On the Road to a Sustainable and Healthy Circular Economy in the Netherlands:

A layered dynamic material flow analysis and a life cycle impact assessment of flame retardant additives in Dutch passenger vehicle plastics

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In partial fulfilment of the requirements for the degree of:

Master of Science in Industrial Ecology

at Leiden University and Delft University of Technology

Leiden, August 2022

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EXECUTIVE SUMMARY

Flame retardants are added to plastics in passenger vehicles to save lives. However, their releases can, in some cases, be accompanied by undesired impacts to human health and the environment. In the context of a transition to a Circular Economy, it is therefore important to keep track of the flow of flame retardants in passenger vehicle plastics. This study focuses on the flame retardant Decabromodiphenyl Ether (decaBDE), whose use in plastics has become banned in recent years.

Through a static layered material flow analysis, the flows of plastics and decaBDE are quantified for passenger vehicles in The Netherlands in 2019. Afterwards, six scenarios are created outlining different possible approaches to plastics and flame retardants: Reference, Recycling, Incineration, Less Cars, No DecaBDE Ban, and No Substitution. The first four of these scenarios acknowledge the progressive phasing out of decaBDE and include the use of an additional flame retardant, triphenyl phosphate (TPP). At the hands of a dynamic layered material flow analysis, the stocks and flows of plastics, decaBDE, and TPP, are calculated from 1980 until 2050. The emissions calculated in this step, as well as the flow of flame retardant directed to incineration, are then used to evaluate the environmental performance of each scenario by means of a life cycle impact assessment. The results from the life cycle impact assessment show that, from a midpoint perspective, no scenario is a clear winner. Moreover, the lack of characterization factors for some categories makes it difficult to assess the reliability of the results. From an endpoint perspective, the best performing scenarios are the TPP-free scenarios No DecaBDE Ban and No Substitution. This is because TPP is shown to have a significant influence in the global warming impacts. From the TPP-using scenarios, the best performing one is the Recycling scenario, which shows the advantages of promoting recycling strategies in the sector. Overall, TPP-using scenarios perform worse environmentally than decaBDE-using scenarios.

According to the findings in this study, some of the key elements that could smooth the transition to a sustainable and healthy circular economy are policy measures that incentivize plastic recycling in the automotive industry and the use of less cars per capita.

Further research is needed on the environmental performance of less harmful flame retardant alternatives to decaBDE. The methodology used in this study could serve as a framework in future research for other type of product applications as it is useful to evaluate different policies involving chemicals or additives embedded in materials over time. With this methodology, a link is made between the product, material, and chemical layers, able to bridge potential gaps in the often data scarce chemical layer.

Acknowledgements

This thesis marks the culmination of an inspiring time as a master student of Industrial Ecology at Leiden University and TU Delft. I am grateful to all the dedicated professors and fellow students that I had the pleasure to learn from throughout these last two thought-provoking years.

Above all, I am extremely grateful to my academic supervisors Ester van der Voet, Willie Peijnenburg, and Janneke van Oorschot, and my RIVM supervisor Erik Dekker. Writing this thesis would not have been possible without the guidance and encouragement that you provided me during the past six months.

I am also indebted to the experts at RIVM that generously offered their advice and shared their knowledge with me: Anne van Bruggen, Lise de Boer, Michiel Zijp, Johannes Lijzen, Susanne Waaijers, and Martien Janssen.

Lastly, to Julia, Trinidad, Guillermo, Piky, and Teun. Thank you for all your love and support.

Milagros

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Abbreviations

ABS	Acrylonitrile Butadiene Styrene
ASR	Automotive Shredder Residue
CF	Characterization Factor
DecaBDE	Decabromodiphenyl Ether
(D)MFA	(Dynamic) Material Flow Analysis
EF	Emission Factor
ELV	End of Life Vehicle
FR	Flame Retardant
LCIA	Life Cycle Impact Assessment
OctaBDE	Octabromodiphenyl Ether
PBDE	Polybrominated Diphenyl Ether
PentaBDE	Pentabromodiphenyl Ether
TPP	Triphenyl Phosphate

1. Introduction

This chapter introduces the thesis by providing a background on the thesis topic and establishing the main goals of the study. In section 1.1, the current context surrounding the thesis topic is covered. This is followed by a description of the relation between passenger vehicles, plastics, and the circular economy in section 1.2 . Then, in section 1.3, the problem statement of the thesis is defined. In section 1.4, the main research question of the study and the research sub-questions are formulated. Lastly, in section 1.5, the selection of the flame retardant to be studied is made.

1.1. Context

Natural resource extraction has tripled since 1970 and is expected to increase by 70% by 2050 (UNEP, 2021). This increasing global extraction of natural resources is putting a serious and concerning strain on the environment, overshooting various planetary boundaries and risking pushing the earth system past its "safe operating space" (Steffen et al., 2015). While maintaining resource use below the critical threshold is indispensable for human well-being, human well-being also relies on everyone being able to use the resources needed for a dignified life (Raworth, 2013). To address this complex issue successfully, a shift must be made from society's current linear take-make-use-dispose economy to a more circular one (Velenturf et al., 2019).

