to the trade in plastic waste. The study placed a spotlight on the European recycling industry, highlighting the poor practices, via international trade, of plastic "recycling" in Europe. To ensure that the global recycling of plastic improves, and thus leakage into the environment decreases, improved reporting by relevant actors along all stages of the recycling chain is required. From this study, the suggested areas which need greatest attention for improved waste reporting are the points of re-export (transfer) of waste within Europe, the breakdown of PE waste into constituent polymer types, and most importantly, the unknown efficiencies of reprocessing facilities and fates of rejected plastic waste in the countries receiving the plastic waste. Improved data at these points in the plastic waste chain would improve the accuracy of the mass flow accounting required to underpin improved waste management practices and policy. This information is essential if plastic and packaging industry claims on environmental efficiency, based on headline recycling rates, are to carry any legitimacy.

6.1.2.3. Key findings pertaining to consumer stakeholders

Within waste management, recycling of plastic waste is typically the most environmentally preferred option owing to lower environmental burdens and reduced resource depletion (Lazarevic et al., 2010). However, if a significant fraction of material reported as recycled ultimately ends up as plastic debris in the oceans (or on land), as **Chapter 3** suggests, then the comparative life cycle environmental efficiency of "circular management" of petrochemical plastic will be reduced. Therefore, the study reveals to the consumer that recycling of plastic waste is not necessarily the "guilt reducing" option for dealing with plastic waste that it is often considered to be (Ma et al., 2019). The results clearly demonstrate to the public that avoidance and prevention of plastic waste remains by far the best approach to reduce environmental impact, in line with the plastic waste hierarchy (European Commission, 2014).

6.1.2.4. Key findings pertaining to policy stakeholders

The results suggested that exporting plastics out of Europe for recycling may not be in line with circular economy objectives (European Commission, 2018). Plastic recycling is supposed to close the technical materials loops. However, potential leakage from these technical material loops shifts recycling away from the fundamental principles of the circular economy (European Commission, 2018; McDonough et al., 2003; McDonough and Braungart, 2002). Without proper understanding of life cycle flows of plastic, reported rates of recycling (at source) are unreliable. As such, there may be implications for monitoring genuine progress towards recycling targets.

Further, **Chapter 3** observed that much of the PE exported from Europe was sent to lower-income countries (**Figure 3.2**), which have smaller budgets and lack the infrastructure to deal with waste streams compared with the higher-income countries from which the waste originated. Consequently, exportation of plastic for recycling outside of Europe was identified to be a major pathway for ocean debris, as indicated by the strong significant relationship between the percentage of PE each country exported out of Europe and the percentage of waste estimated to be lost into the oceans. To reduce the negative fates of exported plastic waste, European countries of origin should take greater responsibility for management of this waste in destination countries. Individual countries should build upon the Basel Convention Plastic Waste Amendments (United Nations, 2021), by restricting the export of plastic waste from Europe to countries which fail to meet high standards of waste management for the recycled material (and waste management of rejected material), and by investing in the receiving countries which are importing the waste, to assist in the improvement of their waste processing efficiencies.

6.1.3. Environmental performance of bioplastic packaging on fresh food produce: a consequential LCA

6.1.3.1. Key findings pertaining to academic stakeholders

Chapter 2 also highlighted the need for application of consequential LCA to represent the environmental outcomes of widespread substitution of petrochemical plastics with bioplastics. It was argued that adopting a forward-looking consequential LCA approach would be critical to more accurately capture the likely effects of displacing petrochemical plastics with bioplastics via specific scenarios of deployment (Bishop et al., 2021a). Thus, **Chapter 4** filled a research gap by investigating the environmental consequences of replacing petrochemical plastic food packaging with compostable bioplastic, within future-orientated scenarios, accounting for the potential diversion of (packaged) food waste streams owing to packaging biodegradability (Bishop et al., 2021b). Specifically, a consequential LCA, utilising the recommendations from **Chapter 2**, was undertaken to rigorously assess the environmental impact of displacing petrochemical plastic of fresh fruit and vegetables with polylactic acid (PLA), where eight end-of-life scenarios of bioplastic packaging were evaluated against a business-as-usual petrochemical packaging scenario, expanding LCA boundaries to include end-of-life impacts of fruit and vegetable food waste within a UK context. The study found that PLA production can have a higher impact compared with petrochemical plastic production across many impact categories, but diversion of PLA-packaged food waste to organic recycling (owing to the characteristics of the compostable bioplastic) can compensate for this, improving the overall environmental performance of bioplastic packaging scenarios.

