SCHOOL OF
ECONOMICS AND
MANAGEMENT

LUND UNIVERSITY

Master's Programme in Innovation and Global Sustainable Development

Closing the Plastic Tap — Global Plastic Waste and the Circular Economy

A Multi-Regional Hybrid Input-Output Analysis of Plastic Waste Footprints

by

Judith Baeta Humanes

ju5328ba-s@student.lu.se

Abstract

Plastic pollution is a cross-national environmental and societal challenge that needs to be addressed from the point of view of global supply chains. The circular economy (CE) has emerged as an alternative paradigm to the traditional "take-make-waste" models of production and consumption to create a closed-loop system so that plastic is trapped in the economy and not in the environment. This research calculates the plastic waste footprint of 48 countries and regions and 164 products by developing a multi-regional environmentally extended input-output model that traces plastic waste generated directly and indirectly by economic activities throughout the global supply chain. A particularity of this study is the implementation of EXIOBASE 3 hybrid-units input-output tables. The assessment of plastic waste footprints provides an opportunity to identify hotspots both upstream and downstream in the supply chain to advance in the CE for plastics. The results show that China holds the largest plastic waste footprint which is equal to 36 Mt of plastic waste. However, when per capita levels are considered, the consumption in high-income countries has a relatively major role in the generation of plastic waste. In addition, this study confirms that, globally, landfilling is the most common treatment method for plastic waste which is detrimental for the CE. The results of this work also show a reallocation of plastic waste from primary and production sectors towards end of value chain sectors such as services and trade when the indirect effects given plastic-intensive processes upstream in the supply chain that generate waste are accounted for.

Key words: Plastic waste; waste footprint: circular economy; multi-regional input-output; hybrid input-output

EKHS34
1st Year Master Thesis (15 credits ECTS)
June 2021
Supervisor: Josef Taalbi
Examiner: Abel Gwaindepi
Word Count: 14.854

Acknowledgements

The topic of this thesis is one very close to my heart and it started as an ambitious attempt to better understand the complexities behind ocean plastics. I am truly grateful to my supervisor Josef Taalbi for his continuous guidance and invaluable feedback. I would also like to thank Nicolai Baumert for kindly sharing his advice and expertise in multi-regional input-output analysis.

I would like to acknowledge my peers and friends in Lund (and around the world) for their support and words of encouragement, I feel extremely lucky to have such a support network around me. Finally, I would like to thank my family for encouraging me to dream big and for always having my back, even in the distance.

Table of Contents

List of Tables

List of Figures

List of Abbreviations

CE	Circular Economy
EEIO	Environmentally-Extended Input-Output
GDP	Gross Domestic Product
IO	Input-Output
IOT	Input-Output Table
LCA	Life-Cycle Assessment
Mt	Mega tonnes
Meuro	Million Euros
MR-EEIO	Multi-Regional Environmentally-extended Input-Output
MR-HIOTs	Multi-Regional Hybrid Input-Output Tables
MR-IO	Multi-Regional Input-Output
PWF	Plastic Waste Footprint
RoW	Rest-of-the-World
TJ	Terajoules
WIO	Waste Input-Output
WSU	Waste Supply-Use

1 Introduction

Every year 8 million tonnes of plastic end up in the ocean (UN Environment, 2020) of which 80% comes from land sources and 20% from marine sources, primarily commercial fishing (Li, Tse & Fok, 2016). Plastic waste that accumulates in landfills or the natural environment can find its way into waterways to end up in the ocean. Plastic has become so ubiquitous in the natural environment that some scientists suggest it as a geological indicator of the Anthropocene era (Jones, 2021; Zalasiewicz et al., 2016).

From an ecological perspective, alongside overfishing and ocean acidification, marine pollution is one of the main stressors of marine ecosystems. Oceans are one of the most important regulators of Earth life systems by regulating the climate and other planetary boundaries (Galaz et al., 2014; Rockström et al., 2009; Steffen et al., 2015). The livelihoods of more than 3 million people in the world depend directly on the health of marine and coastal ecosystems and it is estimated that the contribution to the worldwide economy is approximately 5% of global GDP (Patil et al., 2016). The United Nations Sustainable Development Goal 14 is about "Life Under Water" and it is concerned about conservation and sustainability of the oceans (UN, 2015).

Plastic has been the single fastest growing material in the last decades with an increase in global production from 1.7 million tonnes in 1950 to 368 million tonnes in 2019 (PlasticsEurope, 2020). Today, plastics are used across many industries such as consumer goods, electronics, transports, construction and textiles as it is a durable, versatile and affordable material with many applications. However, most of this growth has been driven by the shift to single-use containers (Geyer, Jambeck & Law, 2017). Packaging has become the largest end market for plastic which has extremely short material life-cycles before its discarded as waste (EEA, 2021; Hahladakis & Iacovidou, 2018; UNEP, 2018). Most plastics are derived from fossil feedstocks which makes them non-degradable and the complexity of the material composition of certain plastics poses challenges for their end-of-life management (OECD, 2008; Hahladakis & Iacovidou, 2018).

Globally, only 9% of total plastic production is recycled (Geyer, Jambeck & Law, 2017) and every year millions of tonnes of plastic waste collected for recycling in high-income countries is systematically exported to developing countries where, typically, waste management infrastructures lack and legislation is weaker (Browning, Beymer-Farris & Seay, 2021; Jambeck et al., 2018; Mwanza & Mbohwa, 2017). In 2017, 98% of plastic waste for treatment was accumulated in Asia (Liang et al., 2021). The mismanagement of waste significantly increases the risk of leakage plastics into the ocean (Jambeck et al. 2015). However, the disproportionate amount of plastic discharged into the ocean in developing countries (Lebreton et al., 2017; Meijer et al., 2021; Schmidt et al., 2017) is only a symptom at the end of the chain, and the roots of the problem of plastic pollution start much earlier.

Plastic pollution is a cross-national environmental challenge that raises issues of social justice as it affects coastal communities in developing countries disproportionately that needs to be addressed from the point of view of global supply chains (Geng, Sarkis & Bleischwitz, 2019; Rodić & Wilson, 2017; UNEP, 2019a; Watson, 2020). The circular economy (CE) presents itself as a potential solution to alleviate the issue, by promoting a systemic shift in paradigm from resource-extractive linear production and consumption systems to circular ones (MacArthur Foundation, 2016; Hahladakis, Iacovidou & Gerassimidou, 2020; Mah, 2021) so that plastic is trapped in the economy and not in the environment.

Developing quantitative assessments of the "circularity" of economies, production systems and value chains is crucial to reduce plastic waste generation (MacArthur Foundation, 2015). Life-Cycle Analysis (LCA) is the state-of-art methodology to analyse material flows and evaluate potential environmental impacts of a product throughout its life-cycle (ISO14040:2006) and it has been widely applied to assess impacts of plastics (Antelava et al., 2019; Lazarevic et al., 2010; Schwarz et al., 2021; UNEP 2021; Wäger & Hischier, 2015). However, the process-driven approach of LCA adds a system boundary that hinders the ability to assess complex supply chains (Genovese et al., 2017).

In today's global economy, international trade flows play a major role and the effects of a country or region's economic activity will extend beyond their borders. Multi-regional environmentally-extended input-output (MR-EEIO) models study environmental pressures *embodied* in international trade, by capturing the direct and indirect effects of economic activities along the whole supply chain which are ultimately driven by final demand. This consumption-driven approach results in the calculation of the so-called environmental *footprints* (Tukker & Dietzenbacher, 2013). Footprints have been calculated for several impacts such as carbon emissions (Baumert et al., 2019; Davis & Caldeira, 2010; Wiebe et al., 2012), water (Hoekstra & Mekonnen, 2012), biodiversity threats (Lenzen et al., 2012) and waste (Fry et al., 2015; Tisserant et al., 2017).

1.1 Research Aim and Questions

The aim of this thesis is to examine the plastic waste footprint (PWF) in the global economy. For this purpose, a MR-EEIO analysis is developed which allows the author to calculate plastic waste *embodied* in international trade. Traditional territorial waste accounting methods are based on the calculation of waste directly generated by production activities that take place in a country or region. Conversely, the consumer-driven approach of MR-EEIO accounts for plastic waste generated both directly and indirectly by production activities throughout the whole supply chain, including intermediate consumption and trade, domestically or by a trading party, which is ultimately allocated to the country where the final consumption takes place (Towa, Zeller & Achten, 2020). Therefore, the assessment of the plastic waste footprint provides an opportunity to identify hotspots in the global supply chain that would enable advancing the CE for plastics (Tisserant et al., 2017).

The scope of this thesis is given by the calculation of the PWF of several countries and products within the global supply chain and addresses the following research questions:

> *RQ1: What are the countries with the largest plastic waste footprint in the global economy and to what extent does it reflect patterns of decoupling?*
>
> *R21: What are the key sectors in the plastics value chain that can help advance the circular economy for plastics?*

By addressing these questions, the contribution of this thesis is three-fold.

First, in a lot of the research studies the linkages between plastic waste and international trade focuses on end of value chain flows of plastic waste (i.e., plastic materials already categorised as waste for disposal). Particularly, the literature in recent years has focussed on assessing the effects of China's ban on plastic waste trade in 2018 (Brooks, Wang & Jambeck, 2018; Liang et al., 2021, Wang et al., 2020). Instead, through the calculation of the PWF, this thesis aims to incorporate an assessment of the generation of plastic waste upstream in the supply chain. The consumption-driven approach of MR-EEIO methods has the potential to derive insights for policy interventions that prevent or reduce plastic waste generation upstream which is more aligned with the CE principles of waste minimisation.

Second, waste input-output models have been developed to assess waste generation patterns at several national or regional levels and are increasingly implemented to model CE interventions; however, the study of a waste footprint that focuses on plastic materials in a multi-regional scope provides a novel approach to the author's knowledge.

Third, this study utilises EXIOBASE 3 multi-regional hybrid input-output tables (MR-HIOTs) (Merciai & Schmidt, 2017) as the source data. A common challenge in the calculation of waste footprints is that conventional EEIO models are displayed using monetary units whereas accounting for waste material flows requires physical units (Aguilar-Hernandez et al., 2018). Hybrid-unit models like the one applied in this work are expressed in a mix of monetary and physical units (Merciai & Schmidt, 2017; Miller & Blair, 2009), and have the potential to be more suitable for the assessment of CE interventions to the extent that transactions are expressed in the appropriate units CE policy is formulated (Towa, Zeller & Achten, 2020). However, the application of hybrid-unit EEIO models is still underdeveloped in the literature.

1.2 Outline of the Thesis

This thesis is structured in seven chapters. The first chapter has introduced the research topic and has identified a research gap that matches the research aim; then, two research questions have been spelled out along with the contribution this author attempts to achieve. Chapter 2 provides an overview of the theoretical background including the CE (section 2.1) and the implications of *telecoupling* for global environmental governance (section 2.2). A review of the literature of EEIO models for the assessment of CE interventions is included in section 2.3 with a particular focus on waste management systems (section 2.3.1). Chapter 3 presents the dataset by providing an overview of MR-HIOTs (section 3.1) and the environmental extension for waste accounts of EXIOBASE 3 (section 3.2). Chapter 4 explains in detail the steps of the EEIO methodology and the implications of working with mixed-units compared with conventional IO monetary tables. Chapter 5 presents the results and Chapter 6 discusses the implications (section 6.1) and limitations of the analysis as well as identifies areas of future research (section 6.2). Finally, Chapter 7 concludes this thesis.

2 Theoretical Framework and Literature Review

This chapter provides an overview of the theoretical background and review of the literature. Section 2.1 introduces the principles of the circular economy (CE) as an alternative to the traditional "take-make-waste" linear systems of production and consumption (MacArthur Foundation, 2017) that can improve long-term sustainability in the plastics industry. Section 2.2 introduces the implications of *telecoupling,* regional environmental issues that occur due to linkages between distant regions, and the implications for governing global sustainability issues (Newig et al., 2020). Section 2.3 reviews the application of EEIO models for the assessment of CE interventions and closes the chapter with a review of input-output models used to assess waste management systems in the context of the CE (section 2.3.1).

2.1 The Circular Economy Paradigm in the Plastics Industry

In recent years, the CE has increasingly gained attention from scholars, practitioners and policymakers as an alternative paradigm to the traditional linear systems of production and consumption with the potential to promote sustainable development (Geissdoerfer et al., 2017). In the CE, the focus is put on extending the useful life of resources (Blomsma & Brennan, 2017; McCarthy, Dellinik & Ribas, 2018; Kristensen & Mosgaard, 2020) as the means to achieve the goal of creating an economy that functions as a closed-loop system in which waste is eliminated and materials are kept in use infinitely (Ghisellini, Cialani & Ulgiati, 2016).

The work of the Ellen MacArthur Foundation has been influential to advance the CE and has coined one of the most widespread and complete definitions of the CE (Geissdoerfer et al., 2017): *"A circular economy is a systemic approach to economic development designed to benefit businesses, society, and the environment. In contrast to the 'take-make-waste' linear model, a circular economy is regenerative by design and aims to gradually decouple growth from the consumption of finite resources*." (MacArthur Foundation, 2017). There are two main parts to this definition of the CE:

The first part aligns the CE with the definition of sustainable development of the Brundtland Commission (1987) by including the three dimensions of sustainability: environmental, social and economic. One of the main critiques to the CE to date is that, in practice, the social aspect still remains underdeveloped and it has not yet been clearly identified how the social benefits from the CE are captures whereas the linkages to economic prosperity and environmental quality have been assessed more extensively (Deutz, 2020; Kirchherh, Reike & Hekket, 2017; Murray, Skene & Haynes, 2015).

The second part of the definition by the MacArthur Foundation highlights the potential of the CE as an enabler of systemic change for long-term sustainable development. The implementation of "closed loop" production systems that are "regenerative by design" is inspired in a number of interrelated concepts (Geissdoerfer et al., 2017; Tóth, 2019) such as cradle-to-cradle design (Braungart, McDonough & Bollinger, 2007), biomimicry (Benyus, 2002), blue economy (Pauli, 2010), industrial ecology (Duchin & Levine, 2014) which are based on the idea that nothing in nature is wasted, instead resources fulfil several different purposes throughout their life-cycle.

The ambitious transformation of current production and consumption systems that the CE proposes requires an in-depth understanding of its principles and expected impacts on sectors and value chains. However, the literature and research on the CE is currently fragmented across different disciplines (Rizos, Tuokko & Behren, 2017). Kirchherh, Reike & Hekket (2017) developed an analysis of 114 different definitions for the CE and found that it is generally associated with the traditional 3Rs framework for waste management of "Reduce, Reuse and Recycle". In recent years, the framework has been extended to include up to 9Rs (Appendix A) in order to put more attention towards upstream intervention.

One of the main challenges in the implementation of the CE is that many definitions and policy responses tend to prioritise the promotion of recycling (Ghisellini, Cialani & Ulgiati, 2016) over prevention of waste as the former fits better within existing production practices and waste management structures, thus, hindering the transformative systemic potential of the CE (Jones, 2021; Kirchherh, Reike & Hekket, 2017; Watson, 2020). In addition, recycling of waste is oftentimes associated with a decrease in the value of materials (downcycling) which simply delays the disposal of waste and extraction of new input materials (Geyer et al., 2015). This is particularly relevant in the case of plastics as packaging is typically considered a lower quality waste material for recycling which hinders the circularity of a substantial proportion of all plastic materials produced (Hahladakis & Iacovidou, 2018).

The European Commission has identified plastics as one of the key areas of action towards the implementation of the CE (Hahladakis & Iacovidou, 2018) and launched the "European strategy for plastics in a Circular Economy" in 2018 (EC, 2018). Furthermore, included within the framework of the EU Waste Framework Directive (2008/98/EC) (EC, 2008; Watkins & Schweitzer, 2018) and as part of the EU Action Plan for the Circular Economy (EC, 2015), the EU Waste Hierarchy. As illustrated in Figure 1, the waste hierarchy establishes a prioritisation list of how waste should be treated with the aim to channel efforts towards prevention of products and materials from becoming waste (Corvellec, 2016; Jones, 2021; Egüez, 2021). Ultimately, the successful implementation of the CE of plastics requires promoting solutions upstream. This requires rethinking the current plastic systems and putting the focus on solutions that promote redesigning products, packaging and business models so that waste can be prevented in the first place (Bocken et al., 2016; Kristensen & Mosgaard, 2020; MacArthur Foundation, 2020).

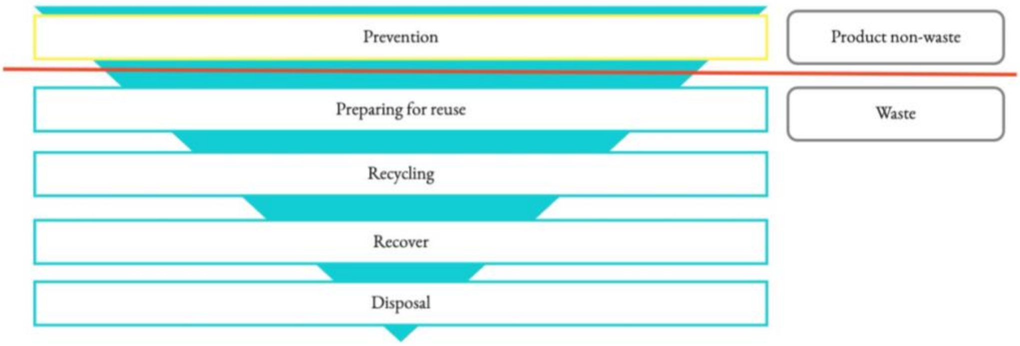


Figure 1. The waste hierarchy. Adapted from EC (n.d., n.p.)

To facilitate the implementation of the CE, it is important to have the ability to monitor progress and the effectiveness of the transition towards circularity (Haupt & Hellweg, 2019; Kristensen & Mosgaard, 2020; Saidani et al., 2019; Tisserant et al., 2017) which has led to the development of several indicators (De Pascale et al., 2021; Saidani et al., 2019). However, to date, there is not yet a universally recognised framework. The assessment of the level of circularity of an economy requires quantitative measurement of the flows and stocks of materials throughout the value chain which belongs to the field of industrial ecology (Tisserant et al., 2017). In turn, this has led to an increased interest in EEIO models for the assessment of environmental impacts of production systems.

2.2 Global Teleconnections and Telecoupling

In sustainability science, global teleconnections describe cause-effect environmental phenomena by which impacts are felt far away from where they are generated, both in time and space (Glantz, Katz & Nicholls, 1991). For example, CO_2 flows worldwide away from where emissions are produced, or ocean plastics can persist in the surface for long periods, travel long distances and accumulate in certain areas such as the Great Pacific Garbage Patch (Lebreton et al., 2018).

Today, we live in a highly globalised world and international trade flows are an example of the globalisation of economic systems. Paradoxically, these global socio-economic connections are given by increasing disconnections between production and consumption processes which tend to occur in very distant places. As a result, global economic transactions can have a significant impact on ecological systems in very distant places too. These globalised environmental issues are referred to as *telecoupling* (Donges et al., 2017; Newig et al., 2020), characterised by adverse environmental effects that are suffered regionally but that have causes that can be traced to global linkages between distant places. Therefore, in a globalised economy, effective environmental policy must take into consideration international trade in order to capture the consequences of global interconnections (Kander et al., 2015; Lenzen et al., 2012; Newig et al., 2020; Peters et al., 2011). Similarly, the concept of telecoupling challenges the decoupling effect between economic growth and environmental degradation generally known as the Environmental Kuznets Curve (EKC) (Panayotou, 1993).

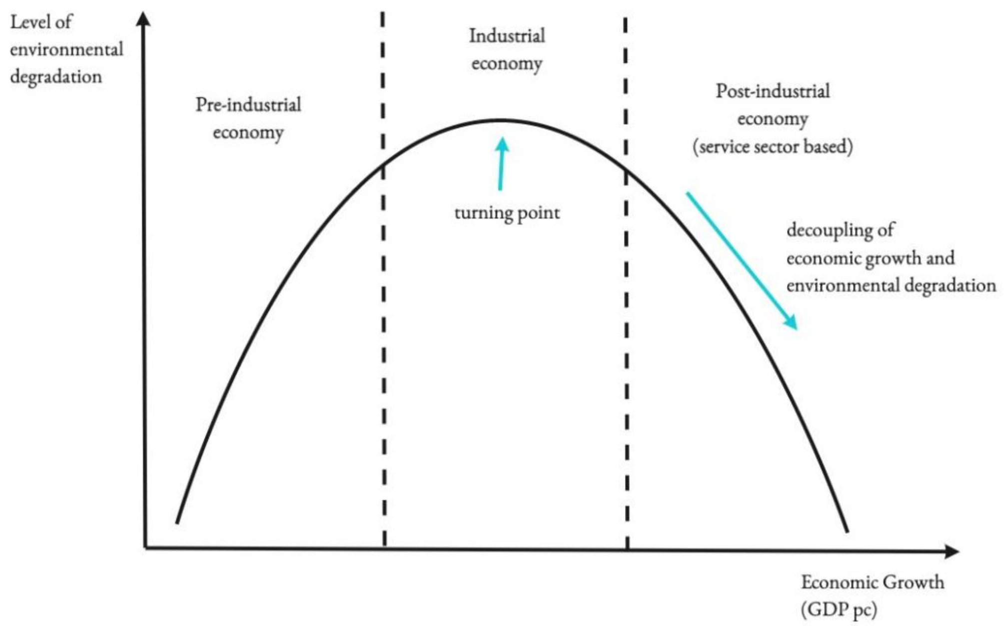


Figure 2. Environmental Kuznets Curve (Panayotou, 1993)

Illustrated in Figure 2, the EKC conceptualises an inverse U-shape relationship between economic growth and environmental degradation. The underlying idea is that in the process of industrialisation, economic growth will lead to increasing levels and environmental degradation. However, there is a turning point in the process of development and, as societies evolve towards post-industrial economies, growth can be decoupled from environmental degradation. This change in pattern is mainly attributed to (1) structural change towards an economy dominated by service sector activities which tend to be less resource-intensive, (2) technological improvements of domestic production systems and (3) more sustainable choices made by consumers (Dinda, 2004).

One of the main critiques to the EKC is that the apparent reduction in environmental degradation in developed countries does not respond to a real decoupling effect, but instead a systematic outsourcing of polluting-intensive economic activities to developing countries which also displaces environmental impacts (Peters et al., 2011) (i.e., telecoupling). Traditional accounting for environmental effects performed at the national or regional level fails to capture the global effects *embodied* in international trade. In response, there has been a generalisation of studies that calculate environmental *footprints* based on the allocation of responsibility over environmental degradation to consumers.

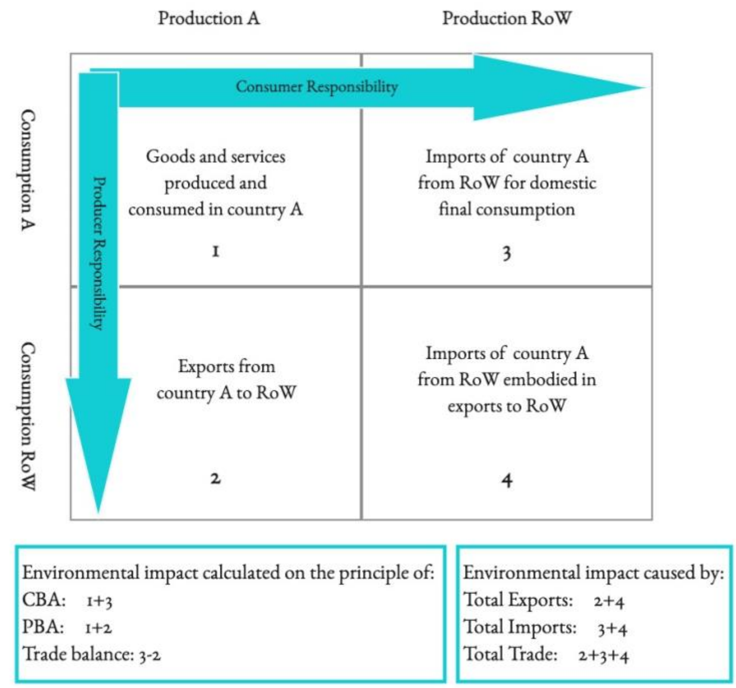


Figure 3. Producer vs Consumer Responsibility. Adapted from Munksgaard et al (2009)

Figure 3 illustrates two different principles to calculate a country's environmental impact balance based on consumer responsibility versus producer responsibility in a two-region model (Country A and Rest of the World). Consumer-based accounting (CBA) allocates the responsibility for the environmental impact (e.g., emissions or waste) *embodied* in trade to the country or region where goods and services are ultimately consumed regardless of where along the value chain these impacts were generated. Based on the CBA principle, environmental impacts that occur due to imports are added to the balance of environmental impacts given by domestic consumption whereas impacts linked to exports are subtracted from a country's account. Conversely, the production-based accounting (PBA) approach employs a territorial-based accounting linking impacts directly to the country where goods and services are produced, regardless of where consumption takes place.