In this light, the government of The Netherlands aims to have achieved a 50 percent reduction of primary raw materials by 2030 and to have realised a fully Circular Economy (CE) by 2050 (Ministerie van Infrastructuur en Milieu & Ministerie van Economische Zaken, 2016). While many varying definitions of CE exist, CE is most often defined as "an industrial system that is restorative or regenerative by intention and design. It replaces the 'end-of-life' concept with restoration, shifts towards the use of renewable energy, eliminates the use of toxic chemicals, which impair reuse, and aims for the elimination of waste through the superior design of materials, products, systems, and, within this, business models." (Ellen MacArthur Foundation, 2012, pp 7). However, renewable energy and the elimination of toxic chemicals are not mentioned in most CE definitions, and are often overlooked (Kirchherr et al., 2017).

Even though the transition to a CE is seen as a promising path to realize sustainable development goals (European Commission, 2019; Ghisellini et al., 2016), CE measures are not always inherently sustainable or healthy. After their disposal, the processing and reincorporation of materials in the system require additional energy, while resulting in lower quality material flows (Castro et al., 2007). Besides being energy-intensive, the recycling process can also be costly and cause the alteration of the material's physical properties, leading to a decrease in material quality (Christensen et al., 2019). Moreover, while a transition to a CE may generate human health benefits thanks to the reduction of

negative environmental impacts, there are potential health risks in using CE measures in processes that involve hazardous materials (World Health Organization, 2018).

Given the opportunities and challenges related to a transition to a CE, the National Institute for Public Health and the Environment (RIVM) of the Netherlands aims to support the Dutch government in creating good quality circular solutions. Through its project QONNECT, RIVM intends to provide decision-makers "insight on possible societal realistic trade-offs" of the CE transition on the micro and macro level (Zijp, 2020). Plastics are a compelling object of study due to various reasons, including the stream's ever-growing use, material composition, material collection, and material recyclability complexities, as well as its associated environmental and health impacts (Geyer et al., 2017).

1.2. Passenger vehicles, plastics, flame retardants and Circular Economy

Since the boom of the plastic industry in the 20th century, plastic has been introduced as a key material in a wide range of applications across all sectors. The automotive sector is no exception. In 2017, it was the third biggest sector making use of plastics in Europe (Plastics Europe, 2018). Figure 1.1 shows the demand of plastics by segments and polymer types.

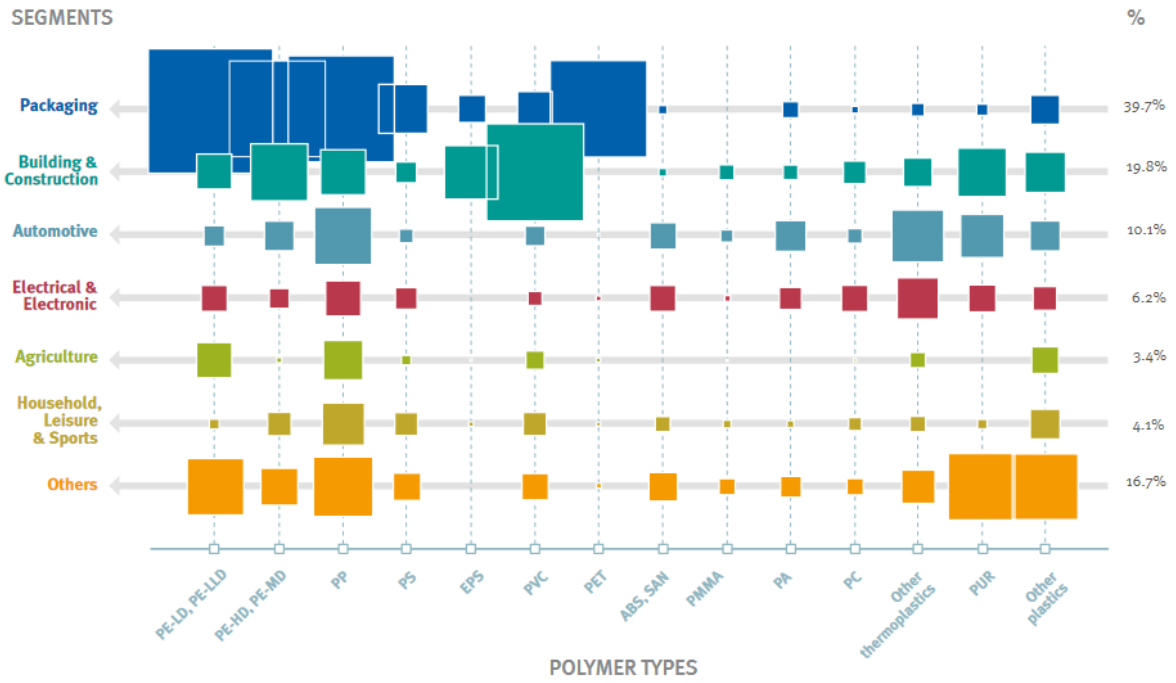

Figure 1.1. European plastic demand by segment and polymer types in 2017 (Plastics Europe, 2018).