Bioplastics have the potential to reduce some impact categories, but the study within **Chapter 4** provided a more rounded view of the environmental performance of PLA, indicating that a switch to PLA from conventional petrochemical plastics could be more environmentally damaging for several impact categories (**Figure 4.2**). The analysis therefore illustrated important trade-offs associated with a transition to PLA food packaging, including notable increases in terrestrial and freshwater acidification, terrestrial and marine eutrophication, ozone depletion, as well as water scarcity. These relate to critical planetary boundaries (Rockström et al., 2009; Steffen et al., 2015) and future sustainability challenges (Mekonnen and Hoekstra, 2016). Thus, as recommended in **Chapter 2**, a comprehensive impact assessment methodology should be adopted to capture the priority environmental challenges.

The findings of **Chapter 4** showed that food waste end-of-life contributed a considerable share of environmental impact across all 16 midpoint impact categories studied. Bioplastic packaging LCA system boundaries are rarely expanded to include possible food waste diversion to biowaste treatment (Kakadellis and Harris, 2020), despite bioplastics potentially facilitating such food waste diversion that is shown in this study to be critical to the overall environmental efficiency of bioplastic use (**Figure 4.2**). Similarly, although the analysis identified that bioplastic production and food waste end-of-life were the major environmental burden contributors (**Figure 4.2**), it was found that, despite plastic packaging only representing 5% fresh weight of the fresh fruit and vegetable food waste, it represented 25% of the dry weight waste due to the high water content of fruit and vegetables. Plastic packaging therefore constitutes a major material flow within food waste streams that strongly influences the energy recovery potential, as well as other end-of-life cycles. As such, any study where (bio)plastic packaging and food waste are intrinsically connected should expand their LCA boundaries to account for both systems, including any diversion of plastic packaging and food waste.

6.1.3.2. Key findings pertaining to industry stakeholders

It is important that the bioplastic industry ensures, via research and development and marketing, that their compostable bioplastic packaging can be placed in organic recycling streams. Further, it has been noted that industrial composting and anaerobic digestion facilities are reluctant to accept bioplastics (Kakadellis and Harris, 2020). This may be largely due to the challenge of distinguishing the biodegradable plastics from the non-biodegradable plastics during screening. However, it has been suggested that optical sorting systems are capable of identifying and separating the bio-based and petrochemical plastics (Kakadellis and Harris, 2020). Therefore, although this chapter highlights the benefits of developing biopolymers fit for anaerobic digestion end-of-life management, waste infrastructure and management businesses may also need to adapt to the evolving composition of waste.

6.1.3.3. Key findings pertaining to consumer stakeholders

Chapter 4 highlighted the importance of consumer behaviour when introducing compostable bioplastic packaging. When the petrochemical plastic packaging material was replaced by the PLA packaging, without any sensitivity analyses or increase in waste diversion, only two of the sixteen impact categories investigated witnessed reductions in environmental impacts. However, by increasing the level of bioplastic and food waste that was placed in a dedicated food waste bin, half (eight) of the environmental impact categories assessed observed environmental savings if 100% of the food waste was diverted to anaerobic digestion (or eleven impact categories if 100% was diverted to insect feed) (**Figure 4.2**). Even when energy efficiency and decentralisation were considered to lower the burdens of bioplastic production, consumer separation of the PLA and food waste into the food waste stream was crucial. Without consumer separation the petrochemical packaging was, overall, more environmentally efficient. However, through extra consumer separation, the bioplastic scenarios were able to outperform the petrochemical plastic packaging scenario for the vast majority of impact categories (**Figure 4.4**).

Ultimately, the more food waste and compostable bioplastic that can be sent to organic recycling, the better it is for the environment (**Chapter 4**) – assuming that the waste cannot be avoided in the first instance. The compostable nature of the bioplastic allows for this waste diversion, but will require support from the consumers to implement this at the source of the waste generation. Consumers will need to be educated to identify bioplastic packaging and recognise how to deal with it (and its contents) appropriately, in order to leverage any environmental savings.

6.1.3.4. Key findings pertaining to policy stakeholders

The results demonstrated the need for policy support to encourage the preferred waste management hierarchy for compostable bioplastic packaging and food waste. For the present, anaerobic digestion was deemed to be the most favourable waste management option, but if the waste can be used for animal feed, e.g., via insects in the future, that would lead to the best environmental outcomes (after avoidance). However, there is a need for regulatory change to allow this to occur, after extensive research where it can be ensured that no significant disease-transmission risks within and across species arises from such recycling. Following the best end-of-life management of insect feed, when scaled up to the total annual fresh fruit and vegetable waste and associated packaging of the UK, annual greenhouse gas (GHG) emissions savings could amount to 754,742,657 kg CO_2 eq. (0.75 Mt CO_2 eq.).