Global MR-EEIO models have enabled the calculation of environmental footprints, or impacts *embodied* in international trade, which trace economic transactions throughout global supply chains and the direct and indirect effects of economic activities which are ultimately driven by final demand (Tukker & Dietzenbacher, 2013). Following a CBA approach, the literature in the calculation of carbon footprints has grown significantly in the last few years to discuss the real decoupling effect between economic development and pollution (Baumert et al., 2019; Davis & Calderia, 2010; Jiborn et al., 2017; Kander et al., 2015; Peters et al., 2011). Similarly, Henriques & Kander (2010) discovered that the real impact in lowering energy-intensity, and therefore emissions, from the transition to a service-based economy is modest.

The CE has identified the transition to a service-based economy as a line of action to potentially reduce the resource-intensity of economic activities (MacArthur Foundation, 2015). However, in a study of waste footprint of different regions in Australia, Fry et al. (2015) found instead a reallocation effect of waste away from agriculture and industrial sectors towards retail, trade and services driven by waste generated in material-intensive processes along the supply chain. Therefore, the transition towards a service economy alone does not necessarily imply a reduction of resource consumption and waste generation. The need for systemic indicators suitable for the assessment of CE policies has increased the interest to adopt EEIO methods that enable the calculation of footprint type of indicators for material and waste flows (De Koning, 2018; Duchin, 2009).

2.3 Environmentally-Extended Input-Output Analysis for the Assessment of Circularity

A fundamental goal of the CE is to achieve the decoupling of economic development and environmental degradation through the creation of closed-loop systems. Therefore, the assessment of CE interventions requires a systemic approach to sustainable supply management through enhancing resource productivity (upstream) (Blomsma & Brennan, 2017) and improvements in waste management systems (downstream) (Watson, 2020), with a particular focus on improving recycling systems and capabilities.

The field of industrial ecology is interested in assessing environmental impacts associated with production and consumption systems. Life-Cycle Assessment (LCA) has been the main method to analyse flows of materials from a bottom-up or process-level approach, i.e., following inputs from resources to products (Duchin & Levine, 2014). However, LCA requires a high level of specificity identifying individual inputs which hinders the ability to analyse complex supply chains often referred to as system boundaries (Genovese et al., 2017). As industrial ecologists have turned their attention towards the assessment of environmental impacts of global supply chains, the need to overcome the systems boundary of LCA (Duchin, 2009; Duchin & Levine, 2014), EEIO models have provided an alternative top-down approach for economy-wide modelling of systems of production.

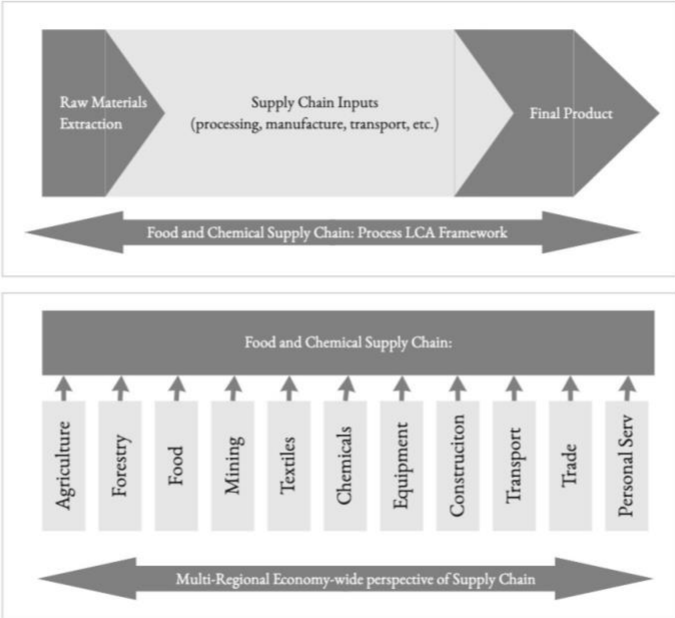


Figure 4. Schematic representation of a typical process LCA system (top) and EEIO framework (bottom) (Genovese et al., 2017, p.346)

A common challenge encountered when implementing EEIO models for the assessment of circularity is, however, that conventional IO tables are displayed using monetary units whereas CE policies are oftentimes articulated in physical units (Aguilar-Hernandez et al., 2018). Nevertheless, in some instances, measuring the impact of CE interventions on employment outcomes requires monetary accounts (Donati et al.,2020; Wiebe et al., 2019; Wijkman & Skånberg, 2017). Thus, hybrid-unit IO tables have been developed (Merciai and Schmidt, 2017) which have the potential to be more suitable to the extent that are expressed in the appropriate units CE is formulated (Towa, Zeller & Achten, 2020).

2.3.1 Review of Waste Environmentally-Extended Input-Output Models in the Context of the Circular Economy

The CE brings a new perspective by which waste becomes a resource and waste management is seen as a system for recovery of resources and environmental impact prevention (Gisellini, Cilani & Ulgiati, 2016). Therefore, a significant portion of materials that nowadays fall into the classification of waste could remain as resources which can be seen as a failure of current systems of production and consumption (Watson, 2020).

EEIO models have been used to assess waste management policy by appending national waste generation data as satellite accounts to conventional monetary IO tables (Towa, Zeller & Achten, 2020). Reynolds et al. (2016) used waste extended IO model to estimate the economic value loss associated with food waste in New Zealand. Barata (2002) modelled the generation, composition and final disposals of solid waste in Portugal using an EEIO model. While EEIO allows assessment of waste generation by economic sector and attributes waste to different sectors and final consumption groups, the disconnection between monetary economic flows and physical waste flows does not allow to assess treatment of waste (Towa, Zeller & Achten, 2020). This has led to the modification of the framework itself to allow for more complete analysis of waste footprints and waste management policies (Aguilar-Hernandez et al., 2018).

The Leontief-Duchin model was the first attempt to extend the Leontief EEIO model for the analysis of waste (Duchin, 1990). The fundamental characteristic of this model was that it maintained the strictly one-to-one correspondence of the conventional EEIO such that each pollutant (waste) is linked to a single abatement (treatment) (Leontief, 1970, p.262). However, products generate different types of waste, and one type of waste can be treated using multiple treatments. To overcome the one-to-one correspondence limitation, the waste input-output model (WIO[1]) developed by Nakamura & Kondo (2002) extends the EEIO model to account for an arbitrary number of waste types and waste treatments. WIO models can follow waste generation to its treatment by the means of the so-called allocation matrix which specifies the share of a particular waste type that is treated using a particular method (Nakamura & Kondo, 2002) and are able to connect monetary flows of products and services to physical flows of waste (Fry et al., 2015; Towa, Zeller & Achten, 2020).

WIO has the potential to be an important tool for studying the CE and waste footprints as it allows to model "downstream" flows of waste and waste treatment at the same time it can trace waste generation "upstream" in supply chains (Tisserant et al, 2017). For example, Nakatami et al. (2020) has studied the intersectoral flows of plastics materials in Japan used for packaging in order to inform recycling policy to meet the government's 2030 goals to reduce the use of single-use plastics. Similarly, Waste Supply-Use (WSU) models are an extension of WIO that represent waste data by type (supply) and treatment (use) simultaneously in one table (Frey et al., 2015; Lenzen & Reynolds, 2014). Fry et al. (2015) develop a multi-regional WSU table for the analysis of regional waste footprint, thus, allowing to calculate waste footprints from a consumption-based accounting.

Physical flows of materials and waste provide important information for optimising waste management systems such as resource and waste intensity for different sectors (Fry et al., 2015). As a result, physical input-output (PIO) models have been derived for Germany (Stahmer et al., 1997; Statistisches Bundesamt, 2001), Denmark (Gravgård-Pedersen, 1999), Italy (Nebbia, 2000), Finland (Mäenpää, 2002) and the UK (Wiedmann et al., 2004). A distinctive feature of PIO models is that, unlike other IO models adapted to study waste flows that append waste data, the generation and treatment of waste is a central part of the accounting system (Dietzenbacher, 2005). However, the main limitation is that given that transactions are exclusively recorded in physical units, sectors with non-physical outputs are not included which may lead to underestimating environmental impacts (Liang et al., 2012).

[1] For clarification, WIO is considered a particular type of EEIO.

Finally, hybrid-unit IO models (HIO) record the flow of economic transactions in mixed units. Thus, the main advantage of HIO models is that it can tackle the limitations of previous models (Towa, Zeller & Achten, 2020) as all types of transactions can be included and material flows can be traced in the most suitable units. Towa, Zeller & Achten (2021) used EXIOBASE 3 Multi-Regional Hybrid Input-Output Table (MR-HIOT) to model the potential environmental pressures of five CE policy interventions. The use of a hybrid model enabled the researchers to study three different environmental pressures: global warming, raw material extraction and waste footprint (Towa, Zeller & Achten, 2021).

This review of the literature has shown that the implementation of IO models for the assessment of environmental pressures linked to waste generation has grown particularly in the last few years as the CE has gained momentum at the policy level. However, the study of waste footprints using Multi-regional IO models is still underdeveloped. In addition, to the author's knowledge, the study of plastic waste embodied in international trade provides a novel contribution to the literature.

2.4 Testable Hypothesis

Based on the research questions proposed by the thesis and the theoretical and literature review presented in this chapter, the author formulates two hypotheses that to be tested in the analysis:

Hypothesis I: The generation of plastic waste is mostly driven by high-income countries who have higher consumption levels. Therefore, when it comes to plastic waste intensity, the EKC is not realised and high-income countries have a larger plastic waste footprint than developing countries who suffer the effect of telecoupling, unless strong CE policy intervention can be observed.

Hypothesis II: The calculation of the plastic waste footprint, by including the direct and indirect effects of waste generation throughout the supply chain, reallocates waste away from primary and manufacturing production sectors towards less material-intensive sectors such as retail, trade and services.

3 Data

The data for this thesis comes from the latest release of EXIOBASE 3 database published in March 2019 and reviewed in February 2020. A particularity of EXIOBASE 3 is that it comes in two versions: monetary and hybrid units. This study uses the multi-regional hybrid input-output tables (MR-HIOTs) which record economy-wide transactions in a combination of monetary (million euros), mass (tonnes) or energy units (terajoules). EXIOBASE 3 provides a variety of environmental extensions such as emissions, resources, land uses, accumulation of new materials and supply and use of waste flow. This study is interested in the extension for supply and use of waste flows for the calculation of plastic waste embodied in international trade, or plastic waste footprint (PWF). The reference year for the analysis is 2011 due to data availability constraints.

Section 3.1 provides a brief explanation about how EXIOBASE 3 dataset was constructed and the structure of the MR-HIOTs used for the analysis is discussed. Section 3.2 describes the waste environmental extension employed for calculating the environmental indicators. Furthermore, each section discusses the assumptions and limitations of EXIOBASE 3 and MR-HIOTs.

3.1 Source Data: EXIOBASE 3

EXIOBASE is a global EE-MRIO database that can be used to model the production-related environmental impacts or footprints associated with final consumption along global supply chains (Wood et al., 2014). The MR-HIOTs have been derived from multi-regional environmentally extended supply-use table (MR-SUT) within the context of the EU FP7 DESIRE project and builds upon the two previous versions of EXIOBASE (Merciai & Schmidt, 2017).

A distinctive characteristic of EXIOBASE is the high resolution in terms of sectoral classification (Tukker & Dietzenbacher, 2013; Tukker, Giljum & Wood, 2018) and a large set of environmental extensions which makes it compatible with the System of Environmental-Economic Accounting (SEEA) (UN, 2017). The hybrid tables of EXIOBASE 3 contain data for 43 countries and 5 rest-of-the-world (RoW) regions, 164 products, 164 production activities. Appendix B provides a complete list of countries and regions (Table 2), products, units of measurement and production activities (Table 3). EXIOBASE allows for both an economy-wide assessment and the identification of environmental hotspots for more targeted policy intervention (Stadler et al., 2018).

Compared to other main global EE-MRIO databases[2], the disproportionate representation of European countries compared to other geographical locations has been identified as a shortcoming (Tukker, Giljum & Wood, 2018). However, it is important to keep in mind that EXIOBASE was developed with the main goal of assessing sustainability matters of the EU, its main trading partners and other major global economies (Stadler et al., 2018). Given that the objective of this study is to calculate the PWF linked to final consumption along global supply chains, this has not been identified as a major limitation as the hypothesis is that high-income countries will have a higher PWF. Furthermore, large developing economies such as China and India, and Indonesia, which previously have been identified as high plastic pollutants (Jambeck et al., 2015; Schmidt et al., 2017) are included in the list of countries. Nevertheless, even if the size of the economy is not as large, the Philippines has been identified as the single country in the world contributing the most to discharges of plastic waste into the ocean (Meijer et al., 2021), thus, it would have been interesting to have data for the country which, unfortunately, it appears aggregated with other countries in the Asia and Pacific region (RoW Asia and Pacific).

Another feature common to all the main MR-IO databases and a comparative strength of EXIOBASE is that all intersectoral flows are represented in monetary units rather than physical units. For transactions of which the value of physical units is volatile, zero or negative, as it is the case in the waste management systems, this may lead to inaccurate estimations (Tukker, Giljum & Wood, 2018). Several single-country PIOs tables have been published (see section 2.3.1), however, EXIOBASE 3 is the first global multi-regional hybrid input-output table. Transactions in EXIOBASE MR-HIOTs are recorded in a combination of units according to the nature of products: services are accounted for in million euros, tangible products are measured in tonnes and energy sectors are recorded in terajoules (TJ). Thus, as previously mentioned, MR-HIOTs have the potential to bridge the gap between IO analysis and the fields of industrial ecology and ecological economy (Merciai & Schmidt, 2017) and to become a powerful tool to inform and assess policy on CE thanks to the detailed waste accounts which link directly physical waste flows and their treatment to economic activities (Merciai & Schmidt, 2017; Towa, Zeller & Achten, 2020).

A limitation of using MR-HIOTs is that while the monetary tables in EXIOBASE 3 consist of a time series ranging from 1995 to 2011, the hybrid version is only available for a single year and the study provides a snapshot of 2011. Similarly, to overcome data inconsistencies across countries and time found in the process of data compilation for the construction of EXIOBASE 3, the authors rely on a manually-constructed concordance matrix (Stadler et al., 2018).

[2] Table 4 in Appendix B provides a brief comparison of the three main MR-EEIO databases. A more thorough comparison is outside of the scope of this thesis, refer to Tukker, Giljum & Wood (2018).

Figure 5 presents a simplified version of the EXIOBASE 3 MR-HIOTs. The objective of this representation is to introduce the empirical material to allow the reader to become familiar with the structure of IO tables and dimensions of the dataset. The IO methodological framework will be explained in detail in Chapter 4.

		Intersectoral Transactions (**Z**)							Final Demand (**F**)			Total Output (**x**)
		Country 1			...	Country 48			C1	...	C48	
		Activity 1	...	Activity 164		Activity 1	...	Activity 164	$\mathbf{f}_1$		$\mathbf{f}_{48}$	
Country 1	Product 1 ... Product 164											
...												
Country 48	Product 1 ... Product 164											
Value Added (**w’**)												
Total Production vector (**x’**)												
Environmental Extension: Plastic Waste (**d’**)												

Typical EE-MRIO tables include five main components represented by matrices or vectors which are presented side by side: the intersectoral transactions (**Z**), final demands (**F**), total output (**x**), value-added (**w’**) and environmental extensions (**d’**). Dimensions of the matrices and vectors in EXIOBASE MR-HIOTs representing the basic components of IO analysis: Z (7872x7872); F (7872x48); x (7872x1); x’ (1x7872); d’ (1x7872). Cells coloured in a darker shade refer to the matrix diagonal entries for domestic transactions

Figure 5. Simplified EE-MRIO table (author's own construction)

The matrix of transactions denominated by **Z** represents all the intersectoral flows of products[3] in the economy which are both delivered and consumed by production activities. Figure 5 presents a multi-regional model where the intersectoral transactions include domestic transactions, imports and exports. The dataset contains 164 products and 164 activities for 48 countries/regions, thus, matrix **Z** has dimensions 7872x7872, where the first number refers to the number of rows and the second is the number of columns. Conventional IO models present the intersectoral transactions by the flow of goods and services from between industry sectors (*industry-by-industry*). A difference in notation of EXIOBASE MR-HIOTs is that tables are provided following a so-called *product-by-product* approach[4]

[3] Product refers to both goods and services throughout the remainder of this thesis.
[4] The alternative approach of industry-by-industry MR-IOTs is only available for monetary tables in EXIOBASE 3.

(Stadler et al., 2018) where rows represent products that are matched to production activities displayed in the columns. The product-by-product table is based on the industry technology assumption and by-products are recorded as negative inputs (refer to Stadler et al. (2018) for a more technical explanation). From a methodological implication, however, this does not affect the way IO analysis is developed.

The original final demand matrix on EXIOBASE MR-HIOTs has dimensions 7872x288 as it records data for 164 products x 48 countries/regions and includes 6 final demand categories given by consumption expenditure of "Households", "Non-profit organisations serving households (NPISH)", "Government", "Gross fixed capital formation", "Changes in inventories" and "Changes in valuables". Final demand represents output demand of each product by final consumers, i.e., not used to produce other goods and services. For the purpose of this analysis, final demand categories have been aggregated to show final consumption for each country in a single column (**f**). The aggregated final demand matrix is denoted by **F** and records the final consumption for 164 products for all final consumers in the 48 countries/regions and has dimensions 7872x48.

The total output vector **x** is obtained by the row-sum of all the intermediate transactions and final demand for each product *i* and country/region (7872x1). The total output vector is oftentimes represented as a transpose vector **x'** which is obtained as the column sum of all primary inputs required for an activity *j* and the corresponding element of the value-added row vector **w'** which captures other production inputs such as labour and capital not included in the matrix of transactions (**Z**). A particularity of HIOTs is that, since transactions for the different products *i* are recorded in mixed units, it would not make sense to calculate the column total for a given activity *j* (Duchin, 2009). Therefore, there is no value-added vector in EXIOBASE 3 MR-HIOTs, but instead the vector of principal productions denoted by **x'** is provided which corresponds to the transpose of the total output vector (**x**). The mention to **w'** has been included in Figure 5 in order to be able to address this methodological difference between monetary and hybrid IO tables.

The final component is the environmental extension which is appended to the IO table as a row vector **d'** and represents the accounts of plastic waste given by production activity. This component is explained in detail in the following section.

3.2 Environmental Extensions: Waste Accounts

A distinctive feature of EXIOBASE 3 and the reason why it has been chosen for the study is the availability of detailed waste accounts. The environmental extension for supply and use of waste flows is divided into 9 different waste fractions (i.e., types of waste) which show the supply and use of waste associated with each of the 164 production activities. For the scope of this thesis only the plastic waste fraction is considered. The physical flows of plastic waste are measured in tonnes.

For the construction of the waste extension, inspiration was taken from the waste input-output (WIO) model introduced by Nakamura & Kondo (2002) (see section 2.3.1) by incorporating the concept of the "allocation matrix" to link the supply of waste flows and the demand of waste treatment services which are then accounted for in the matrix of transactions (Merciai & Schmidt, 2017). Supply of waste refers to the amount of plastic waste generated by all production activities driven by final consumption. Use of waste is mostly driven by waste treatment activities which are also included in the matrix of transactions. EXIOBASE adopts Schmidt et al. (2012)'s definition of waste as material for treatment. Therefore, the use of waste takes place through activities such as "reprocessing of scrap into new materials that can substitute virgin materials", "waste incineration" or "landfill" (Stadler et al., 2018).

The production activities that act as plastic waste users (treatment) have the potential to reduce the total PWF. To account for this positive effect in the analysis, the net plastic waste generated by an activity is calculated by subtracting the amount of waste used from the amount of waste supplied:

$$Plastic\ Waste\ Supply\ by\ Activity - Plastic\ Waste\ Use\ by\ Activity = Net\ Plastic\ Waste\ by\ Activity \tag{3.1}$$

A positive result means that an activity is a net producer of plastic waste. Conversely, a negative result means that an activity is a net user of plastic waste. However, when the implications of different types of waste treatment activities are examined more carefully, it reveals that not all uses of plastic will lead to reducing environmental pressures. This is further discussed in Chapter 5.

4 Methods

This chapter provides an overview of the EE-MRIO framework applied in this thesis for the calculation of plastic waste footprints. Section 4.1 starts with a brief introduction to the IO framework, explains the fundamental elements of IO and discusses the implications derived from working with hybrid tables. Methodological guidance for the development of this section is mostly based on the work of Miller & Blair (2009) and the collection of articles in Suh (2009). Section 4.2 extends the IO model for the calculation of waste footprints. The formulas on this section are based on Fry et al., (2015).

4.1 The Fundamentals of IO Analysis

Input-Output analysis is an analytical framework developed by Nobel-prize laureate Wassily Leontief in 1936 to examine the interdependence of industrial sectors in an economy (Miller & Blair, 2009). The Leontief model conceptualises an entire economy as an accounting system that records the flows of commodities and services as transactions (Leontief, 1936). The model was originally developed at the country level and later it has been expanded to include multiple regions and the global economy.

The Leontief model assesses how changes in one part of the economic system affect the other parts of the system and traces inputs and outputs along the whole supply chain by quantifying the direct and indirect input requirements needed to meet final demand by consumers of a certain product (Duchin, 2009). These characteristics have led to extending the basic IO framework for the study of environmental impacts that occur because of production and consumption systems.

4.1.1 Formalisation of an IO model

In the IO model, production activities are summarised in a set of industry sectors that produce a certain product (output). At the same time, producers act as consumers as they require products (inputs) which are produced by other industry sectors to produce its own output. As introduced in the previous chapter, these intersectoral transactions are captured in the matrix **Z** where rows describe the distribution of a producer's output throughout the economy (i.e., activities for which certain output is used as input) and columns describe all inputs required to produce certain output.

An element z_{ij} of matrix **Z** shows the amount of product i required as input by sector j to produce its output j. *Intra-sectoral transactions* are also included in matrix **Z** when i=j and reflect the purchase of a sector of its own output to be used as input (Miller & Blair, 2009).

Transactions that serve final consumers directly are recorded in the final demand matrix (**F**). As introduced in section 3.1., final demand categories can be aggregated into a single one, expressed by the column vector **f**. The element f_i represents the amount of output i sold to final consumers.

Finally, the total output of an industry sector (**x**), is given by the sum of the intermediate input requirements to serve other sectors (rows of **Z**) plus the total output requirements to serve final demand (rows of **f**). Consider a single-country economic system with *n* production sectors, the basic components of the IO model can represent in its matrix form as:

$$x = \begin{bmatrix} x_1 \\ \vdots \\ x_n \end{bmatrix}, Z = \begin{bmatrix} z_{11} & \cdots & z_{1n} \\ \vdots & \ddots & \vdots \\ z_{n1} & \cdots & z_{nn} \end{bmatrix} \text{and } f = \begin{bmatrix} f_1 \\ \vdots \\ f_n \end{bmatrix} \quad (4.1)$$

The relation described above is known as the standard IO relationship and can be written as:

$$x = Z1 + f \quad (4.2)$$

Here **1** is a column vector of *n* size that contains all 1's known as the "summation vector". The post-multiplication of a matrix **Z** by the summation vector creates a column vector whose elements are the row sums of the matrix **Z** (Miller & Blair, 2009).

