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The dark truth behind plastic waste trade

the consequences of the Chinese import ban & the environmental damage of plastic waste trade using Life Cycle Assessment.

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Abstract

In recent years, plastic waste trade has received increasing attention due to huge amounts ending up in countries with high rates of mismanaged plastic waste and the environmental damage of plastic pollution. The EU has been exporting plastic waste to China for recycling purposes due to insufficient domestic recycling capacities, until 2018, when China decided to ban all imports of plastic waste. This study aims to quantify the direct effect of the Chinese import ban of the biggest EU exporting countries trade flows, domestic treatment methods and environmental effects using Life Cycle Assessment. Given the importance of international trade, there is a need to understand the environmental impact of exporting thousands of tonnes of plastic waste, especially to Southeast Asia. The result indicated changes in trade flows, stagnated domestic recycling rates and that the new trade pattern to Southeast Asia is the least preferred option given 3 out of 6 environmental indicators in the LCA, including human health and the marine environment.

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Abbreviations

FPMF - Fine particulate matter formation

HCT - Human non-carcinogenic toxicity

HS - Harmonized System

ISO - International Organisation for Standardisation

LCA - Life Cycle Assessment

LCI - Life Cycle Inventory

MT - Metric tonnes

PET - Polyethylene terephthalate

PP - Polypropylene

PS - Polystyrene

PVC - Polyvinyl chloride

1. Introduction

Plastic is one of the most used materials, and with its remarkable properties, plastic can be found in all kinds of commercial products. As the use of plastic has expanded quickly, the global production of plastic has increased by 6.3 billion MT (metric tonnes) from 1950 to 2017 and are not expected to slow down (Brooks, Wang & Jambeck, 2018; Geyer, Jambeck, & Law, 2017). Despite the multiple benefits of plastic, it raises several environmental concerns throughout the life cycle, especially when it comes to waste management. The leakage of plastic to the ocean is estimated to be 8 million tonnes every year; the equivalent of pouring one garbage truck of plastic into the ocean every minute (WEF, 2016). With current global recycling rates only at 18% (OECD, 2018) and the high rates of marine litter, there will probably be the same amount of fish and plastic in the ocean by 2050 (EllenMacArthur, 2016). Insufficient recycling capacities has resulted in vast amounts ending up at landfills and two-thirds are being exported to countries with high rates of mismanaged plastic waste (Geyer, Jambeck, & Law, 2017; Jambeck et al. 2015).

The EU has long prided itself of being an environmental leader in the circular economy by turning waste into raw material. However, the truth in their success is that a lot of their waste has been exported abroad for recycling purposes. For decades, China had been one of the leading importing countries for recycling, re-use and disposal of solid waste until January 2017, when they announced an import ban of 24 types of solid waste, including plastic waste (Qu et al. 2019). The abrupt ban turned the global plastic recycling industry into turmoil, and bales of recyclables piled up at landfill due to limited domestic recycling and incineration capacities (Hook & Reed, 2018; Qu et al. 2019). Given China's leading role in the world's recycling industry, global trade flows of waste were redirected to Southeast Asia with lower environmental regulations and higher rates of mismanaged plastic waste. The environmental impact of the redirected trade flow to countries such as Malaysia, Indonesia, Thailand, and the Philippines may result in increased rates of plastic pollution as open dump and open burning are dominated waste management practices (Kaza et al. 2018).

As a consequence, with growing awareness of environmental protection and circular economy, a large number of studies has increased within Life Cycle Assessment (LCA), a commonly used methodology to evaluate the environmental impact of a product's life cycle (Golsteijn, 2020). A majority of LCA studies within plastic waste have compared the different end-of-life treatment scenarios to come up with the optimal treatment method (Lou & Nair, 2009; Rabl, Spadaro & Zoughaib, 2008; Lazervic et al. 2010). However, it is only in the past few years, LCA has been used in the context of plastic waste trade (Qu et al. 2019; Ren et al. 2020; Wen et al. 2021). Few attempts have been made to quantify the environmental impact of exporting plastic waste abroad to countries with high rates of mismanaged plastic waste using LCA (Huang et al. 2020; Qu et al. 2019; Wang et al. 2020; Xu et al. 2020). As the Chinese import ban had a huge impact on the global plastic waste trade flows and the recycling industry, the majority of LCA studies within plastic waste trade also fail to address the environmental impact in Southeast Asia. This study therefore, evaluated three different scenarios of exporting waste to China (as before the ban), to Southeast Asia (as after the ban), with the preferred option of domestic waste treatment within the EU. With new trade data until 2020, waste statistics and an environmental comparison between different trade scenarios, this study contributes with new perspectives and insights of the global plastic waste trade.

1.1 Aim & Research Questions

On the basis of the current research gap, the aim of this study is twofold. Firstly, this study aims to quantitatively analyse and compare the biggest exporting countries' waste management in terms of exports and rates of recycling, incineration and landfill after the Chinese import ban. Secondly, a LCA is conducted to compare the environmental impact between the scenarios of, exporting plastic waste to China (as before the ban), to Southeast Asia (as after the ban) and the most preferred option of domestic waste management. To investigate this matter, the following research questions are answered;

- *What is the direct impact of the Chinese import ban on exporting countries' waste management, in terms of exports and rates of recycling, incineration and landfill?*
- *What are the environmental impacts of exporting plastic waste to China; to Southeast Asia*[1]*, or domestic treatment within the EU?*

To understand these developments, a closer look needs to be taken into countries' waste management systems, trade flows, rates of recycling, landfill and incineration and the life cycle of plastic waste. This study highlights the importance and environmental benefits of domestic waste management and the urgent need to stop exporting waste to Asian countries where landfill and open burning are dominant waste management methods. The consequences of the ban may also reinforce the motivation to prevent, reduce and re-use plastic and increase investments to develop domestic recycling capacities. Further, as plastic waste trade and waste management occurs in a complex network, this study does not aim to deliver final solutions to the problem. It rather aims to provide evidence that the field of plastic waste management should be studied to a larger extent and still has a long way to go to achieve a circular economy.

However, this study has a number of possible limitations when considering the dataset and the chosen methodology of LCA. Even if LCA is widely used to assess the environmental impacts, it relies on assumptions and scenarios. It is also hard to compare one LCA study with another due to the fact that different scopes (when considering a product's life cycle) may lead to different results. In this study, the plastic waste life cycle starts from collection spots to disposal facilities followed by either domestic waste treatment of recycling, landfill and incineration or export for disposal to China or Southeast Asia. Apart from the methodology, waste statistics is often hard to evaluate as there is no international framework that provides a basis for harmonising waste definitions, classifications and methods of calculations (UNECE, 2020). All these possible limitations have been taken into account and official statistics from European authorities and waste management organisations have been reviewed.

1.2 Outline of the study

The outline of this study is divided into nine main chapters, including the introduction. Chapter 2 provides the reader with a contextual insight of the life cycle of plastic and the policy framework within the EU. Chapter 3 reviews existing literature in the field of waste management, trade with plastic waste and current LCA studies. Chapter 4 describes the conceptual framework of LCA and provides insights of the limitations. Chapter 5 covers the data used in this study. Chapter 6 describes

[1] Malaysia, Indonesia, the Philippines, Thailand and Vietnam

the twofold methodology and details about how the LCA has been conducted. Chapter 7 presents the results and a comparison between the scenarios whilst chapter 8 provides the reader with a discussion of the results in relation to the literature and limitations. The final chapter presents concluding remarks, policy recommendations and future research.

2. Contextual framework

In the following chapter, the life cycle of plastic is described briefly, with particular attention given to the end-of-life as plastic waste management. Further, the EU policy framework for plastic waste trade is covered to give the reader an insight into regulations and laws that regulate trade with waste.

2.1 Life cycle of plastic

Given the below-simplified illustration (figure 1), plastic production starts with the extraction of oil and natural gas to obtain the building blocks of plastics, ethylene and propylene. Further, in the process stage, ethylene and propylene are chemically modified to obtain resins, the core ingredient for plastic. The chemical processing can be done in several different ways, leading to a variety of plastic products such as the most common types; Polyethene (PE), Polypropylene (PP), Polyethylenterephthalat (PET), Polystyrene (PS) and Polyvinyl chloride (PVC). For example, polyethene is used to make bottles and plastic bags (see table A1, Appendix A). Meanwhile, polystyrene is commonly known to make styrofoam (Brown, 2020). Plastic remarkable properties have therefore become one of the most used and widespread materials globally and are now present in almost all types of commercial products (Liang et al. 2021).

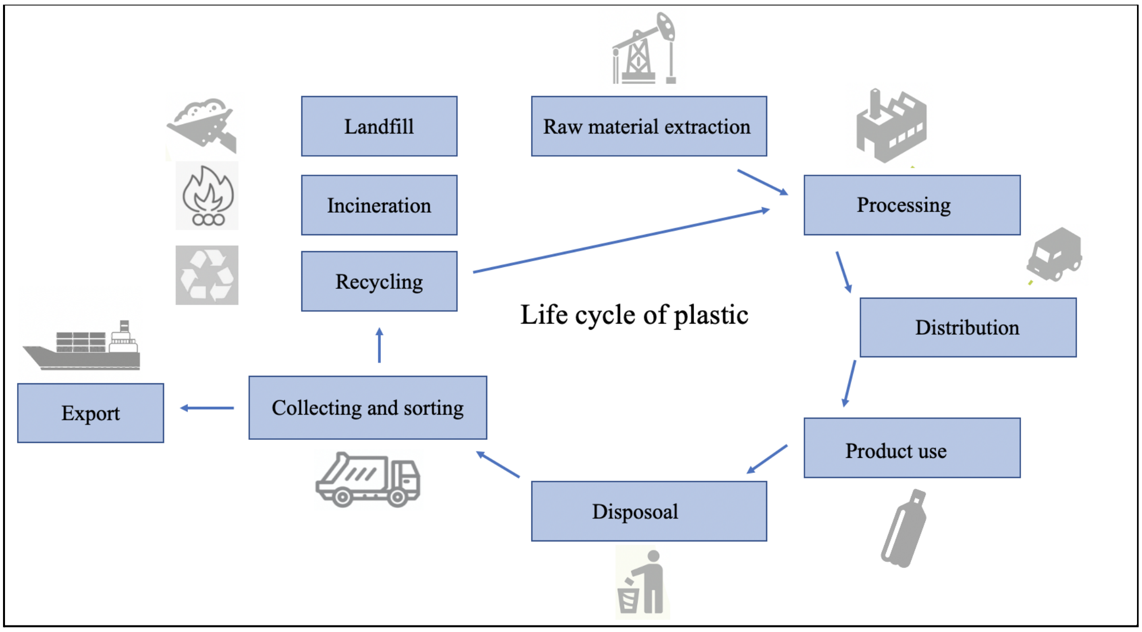

Figure 1 - Simplified flow of the plastic life cycle (Adapted from Ambrières, 2019)

After a plastic product has been disposed of, the most common end-of-life processes are recycling, incineration and landfill, with the latter being the least prefered option. The plastic waste collected but not recycled is often landfilled or incinerated, with the consequences of higher environmental impacts in terms of emissions to air and loss of valuable materials. Landfill may be the easiest way of waste management but the least preferred according to the EU Waste Hierarchy. Sanitary landfills are constructed with different layers and underground pipe systems aiming to reduce methane gasses and leakage into the environment (EPA, 2020). Unsanitary landfills or open dumps, on the other hand, are not regulated, and when water percolating through the landfill (i.e leachate) it contributes to eutrophication and groundwater pollution (Tinmaz, 2006). Another end-of-life method is incineration which includes burning plastic waste and can either be processed with energy recovery or without, depending on technology. This waste treatment process is popular in countries with developed technologies to transform the heat from burning waste to an energy source. However, the downsides with incineration means a production of Co2-emissions and toxic organic gases such as polycyclic aromatic hydrocarbons, dioxins and furans when plastic is burned (Verma et al. 2016).

Recycling is the process of recovering and reprocessing plastic waste into new secondary material that can be used to produce new plastic products. The different methods of plastic recycling include chemical recycling (feedstock recycling), quaternary recycling (energy recovery) and mechanical recycling, which is the most well-established and widely used method (Brachet et al. 2008; Meran, Ozturk & Yuksel, 2008; Yin et al. 2015). Chemical recycling uses a chemical process to recover the petrochemical components in plastics, but this technology is not widely used due to the need of sizable energy inputs (Rahimi & García, 2017). However, before plastic waste is mechanically recycled, it is collected through curbside collection, refund programs, or drop-off centres (EPA, 2020). After the collection phase, plastic waste is sent to a recovery facility, where it is sorted by identification codes. The different types of plastic are then taken to the correct recycling facility, where they get broken down into small pieces before being washed, dried and melted down into pellets that can be transformed into the desired material (Leblanc, 2020). Although mechanical recycling is the most established recycling method of plastic, the recycling industry is currently facing a wide range of bottlenecks (Hahladakis & Iacovidou, 2019).

Current recycling capacities and technologies cannot deal with the millions of tonnes of plastic waste containing contaminated, multi-layer or mixed plastics (Ragaert et al. 2017). Mixed plastic waste from households or highly contaminated waste is hard to recycle and highlights the importance of sorting. This has also led to low demand for recycled plastic, and together with the low cost of producing virgin plastic, it makes it both easier and less uncertain for firms to buy virgin plastic as raw material. Given the limited capacity and high cost of recycling, a significant part of recyclable plastic is being exported abroad (Geyer et al. 2017; Jambeck et al. 2015).

2.2 The EU - Policy framework for waste management

Waste management systems within the EU are embedded within the EU waste policy framework aiming to protect the environment and human health in the transition to a circular economy (European Commission, 2020a). The Waste Framework Directive is the legal framework for waste management where the five-step *waste hierarchy* acts as the foundation of EU waste management and applies to all Member States when developing domestic waste management laws (see figure A1, Appendix A). It establishes an order of priority for managing and disposing of waste consisting of; prevention, re-used, recycled, processed for energy recovery, and the slightest option of landfill/disposal (EU Commission, 2020a).