Over time, significant part of the metal used to manufacture passenger car parts was replaced by plastics. Pradeep et al. (2017) documents an increase from 9 kg of plastics per car in 1960 to 162 kg of plastics per car in 2010, while an analysis made by A.T. Kearney reports an increase in plastics content in cars form 66kg in 1970 to 224kg in 2010 (Emilsson et al., 2019; Rouilloux & Znojek, 2012). While the average content of plastics per cars throughout the years varies per source, one thing is certain: the incorporation of the material in vehicles has grown steadily. Figure 1.2 shows a projection made by AT Kearney of the shares of materials used in passenger vehicles from 1970 until

2020. While the share of metals in the car goes down in time, the share of plastics in the car increases considerably. This is attributed to plastics' light weight, low costs, versatility, durability, and corrosion resistance (Mitrano & Wagner, 2022).

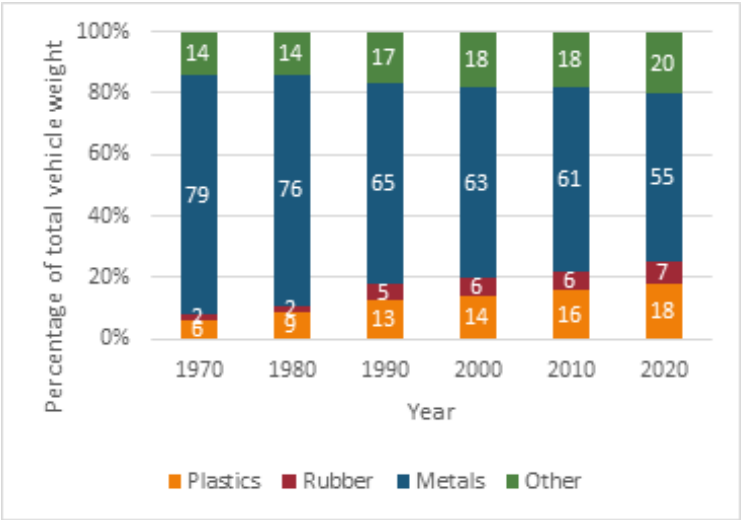

Figure 1.2. Material shares in passenger vehicles throughout the years. Adapted from AT Kearney (2012).

Due to the increased use of plastics in passenger vehicles, it has become increasingly relevant to discuss the transition to a CE of plastics in the automotive industry. The great variety of polymers used in passenger vehicles makes the recycling of vehicles challenging, which is further exacerbated by the lack of economic incentives to implement CE strategies such as recycling, extending the lifespan of a product, or investing in suitable substitutes (Bocken et al., 2016; van Bruggen et al., 2022). Moreover, plastics in passenger vehicles are partly composed by flame retardant additives that can pose risks to human health and the environment, which adds an extra layer of complexity to the End of Life (EOL) treatment of passenger vehicles. This makes it of great importance to carefully consider the trade-offs regarding environmental and health impacts before implementing circular measures that aim to improve resource efficiency (Leslie et al., 2016).

Despite the ubiquity of flame retardant use, there is limited information on their fate, transport, human exposure and transport, as well as little knowledge on their uses and production volumes (Wong et al., 2018). Therefore, it makes sense to study the flame retardants together with their product application: plastics. While plastics and flame retardants have been studied separately in the past, added insights might result from studying them together, considering the interlinkages between the materials, to better understand the repercussion of different circular strategies.

1.3. Problem statement

Many considerations must be made in the pursuit of designing and achieving a circular economy. Circular solutions come at the hand of recycling, reuse, and substitution strategies, among others. However, these should be carefully analyzed when the use of hazardous substances comes into play. In this context, it is of interest to evaluate what the best circular solutions are, considering health and environmental impacts, to achieve a transition to a circular economy in the automotive sector of The

Netherlands. In this study, the focus lies on flame retardants of concern used in passenger vehicle plastics.

1.4. Research questions

The main objective of this thesis is to gain insights into the consequences and trade-offs of implementing different circular strategies when it comes to plastics and flame retardants in passenger vehicles in The Netherlands. This leads to the main research question:

What might be the health and climate effects of flame retardants in passenger vehicle plastics according to different circular approaches in The Netherlands?

Five sub-questions are formulated to answer the main research question. The first sub-question aims to gain an understanding of the current situation regarding the use and treatment of plastics in passenger vehicles. This results in a quantified map of the main plastic flows of passenger vehicles in The Netherlands.

1. *What do the major plastic material flows of passenger vehicles currently look like in The Netherlands?*

These results help to answer the second sub-question, which intends insights into flame retardant flows in passenger vehicle plastics. Also, it aims to account for the behaviour of the additive throughout the different life-phase processes (i.e., potential emissions).