4.1.2 The Technical Coefficient Matrix

The technical coefficient matrix denoted by **A** represents the economy's production structure. Elements a_{ij} are the so-called direct input coefficients, or technical coefficients, which represent the amount of direct inputs of product *i* required for the production of one unit of product *j*.

This relationship is established by the following ratio:

$$a_{ij} = \frac{z_{ij}}{x_j} \quad (4.3)$$

In hybrid IO models, transactions in **Z** are recorded using multiple units, therefore, the technical coefficient matrix (**A**) will reflect hybrid units as well (Duchin, 2009; Miller & Blair, 2009). A simplified two-sector model is presented below to exemplify this scenario. The first sector corresponds to a tangible product measured in tonnes and the second sector is a service measured in euros.

$$Z = \begin{bmatrix} \text{tonnes} & \text{tonnes} \\ € & € \end{bmatrix}, f = \begin{bmatrix} \text{tonnes} \\ € \end{bmatrix} \text{and } x = \begin{bmatrix} \text{tonnes} \\ € \end{bmatrix} \quad (4.4)$$

A can be obtained as follows:

$$A = Z(\hat{x})^{-1} = \begin{bmatrix} \text{tonnes}/\text{tonnes} & \text{tonnes}/€ \\ €/\text{tonnes} & €/€ \end{bmatrix} \quad (4.5)$$

The technical coefficients are obtained by dividing the element z_{ij} of the transactions matrix (**Z**) divided by the element x_j of the total output vector **x** that is obtained as the j row total of **Z** and **f**. In equation (4.5) the "hat" over the vector **x** denotes the diagonal matrix which contains all zeros except for the elements of the output vector along the main diagonal and it has the same dimensions as **Z**. The expression $(\hat{\boldsymbol{x}})^{-1}$ denotes the inverse of the diagonalised output vector which is the equivalent in matrix algebra to a division. The resulting technical coefficient matrix (**A**) also has the same dimensions as the transaction matrix (**Z**) and in this study it is a square matrix of 7872 rows and 7872 columns.

4.1.3 The Leontief Inverse

The technical coefficient matrix derived in the previous section reflects the direct input requirements needed to meet the final demand of a certain product, however, supply chains are complex and interconnected and the production of an input requires further inputs. The Leontief inverse (**L**) also referred to as *total requirements matrix* captures these second-order effects. It can be obtained through the following expression:

$$\mathrm{L} = (\mathrm{I} - \mathrm{A})^{-1} \tag{4.6}$$

Where $\mathrm{I} = \begin{bmatrix} 1 & \dots & 0 \\ \vdots & \ddots & \vdots \\ 0 & \dots & 1 \end{bmatrix}$ is the identity matrix of the same dimensions of A which contains ones on the main diagonal and zeros for every other element. An element l_{ij} in the Leontief inverse captures the direct and indirect change in production of input i needed for one additional unit of output in sector j.

The implications of working with hybrid units for the calculations of **L** are the same as for **A**. The matrix **L** will be expressed in the same units as **A**, except that the interpretation of **L** is in terms of input requirement per unit of final demand.

To conclude this section, the IO relationship can be expressed as the total production requirements (i.e., direct and indirect) needed from all sectors to meet final demand by consumers:

$$\mathrm{x} = (\mathrm{I} - \mathrm{A})^{-1}\mathrm{f} = \mathrm{Lf} \tag{4.7}$$

4.2 EE-MRIO Analysis for the Calculation of Waste Footprints

Growing concerns about environmental degradation caused by uncontrolled pollution led to the development of the first environmentally-extended Input-Output (EEIO) model motivated by the need to incorporate the "undesirable by-products of economic activities" (i.e., pollutants) to conventional IO models (Leontief, 1970, p.262). Today, EEIO models are one of the state-of-the-art methodologies to trace economy-wide environmental pressures linked to production activities (De Koning 2018; Genovese et al., 2017).

The environmental impact this thesis analyses is the plastic waste footprint (PWF) — i.e., plastic waste embodied in international trade. To calculate the PWF, the standard IO relationship holds (Fry et al., 2015) and it is extended to include the environmental account in the following way:

$$P = \hat{q}(I - A)^{-1}f = \hat{q}Lf \tag{4.8}$$

In this study, **q'** is a 1x7872 vector that represents the direct (net) plastic waste production intensities for 164 production activities and 48 countries/regions, and it is expressed in tonnes of plastic waste per unit of output. The direct plastic waste intensities are obtained from the environmental extension **d'** (section 3.2) following the same approach as the technical coefficients in matrix **A**.

$$q = d'(x)^{-1} \tag{4.9}$$

The PWF is denoted by **P** and represents the net plastic waste associated with production activities in order to serve the final demand, measured in terms of tonnes per unit of final demand. The Leontief inverse enables the calculation of the PWF by capturing the direct plastic waste generated from consumption and the plastic waste generated along the supply chain.

5 Results

This section presents the results obtained from the multi-regional environmentally-extended input-output (EE-MRIO) method described in the previous section to quantify the plastic waste footprint (PWF). All the results are for reference year 2011. The analysis of results begins by assessing the crude PWFs obtained considering the net effects of products that generate (supply) and treat (use) plastic waste (see Section 3.2). Then, the drawbacks of this measure are discussed and an alternative, in the author's view, more accurate measure of PWF is provided (section 5.1.1).

This chapter is organized into four parts. Section 5.1 presents the PWF by countries. Section 5.2 addresses the impact of plastic waste treatment product categories. Section 5.3 assesses the PWF of several products. Section 5.4 concludes with an analysis of products by country with higher PWF.

PWFs are expressed in tonnes, mega tonnes (Mt) (1 Mt = 1.000.000 tonnes) of plastic waste or kilograms per capita (kg p.c.) (1kg = 1.000 tonnes), as specified. All tables and figures in this section represent the author's own calculations. To improve the readability, names of countries/regions and products have been abbreviated (see Appendix B for the reference list).

5.1 Plastic Waste Footprint by Country

When accounting for waste generated in a country, two different approaches can be considered: territorial waste accounting and waste footprint (Towa, Zeller & Achten, 2020). The territorial waste accounting is equivalent to a production-based accounting (PBA) approach where waste is calculated as a direct result of production activities that take place in a country or region. Alternatively, the calculation of the waste footprint follows a consumption-based accounting (CBA) approach where plastic waste derived from economic activity that is performed by a trading party is allocated to the country or region where final consumption takes place. Thus, the calculation of the waste footprint provides a more holistic approach as it captures plastic waste generated directly and indirectly throughout the whole supply chain, including intermediate consumption and trade, as well as final consumption. As this study is interested in the PWF in order to capture the effects of plastic waste accumulated throughout the entire supply chain, the analysis follows a CBA approach.

Previous studies have found that the calculated waste footprints to be higher than reported territorial waste accounts (Fry et al., 2015: Towa et al., 2020), however, the results obtained in this study do not show a clear pattern. While China, India and South Korea show higher values of plastic waste by PBA, Canada, Indonesia, RoW Europe and RoW Africa have higher values of plastic waste accounting by CBA (Figure 6). A larger CBA means that a country is a net "outsourcer" of plastic waste whereas a larger PBA means that a country is a net *insourcer* of plastic waste. Appendix C (Table 5) provides a complete table of the results, the figures in this section comment on the main insights.

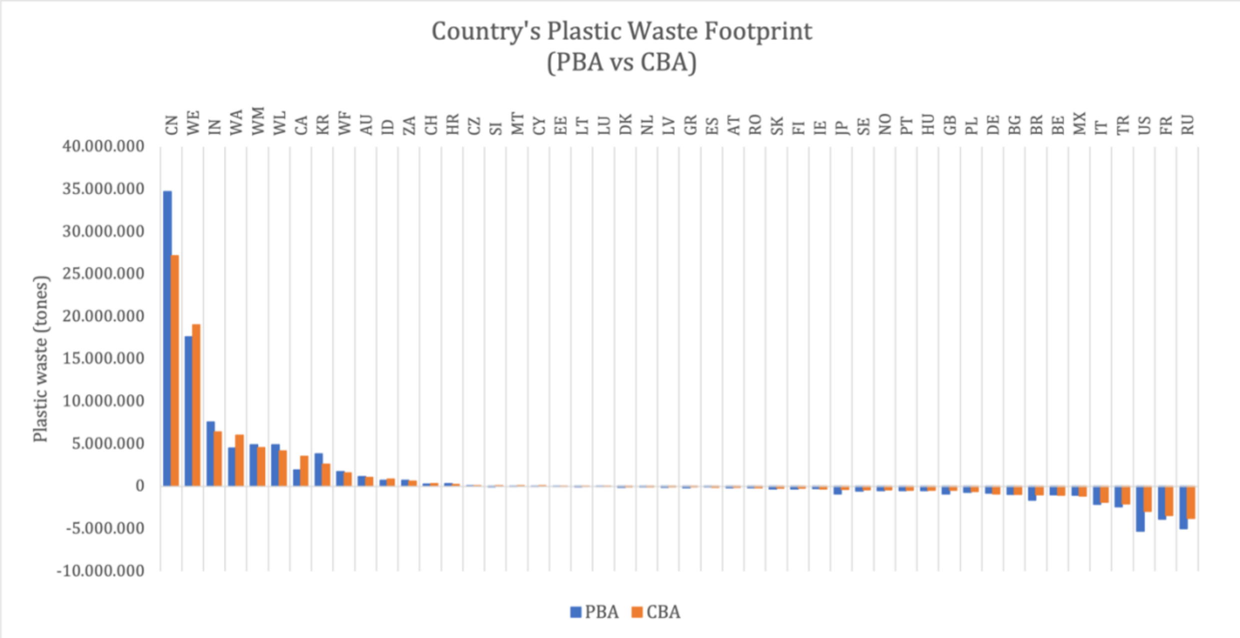


Figure 6. PBA vs CBA comparison: Plastic waste footprint of countries and regions.

China is the country with the largest PWF equal to 27 Mt of plastic. The PBA result for plastic waste generated in China is significantly larger with nearly 35 Mt of plastic waste. This could be attributed to the country's role as "factory of the world" (Wang, 2020), however, more detailed analysis of the products driving the generation of plastic waste in the country will be provided in the next section. The second largest PWF is given by the region representing the aggregation of European countries ("RoW Europe" or WE) not specified in the dataset with a PWF equal to 19 Mt. India's PWF is the third largest and it is equal to 6,4 Mt of plastic waste. Looking at the bottom of the distribution of countries by PWF in Figure 6, Russia, France, the US, Turkey and Italy appear as the countries with the smallest (negative) PWF.

A crude interpretation of these results is that countries with a PWF close to zero in Figure 6 show that through production activities along the whole supply chain, the amount of plastic waste generated is almost compensated by plastic waste used for treatment. Similarly, countries with negative footprint are treating more plastic waste than the amount generated as a result of economic activities. Thus, potentially reducing the amount of plastic waste in the economy.

Although this might seem counterintuitive, the explanation for these results is that the product breakdown in EXIOBASE includes accounts that refer to the treatment of plastics such as "Plastic waste for treatment: incineration", "Plastic waste for treatment: landfill" and "Secondary plastic for treatment, Re-processing of secondary plastic into new plastic" which are considered accounts that use plastic waste, therefore, reducing the overall amount of plastic waste linked to production activities in an economy. However, under the lenses of the CE this interpretation may be flawed. For example, since plastic is not a biodegradable product, the disposal of waste in landfill should not be associated with a reduction of the PWF. Similarly, incineration appears further down in the waste hierarchy if compared to recycling or re-processing of plastic waste materials (EC, 2008; Lazarevic et al., 2010; MacArthur Foundation, 2016), not only because it does not prevent from reducing the use of new virgin plastics, but it also contributes to high emission levels.

5.1.1 Revised Plastic Waste Footprint

The results discussed in the previous sections represent the "net" PWF calculated from the difference between the supply and use of plastic waste (see section 3.2). Treatment of plastic waste activities have the potential to reduce the PWF of a country and it reflects how a country is (at least partially) dealing with the plastic waste generated throughout its supply chain. In an attempt to show a more accurate measure of the PWF that is better aligned with the CE principles, a revised calculation is provided, excluding the counter-effect of plastic waste treatment accounts for landfill and incineration. The account for "Re-processing of secondary plastic into new plastic" is kept as, from the perspective of the CE, recycling would be the preferred way of dealing with plastic waste once it has already been created (Lazarevic et al., 2010).

Figure 7 shows that, in total values, China continues to be the country with the largest PWF with nearly 36 Mt of plastic waste. By eliminating the counter effect of plastic waste sent to landfill and incineration, the revised CBA value has increased and is almost the same magnitude as the previous PBA result obtained for the country. At the same time, the US, with a PWF equal to 15,15 Mt, becomes the third country with the largest PWF, only behind the RoW Europe region. It is particularly interesting to note that the US, Russia and Brazil which ranked as countries with the largest negative PWF move dramatically to the top of the list. Similarly, as total values for plastic waste are considered, the other rest-of-the-world aggregated regions for RoW Asia and Pacific (WA), RoW Middle East (WM) and RoW America (WL) present relatively high numbers whereas RoW Africa comes after several individual countries including China, the US, India, Russia, Canada Brazil and South Korea. West Africa is one of the regions in the world with more emissions of plastic waste into the natural ecosystem (Jambeck et al., 2018; Meijer et al., 2021), however, when analysing PWF from a consumption-driven approach, other countries appear to play a more important role in the generation of plastic waste in the first place.

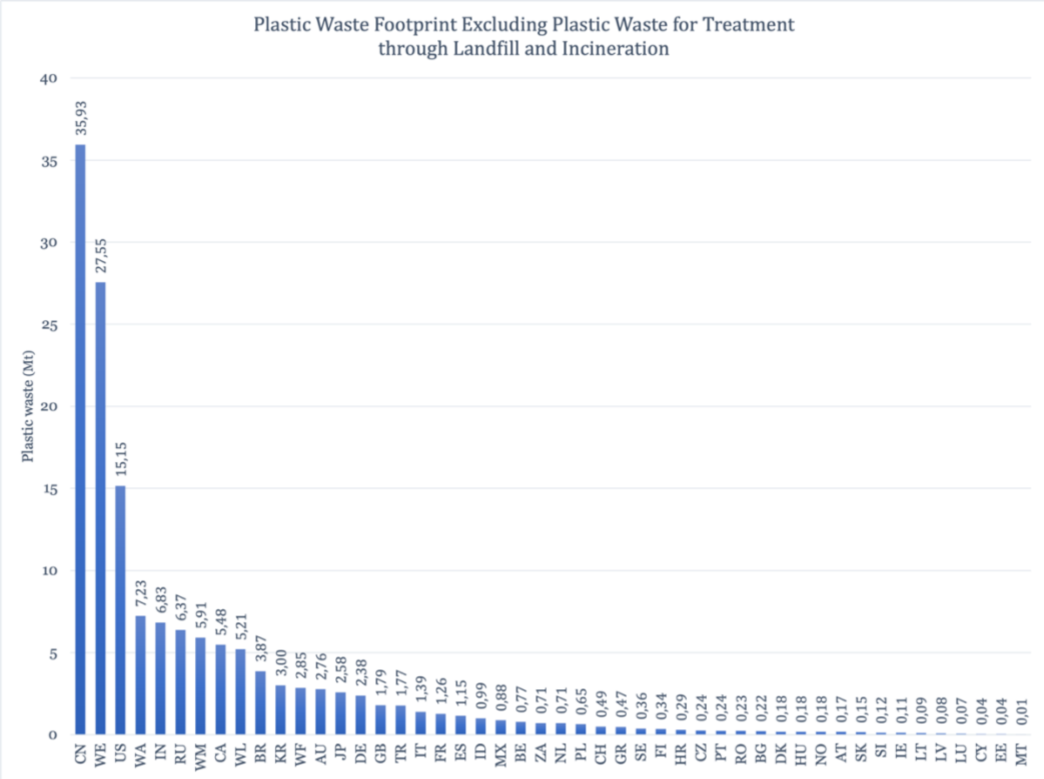


Figure 7. Revised Plastic waste footprint by country/region (in Mt)

As EE-MRIO models evaluate environmental impacts from a consumer-driven approach, countries and regions with larger population sizes such as China may be overrepresented when comparing footprints in total plastic waste values. Similarly, high-income countries may have higher consumption levels and therefore generate more plastic waste. To capture these potential effects population and income data were incorporated in the analysis[5]. Figure 8 shows the relationship between the revised measure of the PWF and income per capita for the different countries, therefore, it is possible to assess whether the environmental Kuznets Curve (EKC) holds. If decoupling between economic growth and environmental degradation (in this case plastic waste pollution) exists, high-income countries will show a tendency towards smaller PWFs compared to other countries with lower income. Conversely, the disproportionate effects of plastic pollution in developing countries (UNEP, 2019a) may be explained by telecoupling. In general, the results obtained point towards the latter and high-income countries tend to have relatively higher PWFs.

[5] The statistics have been extracted from The World Bank's open database. For consistency with the rest of the analysis, 2011 was taken as the reference year for population and income levels.

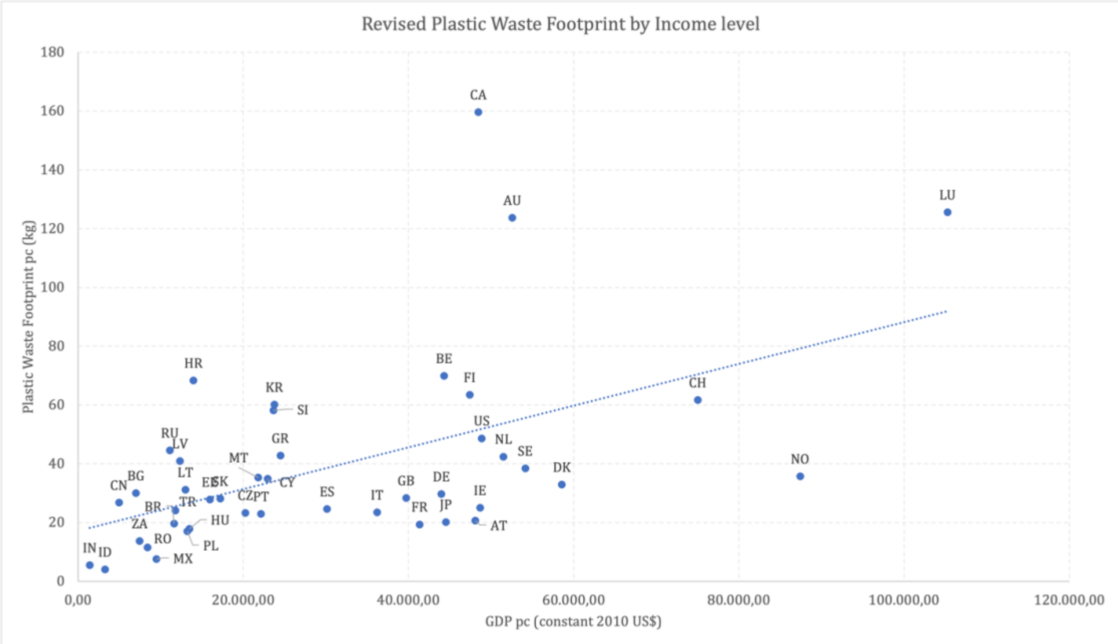


Figure 8. Revised plastic waste footprint and income per capita. excluding Rest-of-the-world regions

Canada, Australia and Luxembourg have a very large PWF compared with other high-income countries and do not show patterns of decoupling. Norway and Switzerland have higher income per capita but show a PWF similar to other EU countries with relatively lower income levels. At the same time, a group of EU countries including Spain, Italy, France, Austria, Great Britain[6], Germany, Ireland plus Japan show a mild pattern of decoupling. A second group of EU countries including Belgium, Finland, the Netherlands, Sweden and Denmark plus the US show a decreasing PWF as income increases, however, compared to the footprints of other lower-income countries, these rich countries, particularly Belgium and Finland, have significantly higher footprint and the EKC is not realised. India and Indonesia are the two countries with lowest income level and present the lowest PWF, however, plastic waste pollution is a much larger challenge and both countries are high contributors to ocean plastic pollution (Meijer et al., 2021). If the EKC was to hold, it is expected that the PWF in these countries will increase significantly as high population levels and rising living standards have been associated with driving plastic consumption up (Phelan et al., 2020; Geyer, Jambeck & Law, 2017). China appears grouped with countries with relatively lower income levels, although in total values China is distinctively the country with the largest waste footprint trends in this relationship leads to the belief that unless radical change in the patterns of consumption, China's PWF will continue to increase which is worrisome in terms of the implications for waste management systems. Croatia, Slovenia and South Korea have large PWFs compared to other countries in their income level and the same can be observed for Russia, Latvia and Greece although at a lower extent.

[6] In 2011, Great Britain was an EU member, therefore, in the development of this analysis Great Britain is considered an EU country.

5.2 Treatment of Plastic Waste

This section focuses on the product categories identified as "plastic waste for treatment" in order to analyse the treatment methods (landfill, incineration and re-processing) that are driving the "reduction" of a country's PWF. Appendix C (Table 6) displays in relative percentages how countries handle the treatment of plastic waste generated across their supply chains.