According to the waste hierarchy, Directive 2008/98/EC (see figure A1, Appendix A), *Prevention* includes those measures taken before a product has become waste that reduce the overall quantity of waste. This is the top-priority from an environmental perspective. *Re-use* is those methods by which products are being re-used again for the same purpose they were conceived. *Recycling* comes after and it includes the recovery measures of waste and the recycling process of turning waste into new products with the original or other purposes. *Recovery* is the waste-to-energy option. It refers to transforming non-recyclable waste into usable heat, electricity, or fuel through different processes such as incineration, gasification or pyrolization. The least preferred option is *disposal* and includes landfill or incineration without energy recovery (EU Commission, 2020a). The EU waste policy aims to protect the environment and human health by boosting recycling, stimulating innovation, limiting the amount ending up at landfill, and creating incentives to decrease the consumption of plastics (EU Commission, 2020b).

Trade with waste within and outside the EU is regulated by the The Waste Shipment Regulation and as post-consumer plastic waste is categorized as a non-hazardous (green listed) waste, it can be exported to non-OECD countries for recycling purposes. The Waste Shipment Regulation prohibits shipments of hazardous waste and waste destined for disposal to non-OECD countries outside the EU (EEA, 2019). At international level, transboundary shipments of waste are governed by the UN via the Basel Convention on the *Control of Transboundary Movements of Hazardous Wastes and their Disposal.* The Basel Convention, establishes standards for the transboundary movement of hazardous and solid waste where countries ratifying the convention need to comply with standards for the transboundary movement of hazardous and solid waste. In 2019, Governments amended the Basel Convention to significantly restrict plastic waste trade to help address the problems of plastic waste disposal and reduce its leakage into the environment. Plastic waste trade destined for recycling or disposal is allowed only with the prior written consent of the importing country (and any transit country) and the amendments took effect as of January 1, 2021(EPA, 2021).

3. Literature Review

The following literature review provides an overview of previous research and insights in the field of plastic waste trade and waste management. The first part covers trade with plastic waste with special attention to the Chinese import ban and the consequences. Second, given plastic waste damage to the environment, several previous LCA studies are reviewed, followed by other studies covering environmental issues within the context of plastic waste trade and waste management.

3.1 Trade with plastic waste

The plastic waste trade is a big part of the global recycling industry and accounted for over 15 million tonnes in 2016 (ZeroWasteEurope, 2018). Since 1990 global plastic waste trade has continued to rise as China started to import waste for recycling purposes. Foreign waste is a cheap input material in the Chinese manufacturing industry (Brooks, Wang & Jambeck, 2018; Qu et al. 2019), and the low freight rates meant that ships could efficiently deliver the material back to China when they had offloaded cargo from China at ports in Europe and the United States. For example, shipping a container from China to the US could be $2,400, whilst the return trip was only $300 (Xia et al. 2018). Consequently, it could be seen as a de facto subsidy for scrap exporters and therefore an easy way to get rid of their waste and prevent it from going to landfill or incineration (Velis, 2014). The limited capacities and technologies in developed countries' waste management systems and the high cost of dealing with contaminated and multi-layer plastic products, making it both hard and expensive to recycle all plastic domestically. Plastic waste is traded just like other commodities given the demand and includes many actors such as recycling companies who sell their collected waste to waste traders who also need to involve transport companies.

In order to understand these developments and trade patterns, one needs to look into what variables determine waste trade across borders. In the literature, several socioeconomic variables have been discussed (Kellenberg, 2012; Kellenberg, 2007; Mazzanti & Zoboli, 2013; Higashida, 2014). Kellenberg (2007) relates to differences in income, environmental regulations, and the presence of organized crime. Differences in environmental regulation have a significant effect meaning that waste will flow to countries with lower levels of environmental regulations since the cost of managing waste is therefore lower. Higashida et al. (2014) focuses more on economic variables, such as wages and GDP, and argues that both higher wage/per capita and GDP of an importing country will increase the imports of recyclable wastes. Their research highlights that Southeast Asian countries have a labour-abundance and low wages that contribute to their comparative cost advantage in handling plastic waste (Higashida et al. 2014). Income, wages and environmental regulations are not the only factors explaining transboundary movements of waste and scrap. Mazzanti & Zoboli (2013) argues that transportation costs, trade barriers such as tariffs and legislation and incentives for recycling are the main drivers of trade with plastic waste.

3.2 The Chinese import ban

China's economy has partly benefited from the global trade with plastic waste (Qu et al. 2019). Due to the environmental damage that imported waste contributes to and to the desire to become a less polluted country (Wang et al. 2019), China announced in 2017 to the WTO that they will permanently ban the import of non-industrial plastic waste as of January 2018. The sudden ban of 24 different types of solid waste turned the global recycling industry into turmoil as countries could not rely on China taking care of their waste anymore. Brooks, Wang & Jambeck (2018) investigated historical and regional trends of the global plastic waste trade and estimated that 111 million tonnes needs to be displaced by 2030. The key implication drawn from this is how much countries have relied on China and the large amount that is being traded all over the world. Given countries' lack of managing their waste, countries changed export destinations as a result of the ban, from China to Southeast Asian countries such as Malaysia, the Philippines, Indonesia and Thailand (Brooks, Wang & Jambeck, 2018; Qu et al. 2019; Wang et al. 2020).

A growing number of studies have been published trying to quantify these changed trade patterns (Qu et al. 2019; Huang et al. 2020; Wang et al. 2020). In 2017, the import quantity to China was reduced from 7 Mt to 5.8 Mt of plastic waste and dropped even more to 52 kt in 2018. The same pattern has shown up in Hong Kong, from imports over 2.8 Mt to 0.6 Mt of plastic waste in 2018, as they have acted as the main transfer hub (Liang et al. 2021). For Southeast Asian countries, Malaysia has witnessed a rapid increase in plastic waste imports in 2017 and 2018 compared to previous years. In addition, the imported amount to Indonesia and Thailand increased by 150% and 260%, respectively, in 2018. Liang et al. (2021) concluded that Indonesia and Malaysia acted as transferers in 2016, but after the ban, they became main importers in 2018. The huge amount of imported waste, especially to

Malaysia, has resulted in stricter import regulations to avoid further imbalance of domestic plastic pollution. In October 2018, Malaysia announced that all imports reported under HS 3915 require an approval permit to demonstrate that they follow the Basel convention conditions and regulations (Liang et al. 2021). Other countries in Southeast Asia such as Vietnam, Thailand and Indonesia have also strengthened their import regulation of plastic waste after the Chinese import ban; Indonesia only imports plastic waste in the form of flakes, chips or pellets, whilst the imported plastic waste to the Philippines can be in any form only if it is recyclable (Liang et al. 2021).

3.3 The problem of exporting plastic waste to Southeast Asia

The waste management systems in countries such as Malaysia, Indonesia and the Philippines are not ideal (Liang et al. 2020). Southeast Asia has the highest rates of mismanaged plastic, accounting for 70 per cent of the world total as of 2010 (Ritchie & Roser, 2018). Mismanaged plastic waste is defined as the material that enters the environment by littering or inadequate disposal methods such as open burning, uncontrolled landfills and open dump. China, Indonesia, the Philippines, Vietnam, Thailand, and Malaysia are all in the top 10 when it comes to mismanaged plastic waste, where open dumping and burning of waste are common treatment methods (Jambeck et al. 2015). In the report “What a Waste 2.0” from the World Bank (2018), the estimated average waste treatment rates for Southeast Asian countries shows that a majority is managed by open dump (75%), open burning (16%), landfill (4%) and recycling (5 %) (World Bank, 2018). The imported amount of plastic waste is generally mixed with other solid and contaminated waste such as metals, batteries and paper, which raises even more environmental concerns (Wen et al. 2021). A recent study from NUI Galway and the University of Limerick (2020) was first quantifying how much of the European plastic exports end up as marine litter. Their estimated result indicated that between 1-7% of all exported European polyethene ends up in the ocean. In addition, they also concluded that 31% of the exported plastic waste is not recycled in Asia, which implies that the reported recycling rates may deviate from the actual rates (NUI Galway & Limerick, 2020).

Jambeck et al. (2015) highlight the issue of mismanaged waste, and countries which import the majority of plastic waste are also responsible for the highest amount of marine littering. Eight out of the ten rivers which are responsible for 90% of the plastic waste in the oceans can be found in Southeast Asia (UN, 2019). Jambeck et al. (2015) estimated the annual input of plastic to the ocean by looking at solid waste data together with population density and economic status. They found that out of 275 million MT waste generated in 192 coastal countries in 2010, between 4.8 to 12.7 million MT are estimated to enter the ocean. Without waste management improvement the authors argue that the cumulative quantity of plastic waste entering the ocean by 2025 would be multiplied by 10 (Jambeck et al. 2015). Marine litter of plastic can also degrade into small plastic fragments or “microplastic” (<5mm) from the UV light from the sun which has been an emerging field of research when considering plastic waste (Wang et al. 2018; Li et al. 2016; Borrelli et al. 2017; Worm et al. 2017). The ingestion of microplastics by fish and smaller aquatic organisms has been proven to have negative impacts on their growth rate and morbidity. Microplastics can also carry toxic chemicals such as heavy metals, (Verla et al. 2019) and can be found in marine, freshwater, and even in polar regions which highlight the global problem of marine littering (Fu & Wang, 2019).

Despite the environmental concerns with mismanaged plastic waste to marine wildlife, human health is also put in danger as open burning releases toxic gases such as carbon monoxide, furans and dioxins. In the report 'The Recycling Myth' by Greenpeace Malaysia (2018), plastic waste disposed of at several dumpsites in Malaysia contain a range of metals that can potentially cause respiratory diseases, Alzheimer's disease, Parkinson's disease and cancer. Since the end of 2018, there has been increased news reporting relating to the imported plastic waste trade on plastic pollution and the damage to the environment and human health (Greenpeace Malaysia, 2021). Despite the serious health problems caused by handling plastic waste, people still continue to work in the waste management sector for their livelihoods (GRID-Arendal, 2017).

With an increased interest in studying the consequences of the rapid increase in imports to Southeast Asia, it is also believed that illegal shipments and dumping of wastes have grown significantly in recent years. Huang et al. (2020) argues that it is of high risk that illegal dumping occurs when China, as the dominant and essential player, bans all imports of plastic waste. Although most plastic waste trade is legally shipped for resource recovery purposes, estimates suggest that 25% of the overall shipments do not comply with the regulations (European Commission, 2020). In Interpol's Strategic Analysis Report 2018, they investigated how trade with plastic waste has shifted over the recent years. The organization highlights the case of Malaysia, with a substantial increase in imports of plastic waste, which has resulted in a growing number of illegal recycling facilities. 170 illegal recycling facilities have been shut down in the country (INTERPOL, 2020), and in May 2019, Malaysia announced that they would send back 3,000 MT to the US, Canada, Australia and the UK due to illegally shipped contaminated plastics. Other Southeast Asian countries witnessed the same trend in 2019 as the Philippines returned 1,500 tonnes of illegally dumped waste back to Canada (Petersen-Ellis, 2019). However, due to low transparency and the lack of data on illegal activity, the emerging trends have not been studied widely (GRID-Arendal, 2017).

3.4 LCA studies within plastic waste

The growing environmental awareness of the negative externalities arising from the plastic waste trade has resulted in many studies investigating the environmental impact of waste management and plastic pollution. Life Cycle Assessment (LCA) has been widely used within this context as LCA is the leading methodology in assessing the environmental impact of products and processes throughout the entire life cycle. LCA studies in the context of plastic waste have mainly developed within the fields of waste management and focused on evaluating the environmental impact of different end-of-life treatment methods such as recycling, incineration and landfill (Lou & Nair, 2009; Rabl, Spadaro & Zoughaib, 2008; Shonfield, 2008) and try to identify solutions for managing waste (Barton et al. 2000; Finnveden, 2000; Saner et al. 2012). Most of the results agreed with the waste hierarchy, meaning recycling is the environmental prefered option over incineration and landfill (Finnveden et al. 2001; Morris, 2005; Björklund & Finnveden, 2005; Schmidt et al. 2007; Perugini et al. 2005).

LCA studies connected to trade with plastic waste (Qu et al. 2019; Ren et al. 2020; Wen et al. 2021) put special attention to the role of China in the global recycling industry and the implication of the ban using different post-ban-scenarios. For China, the ban created a shortage of recycled material in their manufacturing industry, especially for PET. Ren et al. (2020) concluded that the shortage of recycled material for PET fibre production in China increased the PET production of virgin plastics. Therefore, the unintended environmental consequences of the ban are essential to analyse from different perspectives and consider different types of plastics. Qu et al. (2019) evaluated different end-of-life treatments in China, and as previous studies confirm, mechanical recycling showed overwhelming advantages for environmental impacts over landfill and incineration. Further, the result also indicated that the ban would decrease transportation distances and reduce the environmental impact, most notably a reduction of 85% for marine ecotoxicity potential. However, Chet et al. (2019) focus only on China and do not consider the environmental effects of changing trade patterns and the potential additional environmental consequences. Given the reduction of marine ecotoxicity in China, it may rise in Southeast Asia as these countries were the new destination for plastic waste exports.

Wen et al. (2021) used LCA to investigate the environmental impacts, in a short-term perspective, of the ban and the impacts of the trade flow changes and treatment methods of plastic waste (by type) in developed and developing countries. Their trade data is divided into a Baseline Scenario and 2018 Scenario, where the Baseline scenario (2008-2016) does not include figures from 2017 and therefore exclude data from the year that the ban was announced. Further, Wen et al. (2021) use only data from 2018 to represent the “post-ban” scenario, meaning that their assumptions about the impact of the China ban are not fully supported in reality. Even if trade data from 2018 give a snapshot of the effect after the ban, it does not account for the potential waste market restructuring delay and the redirection of plastic waste to Southeast Asia. Data for 2019 should instead be used to give a more accurate impact of the trade flow changes after the ban.