Chapter 4 also indicated the potential for reducing environmental emissions from bioplastics through improved energy efficiency in PLA production. Further, the findings from **Chapter 4** suggested that, in order to realise greater environmental savings, bioplastics should be produced within developed countries where demand is initially greatest, owing to more efficient feedstock cultivation and processing technologies, and cleaner energy supplies (Victoria et al., 2020). This may require "on-shoring" of plastic production from low-cost developing and transitioning countries.

6.1.4. Land-use change and valorisation of feedstock side-streams determine the climate change mitigation potential of bioplastics

6.1.4.1. Key findings pertaining to academic stakeholders

Chapter 4 found that large inputs required for bioplastic feedstock production, such as fertiliser and land-use, can negate the sought environmental efficiency of the bio-based materials (Bishop et al., 2021b). Despite often being presented as a more sustainable material than petrochemical plastic (Bishop et al., 2021a), bioplastic production emissions can be greater than the potential credits earned through avoided petrochemical plastic production and end-of-life management, and/or long-term carbon sequestration. **Chapter 5** quantified the environmental envelopes of bioplastic production from multiple feedstocks in relation to net-zero GHG targets, using a consequential LCA approach (Bishop et al., in review). Lignocellulosic biomass from forestry, maize biomass, food waste digestate, and food waste were evaluated as indicative feedstocks for potential bioplastic production. Upstream and end-of-life emissions for these three main feedstock types equated to GHG balances ranging from -16.3 to +23.5, 0.3 to 1.0, 1.0 to 4.8, and -0.1 to +0.4 kg CO_2 eq. per kg bioplastic, respectively. The chapter found that land-use change (specifically indirect land-use change) could have a considerable negative impact on the embodied burdens of maize-based plastic, but a positive impact, via terrestrial carbon sequestration within new commercial forest plantations, for lignocellulosic-based plastic. It was also found that appropriate use of residues and side-streams is critical to the environmental performance of bioplastics.

6.1.4.2. Key findings pertaining to industry stakeholders

Chapter 5 shows that not all compostable bioplastic feedstocks are environmentally equal, and that feedstocks come with highly variable upstream burdens. The chapter highlighted that embodied burdens of feedstocks are highly connected with land-use change and how residues are utilised (**Figure 5.6**). The results emphasised the need to consider the entire bioplastic value chain, in order to generate bioplastics compatible with climate neutrality. **Chapter 5** demonstrates that there is high potential to achieve net-zero GHG emissions over the whole life cycle of compostable bioplastic, but that careful consideration is required for feedstock acquisition to reduce the potential major environmental hotspots, so that one environmentally poor material is not simply replaced with another.

It was suggested that efficient utilisation of residues may require decentralisation of production, supported by the conclusions in **Chapter 4** (see **Chapter 6.1.3.4**). Decentralisation of production was crucial to reduce the large environmental hotspot of feedstock transport, especially for feedstocks which have a high water content or are required in large quantities to produce bioplastic. Though in reality, economics may constrain such transport anyway. The study also highlighted that producers may also require the implementation of biorefinery and circular economy concepts into their business models to valorise the (sometimes high quantity of) side-streams.

6.1.4.3. Key findings pertaining to policy stakeholders

Regulation and incentives will be required to ensure that the bioplastic industry development supports net-zero GHG emissions targets. This may involve safeguarding against environmentally poor feedstock acquisition. For example, results from **Chapter 5** suggest that the use of lignocellulose for bioplastic production should only be considered where demand for bioheat (e.g., to decarbonise industrial processes such as cement production) can first be met, and deforestation avoided. Policy should also be developed to reduce risks of indirect land-use change via crop displacement and underutilisation of residue waste, the identified largest contributors of the upstream and end-of-life emissions from the studied feedstocks.

Table 6.1 – Key recommendations from the chapters of the present thesis, relating to the main stakeholders