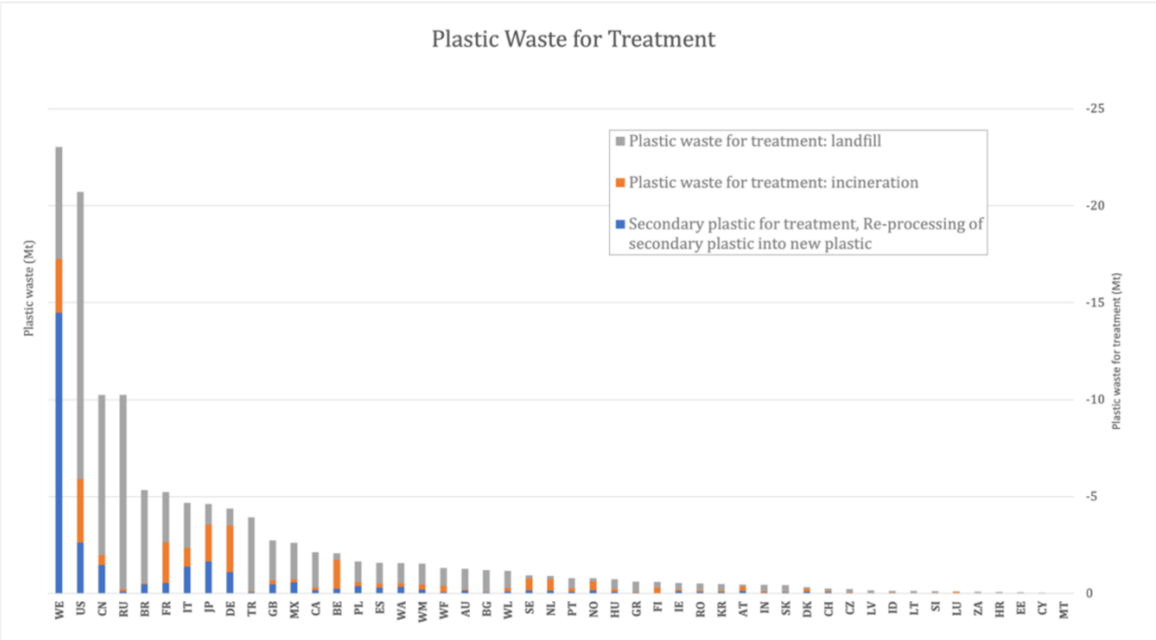
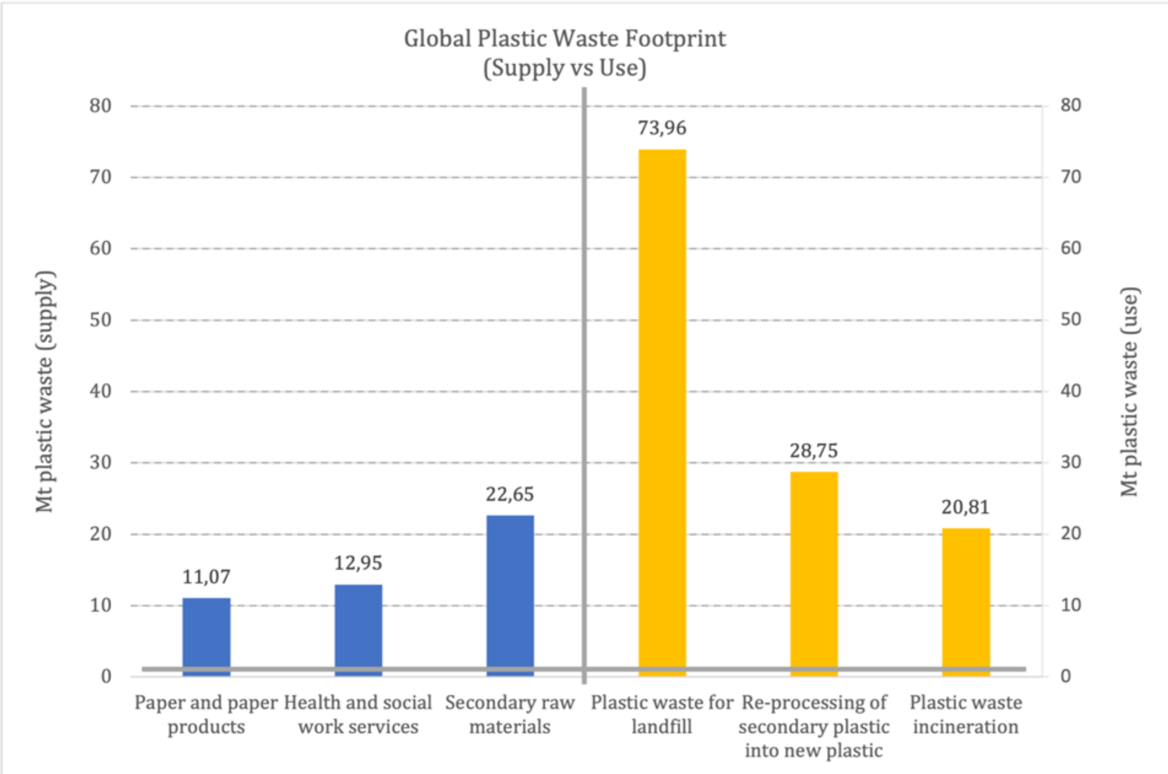


Figure 9. Plastic Waste for treatment, by treatment method: landfill, incineration and re-processing (Mt)

Figure 9 shows that landfill and incineration are the two methods most countries were reducing their PWF, except for RoW Europe where the "Re-processing of secondary plastic into new plastic" is substantial. Japan, Italy and Germany also present relatively high levels of plastic recycling; however, incineration is still more common in Japan and Germany, and landfill in the case of Italy. The large amount of plastic waste that is sent to landfill in the US and Canada is alarming, even if from a waste mismanagement perspective, the risk of plastic entering the natural environment is much smaller in high income countries (Jambeck et al., 2015), from a resource use perspective this is not a desirable outcome. China, Russia, Brazil, Turkey and Bulgaria also present high rates of landfilling when it comes to treatment of plastic waste whereas Belgium, Sweden, Netherlands, Norway, Luxembourg and Germany show high relative rates of incineration. Japan, Italy, Austria, Denmark, Switzerland, Malta, and Germany are the countries with the highest relative percentage of recycling plastic waste into secondary materials for new plastic.

Finally, Figure 10 shows a comparison in absolute values of the PWFs obtained by the three products with the highest impact on plastic waste generation (supply) on the left-hand-side and three plastic waste treatment (use) categories on the right-hand-side. According to the revised measure proposed (section 5.2.1), if plastic waste sent to landfill is considered as increasing the PWF of the economy, it will be the single sector with the largest impact in the global supply chain. The implications are similar for incineration of plastic waste which in absolute terms is the product with the third largest footprint.



To the left of the vertical line, the three products that generate the largest footprint (supply). To the right of the vertical line, the three products with negative footprint (use) with values presented in absolute terms.

Figure 10. Comparison of Global Footprint: Supply and Use of Plastic Waste

5.3 Plastic Waste Footprint by Product

After analysing the impact of the plastic waste for treatment categories, this section turns to examine the PWF of the remaining products included in EXIOBASE 3 MR-HIOTs in the global supply chain. Figure 11 summarises the 30 product categories with the largest PWF in the global supply chain based on the author's calculations. From these, four product categories can be highlighted: "Secondary raw materials" represents a 12.7% of the total global PWF, followed by "Health and Social Services" with a 7.2%. While at first it might come as a surprise for a service sector to be the second largest contributor to the plastic waste, there is a large amount of single-use personal protective equipment (PPE) that is made of plastic. The recent events of the Covid-19 pandemic have made this reality more evident due to a significant increase in the use of PPE, masks, gloves and other biomedical plastic waste that has the potential to undermine the efforts to reduce global plastic pollution in previous years (Benson, Bassey & Palanissami, 2021; EEA, 2021; Selvaranjan et al. 2021). "Paper and paper products" (6.2%) and "Basic iron and steel and of ferro-alloys and first products thereof" (4.3%) are the third and fourth products, respectively, with a higher contribution to the global PWF.

"Construction work" is another product account identified as a high generator of plastic waste as plastics have indeed several applications in the construction sector such as pipes, cables, and insulation and as a result waste is generated. Similarly, plastics have several applications in the automotive industry and this is reflected by the waste footprint in product categories such as "Motor vehicles, trailers and semi-trailers" and "Sale, maintenance, repair of motor vehicles, motor vehicles parts, motorcycles, motorcycles parts and accessories" and the production of machinery and electronics (EEA, 2021). Products such as "Rubber and plastic products" and "Plastics, basic" itself generate a high PWF. Similarly, products related to the extraction and refinement of petroleum such as "Refined Petroleum", "Crude petroleum and services related to crude oil extraction, excluding surveying" also leave a significant PWF. It is relevant to keep in mind that plastic materials are derived from petroleum.

In recent years, awareness has been raised about the environmental impacts related to the fashion industry, one of them being the millions of plastic microfibers released to the oceans (UNEP, 2019b). The results here show "Textiles" products and "Leather and leather products" as one of the main contributors to the global PWF. In addition, the consumption-driven approach of EEIO enables to unveil the reallocation of plastic waste towards services activities such as "Public administration and defence services; compulsory social security services" and "Hotel and restaurant services", "Education services", "Research and development services" and "Membership organisation services" which reflect that even products that do not have an obvious component of plastic in the moment of consumption, upstream in the supply chain there might be processes that are intensive in the use of plastics and create waste (Fry et al., 2015).

Finally, "Food products" also play an important role in consuming plastic. Since EXIOBASE offers a highly detailed breakdown of primary products used in the farming industry, it is possible to identify specific accounts such as "Pigs", "Poultry", "Raw Milk" or "Cattle" and "Fish products" as being relatively high plastic-intensive products as well as "Beverages" and "Collected and purified water, distribution services of water" may reflect the use of plastics as packaging to transport water. The intuition is that the PWF related to these activities is driven by the use of packaging to a larger extent (Iacovidou & Gerassimidou, 2018).

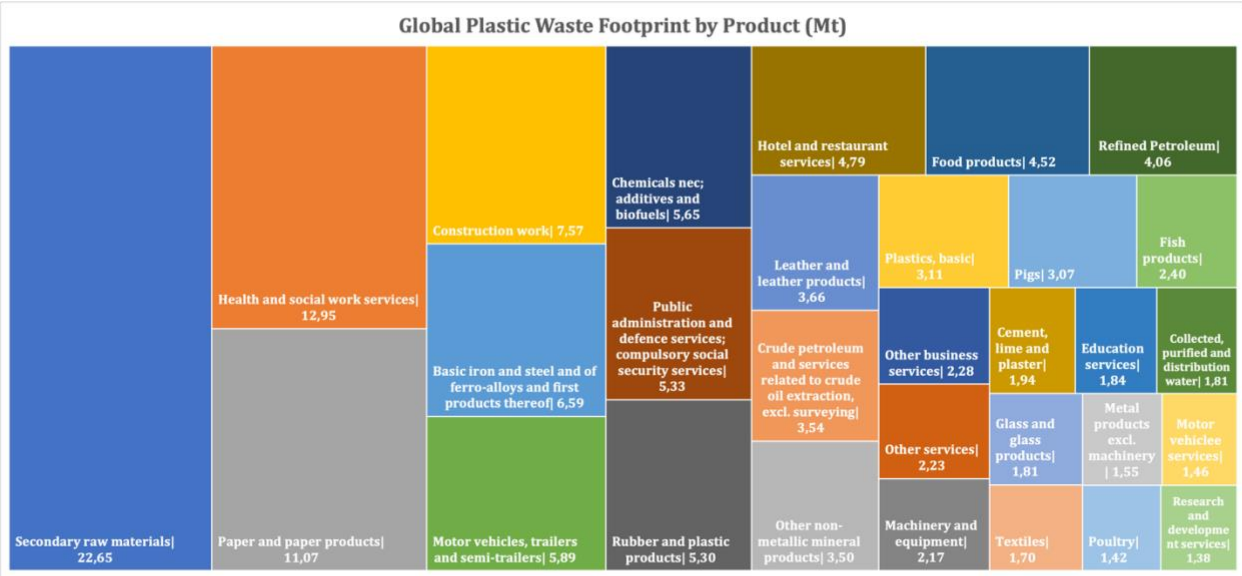


Figure 11. Global Plastic Waste footprint by Product (in Mt)

5.4 Plastic Waste Footprint by Product by Country

The results in the previous section show that four main product categories generate a significant proportion of the global waste footprint. Figure 12 shows which countries are behind the consumption of these plastic-intensive products and therefore driving the generation of plastic waste at the global level. Figure 12a shows that it is mostly solely RoW Europe (WE) contributing to the generation of plastic waste in the "Second raw materials" product category which can be matched to the treatment "Re-processing of secondary plastic materials" identified in the previous section. This result is surprising and it makes up almost the total amount of the calculated PWF in RoW Europe, although it is not the only product contributing to it (Table 1). In addition, most European economies appear disaggregated in EXIOBASE, therefore, it is difficult to infer what is driving this result.

In Figure 12b it can be observed that a very significant proportion of the global PWF attributable to "Health and social work services" is mostly given by China's impact. Similarly, the plastic footprint generated by "Paper and paper products" is mostly a result of consumption in China and the US (Figure 12c). Finally, the footprint of "Construction work" (Figure 12d) is much more spread across several countries such as China, the aggregation of countries in Latin America, Japan, the US and RoW Middle East (WM).

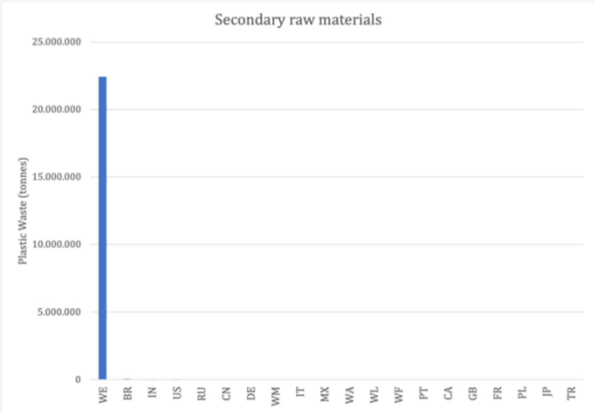


Figure 12a. PWF for "Health and social work services" by country/region.

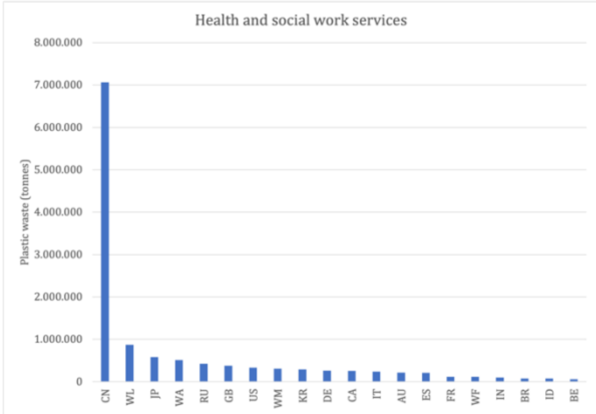


Figure 12b. PWF for "Health and social work services" by country/region.

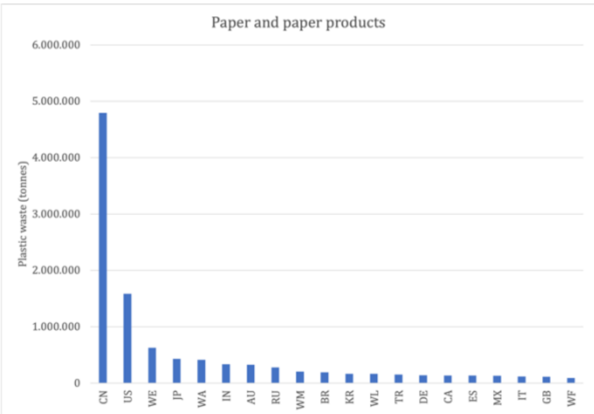


Figure 12c. PWF for "Paper and paper products" by country/region.

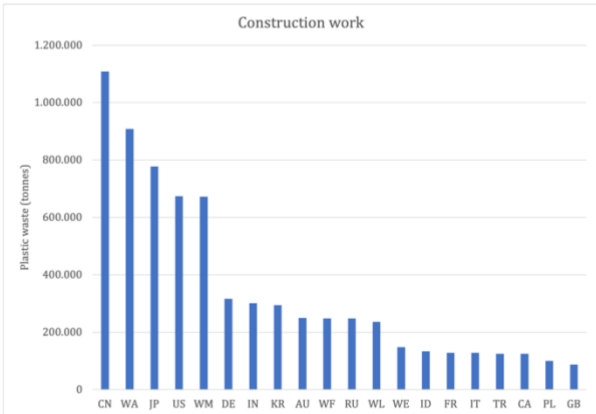


Figure 12d. PWF for "Construction work" by country/region.

Figure 12. Country breakdown for plastic-intensive products

As several service categories in this study have appeared as driving significant plastic waste generation in their supply chains, Figures 13-15 illustrate which countries are responsible for driving plastic waste in four service categories with the largest footprints combining the total the per capita values in each case. Overall, China continues to be the country with the largest impact for almost all service categories in absolute terms, however, considering PWF per capita, positions China in a much more moderate position. The service category "Public administration and defence" is an exception as the US and Canada have a higher impact, both in absolute and per capita terms, while all the other countries show much lower levels. Furthermore, Luxembourg, Australia and Canada are the countries with the largest per capita PWF in "Health and social worker services" whereas in the other side of the spectrum appear countries with lower income levels such as Indonesia, India and Mexico. In the category "Hotel and restaurant" services, Croatia generates a relatively large amount of plastic waste of over 16kg of plastic waste per inhabitant, followed by Australia and Luxembourg which generate 9kg and 7,5kg of plastic waste per inhabitant, respectively. A possible explanation may be that while these countries have relatively small populations, they receive many visitors through tourism.

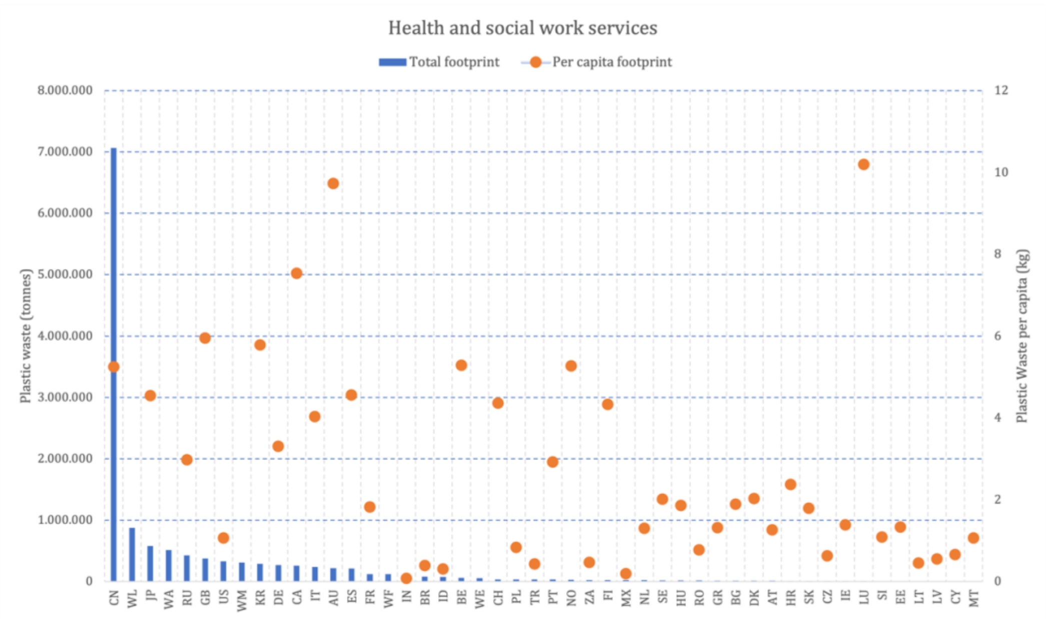


Figure 13. PWF Health and social work services by country (in total and per capita values)

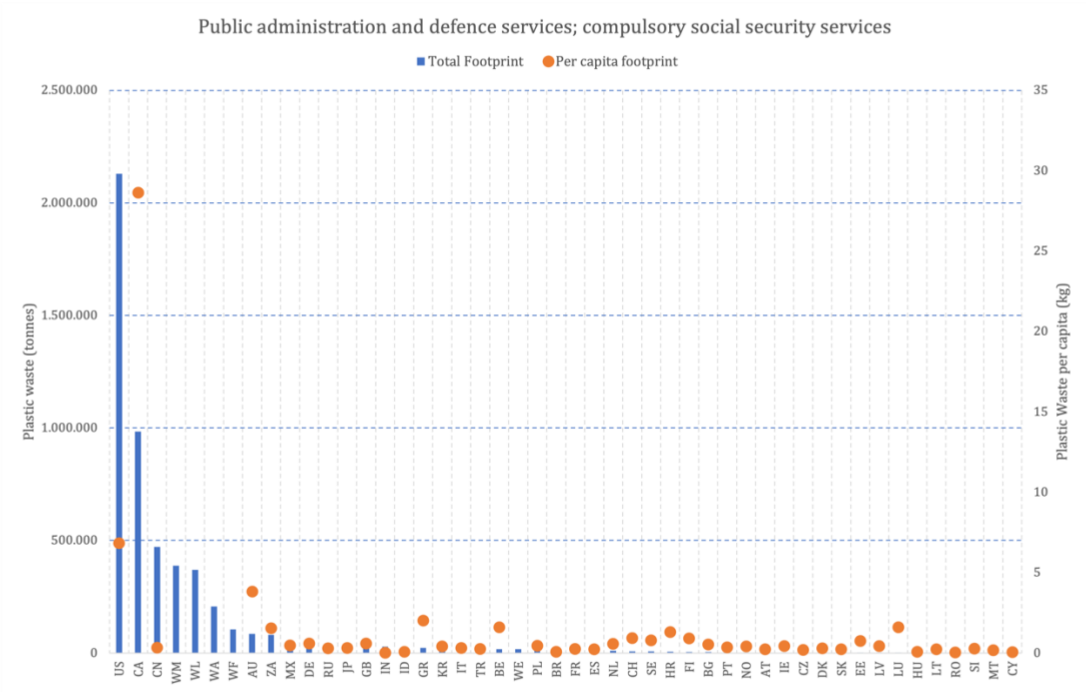


Figure 14. PWF Public administration and defence services by country (in total and per capita values)

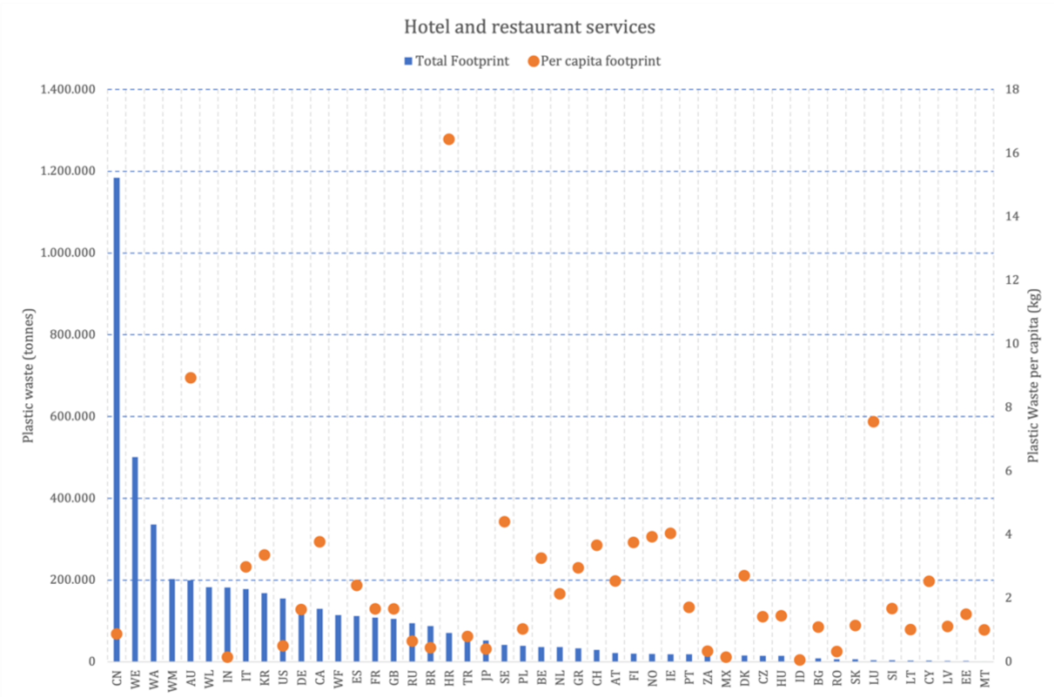


Figure 15. PWF Hotel and restaurant services by country (in total and per capita values)

Table 1 summarises the 20 products with the highest PWF by country whose consumption drove the production of plastic waste, omitting the effect of plastic waste for treatment in landfill and incineration. Interestingly, RoW Europe, US and China are the three main generators of plastic waste even at the product level. Furthermore, the list of products in Table 1 are aligned with the observations in Figures 12a-12d. At the product level, the breakdown by country shows that the production of "Secondary raw materials" in the WE region is the product with the highest impact, followed by "Health and Social services" and "Paper and paper products" in China and "Public administration and defence services; compulsory social security services" and "Basic iron and steel and of ferro-alloys and first products thereof" in the US. Thus, the information in this table reiterates what this analysis has found so far, the production of few products in a few countries and regions dominate the global plastic waste footprint.

Table 1. Plastic Waste Footprint by Product by Country

Country / Region	EXIOBASE Product	Plastic waste footprint (Mt)	Country/ Region	EXIOBASE Product	Plastic waste footprint (Mt)
RoW Europe	Secondary raw materials	22,4345	China	Plastics, basic	1,5628
China	Health and social work services	7,0628	United States	Cement, lime and plaster	1,1946
China	Paper and paper products	4,7932	China	Hotel and restaurant services	1,1839
RoW Europe	Motor vehicles, trailers and semi-trailers	2,2958	China	Other services	1,1555
China	Chemicals n.e.c.; additives and biofuels	2,1645	China	Crude petroleum and services related to crude oil extraction, excluding surveying	1,1354
United States	Public administration and defence services; compulsory social security services	2,1296	China	Other non-metallic mineral products	1,1164
United States	Basic iron and steel and of ferro-alloys and first products thereof	2,1079	China	Construction work	1,1080
RoW Europe	Collected and purified water, distribution services of water	1,6930	RoW Europe	Precious metal ores and concentrates	1,0750
China	Pigs	1,6225	RoW Europe	Rubber and plastic products	1,0289
United States	Paper and paper products	1,5874	Canada	Public administration and defence services; compulsory social security services	0,9839

To close this section, and since the list of countries included in the analysis is extensive, the author extends the analysis of the PWF by product by country to the next four countries that appeared as having the larger impact: Canada, India, Russia and Brazil.