However, even if a number of LCA studies (Qu et al. 2019; Ren et al. 2020; Wen et al. 2021) have focused on the Chinese import ban and the effects, they fail to account for the redirected trade flow and the environmental impact in Southeast Asia. This study uses different scenarios to analyse the environmental impact if one exports to China (as before the ban), to Southeast Asia (after the ban) and compares it with no exports of waste from the EU. In addition, the post ban scenario covers 2019 figures and trade data until 2020 have been considered. This study further gives a broader analysis of the global recycling industry and how the biggest European exporters' rates of recycling, incineration and landfill changed.

4. Conceptual framework

4.1 Life Cycle Assessment

The increased awareness of environmental protection and the impacts associated with a product's life cycles has increased interest in developing methods to better understand and address these impacts. Life cycle assessment (LCA) has been used for over 40 years and is one of the main methods worldwide to evaluate the environmental impact of a product or system process (Golsteijn, 2020). LCA is embedded within a broader framework of Life Cycle Sustainability Assessment (LCSA) and aims to evaluate both the environmental, social and economic impacts throughout a product's life cycle to provide guidelines for decision makers. As LCSA considers all three sustainability dimensions, the broader framework combines the three methods of Life Cycle Assessment (LCA), Life Cycle Costing (LCC), and Social Life Cycle Assessment (SLCA) (Chang et al. 2017). LCSA helps to clarify the different trade-offs between the three sustainability pillars of economic, social and the environment.

LCA can be performed for a product, system process or service with various scopes, *cradle to gate* (from raw material extraction to factory gate), *gate to gate* (focus only on the manufacturing processes) or *cradle to grave* (from raw material extraction to disposal). The most common scope is the latter, and the analysis covers all stages from the extraction of raw material (cradle) to disposal/re-use (grave). Moreover, it helps researchers quantify the resource usage, pollution emissions to the air, water and soil, wastes disposal and other negative externalities of the life cycle of products/activities. LCA includes recognizing inefficiencies across life cycle phases, choosing between materials for product development, strategy planning, and is a valuable tool for policymakers (Golsteijn, 2020).

LCA has historically been discussed between scientists whether LCA is a framework or a practical tool; both views are correct and depend on the context (EEA, 1997). However, according to the International Organisation for Standardisation (ISO) 14040:2006, "LCA is a technique for assessing the environmental aspects and potential impacts associated with a product..." (n.p), including compiling an inventory of data, evaluation of the environmental impacts and interpreting the results in relation to the goal and scope of the study (see figure 2 for a graphic illustration). Over the past decades ISO has developed an internationally standardized methodology for practitioners of LCA (Kloepffer, 2008). The ISO 14040 standard includes the four-step process of; 1) Goal and scope definition, 2) Life cycle Inventory, 3) Impact assessment, and 4) Interpretation (ISO 14040:2006).

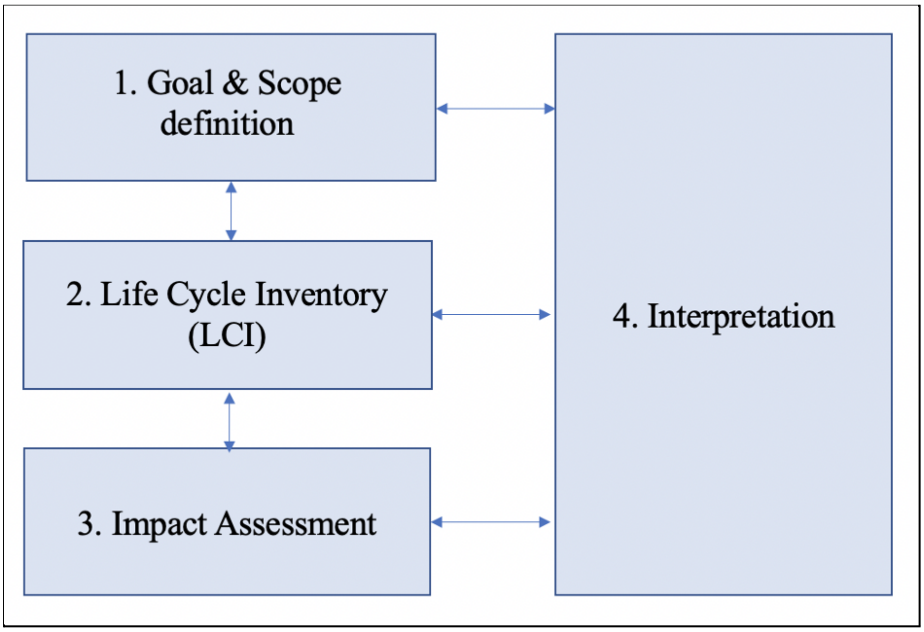

Figure 2 - Life Cycle Assessment, illustrated by the author

The first stage, *goal and scope definition*, is the planning phase, including goal setting, system boundary definition, the assumptions needed in the calculations and what type of functional unit will be used. The system boundaries define the scope of the study, start and endpoint of the research process and depth of analysis which is of high importance because an analysis can, in theory, never be fully finished (Ecochain, 2020). Within this stage, it is also essential to determine a functional unit (FU), "the measure of performance which the system delivers" (Zbicinski et al. 2006, p.92) and are usually numbers of the product (e.g. one bicycle or a book) or amounts of product (e.g. 1 kg paper). Given a well-defined goal and scope, the LCA will be performed consistently (Golsteijn, 2020).

The second stage, called the *life cycle inventory* (LCA) is the data collection process and quantifies each life cycle phase inflow (resources consumed) and their outflow (emissions). This analysis can be highly complex and time-consuming due to the construction of supply chains or different processes (Ecochain, 2020). However, a lot of data today is already collected in big LCI databases integrated into LCA software tools such as SimaPro or GaBi. The different databases are usually called EcoInvent, Agri-footprint or Exiobase and consist of thousands of LCI datasets at product, process or industry level (SimaPro, 2020). All different processes within the system boundaries need to be collected for all activities.

The third stage is the *impact assessment* and links the LCI data to selected environmental impact categories given a chosen methodology. ReCiPe is the most commonly used method developed in 2008 between RIVM, Radboud University Nijmegen, Leiden University and PRé Sustainability (RIVM, 2016). The ReCiPe methodology for impact assessment in LCA studies aims to transform the long list of life cycle inventory results (the data collected) into a number of indicator scores. To measure the environmental impact of the life cycle, the ReCiPe methodology has 17 different impact categories (see figure B1, Appendix B) covering human health, climate change, ecosystem and resources. Within this method, three mandatory steps need to be taken; select impact indicators, classification, and characterization to evaluate how significant the environmental impacts are given the LCI (Pajula et al. 2017). First, a selection of the impact categories need to be made based on the goal and scope of the study (Zbicinski et al. 2006). After selection, the classification step includes assigning the LCI to the impact categories and are made by the chosen LCA software. Different emission types contribute to the same impact category, e.g. methane and nitrous oxide also play a role in global warming potential measured in CO2-equivalent. This grouping of the LCI results into common units, and impact categories refer to the last part of the impact assessment, characterization (Ecochain, 2020).

The final stage, called *interpretation*, includes a presentation and evaluation of the results followed by a discussion of recommendations. In addition, the result is compared with the goal of the study. However, this stage of interpretation can happen at any time during the process and is especially helpful when doing complex LCA studies (Ecochain, 2020).

4.2 Limitations with LCA

As with all complex assessment tools, LCA has its limitations. Although the LCA methodology is standardized according to the ISO and provides a general framework, a lot of the interpretation of the results is up to the person conducting the assessment (Chang et al. 2017). As most LCA studies rely on assumptions, scenarios and different scopes, it is hard to compare them. One study may leave out critical environmental indicators or processes that another study has included, leading to different LCA results. In addition, the data collected during the LCI is both time consuming and resource-intensive if one needs additional datasets that are not already included in the software tool used. However, the different datasets in, for example, the EcoInvent database, do not always provide detailed information about assumptions made in the data collection process (Curran, 2012). When analysing different scenarios, the LCA results do not always identify a clear "winner" since one scenario can result in high emissions to air but low impact on the marine environment. Marine littering is also hard to capture when doing an LCA due to lack of underlying data and limitations in impact categories (Chang et al. 2017). Although LCA is a transparent method for estimating impacts and provides useful environmental analysis to decision-makers, it does not consider the social or economic dimensions. The wider framework of LCSA needs to be considered if one aims to include all dimensions of sustainability, however as this study is concerned with the environmental aspect, LCA will only be considered.

5. Data

The following chapter aims to provide the reader with a detailed picture of the data used in this study. Table 1, gives an overview of the different data sources, where trade data is taken from the UN Comtrade database, waste statistics of end-of-life methods from multiple different sources and LCA data from the EcoInvent database.

Table 1 - Data overview

Data of	Focus	Data source	Time period
Trade	Exports to China	UN Comtrade database	2008-2020
	Exports to Southeast Asia		
End-of-life methods	China	China Plastics Industry Yearbook, 2017	2017
	Southeast Asia	"What a Waste 2.0" report	2019
	EU countries	"Plastic - The facts 2018" report	2017
		"Plastic - The facts 2020" report	2019
LCA	Scenario 1	EcoInvent database* in SimaPro	
	Scenario 2		
	Scenario 3		

*Note: *all datasets used within the EcoInvent database can be seen in table B1, Appendix B.*

5.1 Trade data

The plastic waste trade data have been taken from the UN Comtrade database since it is the most comprehensive international trade database and includes detailed import and export trade data reported by country governments (Brooks, Wang & Jambeck, 2018; Weng et al. 2021). Trade with plastic waste is reported under the Harmonized System, HS commodity code 3915 “waste, parings and scrap, of plastics”. It includes waste consisting of PE, PS, PVC and "other plastics", which are PET and PP, not yet harmonized by the UN (Brooks, Wang & Jambeck, 2018). The trade data in this study is measured in net weight (tonnes) and not in trade values (US$) to exclude potential problems if countries have reported differently based on currency conversion. The time period covers data from 2008 until 2020 in order to provide the reader with a historical trend line and add to previous studies.The sample consists of 17 countries, where Japan, the US, Mexico, Germany, the UK, Belgium, Spain, Italy, France, and the Netherlands act as the main exporting countries and China, Hong Kong, Thailand, Indonesia, the Philippines, Vietnam and Malaysia as the main importing countries. These 17 countries accounted for 92% of the global plastic waste trade before the ban (Wen et al. 2020).

However, there are a few possible limitations with the trade data obtained from the UN Comtrade database. It reports only bilateral trade flows, meaning that we cannot accurately track the movement of waste between different countries until it reaches the final destination. As the dataset is based on HS 3915 “waste, parings and scrap of plastics” the data does not show how much PE, PVC, PP, PET and PS the traded waste consist of, meaning that it may not represent our functional unit based on the

EU demand in 2019 (see figure B2, Appendix B for illustration). There is missing trade data to the Philippines and Vietnam before 2015 but as this study wants to identify the trade pattern before and after the ban the missing trade data will not be an issue and therefore not further discussed. However, another data problem that has been noticed is that trade values in US$ can be reported although that the net weight is zero, e.g the UK export to Malaysia in 2019 where according to the UNComtrade database 0 tonnes in net weight, but almost 12 million in US$ (UNComtrade, 2021). This is probably due to lack of data but needs to be taken into consideration when analysing the results.

5.2 Waste data

Data of most common end-of-life treatment methods (recycling, incineration, and landfill) has been collected from the biggest exporting EU countries, Germany, the UK, Italy, France, Spain, Netherlands, and Belgium. To compare waste management data before and after the ban, data for 2017 have been compiled from the report "Plastic - The facts 2018" and aim to represent the before the ban period whilst data for 2019 represent the post-ban period and have been compiled from the report "Plastic - The facts 2020" by PlasticEurope (the Association of Plastics Manufacturers in Europe) and EPRO (the European Association of Plastics Recycling and Recovery Organisations). 2019 figures are used instead of waste data from 2018 to account for the potential waste market restructuring delay due to the ban and give a more accurate picture of the effect of countries' domestic recycling, incineration and landfill rates. The underlying data in the reports of waste recovery data has been collected by Conversio Market & Strategy GmbH given official statistics from European authorities and waste management organisations. Data for 2020 has not yet been available and that is why “Plastic - The facts 2020” only look at figures from 2019.

Data of recycling, incineration, and landfill rates (%) for China have been taken from the China Plastics Industry Yearbook, 2017 and aims to mirror the statistics from the year before the ban was implemented. The recycling rate for plastic was estimated to be 30% based on the average statistics on imported recycling rates for PVC and PET. Data of detailed recycling information for each plastic category has not been found. Statistics for rates of incineration and landfill in China were directly taken from the Yearbook and set to be 30% and 40% respectively. Data for Southeast Asian countries have been compiled from the "What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050” report from the World Bank in 2018 (open dump, 75%; landfill, 4%; recycling, 5% and open burning,16%) (Kaza et al. 2018). The data from the World Bank report representing Southeast Asia waste management has been collected led by World Bank solid waste experts and is based on a global project aimed to aggregate global solid waste data.

Although waste statistics used in this study is based on official statistics from reliable sources, there are some general difficulties with waste statistics that have been taken into account during the data collection. First, there has for a long period of time not been an international framework that can provide a basis for harmonising waste definitions, classifications and methods of calculations (UNECE, 2020). National official statistics do not include illegal imports and exports and illegal dumping. The reported annual illegal shipments (discovered instances) can vary between 6,000 and 47,000 tonnes according to a report of the European Environment Agency only within Europe (UNECE, 2020). Lack of transparency and data reporting within the recycling industry are a growing concern especially outside the EU.

5.3 LCA data

To analyse the environmental impact of the three scenarios, data have been taken from the Ecoinvent v3 database and contains over 15,000 interlinked datasets of LCI at process level. The data covers all kinds of sectors such as transport, energy supply, agriculture, textile, electronics, wood, chemicals, construction and waste treatment (Ecoinvent, 2020). Ecoinvent is the most comprehensive, transparent and global leading LCI database and integrated into SimaPro, the LCA tool used in this study (SimaPro will be further discussed in chapter 6). For a detailed overview of the datasets taken from Ecoinvent 3.6, see table B1 (Appendix B). Additional data of transport distances have been obtained from Seadistances.org and are estimated to be 18,000 km from the EU to China/Southeast Asia. The estimated sea distance is assumed in this study to be the same for both scenario 1 (export to China) and scenario 2 (export to Southeast Asia).