2. *What do major flame retardant flows in plastics of passenger vehicles currently look like, and how do they behave throughout the different processes?*

The insights gained in first and second sub-questions contribute to the next steps. The goal of the third sub-question is to analyze the evolution of plastic flows according to different hypothetical circular approaches.

3. *How would the major plastic material flows of passenger vehicles evolve under different circular economy scenarios in The Netherlands?*

The outcomes of the third sub-question are partially used as inputs to answer the next sub-question, which aims at quantifying the flame retardant flows according to the same previously mentioned circular approaches.

4. *How would the flame retardant flows in plastics of passenger vehicles evolve under different circular economy scenarios in The Netherlands?*

Lastly, the fifth sub-question is concerned with quantifying the health and climate impacts of the flame retardant flow values obtained as a result in the fourth sub-question.

5. *How can the flame retardant flow values be translated into health and climate impacts?*

1.5. Flame retardant selection: DecaBDE

There are a vast variety of different flame retardants used in plastic in passenger vehicles, which vary across manufacturers and year of production. In this research, a representative and relevant flame

retardant is chosen as the subject of study: decaBDE. In the following paragraphs, the reasoning behind the selection is elaborated.

Specifically, the focus of this study is on a flame retardant POP, which cause adverse effects impacts in human health and the environment. According to the UNEP, POPs can be defined as *"… a particular combination of physical and chemical properties such that, once released into the environment, they (i) remain intact for exceptionally long periods of time, (ii) become widely distributed throughout the environment as a result of natural processes involving soil, water, and, most notably, air, (iii) accumulate in the fatty tissue of living organisms including humans and are found in higher concentrations at higher levels in the food chain, and (iv) are toxic to both humans and wildlife*" (UNEP, n.d.). Since its creation in 2004, the Stockholm Convention on POPs, an international environmental treaty overseen by the UNEP, has aimed to eliminate and restrict the use of POPs.

Brominated Flame Retardants (BFRs) are bromine-based organic chemicals that have been widely incorporated in consumer products to prevent the ignition and slow down the spread of flame in the material (British Plastics Federation, 2022). BFRs are additives that are not chemically bound to plastic. Therefore, they are more easily released to the environment than their counterparts, reactive flame retardants, which are chemically bound to the material.

In 2004, several BFRs were listed in the Stockholm Convention. Among them were two of the most important families of Polybrominated Diphenyl Ethers (PBDEs). Namely, Commercial Pentabromodiphenyl Ether (c-pentaBDE) and Commercial Octabromodiphenyl Ether (c-octaBDE). Its use in new products was banned – while its use in recycled products was still permitted. Through the EU POPs Regulation, the Stockholm Convention is implemented in all EU member states. Since 2019, the POPs Regulation has included a stronger focus on waste management, setting thresholds for POPs within waste. For instance, waste articles exceeding 1,000mg/kg concentrations of sum of PBDEs must be disposed through incineration, and articles newly introduced to the market should not exceed 500mg/kg of PBDEs (Sharkey et al., 2020). The use of c-pentaBDE and c-octaBDE was mainly replaced by Decabromodiphenyl Ether (decaBDE), which was the latest BFR to be added to the Stockholm Convention, in 2019.

The implementation of the POPs Regulation has had a significant impact in the polymer production of passenger vehicles. Fire safety requirements apply to passenger vehicle plastics, which are met by the application of flame retardants (European Flame Retardants Association, 2007). Due to their high effectiveness and flexibility, c-pentaBDE and c-octaBDE were widely used in passenger vehicle plastics from the 1980s until their ban in the 2000s, when they were mainly replaced by decaBDE (Sharkey et al., 2020).

Despite decaBDE's late ban in 2019, the European Automobile Manufacturers Association (ACEA) states that its phasing out began a few years earlier. They conclude this by looking at the number of times decaBDE is reported in component manufacturing data sheets: by 2015, very few new passenger vehicle applications contained decaBDE (ACEA, 2015; Melhart et al., 2018). This is likely due to the sector's suspicions that decaBDE was about to be listed in the Stockholm Convention, due to its close relation to pentaBDE and octaBDE. According to a Finnish study, it is estimated that pentaBDE and octaBDE will be found in End of Life Vehicles (ELVs) until approximately 2024 – while decaBDE is expected to be found in ELVs until the late 2030s (Ministry of the Environment Finland, 2016).

Due to its POP status, and as the latest BFR to be included in the Stockholm Convention, decaBDE is selected as the flame retardant of choice to study in thesis. It is likely that it will still be present in the Dutch passenger vehicle fleet for a few more years, making relevant the study of its emissions and its impact on the environment.

2. Methods and data collection

This chapter presents the methods applied in the study and describes the collection of the required data to answer each research sub-question. First, the layered static material flow analysis (MFA) method is explained in section 2.1. Then, the layered dynamic material flow analysis is covered in section 2.2. Lastly, the life cycle impact assessment method is presented in section 2.3.