As shown in Table 1, “Public administration and defence services; compulsory social security services'' is single product with the largest PWF in Canada. Furthermore, characteristic of a high-income nation, Canada’s PWF (Figure 16) is mostly driven by service sectors such as health and social services, hotel and restaurants and other retail services. In India (Figure 17 17), “Rubber and plastic products” is the product driving the PWF by a difference and plastic waste for landfill appears as the second category. In most other countries, however, the value for landfill is much larger which may indicate that a lot of the plastic waste disposal in India is taking place through uncontrolled methods or open dump sites. This is problematic as mismanagement of waste is not only a potential environmental risk but also a health hazard (Ferronato & Torretta, 2019). Unlike in Canada products with higher plastic waste impact in India are given by manufacturing activities. In Russia (Figure 18) and Brazil (Figure 19), the generation of the plastic waste footprint is distributed more heterogeneously across several products. In Russia, the plastic waste footprint is generated by a mix of services (health and social work; research and development, education) and industrial sectors (refined petroleum, rubber and plastic products, iron and steel) with the particularity of sectors such as “Steam and hot water supply services and “Distribution services of gaseous fuels”. In Brazil the distribution of products is more like what was observed in India in the sense that plastic waste is mostly allocated to manufacturing products such as the leather industry and agricultural products such as cattle, raw milk, pigs and poultry have a significant footprint and food products.

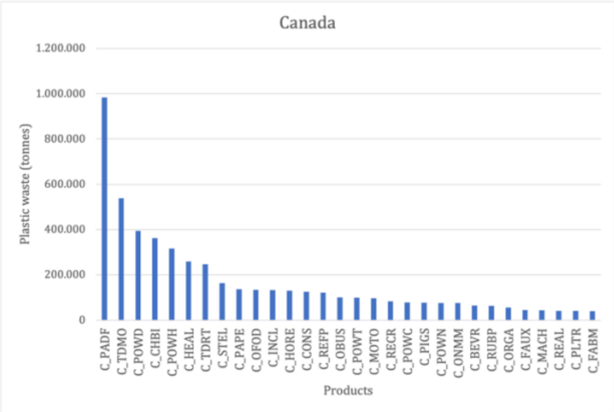


Figure 16 Main products driving the plastic waste footprint in Canada (in tonnes)

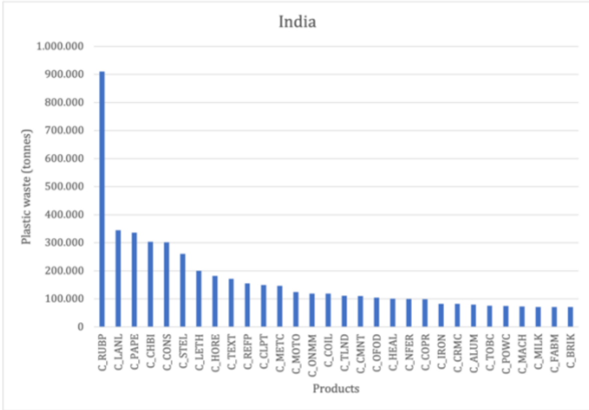


Figure 17. Main products driving the plastic waste footprint in India (in tonnes)

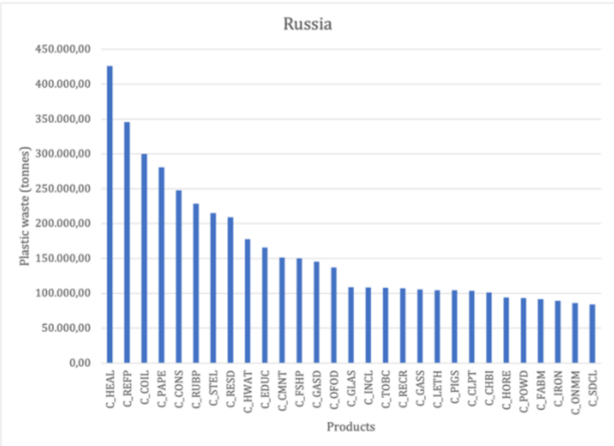


Figure 18. Main products driving the plastic waste footprint in Russia (in tonnes)

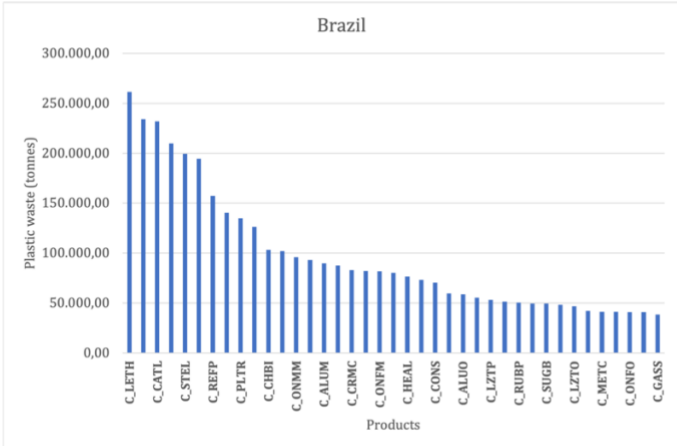


Figure 19. Main products driving the plastic waste footprint in Brazil (in tonnes)

6 Discussion

This study has calculated the PWF of 48 countries and regions for the year 2011. In this chapter, section 6.1 connects the results obtained in the analysis to the research questions proposed at the beginning of this thesis, evaluates the hypotheses and discusses the implications of the main findings. Section 6.2 discusses the limitations of the study and identifies potential lines for future work.

6.1 Discussion of Results and Implications

The first research question of this thesis aimed to assess which countries have the larger impact in terms of generation of plastic waste and whether signs of decoupling between economic development and the PWF exist.

Hypothesis I established that high-income countries would present a higher PWF, therefore, the decoupling effect suggested by the EKC would not be achieved. Instead, developing countries are disproportionately suffering from plastic waste generated as a result of consumption in high-income countries. The results of the analysis show that China is the country with the largest PWF in the global economy, and while in 2011 it was still classified as a developing economy, it is also the country with the largest population. The calculation of per capita PWFs has revealed that consumption in high-income countries, particularly in Canada, Luxembourg and Australia are much more plastic waste-intensive and these countries do not show patterns of decoupling. Nevertheless, EU countries where CE policy intervention has increased over the last years, show a mild pattern of decoupling and relatively higher rates of recycling. While the CE Action Plan in EU was approved in 2015 (EC, 2015) and "Europe's New Plastics Economy" in 2018 (Crippa et al., 2019; EC, 2018) and the data used from this analysis is from a previous year, the EU has been on a journey since the 1970s towards improving environmental standards (Deutz, 2020), including plastic waste.

On the other side of the spectrum, while plastic pollution is most problematic in developing countries and particularly in the Asian continent (Jambeck et al. 2015; Meijer et al., 2021), Indonesia and India are the two countries with the lowest per capita PWF. Nonetheless, like China, due to its high population, India appears as one of the countries with the largest impact when measured in absolute values. If these countries follow a development pattern similar to that of high-income countries, the amount of plastic waste that can potentially leak into the natural environment can be catastrophic (Browning, Beymer-Farris & Seay, 2021). According to the results, although taking into consideration that only a few developing countries are represented in the study, it appears that the most significant challenge related to plastic pollution in developing countries may not be as much a consequence of consumption patterns leading to relatively higher levels of plastic waste, but the result of having underdeveloped or inefficient waste management systems (Jambeck et al., 2018; Jones, 2021; Watson, 2020) which lead to recycling rates being particularly low and high risk of mismanagement of plastic waste (Jambeck et al., 2015).

Therefore, it leads to the conclusion that generally speaking consumption in high-income countries tends to generate relatively higher levels of plastic waste. As decoupling between economic growth and PWF is not overwhelmingly clear, it is important that these countries focus on reducing their impact. The transition to the CE of plastics has the potential to address this challenge by prioritising measures of plastic waste prevention and channelling solutions higher up in the waste hierarchy. Another way developed countries can support developing countries alleviate environmental and social challenges linked to plastic pollution is by stopping plastic waste exports such as the 2019 Plastic Waste Amendment of the Basel Convention (Basel Convention, 2019), and by helping develop waste management capabilities to mitigate for the negative effects linked to international trade.

The second research question of this thesis hinges on the economy-wide approach that the calculation of footprints provides in order to identify opportunities to improve circularity in the plastic industry by highlighting plastic-intensive products in the supply chain.

Hypothesis II was interested in testing whether a reallocation effects of plastic waste at the sectoral level, from products in the primary and manufacturing sectors towards trade and services occurs as a result of accounting for both, direct and indirect effects of waste generation throughout the supply chain as discovered on Fry et al., (2015)'s study of Australian waste footprints. The results obtained by this work show a similar reallocation pattern. The service category with the higher plastic waste impact is health and social work, which can be associated with personal protective equipment (PPE) typically made of single-use plastics (Benson, Bassey & Palanissami, 2021; Selvaranjan et al. 2021). In addition, public administration and defence, particularly in the US and to a lesser extent Canada; hotels and restaurants; education, research and developments represent a larger proportion of the PWF than other manufacturing activities both at the global level and particularly in high-income countries. This effect may also be explained by these products typically having longer supply chains which creates more opportunities to accumulate plastic waste (Ruíz-Peñalver, Rodríguez & Camacho, 2019). In contrast, in developing countries agricultural and manufacturing products still account for most of the PWF as exemplified by the product PWF analysis of in India and Brazil. Therefore, a transition towards a service economy does not guarantee reduced environmental pressures and it requires strong policy and governance (Henriques & Kander, 2010).

The product analysis shows that, generally, products associated with main end-use categories of plastic production such as construction, automotive, electronics and textiles (EEA, 2021) rank amongst those with a larger PWF, apart from the service sectors explained above. In that sense, the approach provided by EE-MRIO analysis reveals that even when plastic in not apparent in a product or service, upstream in the supply chain there might be plastic-intensive processes that generate waste, which otherwise would go unnoticed (Fry et al., 2015). Similarly, plastic waste related to food consumption categories can be explained by the heavy use of packaging, the single largest end-use market for plastic production, and other single-use plastics, which has become one of the main areas of action to prevent plastic pollution (MacArthur Foundation, 2016; Iacovidou & Gerassimidou, 2018; Nakatami et al. 2020). It is important to recognise that plastic as a material brings many benefits and eliminating the use of plastic materials completely from production systems may not necessarily be a desirable objective. However, re-evaluating the need of plastics for certain products, the types of plastics used so that recyclability can be improved, looking for alternative biodegradable materials with similar characteristics to plastics and ensuring that there are mechanisms in place to minimise the disposal of plastic waste are some of the pillars the CE is built upon (Deutz 2020; Hahladakis & Iacovidou, 2018; Watson 2020).

EXIOBASE dataset includes three main categories relevant for the treatment of plastic waste which allows to examine the potential to reduce or even “offset” the PWF. The revised measure proposed by this study excluded landfilling and incineration to better align the analysis of PWFs with the principles of the CE. The results show that most plastic waste is “treated” through landfilling, and not taking this effect into consideration could underestimate the real PWF in an economy. The analysis of plastic waste treatment methods is aligned with the study by Geyer, Jambeck & Law (2017) who estimated that 12% of global plastic waste is incinerated and 79% accumulates in landfill. Treatment of plastic waste through incineration warrants further considerations as, in the context of the CE, although placed at a lower level of the waste hierarchy, this treatment method is often presented as a “waste-to-energy” mechanism for dealing with solid waste such as plastic and is common practice in a number of European countries, particularly in the Nordics as the results show. Similarly, for certain plastic materials which are too complex for current recycling systems, incineration may be a more sustainable approach as it reduces the risk of contamination of recycling materials (Hahladakis & Iacovidou, 2018). However, concerns over emissions generated by the incineration of plastic waste along with the inherent loss of material efficiency, place recycling of plastic waste as a preferred method according to the waste hierarchy.

Over the last decades, Europe has been pioneering the adoption of policies aiming towards waste management strategies at higher levels of the waste hierarchy (Deutz, 2020) and this is already reflected in this analysis as in general most European countries do not stand out as countries with the largest waste footprints, Luxembourg being an exception in some cases. Similarly, RoW Europe presents the largest amount of plastic waste re-processing of secondary materials, which has the potential to displace the consumption of virgin plastic materials.

6.2 Limitations of the Study and Areas of Future Research

Although this research has been carefully carried out and the research aims presented at the beginning of the process have been met, there are several weaknesses some of which have been identified throughout this thesis and opportunities to expand this line of work in the future.

The first limitation is related to the methodological approach employed for the analysis. While EEIO analysis has become a widely used method to perform environmental impact assessments derived from economic activities by scholars, the basic IO framework relies on three main assumptions that simplify the complexity of economic systems to be able to arrive the standard IO mathematical relationship (section 4.1) (Munroe & Biles, 2005). First, the assumption of constant returns to scale establishes that the technical relationship between input and output given by the coefficients in matrix **A** is fixed for a given sector, ruling out the potential for industries to operate under economies of scale. The second assumption of IO is proportionality which assumes that all the different production agents within a sector in an economy use inputs in fixed proportions (i.e., *aij* is the same for all companies). The third IO assumption requires that all companies use the same input-mix and there is no substitution between different inputs. Another controversial assumption of MR-IO models is similar technology in all countries. Extended models that consider country-specific technological capabilities have been implemented in IO assessment of emissions embodied in international trade (Baumert et al., 2019; Kander et al., 2015). However, in the calculation of waste footprints it remains a relevant limitation (Towa, Zeller & Achten, 2020).

A second limitation characteristic of EEIO models is related to uncertainties that arise from the complexity and data-intensive processes required to build IO tables. For this study in particular, the collection of physical IO data and waste data would benefit from a global common framework that defines and classifies waste consistently across territories (Watson, 2020) and establishes harmonized standards for waste statistics compilation, like the efforts made by the FORWAST project of the EU which has enables the development of EXIOBASE 3 hybrid tables (Towa, Zeller & Achten, 2020). In addition, EEIO is given by the representation of an economic system as an aggregation of sectors. Low resolution IO models in terms of economic sectors can lead to inaccurate calculations of material footprints (De Koning, 2018). De Koning (2018) consider 60 sectors still low-resolution model, however, the author has not considered this a major shortcoming for this study as EXIOBASE 3 includes 164 different products

Another data-related potential limitation of this analysis is based on the data availability for waste extensions as of the main MR-IO databases, only EXIOBASE includes waste accounts in its hybrid-unit version. In addition, the hybrid version of EXIOBASE is only available for the year 2011. Therefore, this study has been limited to provide a snapshot of PWFs for the year 2011. If EXIOBASE's MR-HIOTs are extended to cover more years, further research work could extend the analysis of PWFs on time series data to identify trends and patterns over time. Similarly, the data might be outdated as since 2011 there has been a significant uptake in CE strategies both from practitioners and policymakers. Another shortcoming of EXIOBASE is that EU countries are overrepresented and most developing countries are represented in rest-of-the-world (RoW) aggregated regions. Thus, while there is increasing interest to assess CE interventions with IO models, more comprehensive physical and hybrid representations of the economy are still lacking (Aguilar-Hernandez et al., 2018; Towa, Zeller & Achten, 2020).

Finally, MR-IO analysis entails a considerable level of complexity given by the large amount of data that is involved. Given time constraints, the scope of this work has focused on summarising the results for countries and products that create the largest PWF at a global level. However, the product classification of EXIOBASE is quite detailed and future work could analyse footprints for other products that might have been overlooked by this work more carefully. At the same time, while this work has attempted to cross-analyse which products have the largest impact for several countries, further studies could be extended to analyse more thoroughly trade flows between trade partners and the implications for PWFs. This work has provided a novel approach by developing an EE-MRIO analysis to examine the PWF using hybrid units IO tables. The study has focused on the environmental impacts of plastic pollution given by the material flows of waste; however, the environmental impacts of plastics have several dimensions that need to be considered to achieve a fully CE of the plastic industry. This work could be extended with an assessment of emissions generated throughout the plastics life cycle, from the extraction and production of new virgin plastic materials all the way to end-of-life management (EEA, 2021) as data for emissions is available on the environmental extensions of EXIOBAE MR-HIOTs.

7 Conclusion

Plastics have become an essential material in modern day society thanks to its unique characteristics and many applications. However, the boom of plastics has led to uncontrolled amounts of plastic waste accumulating in landfills and the natural environment. Plastic pollution is one of today's grand environmental and societal challenges as it poses a major threat to marine and coastal ecosystems affecting disproportionately coastal communities in developing countries. In an attempt to mitigate the negative impacts of plastic waste pollution, the CE has emerged as an alternative paradigm to linear models of production and consumption with the ultimate goal of creating a closed-loop system where waste flows are minimised and instead turned into resources.

In today's globalised world, international trade flows are an essential feature of economic systems in which consumption and production have become increasingly distant. As a result, the environmental impacts related to a country's economic activities are oftentimes suffered beyond its borders. Thus, plastic waste pollution needs to be addressed from the point of view of global supply chains. This research work has developed a MR-EEIO analysis utilising EXIOBASE 3 hybrid input-output tables and waste extensions to calculate the plastic waste footprint of 48 countries and regions, or plastic waste *embodied* in international trade. The calculation of footprints provides a consumption-driven approach to the allocation of waste that traces plastic waste generated by economic activities throughout the global supply chain both directly and indirectly.

Two research question have guided this study:

RQ1: What are the countries with the largest plastic waste footprint in the global economy and to what extent does it reflect patterns of decoupling?

R21: What are the key sectors in the plastics value chain that can help advance the circular economy for plastics?

Plastic waste footprints (PWF) were obtained considering the net effects of products that generate (supply) and plastic waste collected for treatment (use), where the latter has the potential to reduce a country's impact. Treatment methods for plastic waste as provided in EXIOBASE include landfilling, incineration and re-processing of materials. This study has proposed a revised measure better aligned with CE principles where landfilling and incineration do not count towards reducing a country's impact. The results have shown that China is the country with the largest PWF equal to nearly 36Mt of plastic waste, followed by the US with a footprint equal to 15,15 Mt of plastic waste. The region including an aggregate of European countries also has a large PWF equal to 27,55 Mt. A more detailed analysis has incorporated population and income data in order to examine the relationship between plastic waste pollution per capita and economic development of countries. In general, high-income countries have relatively higher PWFs, particularly in Australia, Canada and Luxembourg. Nevertheless, a group of EU countries and Japan show a slight tendency towards decoupling. The results obtained in the analysis of plastic waste treatment methods are aligned with previous studies showing that landfill is the most used method for plastic disposal (Geyer, Jambeck & Law, 2017). Since most plastic waste is non-degradable, this study does not consider landfilling as a method to reduce the PWF, instead it contributes to the global footprint with 74Mt of plastic. This number is mostly driven by the US and Canada while other countries with high rates of plastic waste to landfill are China, Russia, Brazil and Turkey. Incineration of plastic waste is most common in Northern European countries whereas Japan and other EU countries show the highest rates of plastic waste re-processing. To partially answer the first research question, this study concludes that economic development is not necessarily linked to a reduction of plastic-intensity of economic activity. However, if policy intervention incentivises the reduction of plastic consumption and favours domestic recycling over other methods of waste disposal, the PWF can be reduced.

To answer the second research question, the calculation of plastic waste footprints as an alternative to territorial waste accounting provides a more holistic approach to the evaluation of supply chains and it allows to identify hotspots for CE policy intervention. For example, the analysis of the plastic waste by product shows that a small number of products had a significant impact in the global PWF, including "Secondary raw materials" (mostly driven by European consumption), "Health and social services" (mostly driven by China), "Paper products" (mostly driven by China and the US) and "Construction work". On the one hand, the concentration of plastic waste in a few products provides an opportunity for targeted action. On the other hand, while it may seem surprising that a service sector is associated with high levels of plastic waste, the calculation of footprints reveals a reallocation of plastic waste towards end of supply chain sectors (Fry et al., 2015) and may slowdown the decoupling of economic growth and environmental degradation. This is explained by the potential indirect effects of all other production activities upstream in the supply chain that may be plastic-intensive that otherwise could go unnoticed by policymakers.

Furthermore, the calculation of PWFs has revealed that plastic pollution is not only an issue of developing countries where the effects are most suffered, but instead the generation of plastic waste is driven by global demand, particularly from developed countries where consumption levels are higher. Thus, high-income countries can reduce their PWF by (1) eliminating or reducing the consumption of plastic from economic activities when possible and (2) improving their domestic plastic waste methods to significantly reduce the disposal of plastic waste to landfill and increase recycling rates. In that line of action, part of the EU Action Plan for the Circular Economy (EC, 2015), a strategy for plastics in a circular economy was launched in 2018 which has the potential to enable systemic change.

References

Aguilar-Hernandez, G. A., Sigüenza-Sanchez, C. P., Donati, F., Rodrigues, J. F. D. & Tukker, A. (2018). Assessing Circularity Interventions: A Review of EEIOA-Based Studies, *Journal of Economic Structures*, vol. 7, no. 1, Available Online: https://doi.org/10.1186/s40008-018-0113-3 [Accessed March 20, 2020].

Antelava, A., Damilos, S., Hafeez, S., Manos, G., Al-Salem, S. M., Sharma, B. K., Kohli, K. & Constantinou, A. (2019). Plastic Solid Waste (PSW) in the Context of Life Cycle Assessment (LCA) and Sustainable Management, *Environmental Management*, [e-journal] vol. 64, no. 2, pp.230–244, Available Through: SpringerLink https://doi.org/10.1007/s00267-019-01178-3 [Accessed May 1, 2021].

Barata, E.J.G. (2002). Solid Waste Generation and Management in Portugal: An Environmental Input-Output Modelling Approach, Conference Paper: Tunisia

Basel Convention. (2019). Basel Convention Plastic Waste Amendments, , Available Online: http://www.basel.int/Implementation/Plasticwaste/PlasticWasteAmendments/Overview/tabid/8426/Default.aspx [Accessed April 22, 2021].

Baumert, N., Kander, A., Jiborn, M., Kulionis, V. & Nielsen, T. (2019). Global Outsourcing of Carbon Emissions 1995–2009: A Reassessment, *Environmental Science & Policy*, vol. 92, pp.228–236, Available Online: https://doi.org/10.1016/j.envsci.2018.10.010 [Accessed March 18, 2021].

Benson, N. U., Bassey, D. E. & Palanisami, T. (2021). COVID Pollution: Impact of COVID-19 Pandemic on Global Plastic Waste Footprint, *Heliyon*, vol. 7, no. 2, p.e06343, Available Online: https://doi.org/10.1016/j.heliyon.2021.e06343 [Accessed May 21, 2021].

Benyus, J. M. (2002). Biomimicry : Innovation Inspired by Nature, New York, Ny: Perennial.

Blomsma, F. & Brennan, G. (2017). The Emergence of Circular Economy: A New Framing around Prolonging Resource Productivity, *Journal of Industrial Ecology*, vol. 21, no. 3, pp.603–614, Available Online: https://doi.org/10.1111/jiec.12603 [Accessed May 2, 2021].

Bocken, N. M. P., de Pauw, I., Bakker, C. & van der Grinten, B. (2016). Product Design and Business Model Strategies for a Circular Economy, *Journal of Industrial and Production Engineering*, vol. 33, no. 5, pp.308–320, Available Online: https://www.tandfonline.com/doi/pdf/10.1080/21681015.2016.1172124 [Accessed May 2, 2021].

Braungart, M., McDonough, W. & Bollinger, A. (2007). Cradle-To-Cradle Design: Creating Healthy Emissions – a Strategy for Eco-Effective Product and System Design, *Journal of Cleaner Production*, vol. 15, no. 13-14, pp.1337–1348, Available Online: https://doi.org/10.1016/j.jclepro.2006.08.003 [Accessed April 22, 2021].