	Stakeholders			
Thesis Chapter	**Academic**	**Industry**	**Consumer**	**Policy**
Chapter 2	• A set of nine distinct recommendations derived from the learnings of the study, should be adopted for comprehensive, comparative LCA evaluation of (bio)plastic environmental performance	• Recommendations proposed in this chapter should be followed to avoid "greenwashing", or the perception of it, when promoting bioplastic products	-	• Policy makers should use the proposed recommendations to benchmark and validate LCA studies from various stakeholders, helping to screen out less robust evidence from decision making processes
Chapter 3	• The accuracy of future LCA studies comparing waste management options could be improved by reflecting the true fates of waste collected for recycling	• There should be improved waste reporting at the points of re-export (transfer) of waste within Europe, the breakdown of PE waste into constituent polymer types, and most importantly, the unknown efficiencies of reprocessing facilities and fates of rejected plastic waste in the countries receiving the plastic waste	• Avoidance and prevention of plastic waste remains by far the best approach to reduce environmental impact	• Individual countries should build upon the Basel Convention Plastic Waste Amendments, by restricting the export of plastic waste from Europe to countries which fail to meet high efficiency waste management practices for the recycled material and by investing in the receiving countries which are importing the waste
Chapter 4	• A comprehensive impact assessment methodology should be adopted to capture the priority environmental challenges • Any study where (bio)plastic packaging and food waste are intrinsically connected should expand their LCA boundaries to account for both systems, including any diversion of plastic packaging/food waste	• It is important that the bioplastic industry ensures, via research and development and marketing, that their compostable bioplastic packaging can be placed in organic recycling streams • Waste infrastructure and management businesses need to adapt to the evolving composition of waste	• Consumers will need to be educated to identify bioplastic packaging and recognise how to deal with it (and its contents) appropriately (e.g., placing into a dedicated food waste bin if appropriate)	• There is a need for policy support to encourage the preferred waste management hierarchy for compostable bioplastic packaging and food waste • Regulatory change would be required to allow some organic waste to be diverted to animal feed, e.g., via insect feed • Bioplastics should be produced within developed countries where demand is initially greatest
Chapter 5	• Indirect land-use change should be accounted for within bioplastic studies (where relevant)	• Careful consideration is required for feedstock acquisition to reduce potential major environmental hotspots • Producers should consider decentralisation of production • Side-streams could be valorised via biorefinery and circular economy concepts	-	• There should be safeguarding against environmentally poor feedstock acquisition • Policy should be developed to minimise the risks of negative land-use change and underutilisation of residue waste occurring

6.2. Limitations of the research

Chapter specific limitations have been acknowledged within those chapters. However, limitations in relation to scope and context of the thesis are described below.

6.2.1. Scope limitations

This thesis has attempted to evaluate the comparative environmental performance of bioplastic production, use, and disposal against conventional petrochemical plastic production, use, and disposal. Though this has been accomplished to a certain degree, the broad nature of bioplastic materials on the (future) market means that the analysis undertaken can only offer a snapshot of the environmental efficiency of some bioplastic value chains, and how they are benchmarked against petrochemical plastic value chains. The themes that have been explored within the previous chapters were the research gaps that were considered to be the most important, relevant, and feasible to explore. This thesis may contribute to the further understanding of the environmental impacts of plastic recycling value chains and the environmental impact of bioplastics, but it is not a comprehensive exploration of all possible bioplastic materials or every potential aspect of future bioplastic production (and nor does it attempt to be). Each chapter provides some valuable insight into related challenges and highlights opportunities for further development of i) sustainable bioplastic value chain development, and ii) rigorous assessment of prospective bioplastic value chains.

6.2.2. Context limitations

The prospective nature and the brief encounters with the bioplastic materials within this thesis make it difficult to draw a single definitive conclusion with regards to the environmental efficiency of bioplastics. Rather, the studies have provided rigorous sensitivity and uncertainty analyses to provide a range of potential outcomes for bioplastic production, use and end-of-life – reflecting the multiple future decisions made within the value chain, as well as consumer behavioural choices. This is both a criticism and a strength, where high impact was gained from modelling potential scenarios of bioplastic life, but all results are provided with a high degree of uncertainty, where it is impossible to have a full exploration for all the potential impacts made from future decisions.

The data underpinning the LCA models used within the previous chapters all came from reliable peer-reviewed sources. However, due to the novel nature of bioplastics, some data on certain aspects of the life cycle were limited, e.g., (future) large-scale bioplastic production emissions. As such, some data had to be adapted to fit within the future, consequential frameworks, adding uncertainty to the results. Similarly, this thesis focussed on consequential LCA to explore the environmental impacts of bioplastics rather than using an attributional approach. A consequential approach was applied to **Chapter 4** and **5** because it was deemed to be a more suitable method, which allowed the chapters to describe how environmentally relevant flows will change in response to possible decisions with regards to the introduction of bioplastics (Bishop et al., 2021a). This methodology was chosen over an attributional LCA approach which would aim to describe the environmentally relevant physical flows to and from a life cycle and its subsystems, using average data and allocation of environmental burdens (Ekvall, 2019). The consequential approach is argued to be a more appropriate choice for exploring product systems, especially when considering consistent socially responsible decision making (Weidema et al., 2018). As stated by Ekvall (2019), LCA should ideally generate results that are as comprehensive, accurate, and precise as possible. And whilst an attributional LCA might be more precise owing to the use of average data for existing well-defined system processes (Ekvall, 2019; European Commission, 2010), in general consequential LCAs are considered to be the more accurate methodology, avoiding incorrect assumptions and bias, where accuracy is related to the absence of systematic errors (Ekvall, 2019; European Commission, 2010) (**Figure 6.1**). For this thesis, a high accuracy was considered to be the more important factor, because even very precise LCA results could guide decisions in the wrong direction. Without following a consequential approach for the **Chapters 4** and **5**, the LCA results may not have been an appropriate representation of the consequences of producing, using, and disposing the explored bioplastic products.