During the data collection in SimaPro, three data quality requirements have been set; validity, reliability and accessibility, which is an important activity when conducting an LCA (Baumann & Tillman, 2004). The data must be consistent with the goal of the study and the aspects of time and geographical scope. Behind the EcoInvent database is a non-profit organization dedicated to promoting and supporting the availability of global environmental data. The database provides high transparency as individual unit process data can be found for all datasets with documentation of how the data has been collected. To ensure reliable data and high quality, all data are subject to internal and external reviews and updated on a regular basis (EcoInvent, 2020). Worth notice is that the Ecoinvent database is not accessible without having a SimaPro license or purchasing an EcoInvent license at their webpage.

6. Methods

As this study is concerned with both plastic waste management at a global level both from the waste management perspectives (research question 1) and from an environmental perspective (research question 2), this chapter aims to describe the methodological approaches used in this study.

6.1 Waste management

Given the first research question; how exporting countries' waste management has changed after the ban, trade data of plastic waste exports and waste data by treatment method have been collected.. Trade data was collected from the UN Comtrade database under the HS commodity 3915 "plastic waste and scrap" from 2008-2020 to analyse the trade patterns. In addition, waste management data of recycling, incineration, landfill have been collected from the european countries, Germany, the UK, Italy, France, Spain, Netherlands, and Belgium from 2016 and 2019 to represent the before and after ban scenarios. The US, Japan and Mexico are only included in the trade graphs and not analysed in terms of domestic waste management due to lack of data and different methods of calculations. All tables and figures have been done in Excel.

6.2 LCA

The second research question is related to the environmental impact of the exporting plastic waste to China, to Southeast Asia, or domestic management within the EU.

- *Scenario 1:* Exporting plastic waste to China with waste management rates at; 30 % recycling, 30% incineration and 40% unsanitary landfill.
- *Scenario 2:* Exporting plastic waste to Southeast Asia with waste management rates at; 5 % recycling, 16 % open burning, 4 % landfill and 75 % open dump.
- *Scenario 3:* Domestic waste management within average EU rates of 35 % recycling, 40% incineration and 25% sanitary landfill.

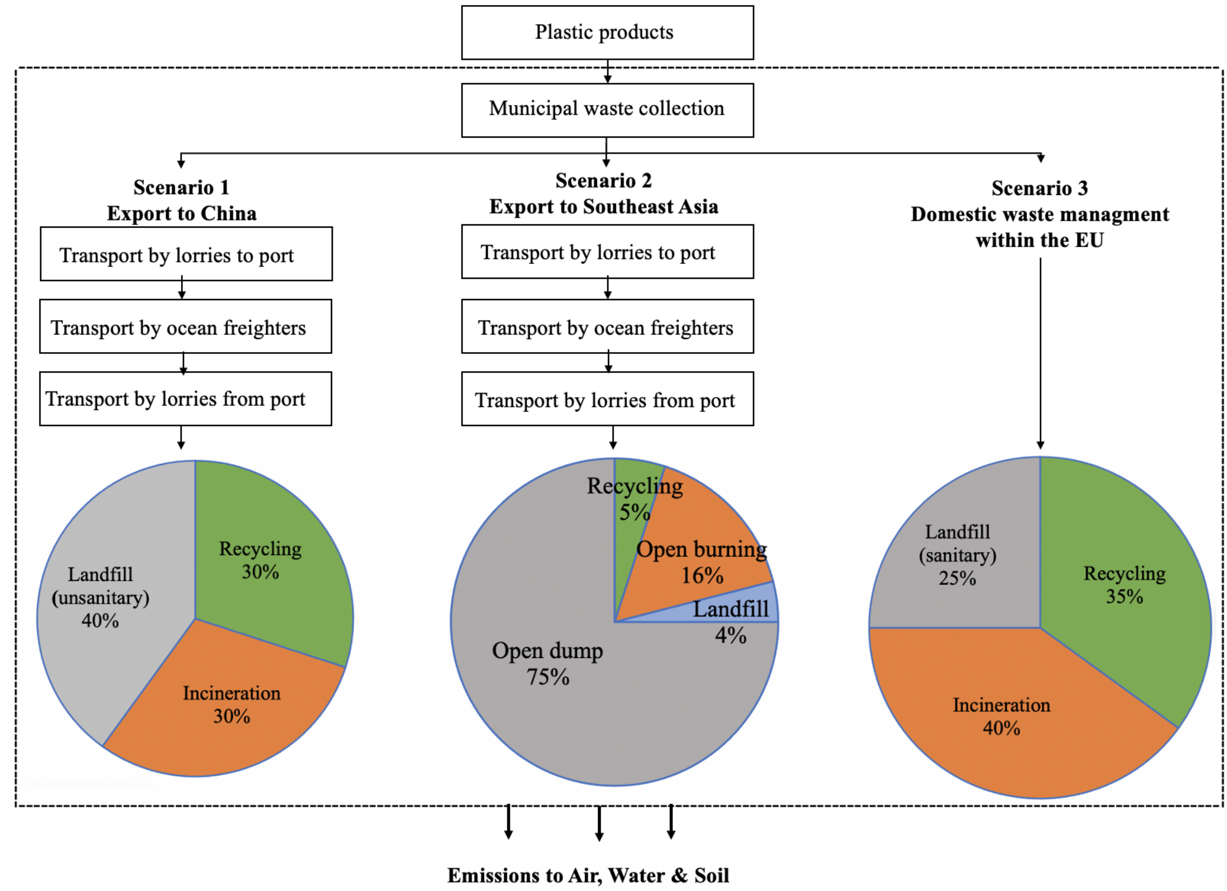

Figure 3 - System boundaries for LCA
Note: See table 1 for sources.

Scenario 1 aims to represent the trade pattern before the ban since China was the dominant importer of plastic waste until 2018 (Jambeck et al. 2015). To give an appropriate view of how waste was managed in China before the ban, data has been compiled from the Plastics Industry Yearbook, 2017. Scenario 2, represents Southeast Asia as these countries became the new import destination for the world's plastic waste after the ban. Due to lack of data on how plastic waste is managed in these countries, statistics of average municipal waste management for 2018 has been taken from the UN report “What a Waste 2.0”. Scenario 3, represent no exports and only domestic waste management within the EU. As waste management differs across member states, average rates of recycling, incineration and landfill have been taken from PlasticEurope “The Facts 2020” report.

Life cycle assessment (LCA) has been used as the methodology framework since it is currently the most scientific and effective approach for environmental impact analysis (Ye et al. 2017). The software used to conduct the LCA is SimaPro as the impact assessment (step three in an LCA) needs a powerful LCA software to calculate the environmental impact given the collected data (LCI). All calculations were done in *SimaPro 9.1.1*, the most well-known LCA tool for calculations. Sima Pro has been developed by PRé Consultants and is the world's leading LCA software to collect, analyse and monitor the sustainability performance data of a product's life cycle (Ecochain, 2020). The assumptions made include exports of plastic waste by ocean freighters and not by other modes of transportation, all plastic waste imports to China and Southeast Asia. Further, LCA for plastic waste has been carried out according to ISO 14040 standards as described in the conceptual framework (ISO, 2006) and included the four-step process; 1) Goal and scope definition, 2) Life cycle Inventory, 3) Impact assessment, and 4) Interpretation (see chapter 7, for step 4).

6.2.1 Goal & Scope definition for LCA

The goal is to compare the environmental impact of the above-mentioned waste management scenarios to guide and make both companies and policymakers aware of the importance of domestic waste management. The functional unit is assumed to be 1 kg of post-consumer plastic waste consisting of 30% PE, 20% PP, 10% PVC, 10% PET, 5% PS and 25% "others" (see figure B2, Appendix B for illustration) and aims to represent the plastic demand within the EU in 2018 (PlasticEurope, 2018). It is important to note that all these different types of plastic vary in their chemical composition, recyclability, and hazardous nature (Thiounn & Smith, 2020), meaning that 1 kg of PET vs 1 kg of PS have a different environmental impact. By weighing the proportions of different plastic types, the result of the functional unit is a good representation of the plastic waste composition within the EU.

The system boundary represents the scope of the LCA and is illustrated in figure 3 above, meaning that this study will not consider the entire life cycle of plastic from production. The cradle to grave, end-of-life phase for domestic plastic waste management begins from the transportation of plastic waste from collection spots to disposal facilities followed by three common treatment methods, landfill, incineration (with energy recovery) and recycling. The alternative end-of-life scenario 2 includes exports to Southeast Asia according to the new trade patterns. Additional transport is needed, and the waste treatment methods consist of open dump, open burning, landfill and recycling, which represent the dominant waste treatment methods for our Southeast Asian countries in 2018 (World Bank, 2018).

6.2.2 Life Cycle Inventory

To construct the three scenarios, an LCI has been conducted for each scenario (Table B2), meaning that the material and energy inputs and emission outputs have been quantified given the functional unit of 1 kg post-consumer plastic waste. As mentioned in the data section, the EcoInvent 3.6 database has been used as the source database in SimaPro (EcoInvent, 2010). The distance of ocean freighter transport was obtained from the website [https://sea-distances.org] and estimated to be 18 000 kilometres between the biggest port in Germany and Hong Kong. Road transport distances by lorries for all scenarios were estimated to be 100 km, which is in line with the previous study Weng et al. (2021).

6.2.3 Impact assessment

The environmental impact of the three different waste management scenarios has been assessed using the well-established ReCiPe method at midpoint level to cover both the environmental dimension of human health damage, ecosystem quality and resource scarcity (see figure B1, Appendix B for illustration). The selected impact categories are; *global warming,* GW (kg CO2 eq), *fine particulate matter formation*, FPMF (kg PM2.5 eq), *freshwater eutrophication* (kg P eq)*, marine eutrophication* (kg N eq)*, marine ecotoxicity (*kg 1,4-DCB eq) and *human non-carcinogenic toxicity*, HCT (kg 1,4-DCB eq)*,* and were chosen based on common environmental damage indicators for, human toxicity[2], marine environmental damage and emissions to air. *Global warming* is a measure for climate change in terms of greenhouse gases and takes into account the total emissions to air in terms of carbon dioxide, methane gas and other substances. *FPMF* refers to tiny particles in the air with a diameter of less than 2.5 µm (PM2.5) consisting of a complex mixture of organic and inorganic substances that can have a substantial negative impact on human health, ranging from respiratory symptoms to hospital admissions (RIVM, 2016). *Freshwater eutrophication and Marine eutrophication* measure the amount of phosphorus and nitrogen-weighted increase in kg P-eq and kg N-eq in relation to species loss (RIVM, 2017). *Marine ecotoxicity* indicates hazard-weighted increase in marine water and the toxicological responses of different species. *Human non-carcinogenic toxicity*, is a calculated index and reflects the amount of dangerous chemicals released into the environment and are toxic for humans through inhalation or ingestion (Aitor, P; Rodriguez, C & Ciroth, A, 2016).

The exact calculations behind these impact categories are highly complex and the reason for not going deep into how they are calculated. The LCA result given the ReCiPe method shows an impact value given the functional unit of 1 kg of post-consumer plastic waste for each impact category. The reason for not choosing ReCiPe at endpoint level is the increased uncertainty in the results due to aggregation of indicators (see figure B1, Appendix B)(Finnveden et al. 2009; RIVM, 2011). ReCiPe at midpoint level has a stronger relation to the environmental flows and a relatively low uncertainty but it is worth mentioning that the impact assessment is a complex process where both the classification and characterization is calculated in SimaPro.

7. Results

This chapter aims to present the result given the research questions. The first part captures research question 1, how exporting countries' waste management has changed after the ban. Both changes in trade flows and domestic rates of recycling incineration and landfill will be presented in graphs. The second part of this chapter captures research question 2 and presents the LCA result based on the three scenarios.