Brooks, A. L., Wang, S. & Jambeck, J. R. (2018). The Chinese Import Ban and Its Impact on Global Plastic Waste Trade, *Science Advances*, vol. 4, no. 6, Available Online: https://advances-sciencemag-org.ludwig.lub.lu.se/content/4/6/eaat0131 [Accessed May 2, 2021]

Browning, S., Beymer-Farris, B. & Seay, J. R. (2021). Addressing the Challenges Associated with Plastic Waste Disposal and Management in Developing Countries, *Current Opinion in Chemical Engineering*, vol. 32, p.100682, Available Online: https://doi.org/10.1016/j.coche.2021.100682 [Accessed April 22, 2021].

Brundtland Commission. (1987). Our Common Future: The "Brundtland Commission" Report, *Ambio*, vol. 16, no. 4, pp.217–218, Available Through: JSTOR Journals https://www-jstor-org.ludwig.lub.lu.se/stable/4313360 [Accessed May 4, 2021].

Corvellec, H. (2016). A Performative Definition of Waste Prevention, *Waste Management*, vol. 52, pp.3–13, Available Online: https://doi.org/10.1016/j.wasman.2016.03.051 [Accessed May 4, 2021].

Crippa, M., De Wilde, B., Koopmans, R., Leyssens, J., Muncke, J., Ritschkoff, A-C., Van Doorsselaer, K., Velis, V. & Wagner, M. (2019). A Circular Economy for Plastics — Insights from Research and Innovation to Inform Policy and Funding Decisions, edited by M. De Smet & M. Linder, Brussels, Belgium: European Commission, Available Online: https://op.europa.eu/en/publication-detail/-/publication/33251cf9-3b0b-11e9-8d04-01aa75ed71a1/language-en/format-PDF/source-87705298 [Accessed May 2, 2021].

Davis, S. J. & Caldeira, K. (2010). Consumption-Based Accounting of CO2 Emissions, *Proceedings of the National Academy of Sciences*, vol. 107, no. 12, pp.5687–5692, Available Online: https://www.pnas.org/content/pnas/107/12/5687.full.pdf [Accessed April 23, 2021].

De Koning, A. (2018). Creating Global Scenarios of Environmental Impacts with Structural Economic Models, Doctoral Thesis, *Leiden University*, Available Online: https://hdl.handle.net/1887/62063 [Accessed May 11, 2021].

De Pascale, A., Arbolino, R., Szopik-Depczyńska, K., Limosani, M. & Ioppolo, G. (2021). A Systematic Review for Measuring Circular Economy: The 61 Indicators, *Journal of Cleaner Production*, [e-journal] vol. 281, p.124942, Available Through: ScienceDirect https://doi-org.ludwig.lub.lu.se/10.1016/j.jclepro.2020.124942 [Accessed May 14, 2021].

Deutz, P. (2020). Circular Economy, *International Encyclopedia of Human Geography*, pp.193–201, Available Online: https://doi.org/10.1016/B978-0-08-102295-5.10630-4 [Accessed May 2, 2021].

Dietzenbacher, E. (2005). Waste Treatment in Physical Input–Output Analysis, *Ecological Economics*, vol. 55, no. 1, pp.11–23, Available Online: https://doi-org.ludwig.lub.lu.se/10.1016/j.ecolecon.2005.04.009 [Accessed April 17, 2021].

Dinda, S. (2004). Environmental Kuznets Curve Hypothesis: A Survey, *Ecological Economics*, vol. 49, no. 4, pp.431–455, Available Through: ScienceDirect https://doi-org.ludwig.lub.lu.se/10.1016/j.ecolecon.2004.02.011 [Accessed May 14, 2019].

Donati, F., Aguilar-Hernandez, G. A., Sigüenza-Sánchez, C. P., de Koning, A., Rodrigues, J. F. D. & Tukker, A. (2020). Modeling the Circular Economy in Environmentally Extended Input-Output Tables: Methods, Software and Case Study, *Resources, Conservation and Recycling*, vol. 152, p.104508, Available Online: https://doi.org/10.1016/j.resconrec.2019.104508 [Accessed April 20, 2021].

Donges, J. F., Winkelmann, R., Lucht, W., Cornell, S. E., Dyke, J. G., Rockström, J., Heitzig, J. & Schellnhuber, H. J. (2017). Closing the Loop: Reconnecting Human Dynamics to Earth System Science, *The Anthropocene Review*, vol. 4, no. 2, pp.151–157, Available Through: SAGE Journals https://doi.org/10.1177%2F2053019617725537 [Accessed May 14, 2021].

Duchin, F. (1990). The Conversion of Biological Materials and Wastes to Useful Products, *Structural Change and Economic Dynamics*, vol. 1, no. 2, pp.243–261, Available Through: ScienceDirect https://doi.org/10.1016/0954-349X(90)90004-R [Accessed May 10, 2020].

Duchin, F. (2009). Input-Output Economics and Material Flows, in S. Suh (ed.), *Handbook of Input-Output Economics in Industrial Ecology*, Springer, Dordrecht, pp.23–41, Available Online: https://doi-org.ludwig.lub.lu.se/10.1007/978-1-4020-5737-3_2 [Accessed May 12, 2021].

Duchin, F. & Levine, S. H. (2014). Industrial Ecology, *Encyclopedia of Ecology*, pp.352–358, Available Online: https://doi.org/10.1016/B978-0-12-409548-9.09407-0 [Accessed April 21, 2021].

EC. (2008). Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on Waste and Repealing Certain Directives, *Eur-Lex.europa.eu*, Available Online: http://data.europa.eu/eli/dir/2008/98/2018-07-05 [Accessed May 10, 2021].

EC. (2015). Closing the Loop - an EU Action Plan for the Circular Economy, *Europa.eu*, Available Online: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52015DC0614 [Accessed March 23, 2021].

EC. (2018). EU Plastics Strategy, Available Online: https://ec.europa.eu/environment/strategy/plastics-strategy_en [Accessed May 2, 2021].

EEA. (2021). Plastics, the Circular Economy and Europe's Environment — European Environment Agency, , Luxembourg: Publications Office of the European Union, Available Online: https://www.eea.europa.eu/publications/plastics-the-circular-economy-and [Accessed May 4, 2021].

EC. (n.d.). Waste Prevention and Management - Environment, Ec.europa.eu, Available Online: https://ec.europa.eu/environment/green-growth/waste-prevention-and-management/index_en.htm [Accessed May 10, 2021].

Egüez, A. (2021). Compliance with the EU Waste Hierarchy: A Matter of Stringency, Enforcement, and Time, *Journal of Environmental Management*, vol. 280, p.111672, Available Online: https://doi.org/10.1016/j.jenvman.2020.111672 [Accessed April 14, 2021].

Ferronato, N. & Torretta, V. (2019). Waste Mismanagement in Developing Countries: A Review of Global Issues, International Journal of Environmental Research and Public Health, vol. 16, no. 6, p.1060, Available Online: https://www.mdpi.com/1660-4601/16/6/1060/htm [Accessed May 21, 2021].

Fry, J., Lenzen, M., Giurco, D. & Pauliuk, S. (2015). An Australian Multi-Regional Waste Supply-Use Framework, *Journal of Industrial Ecology*, vol. 20, no. 6, pp.1295–1305, Available Through: https://doi.org/10.1111/jiec.12376 [Accessed April 9, 2021].

Galaz, V., Österblom, H., Bodin, Ö. & Crona, B. (2014). Global Networks and Global Change-Induced Tipping Points, *International Environmental Agreements: Politics, Law and Economics*, vol. 16, no. 2, pp.189–221, Available Online: https://doi.org/10.1007/s10784-014-9253-6 [Accessed April 22, 2021].

Geissdoerfer, M., Savaget, P., Bocken, N. M. P. & Hultink, E. J. (2017). The Circular Economy – a New Sustainability Paradigm?, *Journal of Cleaner Production*, vol. 143, pp.757–768, Available Online: https://www.sciencedirect.com/science/article/pii/S0959652616321023?via%3Dihub [Accessed April 18, 2021].

Geng, Y., Sarkis, J. & Bleischwitz, R. (2019). How to Globalize the Circular Economy, *Nature*, vol. 565, no. 7738, pp.153–155, Available Online: https://www-nature-com.ludwig.lub.lu.se/articles/d41586-019-00017-z [Accessed April 23, 2021].

Genovese, A., Acquaye, A. A., Figueroa, A. & Koh, S. C. L. (2017). Sustainable Supply Chain Management and the Transition towards a Circular Economy: Evidence and Some Applications, *Omega*, vol. 66 Part B, no. Part B, pp.344–357, Available Through: ScienceDirect https://doi.org/10.1016/j.omega.2015.05.015 [Accessed May 3, 2021].

Geyer, R., Jambeck, J. R. & Law, K. L. (2017). Production, Use, and Fate of All Plastics Ever Made, *Science Advances*, vol. 3, no. 7, Available Online: https://advances.sciencemag.org/content/3/7/e1700782 [Accessed April 22, 2021].

Geyer, R., Kuczenski, B., Zink, T. & Henderson, A. (2015). Common Misconceptions about Recycling, *Journal of Industrial Ecology*, vol. 20, no. 5, pp.1010–1017, Available Online: https://doi.org/10.1111/jiec.12355 [Accessed May 2, 2021].

Ghisellini, P., Cialani, C. & Ulgiati, S. (2016). A Review on Circular Economy: The Expected Transition to a Balanced Interplay of Environmental and Economic Systems, *Journal of Cleaner Production*, vol. 114, pp.11–32, Available Online: https://www.sciencedirect.com/science/article/pii/S0959652615012287 [Accessed 24 April, 2021].

Glantz, M. H., Katz, R. W. & N Nicholls. (1991). Teleconnections Linking Worldwide Climate Anomalies : Scientific Basis and Societal Impact, Cambridge England; New York: Cambridge University Press.

Gravgård -Pedersen, O. (1999). Physical Input-Output Tables for Denmark. Products and Materials 1990, Air Emissions 1990-92, Copenhagen: Statistics Denmark.

Hahladakis, J. N. & Iacovidou, E. (2018). Closing the Loop on Plastic Packaging Materials: What Is Quality and How Does It Affect Their Circularity?, *Science of The Total Environment*, vol. 630, pp.1394–1400, Available Online: https://doi.org/10.1016/j.scitotenv.2018.02.330 [Accessed April 24, 2021].

Hahladakis, J. N., Iacovidou, E. & Gerassimidou, S. (2020). Plastic Waste in a Circular Economy, *Plastic Waste and Recycling*, [e-journal] pp.481–512, Available Online: https://doi-org.ludwig.lub.lu.se/10.1016/B978-0-12-817880-5.00019-0 [Accessed April 21, 2021].

Haupt, M. & Hellweg, S. (2019). Measuring the Environmental Sustainability of a Circular Economy, *Environmental and Sustainability Indicators*, vol. 1-2, p., Available Online: https://doi.org/10.1016/j.indic.2019.100005 [Accessed May 2, 2021].

He, H., Reynolds, C. J., Hadjikakou, M., Holyoak, N. & Boland, J. (2020). Quantification of Indirect Waste Generation and Treatment Arising from Australian Household Consumption: A Waste Input-Output Analysis, *Journal of Cleaner Production*, vol. 258, p.120935, Available Through: ScienceDirect https://doi.org/10.1016/j.jclepro.2020.120935 [Accessed May 12, 2021].

Henriques, S. T. & Kander, A. (2010). The Modest Environmental Relief Resulting from the Transition to a Service Economy, *Ecological Economics*, vol. 70, no. 2, pp.271–282, Available Through: ScienceDirect https://doi-org.ludwig.lub.lu.se/10.1016/j.ecolecon.2010.08.010 [Accessed May 9, 2021].

Hoekstra, A. Y. & Mekonnen, M. M. (2012). The Water Footprint of Humanity, *Proceedings of the National Academy of Sciences of the United States of America*, [e-journal] vol. 109, no. 9, pp.3232–7, Available Online: http://www.ncbi.nlm.nih.gov/pubmed/22331890 [Accessed April 22, 2021].

Iacovidou, E. & Gerassimidou, S. (2018). Sustainable Packaging and the Circular Economy: An EU Perspective, *Reference Module in Food Science*, [e-journal], Available Online: https://doi.org/10.1016/B978-0-08-100596-5.22488-8 [Accessed May 18, 2021].

Jambeck, J., Hardesty, B. D., Brooks, A. L., Friend, T., Teleki, K., Fabres, J., Beaudoin, Y., Bamba, A., Francis, J., Ribbink, A. J., Baleta, T., Bouwman, H., Knox, J. & Wilcox, C. (2018). Challenges and Emerging Solutions to the Land-Based Plastic Waste Issue in Africa, *Marine Policy*, vol. 96, pp.256–263, Available Online: https://doi.org/10.1016/j.marpol.2017.10.041 [Accessed April 22, 2021]

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., Narayan, R. & Law, K. L. (2015). Plastic Waste Inputs from Land into the Ocean, Science, vol. 347, no. 6223, pp.768–771, Available Online: http://science.sciencemag.org/content/347/6223/768 [Accessed April 22, 2021].

Jiborn, M., Kander, A., Kulionis, V., Nielsen, H. & Moran, D. D. (2018). Decoupling or Delusion? Measuring Emissions Displacement in Foreign Trade, *Global Environmental Change*, vol. 49, pp.27–34, Available Online: https://doi.org/10.1016/j.gloenvcha.2017.12.006 [Accessed May 12, 2021].

Jones, S. M. (2021). Waste Management: A Policy Paradox, *Advancing a Circular Economy*, pp.15–40, Available Online: https://doi-org.ludwig.lub.lu.se/10.1007/978-3-030-66564-7_2 [Accessed April 8, 2021].

Kander, A., Jiborn, M., Moran, D. D. & Wiedmann, T. O. (2015). National Greenhouse-Gas Accounting for Effective Climate Policy on International Trade, *Nature Climate Change*, vol. 5, no. 5, pp.431–435, Available Online: https://doi-org.ludwig.lub.lu.se/10.1038/nclimate2555 [Accessed May 11, 2021].

Kirchherr, J., Reike, D. & Hekkert, M. (2017). Conceptualizing the Circular Economy: An Analysis of 114 Definitions, *Resources, Conservation and Recycling*, vol. 127, pp.221–232, Available Online: https://www.sciencedirect.com/science/article/pii/S0921344917302835 [Accessed May 2, 2021].

Kristensen, H. S. & Mosgaard, M. A. (2020). A Review of Micro Level Indicators for a Circular Economy – Moving Away from the Three Dimensions of Sustainability?, *Journal of Cleaner Production*, vol. 243, p.118531, Available Through: ScienceDirect https://doi.org/10.1016/j.jclepro.2019.118531 [Accessed May 2, 2021].

Lazarevic, D., Aoustin, E., Buclet, N. & Brandt, N. (2010). Plastic Waste Management in the Context of a European Recycling Society: Comparing Results and Uncertainties in a Life Cycle Perspective, *Resources, Conservation and Recycling*, [e-journal] vol. 55, no. 2, pp.246–259, Available Online: https://www.sciencedirect.com/science/article/abs/pii/S0921344910002156 [Accessed April 22, 2020].

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, *Nature Communications*, vol. 8, p.15611, Available Online: https://www.nature.com/articles/ncomms15611 [Accessed April 22, 2021].

Lebreton, L., Slat, B., Ferrari, F., Sainte-Rose, B., Aitken, J., Marthouse, R., Hajbane, S., Cunsolo, S., Schwarz, A., Levivier, A., Noble, K., Debeljak, P., Maral, H., Schoeneich-Argent, R., Brambini, R. & Reisser, J. (2018). Evidence That the Great Pacific Garbage Patch Is Rapidly Accumulating Plastic, *Nature Scientific Reports*, vol. 8, no. 1, Available Through: https://www.nature.com/articles/s41598-018-22939-w [Accessed May 14, 2021].

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L. & Geschke, A. (2012). International Trade Drives Biodiversity Threats in Developing Nations, *Nature*, vol. 486, no. 7401, pp.109–112, Available Online: https://doi.org/10.1038/nature11145 [Accessed April 22, 2021].

Lenzen, M. & Reynolds, C. J. (2014). A Supply-Use Approach to Waste Input-Output Analysis, *Journal of Industrial Ecology*, vol. 18, no. 2, pp.212–226.

Leontief, W. (1936). Quantitative Input and Output Relations in the Economic Systems of the United States, *The Review of Economics and Statistics*, vol. 18, no. 3, p.105, Available Through: JSTOR Database https://doi-org.ludwig.lub.lu.se/10.2307/1927837 [Accessed May 6, 2021].

Leontief, W. (1970). Environmental Repercussions and the Economic Structure: An InputOutput Approach, *The Review of Economics and Statistics*, vol. 52, no. 3, pp.262–271, Available Through: JSTOR Journals https://doi-org.ludwig.lub.lu.se/10.2307/1926294 [Accessed May 6, 2021].

Liang, S., Zhang, T. & Xu, Y. (2012). Comparisons of Four Categories of Waste Recycling in China's Paper Industry Based on Physical Input–Output Life-Cycle Assessment Model, *Waste Management*, vol. 32, no. 3, pp.603–612, Available Online: https://doi.org/10.1016/j.wasman.2011.10.020 [Accessed May 13, 2021].

Liang, Y., Tan, Q., Song, Q. & Li, J. (2021). An Analysis of the Plastic Waste Trade and Management in Asia, *Waste Management*, vol. 119, pp.242–253, Available Through: ScienceDirect https://doi.org/10.1016/j.wasman.2020.09.049 [Accessed April 22, 2021].

Liao, M., Chen, P., Ma, H. & Nakamura, S. (2015). Identification of the Driving Force of Waste Generation Using a High-Resolution Waste Input–Output Table, Journal of Cleaner Production, vol. 94, pp.294–303, Available Through: ScienceDirect http://dx.doi.org/10.1016/j.jclepro.2015.02.002 [Accessed May 13, 2021].

Li, W. C., Tse, H. F. & Fok, L. (2016). Plastic Waste in the Marine Environment: A Review of Sources, Occurrence and Effects, *Science of The Total Environment*, vol. 566-567, pp.333–349, Available Online: https://doi.org/10.1016/j.scitotenv.2016.05.084 [Accessed April 8, 2021].

MacArthur Foundation. (2015). Towards the Circular Economy — Economic and business rationale for an accelerated transition, Available Online: https://www.werktrends.nl/app/uploads/2015/06/Rapport_McKinsey-Towards_A_Circular_Economy.pdf [Accessed April 24, 2021].

MacArthur Foundation. (2016). The New Plastics Economy — Rethinking the Future of Plastics, Available Online: http://www.ellenmacarthurfoundation.org/publications [Accessed April 24, 2021].

MacArthur Foundation. (2017). The Circular Economy in Detail, Available Online: https://www.ellenmacarthurfoundation.org/explore/the-circular-economy-in-detail [Accessed April 3, 2021].

MacArthur Foundation. (2020). Upstream Innovation: A Guide to Packaging Solutions, , Available Online: https://www.ellenmacarthurfoundation.org/publications/upstream-innovation [Accessed April 24, 2021].

Mäenpää, I. (2002). Physical Input-Output Tables of Finland 1995 ñ Solutions to Some Basic Methodological Problems, 14th International Conference on iInput-Output Techniques, Montreal, Canada.

Mah, A. (2021). Future-Proofing Capitalism: The Paradox of the Circular Economy for Plastics, *Global Environmental Politics*, vol. 21, no. 2, pp.121–142, Available Through: ProjectMUSE https://muse-jhu-edu.ludwig.lub.lu.se/article/790648 [Accessed April 24, 2021].

McCarthy, A., R. Dellink and R. Bibas (2018), ìThe Macroeconomics of the Circular Economy Transition: A Critical Review of Modelling Approachesî, OECD Environment Working Papers, No. 130, OECD Publishing, Paris. Available Online:http://dx.doi.org/10.1787/af983f9a-e [Accessed April 24, 2021]

Meijer, L. J. J., van Emmerik, T., van der Ent, R., Schmidt, C. & Lebreton, L. (2021). More than 1000 Rivers Account for 80% of Global Riverine Plastic Emissions into the Ocean, *Science Advances*, vol. 7, no. 18, p.eaaz5803, Available Online: https://advances.sciencemag.org/content/7/18/eaaz5803 [Accessed May 5, 2021].

Merciai, S. & Schmidt, J. (2017). Methodology for the Construction of Global Multi-Regional Hybrid

Supply and Use Tables for the EXIOBASE v3 Database, *Journal of Industrial Ecology*, vol. 22, no. 3, pp.516–531, Available Online: https://doi.org/10.1111/jiec.12713 [Accessed April 4, 2021].

Miller, R. E. & Blair, P. D. (2009). Input-Output Analysis : Foundations and Extensions, Cambridge England ; New York: Cambridge University Press.

Munksgaard, J., Minx, J. C., Christoffersen, L. B. & Pade, L. L. (2009). Models for National CO2 Accounting, in *Handbook of Input-Output Economics in Industrial Ecology. Eco-Efficiency in Industry and Science*: Springer, Dordrecht, pp.533–558, Available Online: https://doi-org.ludwig.lub.lu.se/10.1007/978-1-4020-5737-3_26 [Accessed May 12, 2021].

Munroe, D. K. & Biles, J. J. (2005). Regional Science, Encyclopedia of Social Measurement, pp.325–335, Available Online: https://doi.org/10.1016/B0-12-369398-5/00365-0 [Accessed May 20, 2021].

Murray, A., Skene, K. & Haynes, K. (2015). The Circular Economy: An Interdisciplinary Exploration of the Concept and Application in a Global Context, *Journal of Business Ethics*, vol. 140, no. 3, pp.369–380, Available Online: https://doi.org/10.1007/s10551-015-2693-2 [Accessed May 2, 2021].

Mwanza, B. G. & Mbohwa, C. (2017). Drivers to Sustainable Plastic Solid Waste Recycling: A Review, *Procedia Manufacturing*, vol. 8, pp.649–656, Available Through: ScienceDirect https://doi.org/10.1016/j.promfg.2017.02.083 [Accessed April 22, 2021].

Nakamura, S. & Kondo, Y. (2002). Input-Output Analysis of Waste Management, *Journal of Industrial Ecology*, vol. 6, no. 1, pp.39–63.

Nakamura, S. & Kondo, Y. (2009). WIO Analysis, in *Waste Input-Output Analysis. Eco-Efficiency in Industry and Science*, Vol. 26, Springer, Dordrecht, pp.201–248, Available Online: https://doi-org.ludwig.lub.lu.se/10.1007/978-1-4020-9902-1_6 [Accessed April 23, 2021].

Nakamura, S., Nakajima, K., Kondo, Y. & Nagasaka, T. (2007). The Waste Input-Output Approach to Materials Flow Analysis, *Journal of Industrial Ecology*, [vol. 11, no. 4, pp.50–63, Available Online: https://doi.org/10.1162/jiec.2007.1290 [Accessed May 14, 2021].

Nakatani, J., Maruyama, T. & Moriguchi, Y. (2020). Revealing the Intersectoral Material Flow of Plastic Containers and Packaging in Japan, *Proceedings of the National Academy of Sciences*, vol. 117, no. 33, pp.19844–19853, Available Online: https://www.pnas.org/content/117/33/19844 [Accessed May 13, 2021].

Nebbia, G. (2000). Contabilité monetaria e contabilità ambientale. *Economia Pubblica*, *30*(6), 5-33.

Newig, J., Challies, E., Cotta, B., Lenschow, A. & Schilling-Vacaflor, A. (2020). Governing Global Telecoupling toward Environmental Sustainability, *Ecology and Society*, vol. 25, no. 4, Available Online: https://doi.org/10.5751/ES-11844-250421 [Accessed May 14, 2021].

OECD (2018). Improving Plastics Management?: Trends, policy responses, and the role of international co-operation and trade. *OECD Environment Policy Papers*. Available Online: https://doi-org.ludwig.lub.lu.se/10.1787/c5f7c448-en [Accessed May 2, 2021]

Panayotou, T. (1993). Empirical Tests and Policy Analysis of Environmental Degradationat Different Stages of Economic Development. Technical Report, International LabourOrganization.

Patil, P. G., Virdin, J., Diez, S. M., Roberts, J. & Singh, A. (2016). Toward a Blue Economy: A Promise for Sustainable Growth in the Caribbean, Washington D.C.: The World Bank, Available Online: http://documents1.worldbank.org/curated/en/965641473449861013/pdf/AUS16344-REVISED-v1-BlueEconomy-FullReport-Oct3.pdf [Accessed April 21, 2021].