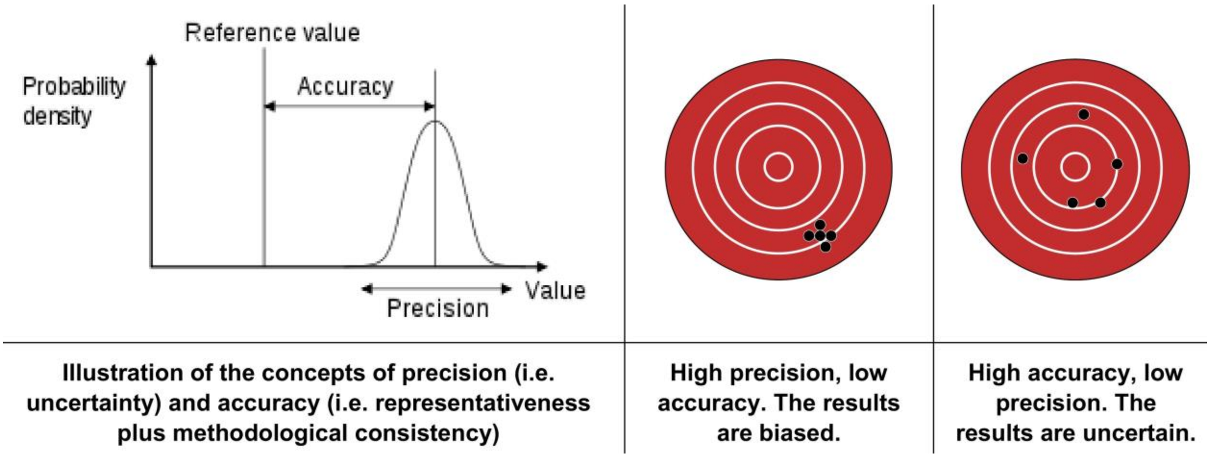

Figure 6.1 – Precision and accuracy linked to attributional (high precision) and consequential (high accuracy) LCA approaches (European Commission, 2010)

As previously mentioned, the actual effects of a decision on global environmental impacts are in most cases highly uncertain. It is impossible to know how close the consequential LCA results are to reflecting actual future consequences. For this reason, the consequential LCA results presented within this thesis from **Chapters 4** and **5** are framed as potential impacts from scenarios pertaining to *some* of the most logical future decisions and consequences, but do not represent every possible consequence that could occur to the product system as a result to the change in demand for the functional unit. This relates to a further limitation of the thesis, where the interpretation of the consequential LCA results produced are difficult to convey to the stakeholders. As consequential models only include activities that change as a result of a decision, the activities involved with the life cycle inventory may not all be the activities that one would intuitively think, and thus may appear counter-intuitive until the context is communicated, and the models are investigated in more detail.

6.3. Potential future research

In order to improve on the understanding developed during this PhD study, specific future research has been acknowledged within the individual chapters (synthesised within **Figure 6.2**). More generally, future research could make specific contributions to the areas outlined below.

6.3.1. Improving the LCA of bioplastics

The main driver towards bioplastics is mandatory bans being placed on single-use petrochemical plastics. These bans are ultimately arising due to the pervasive nature of plastic at the end-of-life stage. If biodegradable bioplastics are being substituted in to reduce the plastic pollution within the environment, this benefit should be considered within a holistic environmental assessment of the material. However, as discussed within **Chapter 2** and **Chapter 6.1.2.1**, the full impacts of plastic release into the environment are not realised within LCA (Bishop et al., 2021a). Without this inclusion, the true benefits of biodegradable bioplastics cannot be captured. Hypothetically, if weighting were to be undertaken, "plastic persistence in the environment" could arguably be one of the most important impact categories for (bio)plastics. Therefore, as a priority, there is a need to identify how plastic littering effects can be integrated into existing impact categories or represented as a new impact category.