2.1. Layered Static Material Flow Analysis

In this section, the concept of a static MFA is described (subsection 2.1.1) and the layered nature of this study's static MFA is explained (subsection 2.1.2). Next, the goal and system definition step, and inventory and modelling step of the static material flow analysis, for both plastics and decaBDE, are presented (subsections 2.1.3 and 2.1.4).

2.1.1. Static Material Flow Analysis

An MFA is a method used to quantify the flows, stocks and inputs and losses of a desired resource within a defined spatial and temporal system, where at every node the law of mass-balance applies (Graedel, 2019). Often used for decision-making, MFAs are useful in assessing the efficiency of industrial practices and overseeing the management of resources and environmental impacts (Brunner & Rechberger, 2016). While they can be employed to study bulk materials, they can also be used for a specific substance of choice (Graedel & Lifset, 2016). This makes the method ideal to quantify both plastics (a conjunction of different polymers) and flame retardants (individual substances). MFAs typically include a graphical representation of the results, as well as a numerical one, and a discussion of the reliability of the results (Graedel, 2019). The MFA can visualize and identify potential hotspots: where a lot of plastic or flame retardant are lost, technical problems that may hinder further treatment of the plastics or flame retardant, etc.

Different types of quantification can be carried out: accounting, static modelling, and dynamic modelling. To answer the first and second research question of this study, a static approach is taken. The static model is able to provide insights into the system at a specific time, which allows for an assessment of the system at the current state (Allesch & Brunner, 2017). The initial flow calculations of this study were carried out in Microsoft Excel, and the MFA static modelling was performed with the open source free software STAN v2.6.8. (STAN, 2012).

Conducting an MFA consists of the three main steps:

- Goal and system definition
- Inventory and modelling
- Interpretation of results

The first step involves drafting the boundaries and formulating the goal of the system (subsection 2.1.3). The second step is concerned with the quantification of the flows and stocks of the system (the procedure is explained in subsection 2.1.4). Finally, the third step consists of discussing the results (section 3.1).

2.1.2. Layered: How?

The MFA carried out in this study has three layers: the product layer (passenger vehicles), the material layer (plastics), and the additive layer (DecaBDE). The product, material, and additive level are simultaneously considered and are connected. The material level depends on the product level, and the additive level depends on the material level. This product-material-additive level connection is useful to determine material and additive flows where data is scarce.

The result of the connected MFA layers can be understood as a layered static MFA, where the different products, materials, and additives are studied within the same spatial and temporal boundaries.

2.1.3. Goal and system definition

The goal of the layered static MFA is to find the answer to sub-question 1 and 2:

1. *What do the major plastic material flows of passenger vehicles currently look like in The Netherlands?*
2. *What do major flame retardant flows in plastics of passenger vehicles currently look like, and how do they behave throughout the different processes?*

The analysis focuses on plastics used in passenger vehicles and decaBDE used in plastics of passenger vehicles. While the focus is on one specific flame retardant, plastics are studied as a bulked material, considering all polymers used in a passenger vehicle.

The system is bounded by the Dutch national borders, as the goal is to obtain a snapshot of the passenger vehicles plastics' and decaBDE within The Netherlands. In addition to the vehicle use phase, the main processes regarding the treatment of ELV that occur in The Netherlands are included in the system: car dismantling, shredding, post-shredder technology, incineration, etc. Moreover, while the mechanical recycling process takes place in Belgium, it is also included, as it is of special interest to understand the EOL options of plastics and decaBDE. It is assumed that all plastics (and flame retardants) that are recycled can be used again in newly produced vehicles, replacing the production of new plastics and flame retardants. On the other hand, while most vehicle production of passenger vehicles used within The Netherlands also takes place abroad (mainly Germany, France, and Japan), this life phase has been chosen to keep outside of the system boundaries to remain within the Dutch borders as much as possible. In the decaBDE system, emissions of the flame retardant to the atmosphere are considered. In the case of plastics however, they are not.

The chosen year for the study is 2019, which is the latest year for which there is data available for the number for all passenger vehicle exports, imports, and dismantling (to be further discussed in section 2.1.4). The systems resulting from these conditions are shown in Figure 2.1 and Figure 2.2. A detailed explanation of the plastic flows of passenger vehicles in the Netherlands over the years can be found in Appendix A.