Pauli, G. A. (2010). The Blue Economy, Taos, New Mexico: Paradigm Publications.

Peters, G. P., Minx, J. C., Weber, C. L. & Edenhofer, O. (2011). Growth in Emission Transfers via International Trade from 1990 to 2008, *Proceedings of the National Academy of Sciences*, vol. 108, no. 21, pp.8903–8908, Available Online: https://www.pnas.org/content/108/21/8903.full [Accessed May 12, 2021].

Phelan, A. (Anya), Ross, H., Setianto, N. A., Fielding, K. & Pradipta, L. (2020). Ocean Plastic Crisis—Mental Models of Plastic Pollution from Remote Indonesian Coastal Communities, *PLOS ONE*, vol. 15, no. 7, p.e0236149, Available Online: https://doi.org/10.1371/journal.pone.0236149 [Accessed May 21, 2021].

PlasticsEurope. (2020). Plastics — the Facts 2020. An Analysis of European Plastics Production, Demand and Waste Data., Brussels: PlasticsEurope, Available Online: https://www.plasticseurope.org/en/resources/publications [Accessed April 22, 2021].

Potting, J., Hekkert, M., Worrell, E. & Hanemaaijer, A. (2017). Circular Economy: Measuring Innovation in the Production Chain. Policy Report, The Hague: PBL Netherlands Environmental Assessment Agency, Available Online: https://www.pbl.nl/sites/default/files/downloads/pbl-2016-circular-economy-measuring-innovation-in-product-chains-2544.pdf [Accessed May 9, 2021].

Reynolds, C., Mirosa, M. & Clothier, B. (2016). New Zealand's Food Waste: Estimating the Tonnes, Value, Calories and Resources Wasted, *Agriculture*, [e-journal] vol. 6, no. 1, p.9, Available Online: https://www.mdpi.com/2077-0472/6/1/9/pdf [Accessed May 12, 2021].

Rizos, V., Tuokko, K. & Behren, A. (2017). He Circular Economy: A Review of Definitions, Processes and Impacts. CEPS Research Report No 2017/8, *CEPS*, Available Online: http://aei.pitt.edu.ludwig.lub.lu.se/85892/ [Accessed May 12, 2021].

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F. S. I., Lambin, E., 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. (2009). Planetary Boundaries: Exploring the Safe Operating Space for Humanity, *Ecology and Society*, vol. 14, no. 2, Available Online: https://www.ecologyandsociety.org/vol14/iss2/art32/ [Accessed April 22, 2021].

Rodić, L. & Wilson, D. (2017). Resolving Governance Issues to Achieve Priority Sustainable Development Goals Related to Solid Waste Management in Developing Countries, *Sustainability*, vol. 9, no. 3, p.404, Available Online: https://doi.org/10.3390/su9030404 [Accessed April 24, 2021].

Ruiz-Peñalver, S. M., Rodríguez, M. & Camacho, J. A. (2019). A Waste Generation Input Output Analysis: The Case of Spain, *Journal of Cleaner Production*, vol. 210, pp.1475–1482, Available Through: ScienceDirect https://doi.org/10.1016/j.jclepro.2018.11.145 [Accessed May 3, 2021].

Saidani, M., Yannou, B., Leroy, Y., Cluzel, F. & Kendall, A. (2019). A Taxonomy of Circular Economy Indicators, *Journal of Cleaner Production*, vol. 207, pp.542–559.

Sangwon Suh. (2009). *Handbook of Input-output Economics in Industrial Ecology. Eco-efficiency in Industry and Science, vol. 23,* Netherlands: Springer

Schmidt, C., Krauth, T. & Wagner, S. (2017). Export of Plastic Debris by Rivers into the Sea, Environmental Science & Technology, vol. 51, no. 21, pp.12246–12253, Available Online: https://doi-org.ludwig.lub.lu.se/10.1021/acs.est.7b02368 [Accessed April 22, 2021]

Schmidt, J., Merciai, S., Delahaye, F., Vuik, J., Heijungs, R., de Koning, A. & Sahoo, A. (2012). CREEA Report: Recommendation of Terminology, Classification, Framework of Waste Accounts and MFA, and Data Collection Guideline — Deliverable 4.1 of the EU FP7-Project CREEA, *2.-0 LCA Consultants*, Available Online: https://lca-net.com/publications/show/creea-report-recommendation-terminology-classification-framework-waste-accounts-mfa-data-collection-guideline/ [Accessed May 10, 2021].

Schwarz, A. E., Ligthart, T. N., Godoi Bizarro, D., De Wild, P., Vreugdenhil, B. & van Harmelen, T. (2021). Plastic Recycling in a Circular Economy; Determining Environmental Performance through an LCA Matrix Model Approach, *Waste Management*, vol. 121, pp.331–342, Available Online: https://doi.org/10.1016/j.wasman.2020.12.020 [Accessed May 2, 2021].

Selvaranjan, K., Navaratnam, S., Rajeev, P. & Ravintherakumaran, N. (2021). Environmental Challenges Induced by Extensive Use of Face Masks during COVID-19: A Review and Potential Solutions, *Environmental Challenges*, vol. 3, p.100039, Available Online: https://doi.org/10.1016/j.envc.2021.100039 [Accessed May 21, 2021].

Stadler, K., Wood, R., Bulavskaya, T., Södersten, C.-J., Simas, M., Schmidt, S., Usubiaga, A., Acosta-Fernández, J., Kuenen, J., Bruckner, M., Giljum, S., Lutter, S., Merciai, S., Schmidt, J. H., Theurl, M. C., Plutzar, C., Kastner, T., Eisenmenger, N., Erb, K.-H. & de Koning, A. (2018). EXIOBASE 3: Developing a Time Series of Detailed Environmentally Extended Multi-Regional Input-Output Tables, *Journal of Industrial Ecology*, vol. 22, no. 3, pp.502–515, Available Through: Wiley Online Library https://doi.org/10.1111/jiec.12715 [Accessed April 4, 2021].

Stahmer, C., M. Kuhn, and N. Braun. 1997. *Physical Input-Output Tables for Germany, 1990* (No. 2/1998/B/1).

Statistisches Bundesambt (2001). Endbericht zum Projekt: A Physical Input-Output Table for Germany 1995. Vetragsnummer 98/559/3040/B4/MM.

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*, vol. 347, no. 6223, pp.1259855–1259855, Available Through: JSTOR Journals http://www.jstor.org.ludwig.lub.lu.se/stable/24746123 [Accessed

April 22, 2021].

Tisserant, A., Pauliuk, S., Merciai, S., Schmidt, J., Fry, J., Wood, R. & Tukker, A. (2017). Solid Waste and the Circular Economy: A Global Analysis of Waste Treatment and Waste Footprints, *Journal of Industrial Ecology*, vol. 21, no. 3, pp.628–640.

Tóth, G. (2019). Circular Economy and Its Comparison with 14 Other Business Sustainability Movements, Resources, vol. 8, no. 4, p.159, Available Online: https://doi.org/10.3390/resources8040159 [Accessed April 27, 2021].

Towa, E., Zeller, V. & Achten, W. M. J. (2020). Input-Output Models and Waste Management Analysis: A Critical Review, *Journal of Cleaner Production*, vol. 249, p.119359, Available Online: https://doi.org/10.1016/j.jclepro.2019.119359 [Accessed March 19, 2021].

Towa, E., Zeller, V. & Achten, W. M. J. (2021). Circular Economy Scenario Modelling Using a Multiregional Hybrid Input-Output Model: The Case of Belgium and Its Regions, *Sustainable Production and Consumption*, vol. 27, pp.889–904, Available Online: https://doi.org/10.1016/j.spc.2021.02.012 [Accessed April 3, 2021].

Tukker, A. & Dietzenbacher, E. (2013). Global Multiregional Input-Output Frameworks: An Introduction and Outlook, *Economic Systems Research*, vol. 25, no. 1, pp.1–19, Available Online: https://doi.org/10.1080/09535314.2012.761179 [Accessed April 27, 2021].

Tukker, A., Giljum, S. & Wood, R. (2018). Recent Progress in Assessment of Resource Efficiency and Environmental Impacts Embodied in Trade: An Introduction to This Special Issue, *Journal of Industrial Ecology*, vol. 22, no. 3, pp.489–501, Available Through: Wiley Online Library https://doi.org/10.1111/jiec.12736 [Accessed April 27, 2021].

UN. (2015). Transforming Our World: The 2030 Agenda for Sustainable Development | Department of Economic and Social Affairs, Available Online: https://sdgs.un.org/publications/transforming-our-world-2030-agenda-sustainable-development-17981 [Accessed April 22, 2021].

UN. (2017). System of Environmental-Economic Accounting 2012. Central Framework, United Nations, p., Available Online: https://doi-org.ludwig.lub.lu.se/10.5089/9789211615630.069 [Accessed May 14, 2021].

UN Environment. (2020). Our Planet Is Drowning in Plastic Pollution. This World Environment Day, It's Time for a Change, *Unenvironment.org*, Available Online: https://www.unenvironment.org/interactive/beat-plastic-pollution/ [Accessed March 19, 2021].

UNEP. (2018). Single-Use Plastics: A Roadmap for Sustainability, Nairobi, Kenya: United Nations Environment Programme, Available Online: https://www.unep.org/resources/report/single-use-plastics-roadmap-sustainability [Accessed May 2, 2021].

UNEP. (2019a). Addressing Marine Plastics: A Systemic Approach – Recommendations for Actions, Nairobi, Kenya: United Nations Environment Programme, Available Online: https://www.unep.org/resources/report/addressing-marine-plastics-systemic-approach-recommendations-actions [Accessed May 21, 2021].

UNEP. (2019b). Fashion's Tiny Hidden Secret, *UN Environment*, Available Online: https://www.unep.org/news-and-stories/story/fashions-tiny-hidden-secret [Accessed May 18, 2021].

UNEP. (2021). Addressing Single-Use Plastic Products Pollution Using a Life Cycle Approach, *Wedocs.unep.org*, Nairobi, Kenya: United Nations Environment Programme, Available Online: https://wedocs.unep.org/20.500.11822/35109 [Accessed May 2, 2021].

Wäger, P. A. & Hischier, R. (2015). Life Cycle Assessment of Post-Consumer Plastics Production from Waste Electrical and Electronic Equipment (WEEE) Treatment Residues in a Central European Plastics Recycling Plant, Science of the Total Environment, vol. 529, pp.158–167, Available Online: https://doi.org/10.1016/j.scitotenv.2015.05.043 [Accessed May 2, 2021].

Wang, C., Zhao, L., Lim, M. K., Chen, W.-Q. & Sutherland, J. W. (2020). Structure of the Global Plastic Waste Trade Network and the Impact of China's Import Ban, *Resources, Conservation and Recycling*, vol. 153, p.104591, Available Online: https://doi.org/10.1016/j.resconrec.2019.104591 [Accessed April 22, 2021].

Watkins, E. & Schweitzer, J.-P. (2018). Moving towards a Circular Economy for Plastics in the EU by 2030, *Think 2030*, Brussels, Belgium: Institute for European environmental Policy (IEEP), Available Online: https://ieep.eu/uploads/articles/attachments/4e8d42a2-c8a8-4d40-bae0-c3429b10fe4d/Think%202030%20Circular%20economy%20for%20plastics.pdf?v=63710011292.

Watson, M. (2020). Waste Management, *International Encyclopedia of Human Geography*, pp.225–231, Available Through: ScienceDirect https://doi.org/10.1016/B978-0-08-102295-5.10761-9 [Accessed April 24, 2021].

Wiebe, K. S., Bruckner, M., Giljum, S., Lutz, C. & Polzin, C. (2012). Carbon and Materials Embodied in the International Trade of Emerging Economies, *Journal of Industrial Ecology*, vol. 16, no. 4, pp.636–646.

Wiebe, K. S., Harsdorff, M., Montt, G., Simas, M. S. & Wood, R. (2019). Global Circular Economy Scenario in a Multiregional Input–Output Framework, *Environmental Science & Technology*, vol. 53, no. 11, pp.6362–6373, Available Online: https://doi.org/10.1021/acs.est.9b01208 [Accessed April 23, 2021].

Wiedmann, T., Barrett, J., Billing, M., & Birch, R. (2004). The REAP Tool - Creating Physical Accounts for the UK and Evaluating Policy Scenarios. Paper presented at the Workshop Driving Forces for and Barriers to Sustainable Consumption, Leeds, UK.

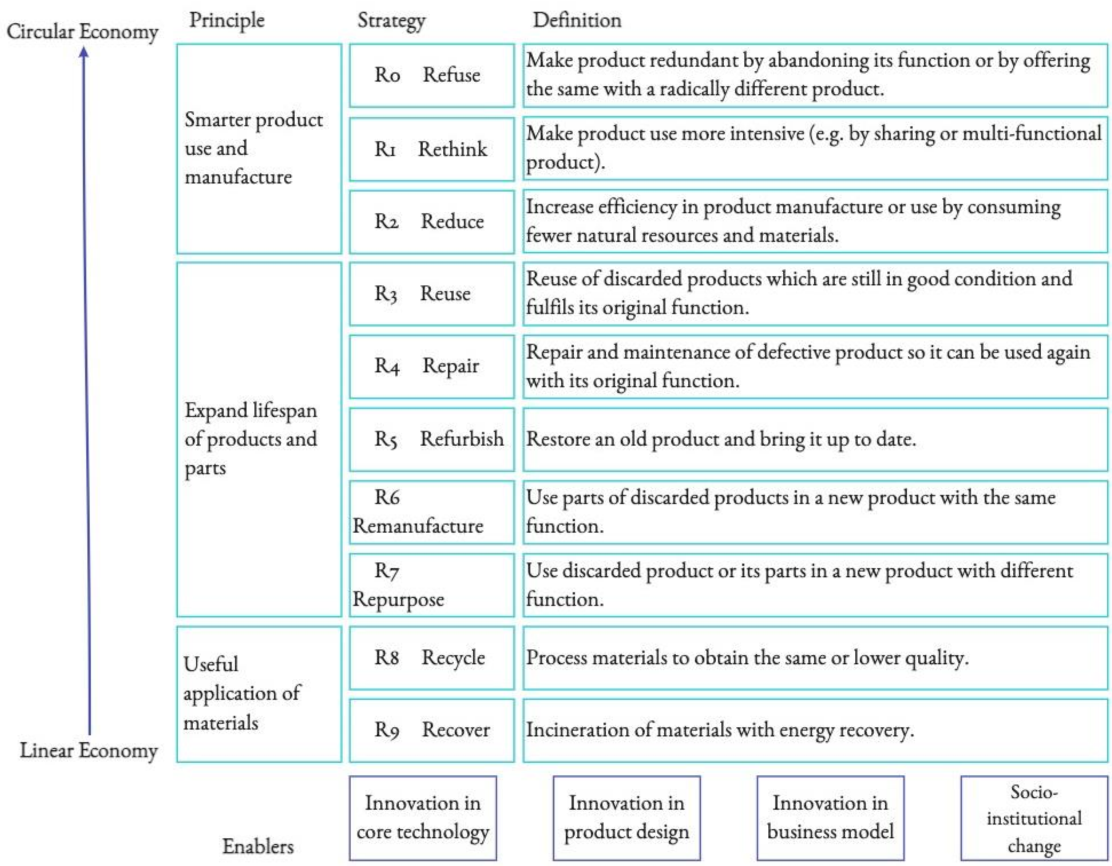
Wijkman, A., & Skånberg, K. (2017). The Circular Economy and Benefits for Society: Jobs and Climate Clear Winners in an Economy Based on Renewable Energy and Resource Efficiency.

Wood, R., Stadler, K., Bulavskaya, T., Lutter, S., Giljum, S., de Koning, A., Kuenen, J., Schütz, H., Acosta-Fernández, J., Usubiaga, A., Simas, M., Ivanova, O., Weinzettel, J., Schmidt, J., Merciai, S. & Tukker, A. (2014). Global Sustainability Accounting—Developing EXIOBASE for Multi-Regional Footprint Analysis, *Sustainability*, vol. 7, no. 1, pp.138–163.

Zalasiewicz, J., Waters, C. N., Ivar do Sul, J. A., Corcoran, P. L., Barnosky, A. D., Cearreta, A., Edgeworth, M., Gałuszka, A., Jeandel, C., Leinfelder, R., McNeill, J. R., Steffen, W., Summerhayes, C., Wagreich, M., Williams, M., Wolfe, A. P. & Yonan, Y. (2016). The Geological Cycle of Plastics and Their Use as a Stratigraphic Indicator of the Anthropocene, *Anthropocene*, vol. 13, pp.4–17.

Appendix A: Theory

Figure 20. The 9Rs Framework (Potting et al., 2017, p.5)



The 9Rs Framework represents, in order of priority, different strategies to advance towards circularity of production systems. These strategies are enabled by different types of innovations, including technological, product design, business and revenue models and systems change.

Appendix B: Data

Table 2. Geographical coverage of EXIOBASE and country classification based on income level

Country/ Region	Code	Country Classification*	Country/ Region	Code	Country Classification*
Australia	AU	High-income	Luxembourg	LU	High-income
Austria	AT	High-income	Malta	MT	High-income
Belgium	BE	High-income	Mexico	MX	Upper middle-income
Brazil	BR	Upper middle-income	Netherlands	NL	High-income
Bulgaria	BG	Upper middle-income	Norway	NO	High-income
Canada	CA	High-income	Poland	PL	High-income
China	CN	Upper middle-income	Portugal	PT	High-income
Cyprus	CY	High-income	Romania	RO	Upper middle-income
Czech Republic	CZ	High-income	Russia	RU	High-income
Denmark	DK	High-income	Slovakia	SK	High-income
Estonia	EE	High-income	Slovenia	SI	High-income
Finland	FI	High-income	South Africa	ZA	Upper middle-income
France	FR	High-income	South Korea	KR	High-income
Germany	DE	High-income	Spain	ES	High-income
Greece	GR	High-income	Sweden	SE	High-income
Hungary	HU	Upper middle-income	Switzerland	CH	High-income
Croatia	HR	High-income	Turkey	TR	Upper middle-income
India	IN	Lower middle-income	United Kingdom	GB	High-income
Indonesia	ID	Lower middle-income	United States	US	High-income
Ireland	IE	High-income	RoW Asia and Pacific	WA	-
Italy	IT	High-income	Row America	WL	-
Japan	JP	High-income	RoW Europe	WE	-
Latvia	LV	High-income	RoW Africa	WF	-
Lithuania	LT	High-income	RoW Middle East	WM	-

*Source: UN (2014). Classification of economies by per capita GNI in 2012.

Table 3. List of the 164 products (and categories of products and services), measurement units and corresponding production activity in EXIOBASE 3 MR-HIOTs

Product code	Product name	Unit	Activity code	Activity
C_PARI	Paddy rice	tonnes	A_PARI	Cultivation of paddy rice
C_WHEA	Wheat	tonnes	A_WHEA	Cultivation of wheat
C_OCER	Cereal grains nec	tonnes	A_OCER	Cultivation of cereal grains nec
C_FVEG	Vegetables, fruit, nuts	tonnes	A_FVEG	Cultivation of vegetables, fruit, nuts
C_OILS	Oil seeds	tonnes	A_OILS	Cultivation of oil seeds
C_SUGB	Sugar cane, sugar beet	tonnes	A_SUGB	Cultivation of sugar cane, sugar beet
C_FIBR	Plant-based fibres	tonnes	A_FIBR	Cultivation of plant-based fibres
C_OTCR	Crops nec	tonnes	A_OTCR	Cultivation of crops nec
C_CATL	Cattle	tonnes	A_CATL	Cattle farming
C_PIGS	Pigs	tonnes	A_PIGS	Pig's farming
C_PLTR	Poultry	tonnes	A_PLTR	Poultry farming
C_OMEA	Meat animals nec	tonnes	A_OMEA	Meat animals nec
C_OANP	Animal products nec	tonnes	A_OANP	Animal products nec
C_MILK	Raw milk	tonnes	A_MILK	Raw milk
C_WOOL	Wool, silk-worm cocoons	tonnes	A_WOOL	Wool, silk-worm cocoons
C_MANC	Manure (conventional treatment)	tonnes	A_MANC	Manure treatment (conventional), storage and land application
C_MANB	Manure (biogas treatment)	tonnes	A_MANB	Manure treatment (biogas), storage and land application
C_FORE	Products of forestry, logging and related services (02)	tonnes	A_FORE	Forestry, logging and related service activities (02)
C_FISH	Fish and other fishing products; services incidental of fishing (05)	tonnes	A_FISH	Fishing, operating of fish hatcheries and fish farms; service activities incidental to fishing (05)
C_CLPT	Coal, lignite and peat	tonnes	A_COAL	Mining of coal and lignite; extraction of peat (10)
C_COIL	Crude petroleum and services related to crude oil extraction, excluding surveying	tonnes	A_COIL	Extraction of crude petroleum and services related to crude oil extraction, excluding surveying
C_GASS	Natural gas and services related to natural gas extraction, excluding surveying, including liquid gas	tonnes	A_GASE	Extraction of natural gas and services related to natural gas extraction, excluding surveying
C_OGPL	Other Hydrocarbons	tonnes	A_OGPL	Extraction, liquefaction, and regasification of other petroleum and gaseous materials
C_ORAN	Uranium and thorium ores (12)	tonnes	A_ORAN	Mining of uranium and thorium ores (12)
C_IRON	Iron ores	tonnes	A_IRON	Mining of iron ores
C_COPO	Copper ores and concentrates	tonnes	A_COPO	Mining of copper ores and concentrates
C_NIKO	Nickel ores and concentrates	tonnes	A_NIKO	Mining of nickel ores and concentrates
C_ALUO	Aluminium ores and concentrates	tonnes	A_ALUO	Mining of aluminium ores and concentrates
C_PREO	Precious metal ores and concentrates	tonnes	A_PREO	Mining of precious metal ores and concentrates
C_LZTO	Lead, zinc and tin ores and concentrates	tonnes	A_LZTO	Mining of lead, zinc and tin ores and concentrates
C_ONFO	Other non-ferrous metal ores and concentrates	tonnes	A_ONFO	Mining of other non-ferrous metal ores and concentrates
C_STON	Stone	tonnes	A_STON	Quarrying of stone
C_SDCL	Sand and clay	tonnes	A_SDCL	Quarrying of sand and clay
C_CHMF	Chemical and fertilizer minerals, salt	tonnes	A_CHMF	Mining of chemical and fertilizer