6.3.2. Evaluating the sustainability of bioplastics

This study only observed a few possible materials and potential scenarios for potential bioplastic deployment. Therefore, future research should continue to explore the growing collective of bioplastic materials and feedstocks (Bishop et al., 2021a). In particular, this thesis failed to include any 3rd generation feedstocks, e.g., algae, which may provide further insight into potential benefits of bioplastics (Devadas et al., 2021). This thesis has shown that lignocellulosic biomass may be one of the more promising feedstocks for bioplastic production (Bishop et al., in review). Further research should expand on this research to develop a complex model which needs to be long-term (e.g., 100 years into the future) to capture the cascading use of wood, and therefore needs to be dynamic, in so far as future processing burdens and substitution credits require consideration of (decarbonising) energy and material value chains (Forster et al., 2021). The effects from different locations should be explored, and scenarios should continue to be developed to encompass more potential decisions through a consequential approach. Due to the uncertainty of future context and decisions, establishing a suit of consequential and dynamic models (Yang and Heijungs, 2018) may represent the most reliable way to approximate a "true" result. The process of developing consequential LCAs elucidates linkages that may otherwise be missed, and is thus important for informing decision making around bioplastic deployment.

In addition to the environmental assessments required via consequential LCA, a full life cycle sustainability assessment should be undertaken to establish the wider sustainability potential of bioplastics. The concept of life cycle sustainability assessment involves an integrated analysis for each of the three pillars of sustainability: environment, economy, and social aspects (Finkbeiner et al., 2010; Kloepffer, 2008). To achieve this, an environmental LCA should be integrated with social LCA and life cycle costing. As it stands, very few social LCAs and life cycle costings of bioplastics exist in the literature (Spierling et al., 2018). However, the concept of the bioeconomy, into which bioplastics fit, aims for a holistic transformation of economy and society (Spierling et al., 2018). Thus, the benefits of bioplastics do not only mean reducing emissions and substituting fossil resources, but also creating benefits for different stakeholders such as workers and consumers. Sustainability is a key factor for a successful transformation, and to develop the potential advantages of bioplastics in a sustainable way, all three pillars of sustainability have to be balanced. As such, life cycle sustainability assessments of bioplastics should be explored in future research.