7.1 Changes in waste management

Figure 4 shows the development of exports to China between 2008 and 2020 for the biggest exporting

[2] Human Toxicity reflects the potential harm of a unit of chemical released into the environment. It is based both on the inherent toxicity of a compound and its potential dose using the reference unit, kg 1,4-dichlorobenzene (1,4-DB) equivalent. Dichlorobenzene is a toxic organic compound with the formula, C6H4Cl2.

countries. The US and Japan export the biggest amount of plastic waste over the entire time period, whilst Germany acts as the biggest exporting country within the EU. Given the result, it is clear that the Chinese import ban had a significant direct impact on countries' export volumes, as the total exports dropped by 96% between 2016 and 2018 (UNComtrade, 2021). Figure 4 also indicates that countries' export volumes started to decrease already in 2017 when the ban was announced by China. Although China played a major role in the global plastic waste trade, Hong Kong also received a huge amount of exports before the ban and has the same trade patterns as China. Hong Kong acted as a transfer hub which can be seen in figure C1 (Appendix C), the total amount of export to China compared to the total amount of export to China and Hong Kong, consequently Hong Kong played a major role in the global plastic waste trade

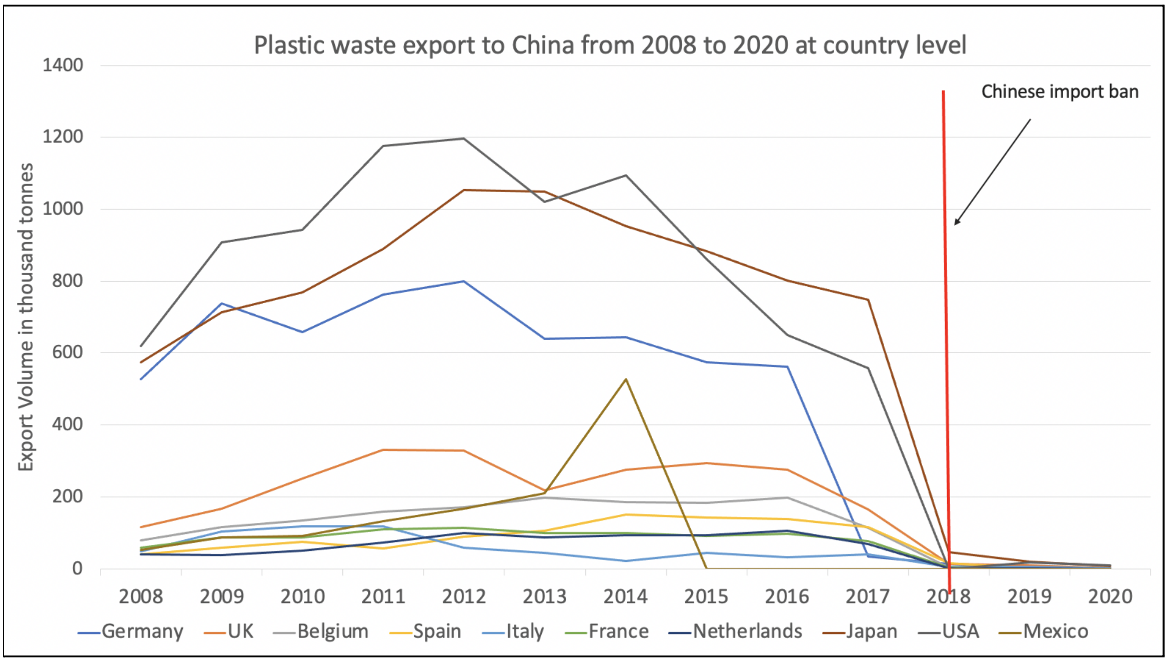

Figure 4 - Plastic waste exports to China between 2008 and 2020 at country level (UNComtrade, 2021)

Germany's export of plastic waste to China dropped by 98% (549 thousand tonnes) between 2016 and 2018 and during this time period the export destination increased remarkable to Malaysia (+161%), Indonesia (+378%), Vietnam (+122%), the Philippines (+767%) and to Thailand (+968%). As the total amount of exports going to these 5 Southeast Asian countries don't represent the total of 549 thousand tonnes show the complexity of plastic waste trade.

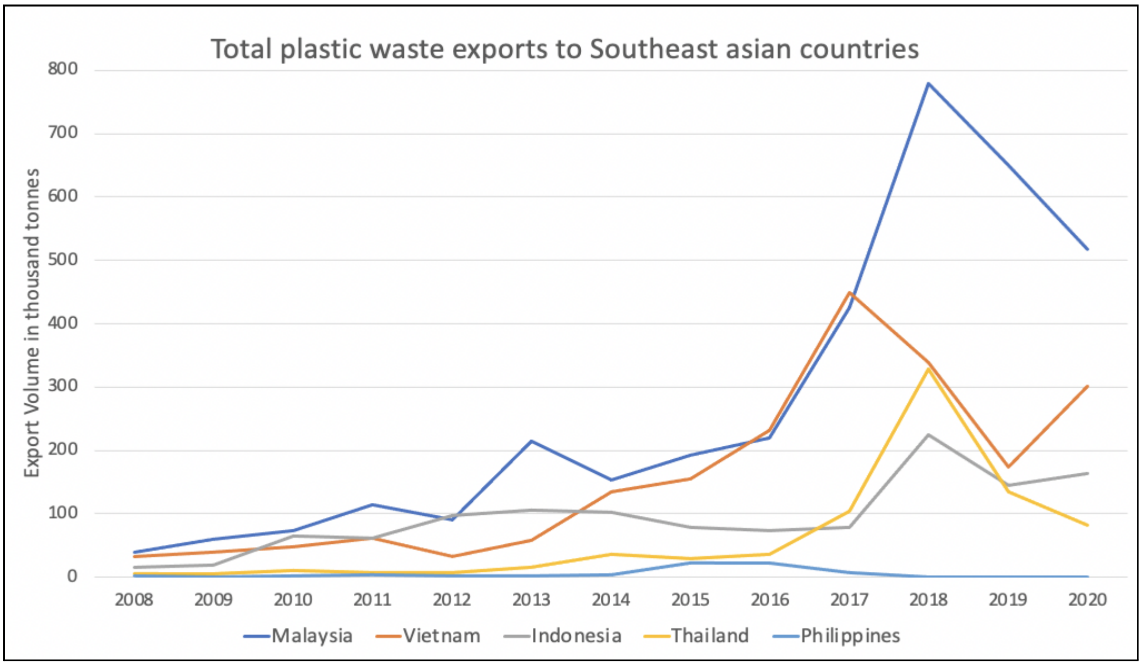

Figure 5 - Total plastic waste exports Southeast Asian countries between 2008 and 2020 (UNComtrade, 2021)

Note: Total plastic waste exports from Germany, UK, Belgium, Spain, Italy, France, Netherlands, Japan, the US and Mexico.

The total exports of plastic waste from the exporting countries in our sample clearly indicate that the Chinese import ban has resulted in an increase in trade flow to Malaysia, Vietnam, Indonesia, Thailand and to the Philippines given figure 5. The total exports to Malaysia increased on average by almost 300% from our exporting countries between 2016 and 2019, to Indonesia (+200%) and to Thailand (+380%). Common for all Southeast Asian countries, except for Indonesia, is that the US and Japan export volumes are significantly higher compared to the European countries, meanwhile Germany acts as the dominant exporter of the European countries. The huge increase of plastic waste to these Southeast Asia is consistent with the result found in previous studies (Liang et al. 2021; Brooks, Wang, & Jambeck, 2018)

As many of our Southeast Asian countries quickly imposed import restrictions around mid-2018 one can see from figure 5, that there is a drop in the amount of exports after 2018, especially for Malaysia, Indonesia and Thailand. Exports to Vietnam started to decrease already in 2017 as the ban was announced but not implemented in China. One surprising fact is that the exported amount started to increase again after 2019 to Vietnam and Indonesia which has not been shown in previous studies. The trade flow trends at country level can further be seen in Appendix C for Malaysia (figure C3), Vietnam (figure C4), Indonesia (figure C5), Thailand (figure C6) and the Philippines (figure C7). As this study has trade data until 2020, it is clear that exports to these countries may start to increase even more although new regulations have been implemented to restrict plastic waste trade.

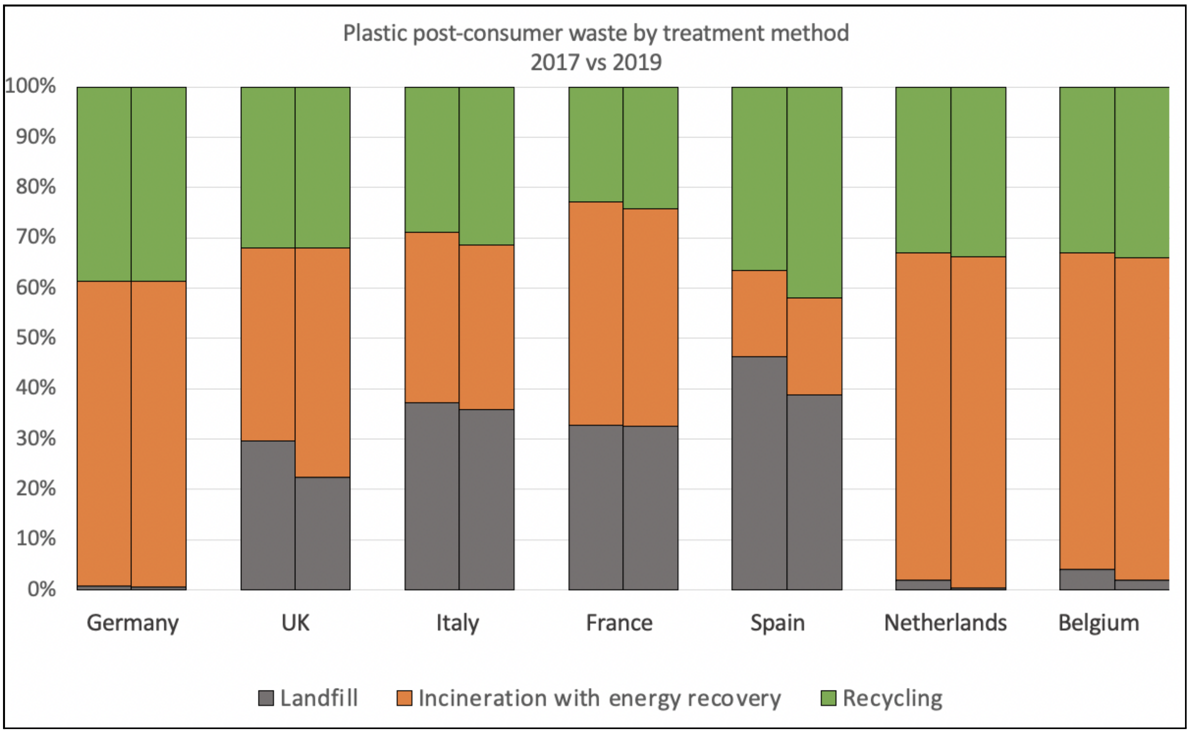

Figure 6 - Waste management by treatment method before (2017) and after the ban (2019)
Note: The left part of each column represents 2016 and the right part, 2019.

As figure 4 and 5 confirms the redirected trade flows of plastic waste trade after the ban, figure 6 indicates that our European exporting countries domestic end-of-life treatment rates have not changed significantly. This is true for almost all of our seven European exporters even if they have different rates of recycling, incineration and landfill. Spain is the only country in our sample of the EU countries that have increased domestic recycling rates (+5.4%) and decreased the amount ending up at landfill (-7.6%) between 2017 and 2019. The amount of plastic waste going to incineration with energy recovery in the UK has increased after the ban with 7.4% (figure 6).

Given figure 6, the Chinese import ban, surprisingly, has not resulted in higher domestic recycling rates with only marginal improvements in waste management when looking into Spain figures. The stagnated trend in recycling rates can be confirmed if one considers figure C2 (Appendix C) and shows an historical trendline for recycling, incineration and landfill for the EU-28. Before 2017, both recycling and incineration with energy recovery had a positive trend and landfill rates decreased. As the ban was announced in 2017, both figure C2 (Appendix C) and figure 6 indicate that recycling rates have stagnated until today and only contributed to changed trade flows to Southeast Asia (figure 5). By comparing domestic waste statistics with the trade patterns, this study further contributes with new perspectives on both how export strategies changed and how our exporting countries' domestic recycling industry reacted to the ban.

7.2 LCA results

The LCA results given the system boundaries can be found in figure 7 where the middle pillar for each impact category represents scenario 1, right pillar scenario 2 and left pillar scenario 3. As shown in figure 7, scenario 2, exporting plastic waste to Southeast Asia is worse for the environment for 3 out of 6 environmental impact categories. As each impact category is measured in different units, figure 7 only represents the LCA result in relative terms. Table 2 instead, shows the total impact values for each impact category and scenario given the functional unit of 1 kg of plastic waste.

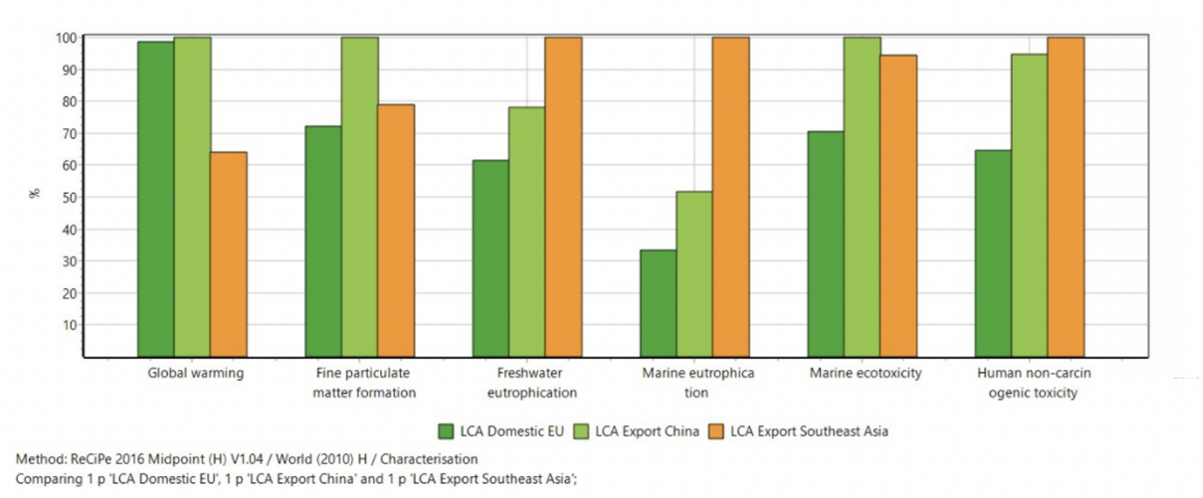

Figure 7 - LCA results of the 3 waste management scenarios

Note: Higher pillar meaning worse environmental damage in terms of the impact category.

Table 2 - The impact values at midpoint level of 1 kg of post-consumer plastic waste

Impact indicator	Scenario 1: Export to China	Scenario 2: Export to Southeast Asia	Scenario 3: Domestic EU	Unit
Global warming	3,548242	2,272682	3,500999	kg CO2 eq
Fine particulate matter formation	0,005586	0,004404	0,004036	kg PM2.5 eq
Freshwater eutrophication	0,000280	0,000358	0,000221	kg P eq
Marine eutrophication	0,000460	0,000890	0,000296	kg N eq
Marine ecotoxicity	0,217551	0,205302	0,153163	kg 1,4-DCB
Human non-carcinogenic toxicity	3,467800	3,662538	2,366810	kg 1,4-DCB

For 1 kg of post-consumer plastic waste being exported and treated in China given their end-of-life treatment methods of 30 % recycling, 30% incineration and 40% landfill (scenario 1), 3,5 kg of Co2-eq will be released to the air (table 2). Taking the total amount of plastic waste exports (2.9 million tonnes) from our exporting countries to China in 2016 (before ban) into consideration, it represents 10 million tonnes of Co2-eq. To understand the quantity, it is approximately one fifth of the total emissions in Sweden in 2020 (Naturvårdsverket, 2021). Compared to scenario 2, export 1 kg of post-consumer plastic waste to Southeast Asia, given their rates of open burning, landfill and open dump, 2.27 kg Co2-eq are emitted (table 2). Considering the total amount of 1.7 million tonnes exported in 2018 (after the ban), it represents 3.8 million tonnes of Co2-eq. Surprisingly, scenario 3 has almost the same impact value as scenario 1, meaning that exporting and treating 1 kg of post-consumer plastic waste in China releases almost the same amount of CO2-eq if it would instead be treated within the EU. This indicates that the transportation by ocean freighters don´t have a significant effect in terms of emissions in Co2-eq and export to Southeast Asia, surprisingly, have the lowest environmental impact in Co2-eq.