Product code	Product name	Unit	Activity code	Activity
	and other mining and quarrying products n.e.c.			minerals, production of salt, other mining and quarrying n.e.c.
C_PCAT	Products of meat cattle	tonnes	A_PCAT	Processing of meat cattle
C_PPIG	Products of meat pigs	tonnes	A_PPIG	Processing of meat pigs
C_PPLT	Products of meat poultry	tonnes	A_PPLT	Processing of meat poultry
C_POME	Meat products nec	tonnes	A_POME	Production of meat products nec
C_VOIL	products of Vegetable oils and fats	tonnes	A_VOIL	Processing vegetable oils and fats
C_DAIR	Dairy products	tonnes	A_DAIR	Processing of dairy products
C_RICE	Processed rice	tonnes	A_RICE	Processed rice
C_SUGR	Sugar	tonnes	A_SUGR	Sugar refining
C_OFOD	Food products nec	tonnes	A_OFOD	Processing of Food products nec
C_BEVR	Beverages	tonnes	A_BEVR	Manufacture of beverages
C_FSHP	Fish products	tonnes	A_FSHP	Manufacture of fish products
C_TOBC	Tobacco products (16)	tonnes	A_TOBC	Manufacture of tobacco products (16)
C_TEXT	Textiles (17)	tonnes	A_TEXT	Manufacture of textiles (17)
C_GARM	Wearing apparel; furs (18)	tonnes	A_GARM	Manufacture of wearing apparel; dressing and dyeing of fur (18)
C_LETH	Leather and leather products (19)	tonnes	A_LETH	Tanning and dressing of leather; manufacture of luggage, handbags, saddlery, harness and footwear (19)
C_WOOD	Wood and products of wood and cork (except furniture); articles of straw and plaiting materials (20)	tonnes	A_WOOD	Manufacture of wood and of products of wood and cork, except furniture; manufacture of articles of straw and plaiting materials (20)
C_WOOW	Wood material for treatment, Re-processing of secondary wood material into new wood material	tonnes	A_WOOW	Re-processing of secondary wood material into new wood material
C_PULP	Pulp	tonnes	A_PULP	Pulp
C_PAPR	Secondary paper for treatment, Re-processing of secondary paper into new pulp	tonnes	A_PAPR	Re-processing of secondary paper into new pulp
C_PAPE	Paper and paper products	tonnes	A_PAPE	Paper
C_MDIA	Printed matter and recorded media (22)	tonnes	A_MDIA	Publishing, printing and reproduction of recorded media (22)
C_COPR	Coke oven products	tonnes	A_COKE	Manufacture of coke oven products
C_REFP	Refined Petroleum	tonnes	A_REFN	Petroleum Refinery
C_NUCF	Nuclear fuel	TJ	A_NUCF	Processing of nuclear fuel
C_PLAS	Plastics, basic	tonnes	A_PLAS	Plastics, basic
C_PLAW	Secondary plastic for treatment, Re-processing of secondary plastic into new plastic	tonnes	A_PLAW	Re-processing of secondary plastic into new plastic
C_NFER	N-fertiliser	tonnes	A_NFER	N-fertiliser
C_PFER	P- and other fertilisers	tonnes	A_PFER	P- and other fertilisers
C_CHBI	Chemicals nec; additives and biofuels	tonnes	A_CHEM	Chemicals nec
C_RUBP	Rubber and plastic products (25)	tonnes	A_RUBP	Manufacture of rubber and plastic products (25)
C_GLAS	Glass and glass products	tonnes	A_GLAS	Manufacture of glass and glass products
C_GLAW	Secondary glass for treatment, Re-processing of secondary glass into new glass	tonnes	A_GLAW	Re-processing of secondary glass into new glass
C_CRMC	Ceramic goods	tonnes	A_CRMC	Manufacture of ceramic goods
C_BRIK	Bricks, tiles and construction products, in baked clay	tonnes	A_BRIK	Manufacture of bricks, tiles and construction products, in baked clay

Product code	Product name	Unit	Activity code	Activity
C_CMNT	Cement, lime and plaster	tonnes	A_CMNT	Manufacture of cement, lime and plaster
C_ASHW	Ash for treatment, Re-processing of ash into clinker	tonnes	A_ASHW	Re-processing of ash into clinker
C_ONMM	Other non-metallic mineral products	tonnes	A_ONMM	Manufacture of other non-metallic mineral products n.e.c.
C_STEL	Basic iron and steel and of ferro-alloys and first products thereof	tonnes	A_STEL	Manufacture of basic iron and steel and of ferro-alloys and first products thereof
C_STEW	Secondary steel for treatment, Re-processing of secondary steel into new steel	tonnes	A_STEW	Re-processing of secondary steel into new steel
C_PREM	Precious metals	tonnes	A_PREM	Precious metals production
C_PREW	Secondary precious metals for treatment, Re-processing of secondary precious metals into new precious metals	tonnes	A_PREW	Re-processing of secondary precious metals into new precious metals
C_ALUM	Aluminium and aluminium products	tonnes	A_ALUM	Aluminium production
C_ALUW	Secondary aluminium for treatment, Re-processing of secondary aluminium into new aluminium	tonnes	A_ALUW	Re-processing of secondary aluminium into new aluminium
C_LZTP	Lead, zinc and tin and products thereof	tonnes	A_LZTP	Lead, zinc and tin production
C_LZTW	Secondary lead for treatment, Re-processing of secondary lead into new lead	tonnes	A_LZTW	Re-processing of secondary lead into new lead
C_COPP	Copper products	tonnes	A_COPP	Copper production
C_COPW	Secondary copper for treatment, Re-processing of secondary copper into new copper	tonnes	A_COPW	Re-processing of secondary copper into new copper
C_ONFM	Other non-ferrous metal products	tonnes	A_ONFM	Other non-ferrous metal production
C_ONFW	Secondary other non-ferrous metals for treatment, Re-processing of secondary other non-ferrous metals into new other non-ferrous metals	tonnes	A_ONFW	Re-processing of secondary other non-ferrous metals into new other non-ferrous metals
C_METC	Foundry work services	tonnes	A_METC	Casting of metals
C_FABM	Fabricated metal products, except machinery and equipment (28)	tonnes	A_FABM	Manufacture of fabricated metal products, except machinery and equipment (28)
C_MACH	Machinery and equipment n.e.c. (29)	tonnes	A_MACH	Manufacture of machinery and equipment n.e.c. (29)
C_OFMA	Office machinery and computers (30)	tonnes	A_OFMA	Manufacture of office machinery and computers (30)
C_ELMA	Electrical machinery and apparatus n.e.c. (31)	tonnes	A_ELMA	Manufacture of electrical machinery and apparatus n.e.c. (31)
C_RATV	Radio, television and communication equipment and apparatus (32)	tonnes	A_RATV	Manufacture of radio, television and communication equipment and apparatus (32)
C_MEIN	Medical, precision and optical instruments, watches and clocks (33)	tonnes	A_MEIN	Manufacture of medical, precision and optical instruments, watches and clocks (33)
C_MOTO	Motor vehicles, trailers and semi-trailers (34)	Meuro	A_MOTO	Manufacture of motor vehicles, trailers and semi-trailers (34)
C_OTRE	Other transport equipment (35)	Meuro	A_OTRE	Manufacture of other transport equipment (35)
C_FURN	Furniture; other manufactured goods n.e.c. (36)	tonnes	A_FURN	Manufacture of furniture; manufacturing n.e.c. (36)
C_RYMS	Secondary raw materials	Meuro	A_RYMS	Recycling of waste and scrap
C_BOTW	Bottles for treatment, Recycling of bottles by direct reuse	tonnes	A_BOTW	Recycling of bottles by direct reuse

Product code	Product name	Unit	Activity code	Activity
C_POWC	Electricity by coal	TJ	A_POWC	Production of electricity by coal
C_POWG	Electricity by gas	TJ	A_POWG	Production of electricity by gas
C_POWN	Electricity by nuclear	TJ	A_POWN	Production of electricity by nuclear
C_POWH	Electricity by hydro	TJ	A_POWH	Production of electricity by hydro
C_POWW	Electricity by wind	TJ	A_POWW	Production of electricity by wind
C_POWP	Electricity by petroleum and other oil derivatives	TJ	A_POWP	Production of electricity by petroleum and other oil derivatives
C_POWB	Electricity by biomass and waste	TJ	A_POWB	Production of electricity by biomass and waste
C_POWS	Electricity by solar photovoltaic	TJ	A_POWS	Production of electricity by solar photovoltaic
C_POWE	Electricity by solar thermal	TJ	A_POWE	Production of electricity by solar thermal
C_POWO	Electricity by tide, wave, ocean	TJ	A_POWO	Production of electricity by tide, wave, ocean
C_POWM	Electricity by Geothermal	TJ	A_POWM	Production of electricity by Geothermal
C_POWZ	Electricity nec	TJ	A_POWZ	Production of electricity nec
C_POWT	Transmission services of electricity	Meuro	A_POWT	Transmission of electricity
C_POWD	Distribution and trade services of electricity	Meuro	A_POWD	Distribution and trade of electricity
C_BGAS	Biogas and other gases nec.	tonnes	A_MGWG	Manufacture of gas;
C_GASD	Distribution services of gaseous fuels through mains	Meuro	A_GASD	Distribution of gaseous fuels through mains
C_HWAT	Steam and hot water supply services	TJ	A_HWAT	Steam and hot water supply
C_WATR	Collected and purified water, distribution services of water (41)	Meuro	A_WATR	Collection, purification and distribution of water (41)
C_CONS	Construction work (45)	Meuro	A_CONS	Construction (45)
C_CONW	Secondary construction material for treatment, Re-processing of secondary construction material into aggregates	tonnes	A_CONW	Re-processing of secondary construction material into aggregates
C_TDMO	Sale, maintenance, repair of motor vehicles, motor vehicles parts, motorcycles, motorcycles parts and accessories	Meuro	A_TDMO	Sale, maintenance, repair of motor vehicles, motor vehicles parts, motorcycles, motorcycles parts and accessories
C_TDFU	Retail trade services of motor fuel	Meuro	A_TDFU	Retail sale of automotive fuel
C_TDWH	Wholesale trade and commission trade services, except of motor vehicles and motorcycles (51)	Meuro	A_TDWH	Wholesale trade and commission trade, except of motor vehicles and motorcycles (51)
C_TDRT	Retail trade services, except of motor vehicles and motorcycles; repair services of personal and household goods (52)	Meuro	A_TDRT	Retail trade, except of motor vehicles and motorcycles; repair of personal and household goods (52)
C_HORE	Hotel and restaurant services (55)	Meuro	A_HORE	Hotels and restaurants (55)
C_TRAI	Railway transportation services	Meuro	A_TRAI	Transport via railways
C_TLND	Other land transportation services	Meuro	A_TLND	Other land transport
C_TPIP	Transportation services via pipelines	Meuro	A_TPIP	Transport via pipelines
C_TWAS	Sea and coastal water transportation services	Meuro	A_TWAS	Sea and coastal water transport
C_TWAI	Inland water transportation services	Meuro	A_TWAI	Inland water transport
C_TAIR	Air transport services (62)	Meuro	A_TAIR	Air transport (62)
C_TAUX	Supporting and auxiliary transport services; travel agency services (63)	Meuro	A_TAUX	Supporting and auxiliary transport activities; activities of travel agencies (63)
C_PTEL	Post and telecommunication services (64)	Meuro	A_PTEL	Post and telecommunications (64)

Product code	Product name	Unit	Activity code	Activity
C_FINT	Financial intermediation services, except insurance and pension funding services (65)	Meuro	A_FINT	Financial intermediation, except insurance and pension funding (65)
C_FINS	Insurance and pension funding services, except compulsory social security services (66)	Meuro	A_FINS	Insurance and pension funding, except compulsory social security (66)
C_FAUX	Services auxiliary to financial intermediation (67)	Meuro	A_FAUX	Activities auxiliary to financial intermediation (67)
C_REAL	Real estate services (70)	Meuro	A_REAL	Real estate activities (70)
C_MARE	Renting services of machinery and equipment without operator and of personal and household goods (71)	Meuro	A_MARE	Renting of machinery and equipment without operator and of personal and household goods (71)
C_COMP	Computer and related services (72)	Meuro	A_COMP	Computer and related activities (72)
C_RESD	Research and development services (73)	Meuro	A_RESD	Research and development (73)
C_OBUS	Other business services (74)	Meuro	A_OBUS	Other business activities (74)
C_PADF	Public administration and defence services; compulsory social security services (75)	Meuro	A_PADF	Public administration and defence; compulsory social security (75)
C_EDUC	Education services (80)	Meuro	A_EDUC	Education (80)
C_HEAL	Health and social work services (85)	Meuro	A_HEAL	Health and social work (85)
C_INCF	Food waste for treatment: incineration	tonnes	A_INCF	Incineration of waste: Food
C_INCP	Paper waste for treatment: incineration	tonnes	A_INCP	Incineration of waste: Paper
C_INCL	Plastic waste for treatment: incineration	tonnes	A_INCL	Incineration of waste: Plastic
C_INCM	Intert/metal waste for treatment: incineration	tonnes	A_INCM	Incineration of waste: Metals and Inert materials
C_INCT	Textiles waste for treatment: incineration	tonnes	A_INCT	Incineration of waste: Textiles
C_INCW	Wood waste for treatment: incineration	tonnes	A_INCW	Incineration of waste: Wood
C_INCO	Oil/hazardous waste for treatment: incineration	tonnes	A_INCO	Incineration of waste: Oil/Hazardous waste
C_BIOF	Food waste for treatment: biogasification and land application	tonnes	A_BIOF	Biogasification of food waste, incl. land application
C_BIOP	Paper waste for treatment: biogasification and land application	tonnes	A_BIOP	Biogasification of paper, incl. land application
C_BIOS	Sewage sludge for treatment: biogasification and land application	tonnes	A_BIOS	Biogasification of sewage sludge, incl. land application
C_COMF	Food waste for treatment: composting and land application	tonnes	A_COMF	Composting of food waste, incl. land application
C_COMW	Paper and wood waste for treatment: composting and land application	tonnes	A_COMW	Composting of paper and wood, incl. land application
C_WASF	Food waste for treatment: wastewater treatment	tonnes	A_WASF	Wastewater treatment, food
C_WASO	Other waste for treatment: wastewater treatment	tonnes	A_WASO	Wastewater treatment, other
C_LANF	Food waste for treatment: landfill	tonnes	A_LANF	Landfill of waste: Food
C_LANP	Paper for treatment: landfill	tonnes	A_LANP	Landfill of waste: Paper
C_LANL	Plastic waste for treatment: landfill	tonnes	A_LANL	Landfill of waste: Plastic
C_LANI	Inert/metal/hazardous waste for treatment: landfill	tonnes	A_LANI	Landfill of waste: Inert/metal/hazardous
C_LANT	Textiles waste for treatment: landfill	tonnes	A_LANT	Landfill of waste: Textiles
C_LANW	Wood waste for treatment: landfill	tonnes	A_LANW	Landfill of waste: Wood
C_ORGA	Membership organisation services	Meuro	A_ORGA	Activities of membership organisation

Product code	Product name	Unit	Activity code	Activity
	n.e.c. (91)			n.e.c. (91)
C_RECR	Recreational, cultural and sporting services (92)	Meuro	A_RECR	Recreational, cultural and sporting activities (92)
C_OSER	Other services (93)	Meuro	A_OSER	Other service activities (93)
C_PRHH	Private households with employed persons (95)	Meuro	A_PRHH	Private households with employed persons (95)
C_EXTO	Extra-territorial organizations and bodies	Meuro	A_EXTO	Extra-territorial organizations and bodies

Table 4. Comparison of the main IO databases. Source: Own construction based on Tukker, Giljum & Wood (2018)

	EXIOBASE	Eora	WIOD	GTAP	OECD
Year	2000-2011*	1970-2013	1995-2011	2004-2011	1995-2011
Countries	43+5 RoW	190	40 + RoW	140	61
Industries	164	Variable (20 to 500)	35	57	34
Products	164		59	57	34
Hybrid units	Yes	No	No	No	No
Waste / Recycle extensions	Yes	No	No	No	No

*The IOTs in EXIOBASE provide a time series for the period 2000-2011, however, the hybrid version is only available for year 2011.

Appendix C: Results

Table 5. Plastic Waste Footprints by country and region

Country/Region	PBA (tonnes)	CBA (tonnes)	Revised CBA (tonnes)	Revised CBA pc (Kg plastic pc)
Australia	1.126.353,11	1.028.402,87	2.763.698,33	123,71
Austria	-184.481,03	-142.763,60	172.925,82	20,61
Belgium	-1.007.348,40	-1.064.015,90	771.317,65	69,88
Brazil	-1.651.903,60	-996.657,34	3.869.074,22	19,59
Bulgaria	-967.094,86	-940.078,74	220.004,04	29,94
Canada	1.903.870,81	3.527.778,15	5.481.164,88	159,62
China	34.724.656,50	27.144.922,10	35.931.760,87	26,73
Cyprus	-9.172,09	1.926,75	39.241,39	34,89
Czech Republic	41.392,02	51.785,06	244.317,76	23,28
Denmark	-126.908,09	-52.228,15	183.172,45	32,88
Estonia	-25.981,18	-18.242,11	36.903,96	27,80
Finland	-315.226,30	-244.116,46	341.685,10	63,41
France	-3.881.289,80	-3.425.603,20	1.262.948,23	19,33
Germany	-822.381,98	-885.324,62	2.384.072,27	29,70
Greece	-162.180,44	-83.020,07	474.367,11	42,72
Hungary	-501.890,91	-463.655,34	177.382,46	17,79
Croatia	322.761,72	215.921,55	292.487,83	68,33
India	7.555.114,65	6.426.059,54	6.830.561,03	5,46
Indonesia	684.206,95	865.395,70	987.689,25	4,03
Ireland	-257.569,16	-299.191,25	114.349,57	24,97
Italy	-2.114.084,10	-1.895.613,00	1.391.450,66	23,43
Japan	-907.896,13	-385.172,18	2.576.222,61	20,15
Latvia	-108.707,50	-75.355,42	84.146,92	40,85
Lithuania	-73.465,19	-31.029,07	94.260,65	31,13
Luxembourg	-44.100,13	-35.043,29	65.093,68	125,58
Malta	-3.832,90	3.159,01	14.693,83	35,30
Mexico	-1.062.933,20	-1.170.033,10	876.051,18	7,57
Netherlands	-55.007,59	-60.529,31	707.541,48	42,39
Norway	-494.441,97	-440.481,88	176.814,56	35,70
Poland	-685.534,99	-623.435,36	648.313,42	17,03
Portugal	-510.593,86	-451.938,86	242.032,45	22,93
Romania	-176.589,34	-195.329,34	231.648,31	11,50
Russia	-5.015.068,30	-3.760.386,50	6.369.066,07	44,55
Slovakia	-314.850,99	-231.126,09	151.542,17	28,07
Slovenia	-57.898,63	8.324,34	119.291,23	58,11
South Africa	705.796,57	614.554,81	710.863,50	13,67

Country/Region	PBA (tonnes)	CBA (tonnes)	Revised CBA (tonnes)	Revised CBA pc (Kg plastic pc)
South Korea	3.829.649,06	2.581.615,73	3.001.175,89	60,10
Spain	-74.560,98	-129.895,33	1.149.068,93	24,58
Sweden	-549.260,64	-411.531,05	362.594,82	38,37
Switzerland	273.840,97	296.988,72	487.620,30	61,63
Turkey	-2.405.137,70	-2.084.145,00	1.768.570,31	24,08
United Kingdom	-921.031,61	-468.111,79	1.790.409,85	28,30
United States	-5.287.631,60	-2.926.177,00	15.146.678,40	48,62
RoW Asia and Pacific	4.527.695,52	6.008.053,22	7.233.990,37	-
Row America	4.901.639,84	4.159.656,85	5.209.913,11	-
RoW Europe	17.591.421,50	19.008.740,10	27.554.363,01	-
RoW Africa	1.745.932,95	1.600.323,54	2.849.752,59	-
RoW Middle East	4.887.014,85	4.565.093,94	5.907.422,90	-

Table 6. Plastic waste for treatment, by treatment category

Country/ Region	Total Plastic Waste for Treatment	Re-processing of secondary plastic into new plastic		Plastic waste for treatment: incineration		Plastic waste for treatment: landfill	
		in Mt	%	in Mt	%	in Mt	%
Australia	-1,2777	-0,1606	12,57%	-0,05000	3,91%	-1,0671	83,52%
Austria	-0,4623	-0,1466	31,71%	-0,22179	47,98%	-0,0939	20,31%
Belgium	-2,0817	-0,2463	11,83%	-1,48842	71,50%	-0,3469	16,67%
Brazil	-5,3437	-0,4779	8,94%	-0,05956	1,11%	-4,8062	89,94%
Bulgaria	-1,2150	-0,0549	4,52%	-0,03337	2,75%	-1,1267	92,74%
Canada	-2,1334	-0,1800	8,44%	-0,13200	6,19%	-1,8214	85,37%
China	-10,2511	-1,4642	14,28%	-0,52061	5,08%	-8,2662	80,64%
Croatia	-0,0892	-0,0126	14,14%	-0,00897	10,06%	-0,0676	75,80%
Cyprus	-0,0418	-0,0045	10,79%	-0,00227	5,42%	-0,0350	83,79%
Czech Republic	-0,2309	-0,0384	16,63%	-0,03171	13,73%	-0,1608	69,64%
Denmark	-0,3365	-0,1011	30,04%	-0,15583	46,31%	-0,0796	23,65%
Estonia	-0,0717	-0,0166	23,09%	-0,00887	12,37%	-0,0463	64,53%
Finland	-0,6086	-0,0228	3,75%	-0,27769	45,63%	-0,3081	50,63%
France	-5,2392	-0,5507	10,51%	-2,09755	40,04%	-2,5910	49,45%
Germany	-4,3759	-1,1065	25,29%	-2,40188	54,89%	-0,8675	19,83%
Greece	-0,6106	-0,0533	8,72%	-0,03474	5,69%	-0,5226	85,59%
Hungary	-0,7361	-0,0951	12,92%	-0,13803	18,75%	-0,5030	68,33%
India	-0,4564	-0,0519	11,37%	-0,05976	13,09%	-0,3447	75,53%
Indonesia	-0,1442	-0,0219	15,19%	-0,02662	18,46%	-0,0957	66,35%
Ireland	-0,5469	-0,1334	24,39%	-0,06972	12,75%	-0,3438	62,86%
Italy	-4,6775	-1,3904	29,73%	-0,97770	20,90%	-2,3094	49,37%
Japan	-4,6207	-1,6593	35,91%	-1,89309	40,97%	-1,0683	23,12%
Latvia	-0,1712	-0,0117	6,82%	-0,01334	7,79%	-0,1462	85,39%
Lithuania	-0,1376	-0,0123	8,94%	-0,00813	5,91%	-0,1172	85,15%
Luxembourg	-0,1195	-0,0193	16,19%	-0,06397	53,54%	-0,0362	30,27%
Malta	-0,0162	-0,0046	28,61%	-0,00144	8,92%	-0,0101	62,47%
Mexico	-2,6256	-0,5795	22,07%	-0,15161	5,77%	-1,8945	72,15%
Netherlands	-0,9170	-0,1489	16,24%	-0,59513	64,90%	-0,1729	18,86%
Norway	-0,7899	-0,1726	21,85%	-0,45612	57,74%	-0,1612	20,40%
Poland	-1,6548	-0,3831	23,15%	-0,20174	12,19%	-1,0700	64,66%
Portugal	-0,7934	-0,0994	12,53%	-0,17864	22,52%	-0,5153	64,95%
Romania	-0,5187	-0,0917	17,69%	-0,05894	11,36%	-0,3680	70,95%
Russia	-10,2400	-0,1106	1,08%	-0,10846	1,06%	-10,0210	97,86%
Slovakia	-0,4387	-0,0560	12,77%	-0,03534	8,06%	-0,3473	79,17%
Slovenia	-0,1289	-0,0180	13,94%	-0,01599	12,40%	-0,0950	73,66%
South Africa	-0,1056	-0,0093	8,78%	-0,01337	12,66%	-0,0829	78,55%
South Korea	-0,4882	-0,0686	14,06%	-0,09202	18,85%	-0,3275	67,09%
Spain	-1,5866	-0,3077	19,39%	-0,20601	12,98%	-1,0730	67,63%
Sweden	-0,9387	-0,1646	17,53%	-0,60875	64,85%	-0,1654	17,62%
Switzerland	-0,2636	-0,0730	27,69%	-0,07233	27,44%	-0,1183	44,87%
Turkey	-3,9310	-0,0783	1,99%	-0,04734	1,20%	-3,8054	96,80%

Country/ Region	Total Plastic Waste for Treatment	Re-processing of secondary plastic into new plastic		Plastic waste for treatment: incineration		Plastic waste for treatment: landfill	
		in Mt	%	in Mt	%	in Mt	%
United Kingdom	-2,7423	-0,4838	17,64%	-0,19963	7,28%	-2,0589	75,08%
United States	-20,7082	-2,6353	12,73%	-3,27925	15,84%	-14,7936	71,44%
RoW Africa	-1,3179	-0,0685	5,20%	-0,33079	25,10%	-0,9186	69,70%
Row America	-1,1676	-0,1174	10,05%	-0,17267	14,79%	-0,8776	75,16%
RoW Asia and Pacific	-1,5730	-0,3471	22,06%	-0,17568	11,17%	-1,0503	66,77%
RoW Europe	-23,0324	-14,4868	62,90%	-2,76739	12,02%	-5,7782	25,09%
RoW Middle East	-1,5531	-0,2107	13,57%	-0,26256	16,91%	-1,0798	69,52%