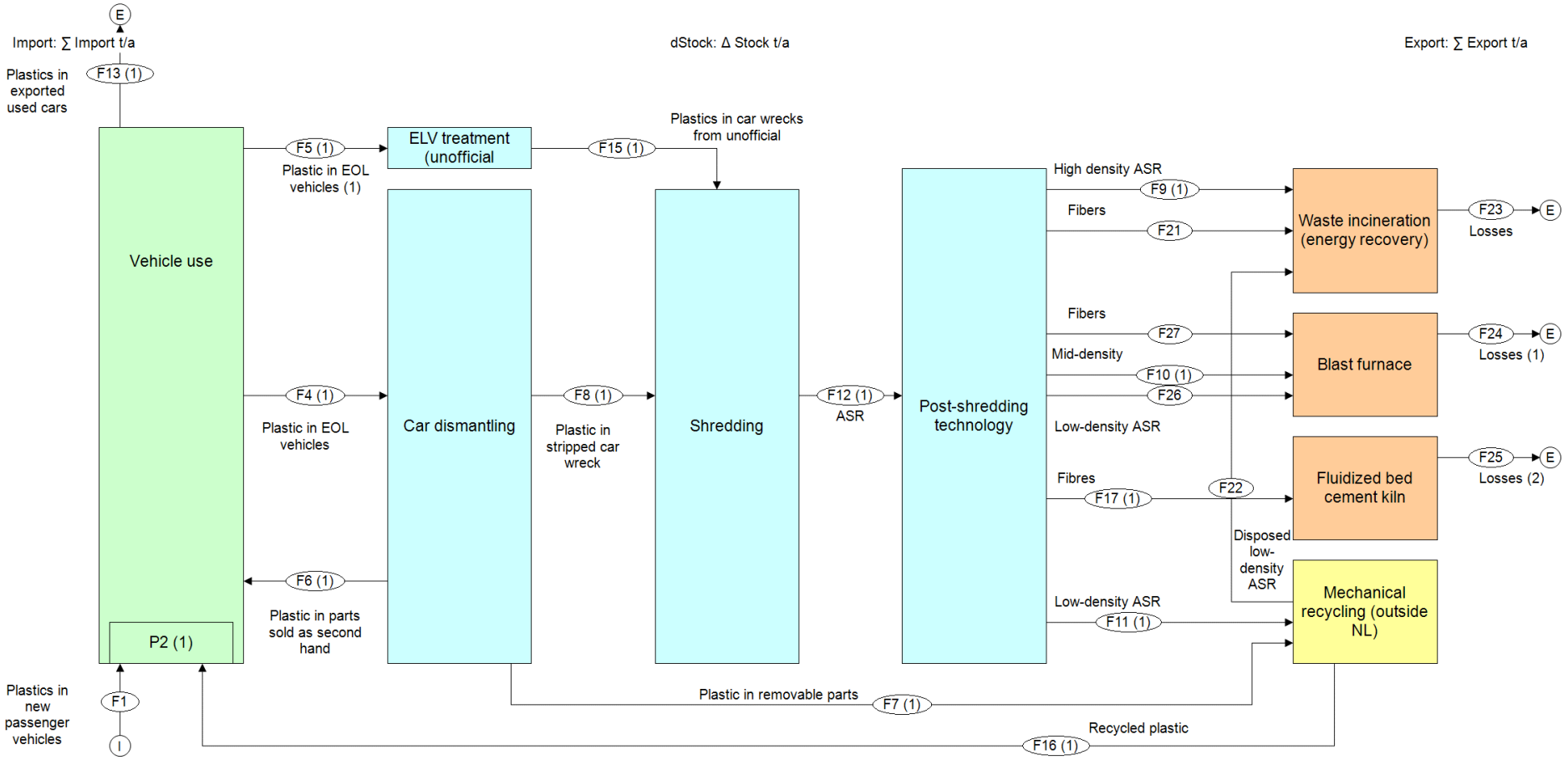

Figure 2.1. Graphical representation of the system of the material flow analysis for plastics in passenger vehicles in The Netherlands in 2019.