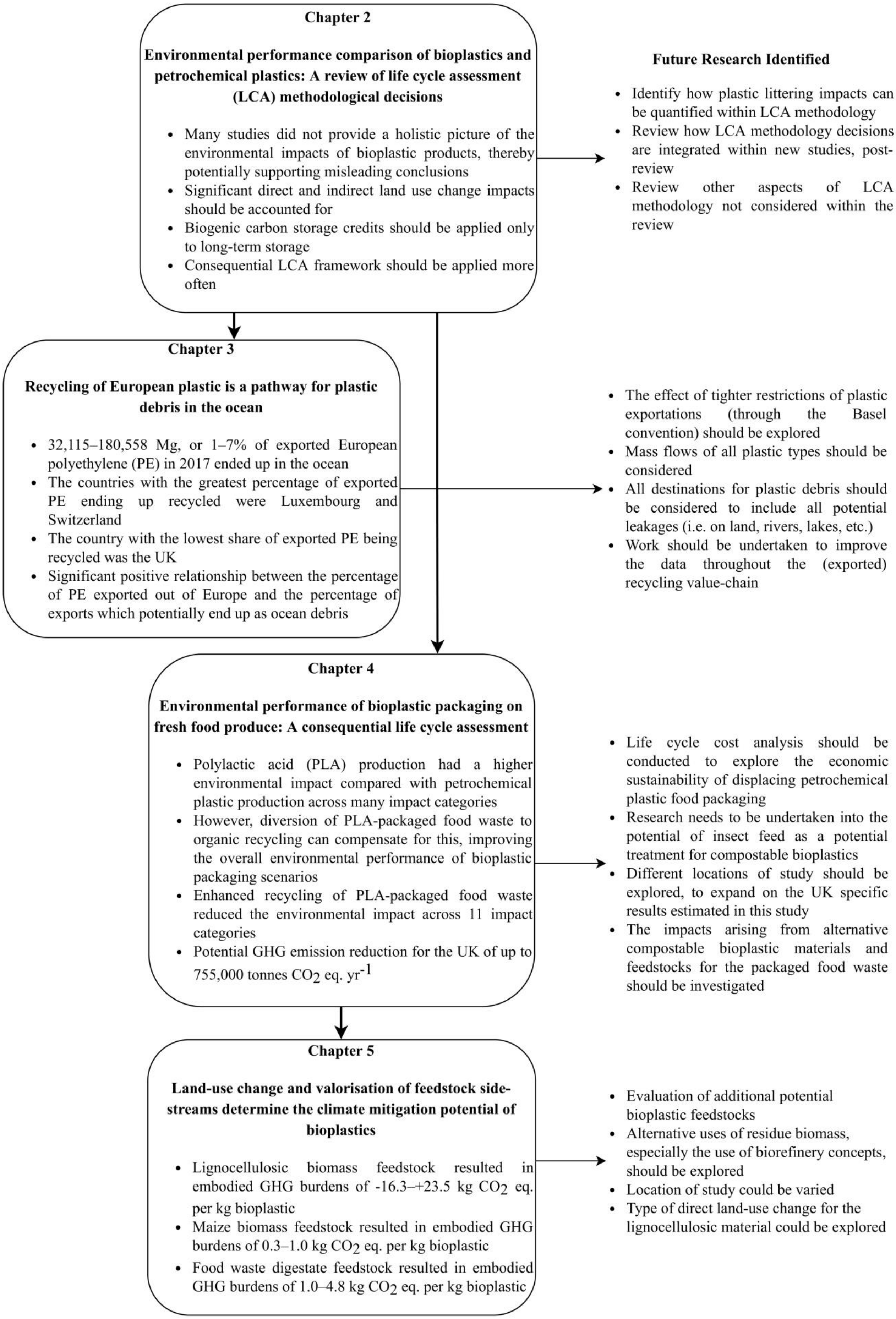

Figure 6.2 – Key results from the research chapters of the present PhD thesis, within the boxes. The chapter-specific future research recommendations are presented on the right

6.4. Prospects for environmentally efficacious bioplastics

Overall, this PhD dissertation showed that bioplastics can play an important role in reducing global greenhouse gas emissions. However, simple substitution of petrochemical plastics with bioplastics will not drive environmental savings unless consumer behaviour and wider value chain logistics also change. Due to the international bans of many single-use petrochemical plastic products, the uptake of bioplastic products is accelerating fast to replace them in the market. It is highly possible that this has occurred without due consideration of the true environmental consequences. Perhaps this early transition could become a learning process, wherein the impacts from the first adopters can be studied, potential solutions can be recommended, and then the processes and regulations of the systems in place can be revised and improved. Indeed, perhaps we need to be willing to make "mistakes" and learn on the go if we are to achieve any significant transformation in line with the current climate emergency. Nevertheless, the uptake of bioplastics represents a great opportunity to design production pathways compatible with net-zero GHG emission targets and waste elimination, in line with a fully circular economy (European Commission, 2018).

If the recommendations derived from this thesis (**Table 6.1**) can be followed, more holistic and rigorous assessments of bioplastics can be performed, providing guidance for more environmentally sustainable bioplastic value chains to be designed. If the academic, industry, consumer, and policy stakeholders can work together, bioplastics could be produced, used, and disposed of in a manner which can be more environmentally efficient than the petrochemical alternatives. However, all stakeholders will need to collaborate and commit fully in order to realise an environmentally successful transition to bioplastics.

6.5. References

Bishop, G., Styles, D., Lens, P.N.L., 2021a. Environmental performance comparison of bioplastics and petrochemical plastics: A review of life cycle assessment (LCA) methodological decisions. Resour. Conserv. Recycl. 168, 105451. https://doi.org/10.1016/j.resconrec.2021.105451

Bishop, G., Styles, D., Lens, P.N.L., 2021b. Environmental performance of bioplastic packaging on fresh food produce: A consequential life cycle assessment. J. Clean. Prod. 317, 128377. https://doi.org/10.1016/J.JCLEPRO.2021.128377

Bishop, G., Styles, D., Lens, P.N.L., 2020. Recycling of European plastic is a pathway for plastic debris in the ocean. Environ. Int. 142, 105893. https://doi.org/10.1016/j.envint.2020.105893

Bishop, G., Styles, D., Lens, P.N.L., in review. Land-use change and valorisation of feedstock side-streams determine the climate change mitigation potential of bioplastics. Resour. Conserv. Recycl.

Devadas, V.V., Khoo, K.S., Chia, W.Y., Chew, K.W., Munawaroh, H.S.H., Lam, M.K., Lim, J.W., Ho, Y.C., Lee, K.T., Show, P.L., 2021. Algae biopolymer towards sustainable circular economy. Bioresour. Technol. 325, 124702. https://doi.org/10.1016/J.BIORTECH.2021.124702

Ekvall, T., 2019. Attributional and consequential life cycle assessment. Sustain. Assess. 21st century 395, 1–22. https://doi.org/10.5772/INTECHOPEN.89202

European Commission, 2018. A European strategy for plastics in a circular economy. COM/2018/028 final. Brussels.