In terms of human toxicity, the impact category, *Human non-carcinogenic toxicity,* measured in

1,4-DCB, has a significantly higher impact value for scenario 1 and 2, compared to scenario 3 (table 2). However, *Marine ecotoxicity* (also measured in 1,4-DCB as human toxicity) has a significantly lower impact value than *Human non-carcinogenic toxicity* meaning that the environmental impact of the life cycle of plastic waste trade is relatively worse for human health compared to marine ecotoxicity.

The impact categories covering marine wildlife and freshwater species are *Marine ecotoxicity*, *Freshwater eutrophication* and *Marine eutrophication.* Given table 2, eutrophication is worse for scenario 2, Southeast Asia. Probably due to high rates of waste ending up at landfill or open dumps, leading to leachate that contributes to eutrophication and groundwater pollution (Tinmaz, 2006). As *Marine ecotoxicity* measures the toxicological responses of different species, it can be seen as an indirect measure of microplastic in marine water. Our result from table 2 indicates that scenario 1, exports to China is the worse option in terms of marine ecotoxicity.

It is clear based on the LCA results found in figure 7 and table 2 that the environmental impacts of exporting plastic waste to China; to Southeast Asia, or domestic treatment within the EU differ a lot when considering different environmental impact categories. Exporting 1 kg of post-consumer plastic to Southeast Asia is worse for the marine impact categories due to high rates of open dump (75%) and no controlled landfills. Their high rates of open burning (16 %) also indicates that this export scenario is worse in terms of human toxicity for the people living in these countries (see table B2, Appendix B for detailed information). Scenario 3, domestic waste management within the EU, is the preferred option in terms of human toxicity and the impact categories; *Freshwater eutrophication*, *Marine eutrophication, Marine ecotoxicity and Human non-carcinogenic toxicity,* due to higher rates of recycling. Not surprisingly, domestic waste management within the EU, is the preferred option in terms of environmental impact given the results in table 2. However, in terms of Co2-eq, scenario 3 (EU) is almost as bad as exporting plastic waste to China. This implies that incineration, which has been confirmed by previous studies (Finnveden et al. 2001; Morris, 2005; Björklund & Finnveden, 2005) is not a preferred end-of-life treatment method in terms of Co2-eq as some people may think.

8. Discussion

This study aims to quantify the direct effect of the Chinese import ban of the biggest EU exporting countries trade flows, domestic treatment methods and analyse the environmental effects of different trade scenarios using LCA. The Chinese import ban led to a huge drop in exports to China (figure 4) for all of our exporting countries, which indicates that the ban fulfilled its aim of restricting imported waste. Our results also indicate the remarkable impacts the ban had on the global plastic waste trade as the total exports dropped by 96 % to China (between 2016 and 2018) and were redirected to Southeast Asia (figure 5). The trade flow changes before and after the ban are broadly consistent with the trends found in the literature (Brooks et al. 2018; Qu et a., Liang et al. 2021; Weng et al. 2021). The total exports to Malaysia increased on average by almost 300% from our exporting countries between 2016 and 2019, to Indonesia (+200%) and to Thailand (+380%). Our results also show that China played a major role in the global plastic waste trade together with Hong Kong who acted as a transfer hub, confirmed by Liang et al. (2021) and Weng et al. (2021). As trade data until 2020 is used, one can also see the increase in exports to especially Vietnam and Indonesia in 2019. Probably due to that additional restrictions to the Basel convention were announced in January 2019 and will be implemented in 2021.

Given that the Chinese import ban had a remarkable impact on the global plastic waste trade, it has not contributed to increased recycling rates for our exporting countries given figure 6. Recycling rates were still the same in 2019 as before the ban in 2017, probably due to the fact that our exporting countries only changed export destinations to Southeast Asia (figure 5). The ban clearly indicated that the global recycling industry is not working without China which is supported by the previous studies (Qu et al. 2019; Ren et al. 2020; Wen et al. 2021). This study highlights the inadequate domestic recycling capacities within the EU and the failure of the global recycling industry as our exporting countries only changed export destinations instead of developing their own. No improvements or investment in domestic recycling can be made if our developed EU countries continue to export their waste abroad or change export destinations when a ban is implemented. One could also question whether exporting waste abroad for recycling purposes is aligned with the waste hierarchy as they do not know the real fate of the plastic waste they sent. As NUI Galway & Limerick (2020) concluded that 31% of the exported plastic waste is not recycled in Asia, implies that there is low transparency, lack of reporting and hard to track what happens to the waste.

To further evaluate the export strategies from an environmental perspective, the LCA result in this study (figure 7, table 2) shows that exporting plastic waste to Southeast Asia is worse for 3 out of 6 impact categories (*Freshwater eutrophication*, *Marine eutrophication,* and *Human non-carcinogenic toxicity*). Although table 2 indicates that *Freshwater eutrophication*, *Marine eutrophication* has relatively low impact values given our functional unit of 1 kg of plastic waste, exporting plastic waste to Southeast Asia is the worst option in terms of environmental damage to the marine environment. This can be confirmed by Jambeck et al. (2015) who argues that those countries in Asia with high rates of mismanaged plastic waste also contribute most to marine litter.

The LCA results given human toxicity is worse for Southeast Asia and is not surprising due to their high rates of open burning, open dump and low recycling rates. However, even if it is hard to compare LCA studies in general due to different functional units, scope and methodology for the impact assessment, the result in this study include trade and confirms that high levels of open burning and open dump (the scenario of Southeast Asia) is worse for human toxicity, marine wildlife and that the high rates of incineration within the EU have a negative environmental impact in terms of CO2-eq. The damage to human health of open burning is confirmed by other studies (Greenpeace Malaysia, 2021; GRID-Arendel, 2017) and how incineration contributes to global warming (Qu et al. 2019; Wen et al. 2021). As no other LCA studies have compared these different trade scenarios and the environmental damage that plastic waste trade causes in Southeast Asia, the result given figure 7 and table 2 provide evidence that Southeast Asia is the worst option for the environment given the marine impact categories and for human health.

As our exporting countries have relied on China for so many years taking care of their waste and not developing their own recycling capacities, it is not surprising that the ban created a recycling crisis. Even with ambitious recycling targets and directives within the EU, major problems still exist. Technological innovation within the recycling industry will be crucial in order to complement current mechanical recycling and prevent plastic waste ending up at landfills. Whether the EU will make a transition to a circular economy and stop exporting waste, will highly depend on if the EU can overcome current obstacles within the recycling industry and push for an attitude change towards seeing all waste as a valuable material.

9. Conclusion

This study has investigated the direct effect of the Chinese import ban on our main exporting countries from the waste management perspectives (research question 1) and from an environmental perspective (research question 2). It is clear that the ban changed the global trade flows of plastic waste to Southeast Asia and given the LCA results, export to these countries is worse for the environment for 3 out of 6 impact categories. In terms of human toxicity, Southeast Asia is the worst scenario, meanwhile there is an environmental advantage in terms of human toxicity, marine pollution and freshwater eutrophication for scenario 3. In addition, this study can confirm that the recycling rates with the EU have stagnated after the ban and supports the argument of inadequate domestic recycling capacities. Given the results, it highlights the need to develop the domestic recycling industry within the EU and work upstream to prevent huge quantities of plastic waste being generated in the first place.

However, due to the high cost of recycling, there is also a risk that the rates of incineration with energy recovery will increase in the long run within the EU. As seen in our LCA results, scenario 3, domestic waste management within the EU with high rates of incineration is not optimal for the environment given high rates of Co2 emissions to air. Emphasis needs to be put on efforts both to invest in R&D and providing a more strict policy framework to ensure that their member states' waste management is aligned with the waste hierarchy, handling their own waste and developing their domestic recycling capacities. Maybe in the long term, the Chinese import ban will create an opportunity and force countries to manage their own waste but as seen in this study, no change in recycling rates. The Chinese import ban was truly a wake up call for the global recycling industry and in order to cope with the rising plastic waste problem we need to include life cycle thinking from the production process to consumer behaviour. As stated in the introduction, we will probably have the same amount of fish and plastic in the ocean by 2050 if no action is taken and countries continue to export plastic waste to Asian countries.

More research in this area of plastic waste trade and the environment is necessary before further conclusions can be drawn about the impact of the Chinese import ban on the environment. LCA studies can be combined with trade flow and waste generation data and network analysis to obtain a more complete picture of the global waste trade flows. Finally, waste statistics need a global framework for calculation and harmonising waste definitions to provide researchers with accurate and reliable data.

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Appendix A

Table A1 - Type of plastics, use, recyclability and toxicity level (AAA Polymer, Ritchi , 2018)

	Types of plastics	Common uses	Recycable?	Toxcity level
1 PET	Polyethylene terephthalate (PET)	Soft Drink, Water bottles, Biscuit Trays	Most commonly recycled plastic.	Most commonly used plastic but can leach the toxic metal antimony if it is exposed to higher temperatures.
2 HDPE	Polyethylene, high density (HDPE)	Shopping bags, milk bottles, shampoo	Most commonly recycled plastic	Appears to be safe
3 PVC	Polyvinyl chloride (PVC)	Cosmetic containers, Plumbing Pipes, Roof sheeting	Difficult to recycle	Most hazardous plasitc. Can leach BPAs, phthalates and other toxins during the life cycle.
4 LDPE	Polyethylene, low density (LDPE)	Garbage bags, Cling wrap, Squeeze bottles	Difficult to recycle	Appears to be safe
5 PP	Polypropylene (PP)	Bottles, Straws, Garden furniture, Packaging Tape	One of the least recycled plastics and a majority ends up at landfill or in the ocean	Appears to be safe
6 PS	Polystyrene (PS)	Styrofoam, CD cases, Plastic cutlery, Toys	Difficult to recycle	Highly toxic if considered as styrofoam. PS can also leach many toxins including styrene when exposed to heat.

Figure A1 - The Waste hierarchy (EU, Directive 2008/98/EC)

Appendix B

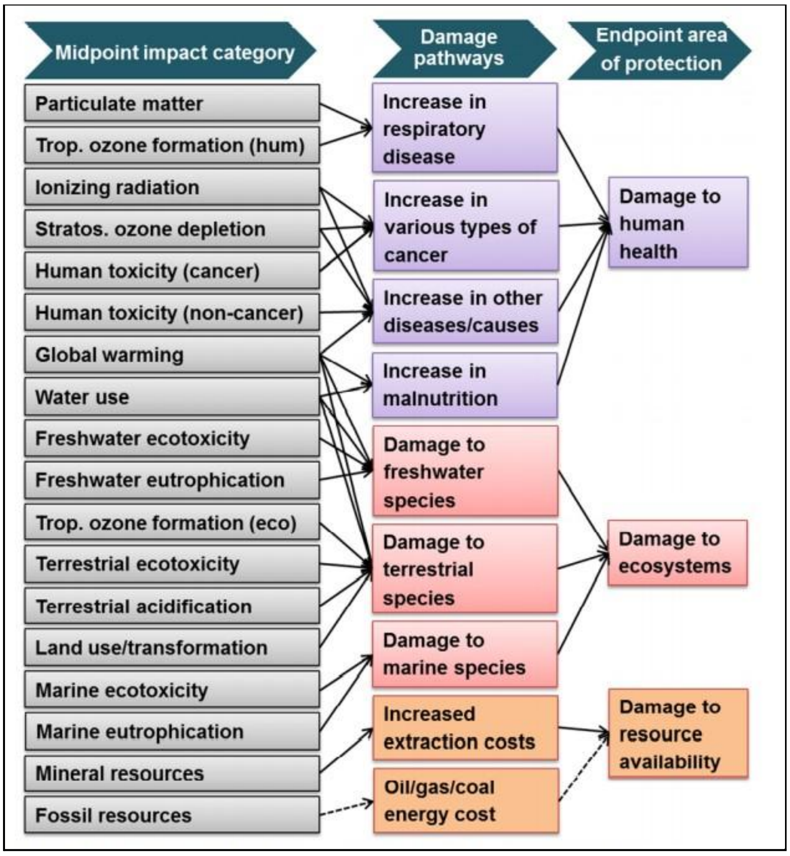

Figure B1 - Overview of the impact categories in the ReCiPe methodology (RIVM, 2016)

Table B1 - LCI data sets

Data of	Selected datasets - Unit processes in Sima Pro	Source database
Transportation	**Road**	
	Municipal waste collection service by 21 metric ton lorry {GLO} |market for| APOS, U	Ecoinvent 3.6
	Transport, freight, lorry 16-32 metric ton, euro4 {GLO} |market for|APOS, U	
	Transport, freight, lorry 16-32 metric ton, euro4 {GLO} |market for|APOS, U	
	Distance for container ship	https://sea-distances.org]
	Ocean	
	Transport, freight, sea, container ship {GLO} |market for|APOS, U	
Waste management	**Landfill**	
	Waste plastic, mixture {Europé without Switzerland}|sanitary landfill| APOS, U	
	Waste plastic, mixture {CH} |sanitary landfill|APOS, U	
	Waste plastic, mixture {GLO} |unsanitary landfill|APOS, U	
	Open dump	
	Waste plastic, mixture {GLO} |open dump|APOS, U	
	Incineration	Ecoinvent 3.6
	Municipal solid waste (waste scenario) {Europé without Switzerland} |incineration| APOS	
	Municipal solid waste (waste scenario) {CH} |incineration| APOS, U	
	Open burning	
	Waste plastic, mixture {GLO} |open burning|APOS, U	
	Recycling	
	Waste plastic, mixture {Europé without Switzerland}|recycling| APOS, U	
	Waste plastic, mixture {CH}|recycling| APOS, U	
	Waste plastic, mixture {GLO} |recycling|APOS, U	