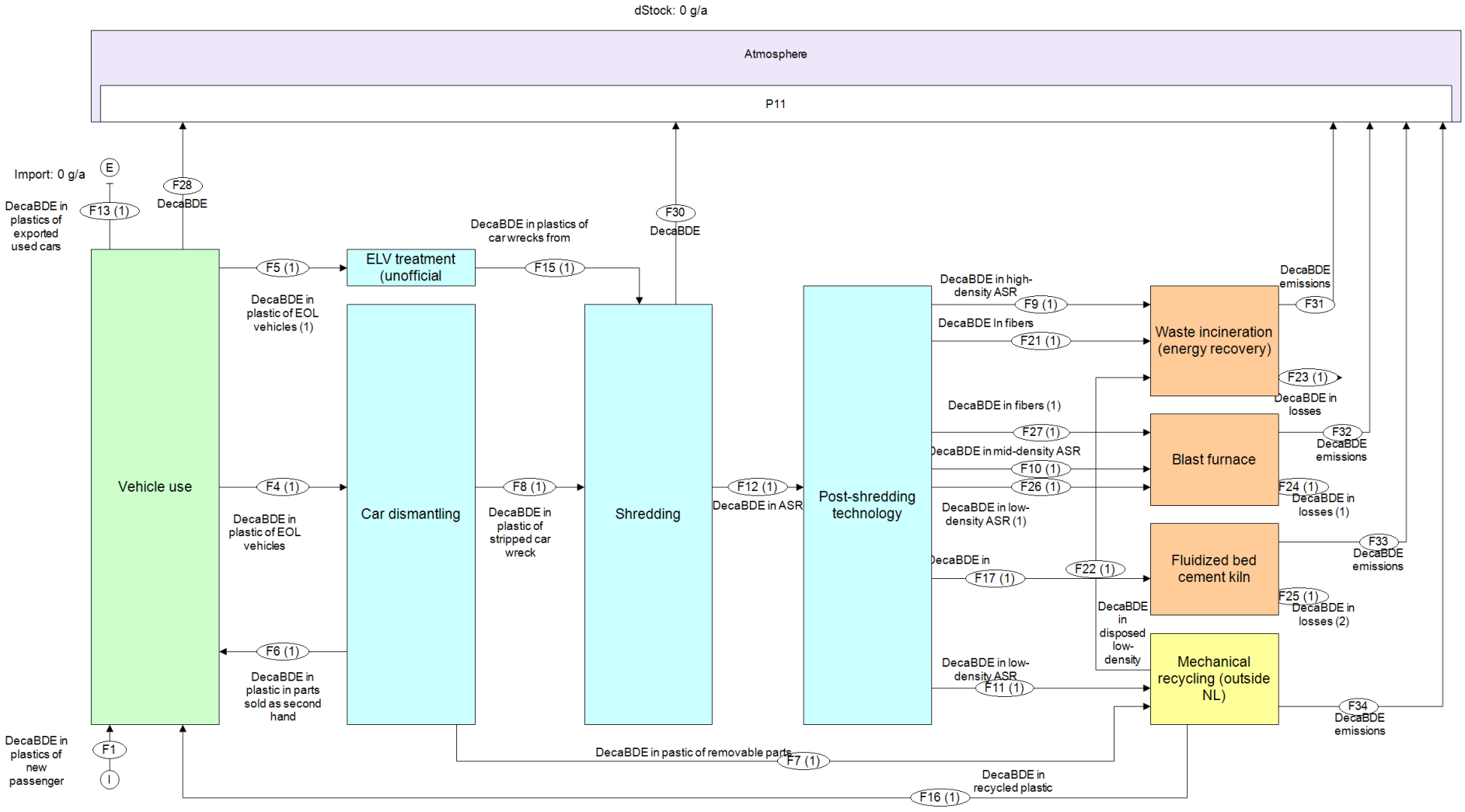

Figure 2.2. Graphical representation of the system of the static material flow analysis for decaBDE in plastics of passenger vehicles in The Netherlands in 2019.

2.1.4. Inventory and modelling

This step involves quantifying and modelling the material flows and stocks of plastics and decaBDE. To this aim, numerous flow data and process data are collected. Where there is no data available, the data gaps are estimated according to specific assumptions. For each system, the fixed flows, dependent flows, and flows to be determined by mass balance and transfer coefficients, are chosen. Based on the assumptions made and the calculations done on the selected data, a model is built in STAN.

2.1.4.1. Plastics

A combination of fixed variable inputs and transfer coefficient inputs are used to build the plastics static MFA model. These inputs can be based on raw data, literature, interviews with knowledgeable sources, and "educated" assumptions. The preferred sources are national statistical database, followed by insights by experts in the field and literature on the topic. The fixed flows in the case of the plastics MFA concern the flows present at the initial stages of the system. This is because the plastic going through these initial stages can be quantified using data regarding the number of vehicles, which is typically available in national statistical databases.

In Table 2.1, the data requirements and the data sources used to estimate the values of the fixed flows in the plastics static MFA are presented: the number of vehicles at different parts of the system, and the average content of plastics in these vehicles. Due to the static nature of the static MFA, it is assumed that the content of plastic in passenger vehicles is the same for passenger vehicles at different parts of the system.

Table 2.1. Data requirements and data sources used to calculate the values of the fixed flows in the plastics static MFA.

Flow name	Variables	Data source
Plastics in new passenger vehicles	Number of newly registered vehicles in the Netherlands in 2019	(Eurostat, n.d.)
	Average plastic content in new vehicles in 2019	Personal communication with Joint Research Centre experts.
Plastics in exported vehicles	Number of exported vehicles from the Netherlands in 2019	(CBS, n.d.)
	Average plastic content in new vehicles in 2019	Personal communication with Joint Research Centre experts.
Plastics in EOL vehicles	Number of EOL vehicles in The Netherlands in 2019	(CBS, n.d.)
	Average plastic content in new vehicles in 2019	Personal communication with Joint Research Centre experts.
Plastics in EOL vehicles via unofficial channels	Number of EOL vehicles treated via unofficial channels in the Netherlands in 2019	(CBS, n.d.)
	Average plastic content in new vehicles in 2019	Personal communication with Joint Research Centre experts.