European Commission, 2014. Towards a circular economy: A zero waste programme for Europe. Brussels.

European Commission, 2010. International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance. Luxembourg.

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. Sustain. 2010, Vol. 2, Pages 3309-3322 2, 3309–3322. https://doi.org/10.3390/SU2103309

Forster, E.J., Healey, J.R., Dymond, C., Styles, D., 2021. Commercial afforestation can deliver effective climate change mitigation under multiple decarbonisation pathways. Nat. Commun. 2021 121 12, 1–12. https://doi.org/10.1038/s41467-021-24084-x

ISO, 2006. ISO 14044: Environmental management -- Life cycle assessment -- Requirements and guidelines. Geneva.

Kakadellis, S., Harris, Z.M., 2020. Don't scrap the waste: The need for broader system boundaries in bioplastic food packaging life-cycle assessment – A critical review. J. Clean. Prod. https://doi.org/10.1016/j.jclepro.2020.122831

Kloepffer, W., 2008. State-of-the-art life cycle sustainability assessment of products. Int J LCA 13, 89–95. https://doi.org/10.1065/lca2008.02.376

Lazarevic, D., Buclet, N., Brandt, N., 2010. The Influence of the Waste Hierarchy in Shaping European Waste Management: The Case of Plastic Waste. Reg. Dev. Dialogue 31, 124–148.

Ma, B., Li, X., Jiang, Z., Jiang, J., 2019. Recycle more, waste more? When recycling efforts increase resource consumption. J. Clean. Prod. 206, 870–877. https://doi.org/10.1016/J.JCLEPRO.2018.09.063

McDonough, W., Braungart, M., 2002. Cradle to cradle : remaking the way we make things. North Point Press.

McDonough, W., Braungart, M., Anastas, P.T., Zimmerman, J.B., 2003. Peer reviewed: applying the principles of green engineering to cradle-to-cradle design. Environ. Sci. Technol. 37, 434A-441A. https://doi.org/10.1021/es0326322

Mekonnen, M.M., Hoekstra, A.Y., 2016. Four billion people facing severe water scarcity. Sci. Adv. 2, e1500323. https://doi.org/10.1126/sciadv.1500323

Nandakumar, A., Chuah, J.-A., Sudesh, K., 2021. Bioplastics: A boon or bane? Renew. Sustain. Energy Rev. 147, 111237. https://doi.org/10.1016/j.rser.2021.111237

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., De Wit, C.A., Hughes, T., Van Der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature. https://doi.org/10.1038/461472a

Spierling, S., Knupffer, E., Behnsen, H., Mudersbach, M., Krieg, H., Springer, S., Albrecht, S., Herrmann, C., Endres, H.J., 2018. Bio-based plastics - A review of environmental, social and economic impact assessments. J. Clean. Prod. 185, 476–491. https://doi.org/10.1016/j.jclepro.2018.03.014

Steffen, W., Richardson, K., Rockstrom, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B., Sorlin, S., 2015. Planetary boundaries: Guiding human development on a changing planet. Science (80-.). 347, 1259855–1259855. https://doi.org/10.1126/science.1259855

United Nations, 2021. Basel Convention Plastic Waste Amendments [WWW Document]. Amend. Overv. URL http://www.basel.int/Implementation/Plasticwaste/Amendments/Overview/tabid/8426/Default.aspx (accessed 10.19.21).

Victoria, M., Zhu, K., Brown, T., Andresen, G.B., Greiner, M., 2020. Early decarbonisation of the European energy system pays off. Nat. Commun. 2020 111 11, 1–9. https://doi.org/10.1038/s41467-020-20015-4

Weidema, B.P., Pizzol, M., Schmidt, J., Thoma, G., 2018. Attributional or consequential Life Cycle Assessment: A matter of social responsibility. J. Clean. Prod. 174, 305–314. https://doi.org/10.1016/j.jclepro.2017.10.340

Wilson, D.C., Rodic, L., Modak, P., Soos, R., Rogero, A., Velis, C., Lyer, M., Simonett, O., 2015. Global Waste Management Outlook. https://doi.org/10.1177/0734242X15616055

Yang, Y., Heijungs, R., 2018. On the use of different models for consequential life cycle assessment. Int. J. Life Cycle Assess. 23, 751–758. https://doi.org/10.1007/s11367-017-1337-4