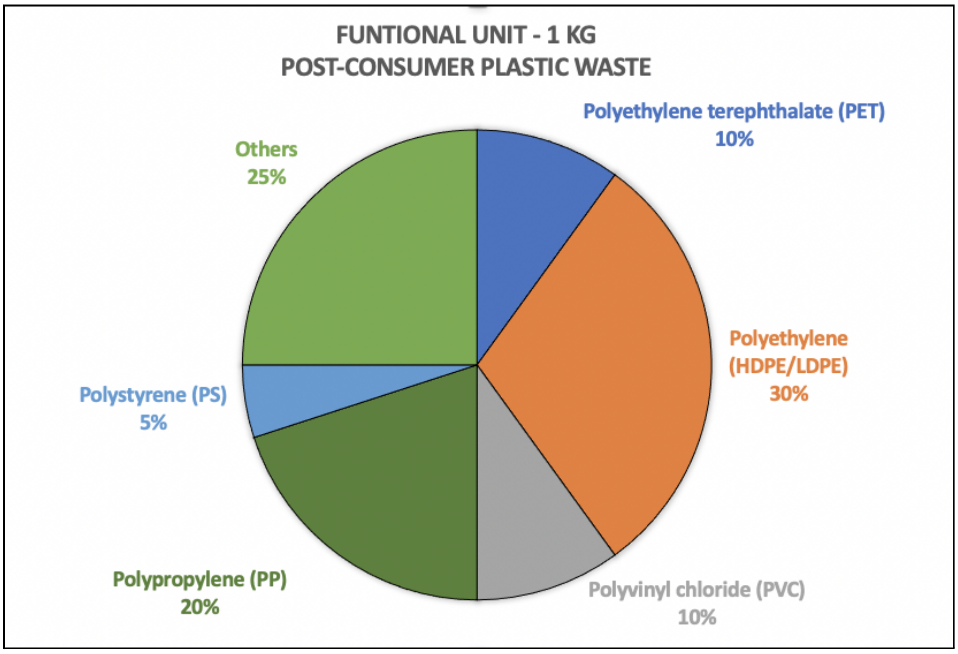

Figure B2 - Functional unit of 1 kg of post-consumer plastic waste (PlasticEurope, 2019)

Table B2 - LCI for waste management scenarios

Emissions to Air	Scenario 1: Domestic EU	Scenario 2: Export China	Scenario 3: Export Southeast Asia	Unit	Indicator
Acrolein	0,0013	0,0028	0,0081	kg 1,4-DCB	Human non-carcinogenic toxicity
Antimony	0,0058	0,0301	0,0176	kg 1,4-DCB	Human non-carcinogenic toxicity
Arsenic	0,0579	0,0663	0,0597	kg 1,4-DCB	Human non-carcinogenic toxicity
Cadmium	0,0076	0,0082	0,0082	kg 1,4-DCB	Human non-carcinogenic toxicity
Carbon disulfide	0,0062	0,0068	0,0060	kg 1,4-DCB	Human non-carcinogenic toxicity
Lead	0,0381	0,0534	0,0500	kg 1,4-DCB	Human non-carcinogenic toxicity
Zinc	0,0398	0,0555	0,0233	kg 1,4-DCB	Human non-carcinogenic toxicity
Antimony	0,0003	0,0015	0,0009	kg 1,4-DCB	Marine ecotoxicity
Copper	0,0016	0,0033	0,0021	kg 1,4-DCB	Marine ecotoxicity
Vanadium	0,0005	0,0004	0,0004	kg 1,4-DCB	Marine ecotoxicity
Zinc	0,0006	0,0008	0,0003	kg 1,4-DCB	Marine ecotoxicity
Ammonia	0,0000	0,0000	0,0000	kg PM2.5 eq	Fine particulate matter formation
Nitrogen oxides	0,0013	0,0018	0,0010	kg PM2.5 eq	Fine particulate matter formation
Particulates, < 2.5 um	0,0013	0,0015	0,0017	kg PM2.5 eq	Fine particulate matter formation
Sulfur dioxide	0,0014	0,0022	0,0017	kg PM2.5 eq	Fine particulate matter formation
Carbon dioxide, fossil	3,1340	3,1592	1,8740	kg CO2 eq	Global warming
Dinitrogen monoxide	0,0316	0,0325	0,0151	kg CO2 eq	Global warming
Methane, fossil	0,3370	0,3561	0,3755	kg CO2 eq	Global warming

Emissions to Water					
Antimony	0,0109	0,0088	0,0053	kg 1,4-DCB	Human non-carcinogenic toxicity
Arsenic	0,3746	0,4018	0,3805	kg 1,4-DCB	Human non-carcinogenic toxicity
Barium	0,0248	0,0306	0,0289	kg 1,4-DCB	Human non-carcinogenic toxicity
Cadmium	0,0176	0,0249	0,0470	kg 1,4-DCB	Human non-carcinogenic toxicity
Lead	0,0599	0,0773	0,0944	kg 1,4-DCB	Human non-carcinogenic toxicity
Mercury	0,0162	0,0209	0,0297	kg 1,4-DCB	Human non-carcinogenic toxicity
Thallium	0,0134	0,0146	0,0134	kg 1,4-DCB	Human non-carcinogenic toxicity
Vanadium	0,0296	0,0163	0,0386	kg 1,4-DCB	Human non-carcinogenic toxicity
Zinc	1,6461	2,6250	2,7969	kg 1,4-DCB	Human non-carcinogenic toxicity
Antimony	0,0032	0,0026	0,0016	kg 1,4-DCB	Marine ecotoxicity
Arsenic	0,0003	0,0003	0,0003	kg 1,4-DCB	Marine ecotoxicity
Barium	0,0005	0,0007	0,0006	kg 1,4-DCB	Marine ecotoxicity
Beryllium	0,0001	0,0058	0,0001	kg 1,4-DCB	Marine ecotoxicity
Cadmium	0,0003	0,0004	0,0007	kg 1,4-DCB	Marine ecotoxicity
Chromium VI	0,0010	0,0011	0,0009	kg 1,4-DCB	Marine ecotoxicity
Copper	0,0677	0,0922	0,0732	kg 1,4-DCB	Marine ecotoxicity
Nickel	0,0021	0,0028	0,0027	kg 1,4-DCB	Marine ecotoxicity
Silver	0,0014	0,0016	0,0013	kg 1,4-DCB	Marine ecotoxicity
Silver	0,0014	0,0016	0,0013	kg 1,4-DCB	Marine ecotoxicity
Vanadium	0,0123	0,0068	0,0161	kg 1,4-DCB	Marine ecotoxicity
Zinc	0,0604	0,0961	0,1018	kg 1,4-DCB	Marine ecotoxicity
Ammonium, ion	0,0001	0,0001	0,0003	kg N eq	Marine eutrophication
Nitrate	0,0000	0,0000	0,0000	kg N eq	Marine eutrophication
Nitrite	0,0000	0,0000	0,0000	kg N eq	Marine eutrophication
Nitrogen, organic bounc	0,0002	0,0003	0,0006	kg N eq	Marine eutrophication
Phosphate	0,0002	0,0003	0,0004	kg P eq	Freshwater eutrophication
Phosphorus	0,0000	0,0000	0,0000	kg P eq	Freshwater eutrophication

Emissions to soil					
Zinc	0,0055	0,0072	0,0163	kg 1,4-DCB	Human non-carcinogenic toxicity
Barium	0,0026	0,0034	0,0082	kg 1,4-DCB	Human non-carcinogenic toxicity

Appendix C

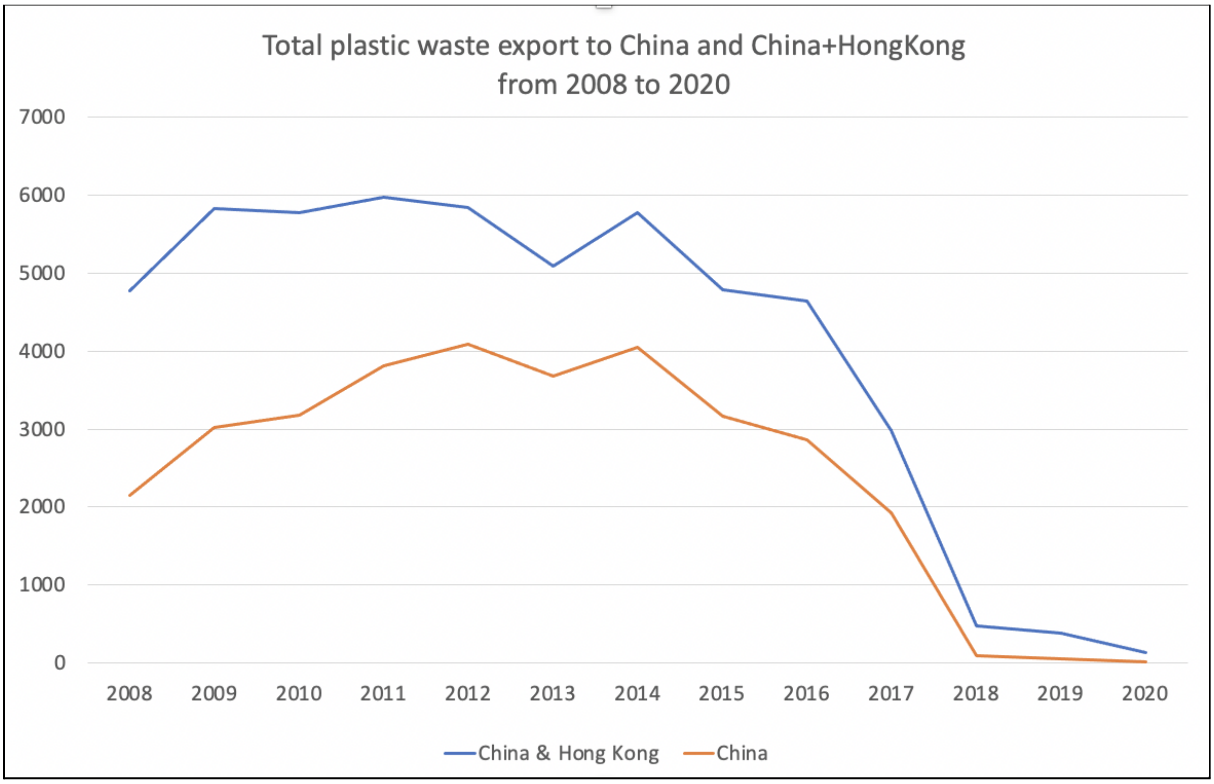

Figure C1 - Total plastic waste to China and China+HK from our exporting countries (UNComtrade, 2021)

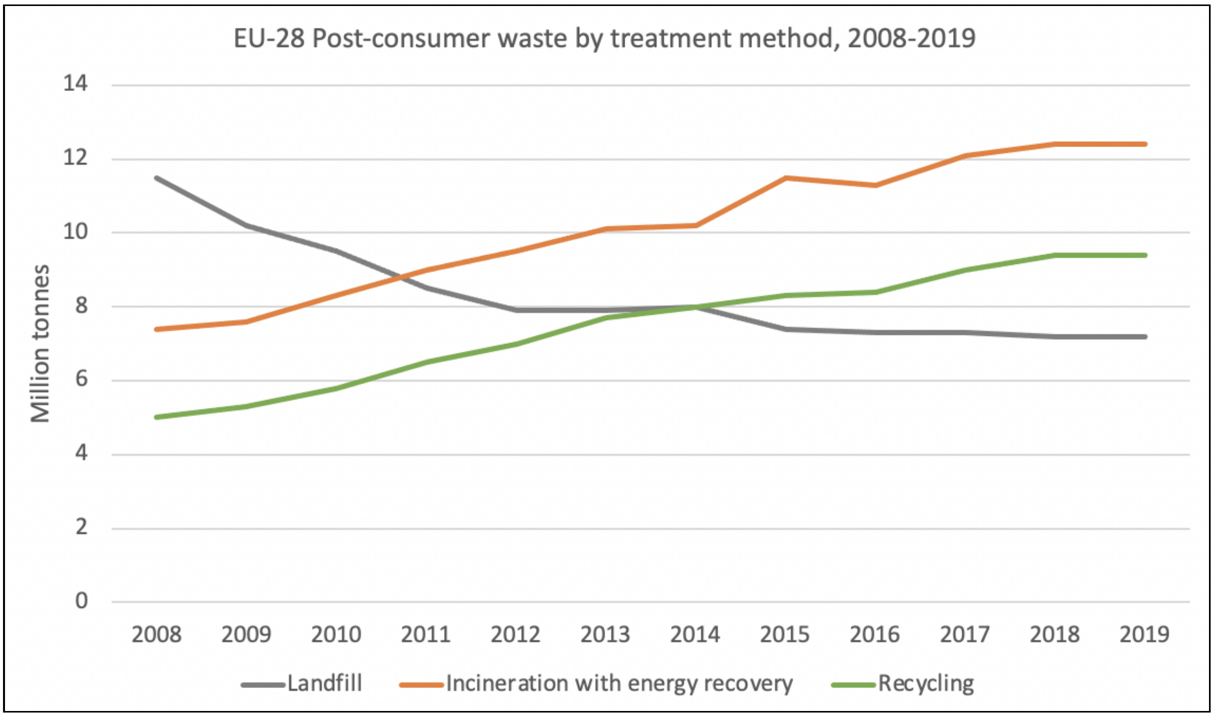

Figure C2 - Post-consumer waste by treatment method, 2008-2019 for the EU-28 (UNComtrade, 2021)

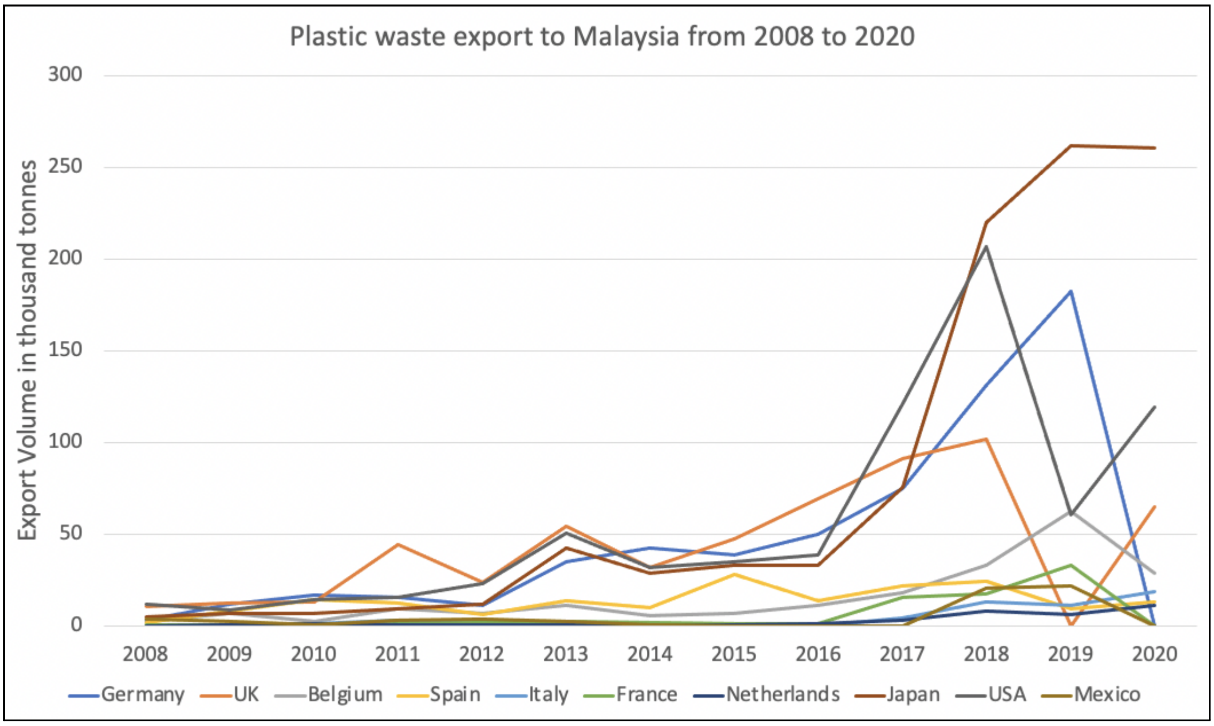

Figure C3 - Plastic waste exports to Malaysia between 2008 and 2020 at country level (UNComtrade, 2021)

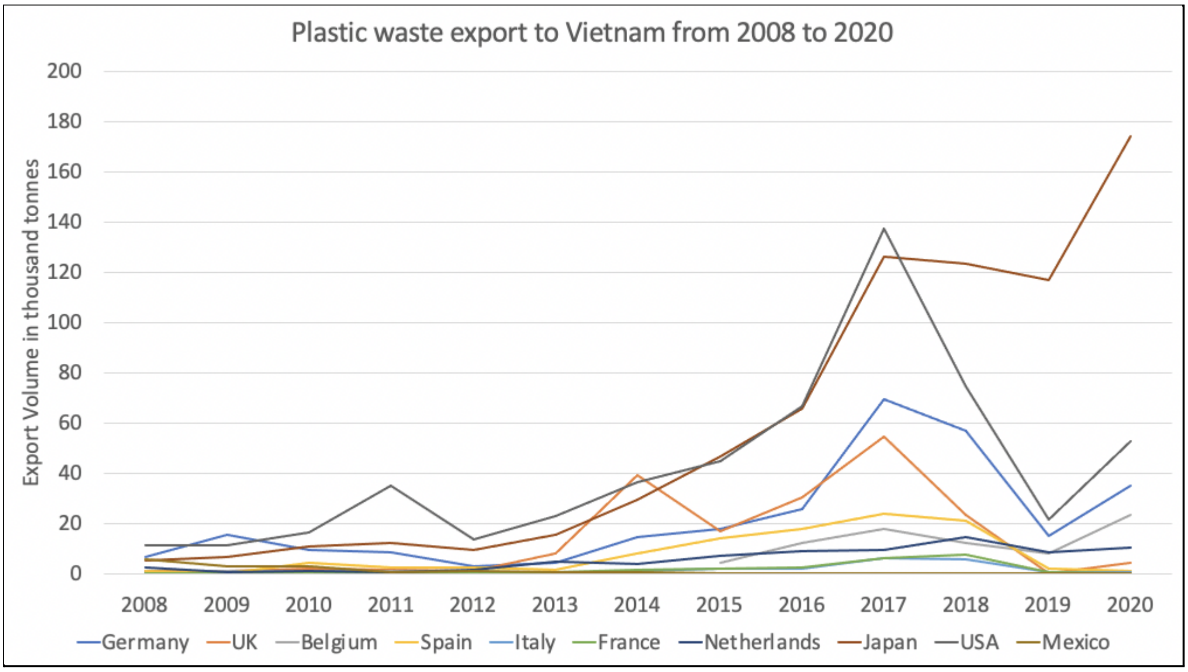

Figure C4 - Plastic waste exports to Vietnam between 2008 and 2020 at country level (UNComtrade, 2021)

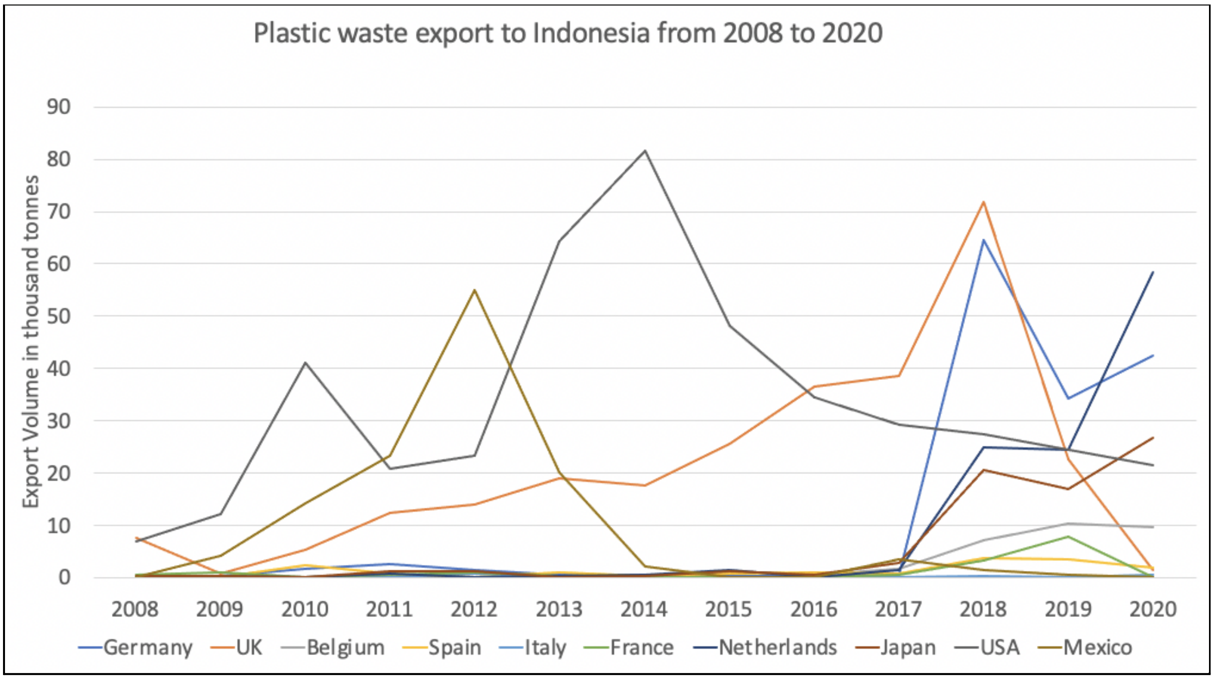

Figure C5 - Plastic waste exports to Indonesia between 2008 and 2020 at country level (UNComtrade, 2021)

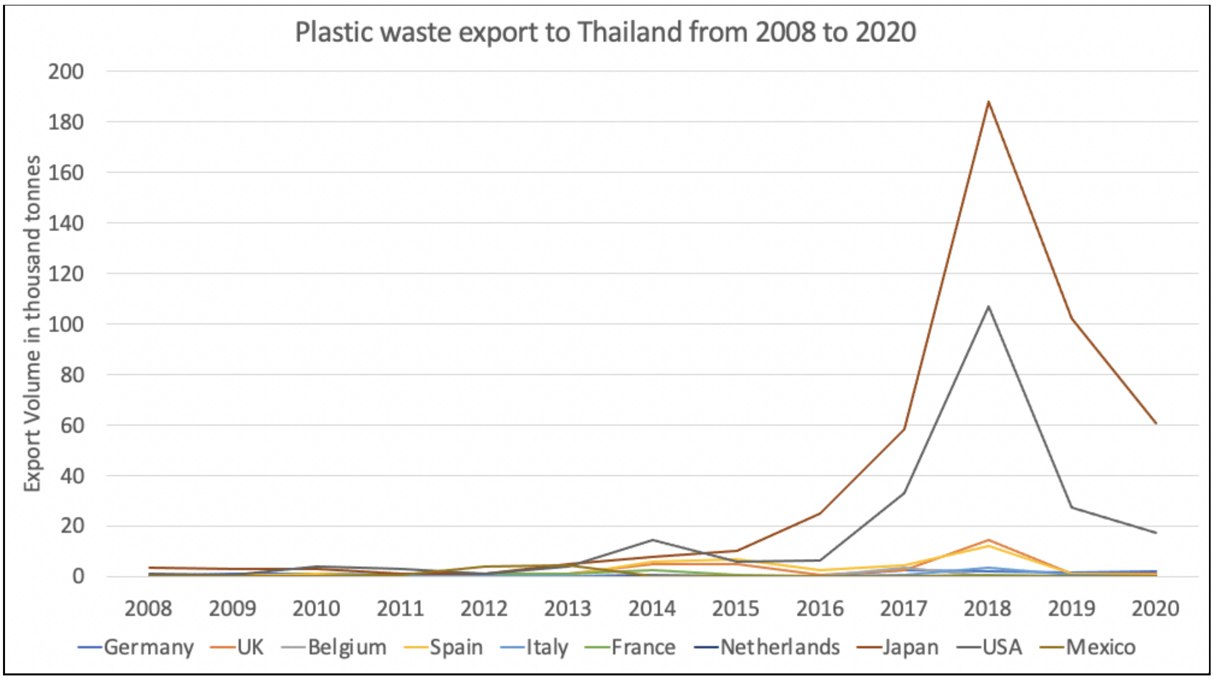

Figure C6 - Plastic waste exports to Thailand between 2008 and 2020 at country level (UNComtrade, 2021)

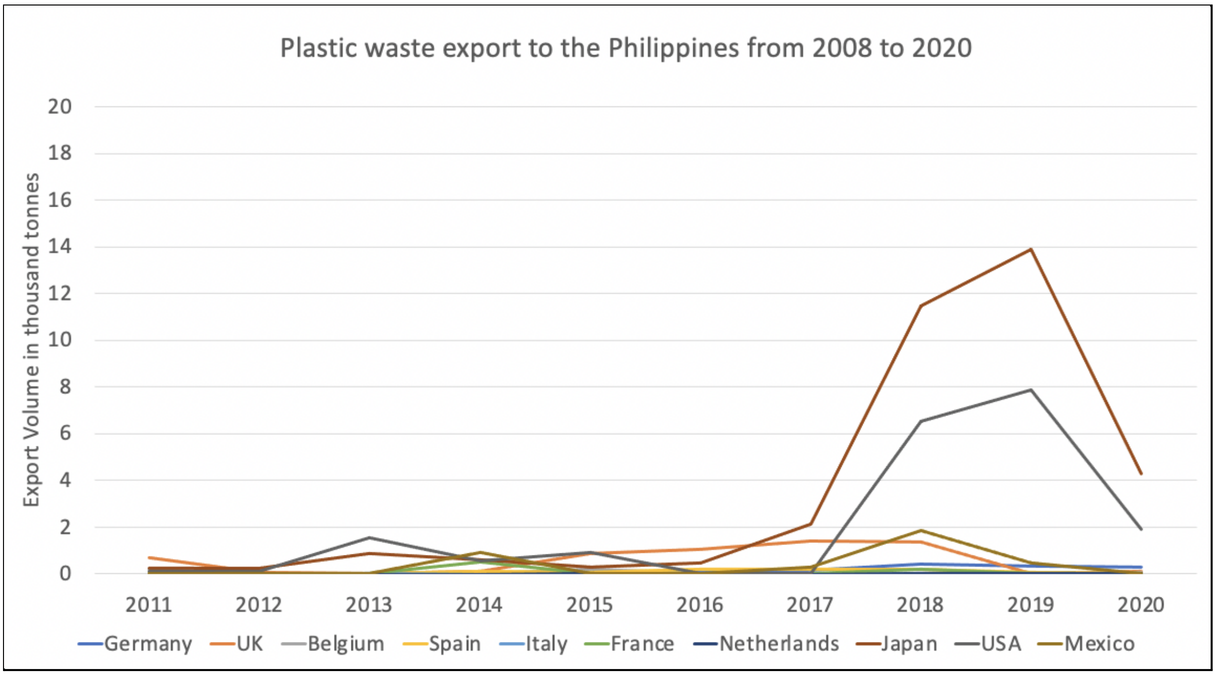

Figure C7 - Plastic waste exports to the Philippines between 2008 and 2020 at country level (UNComtrade, 2021)

Note: No data available for Germany, UK, Belgium, Spain, Italy, Netherlands and Mexico for 2008, 2009 and 2010.