Similarly, in Table 2.2, the data requirements and data sources used to estimate the transfer coefficients of several processes in the plastics static MFA are presented. These transfer coefficients are necessary for the model to calculate the remaining plastic flows of the system. In some cases, these shares or transfer coefficients are obtained directly from knowledgeable experts of the Dutch automotive industry or literature. If that is not possible, estimations are made based on the data available.

Table 2.2. Data requirements and data sources used to estimate the transfer coefficients of several processes in the plastics static MFA.

Process	Variables	Data source
Car dismantling	Share of plastics in car parts that are removed and sent to mechanical recycling in 2019	Estimation based on communications with STIBA professionals, who have first-hand experience of the dismantling process in the Netherlands.
	Share of plastics in car parts that are removed and sold as second-hand	
Post-Shredder Technology	Share of ASR plastics that are high density plastics sent to Incineration	Communications with Auto Recycling Nederland (ARN) and RIVM. While the obtained data is for 2017, it is assumed that these shares did not significantly change between then and 2019.
	Share of ASR plastics that are mid density plastics sent to Blast furnace	
	Share of ASR plastics that are low density plastics sent to Blast furnace	
	Share of ASR plastics that are low density plastics sent to Mechanical recycling	
	Share of ASR plastics that are fibers sent to Waste incineration	
	Share of ASR plastics that are fibers sent to Blast furnace	
	Share of ASR plastics that are fibers sent to Fluidized bed cement kilns	
Mechanical recycling	Yield of mechanically recycled plastics for a mixed plastics flow	(Broeren et al., 2022)

A detailed explanation of the calculations to estimate the final flow and transfer coefficient values is given in Appendix B and a graphical representation of the quantified flows can be found in section 3.1.

2.1.4.2. DecaBDE

To determine the fixed flow values and the transfer coefficients of the static decaBDE MFA, results from the plastics MFA, data provided by Auto Recycling Nederland, and data obtained from literature are used. In Table 2.3, the data required to estimate fixed flow values of the decaBDE static MFA is

presented, as well as the data sources used. The concentrations of decaBDE in the different plastic flows are assumed to be equal to the samples of decaBDE concentrations taken at the Auto Recycling Nederland Post-Shredder Technology facility in 2013. It is assumed that the concentrations of the flame retardant in these plastic flows have not changed significantly between 2013 and 2019. This is one of the few specific data available for the concentration of the flame retardant in plastics. There is no detailed data available yet on the concentration of decaBDE per type of plastic used in passenger vehicles. That is why, in contrast with the plastics MFA, the fixed flows concern end-of-life flows, where decaBDE concentration data is on hand.

Table 2.3. Data requirements and data sources to calculate the fixed flows of the static decaBDE MFA.

Flow name	Variables	Data source
DecaBDE in high density plastics sent to Incineration	High density plastics sent to Incineration	Static plastics MFA
	Concentration of decaBDE in high density plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)
DecaBDE in mid density plastics sent to Blast furnace	Mid density plastics sent to Blast furnace	Static plastics MFA
	Concentration of decaBDE in mid density plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)
DecaBDE in low density plastics sent to Blast furnace	Low density plastics sent to Mechanical recycling	Static plastics MFA
	Concentration of decaBDE in low density plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)
DecaBDE in low density plastics sent to Mechanical recycling	Low density plastics sent to Mechanical recycling	Static plastics MFA
	Concentration of decaBDE in low density plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)
DecaBDE in fibers sent to Waste incineration	Fibers sent to Incineration	Static plastics MFA
	Concentration of decaBDE in fibers plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)
DecaBDE in fibers sent to Blast furnace	Fibers sent to Blast furnace	Static plastics MFA
	Concentration of decaBDE in fibers plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)
DecaBDE in fibers sent to Fluidized bed cement kilns	Fibers sent to Fluidized bed cement kilns	Static plastics MFA
	Concentration of decaBDE in fibers plastics of passenger vehicles in the Netherlands	(Leslie et al., 2013)

To complete the static decaBDE MFA model, the transfer coefficients of the remaining processes need to be determined. To do this, the resulting flows from the plastics static MFA are used. The

decaBDE model also takes the emission of the flame retardant to the atmosphere into account – therefore, the emission factor data for each process are collected. The source for these emission factors is Xue et al., 2017, which focuses on decaBDE present electric and electronic equipment appliances. It is assumed that the behavior of decaBDE release in electric and electronic equipment plastic processes is comparable to that of passenger vehicle plastics. Additionally, the Blast furnace and Fluidized bed cement kilns processes are assumed to have the same emission factors as the Incineration process. Moreover, while the study was published in 2017, there is no information on the age of the emission factors. It is assumed that they are still applicable to the year 2019. In Table 2.4, an overview of the variables and data sources used to estimate the transfer coefficients of the remaining processes is given.

Table 2.4. Variables needed and data sources used to estimate the transfer coefficients of the Vehicle use, Car dismantling, Mechanical recycling, Shredding, Incineration, Blast furnace, and Fluidized bed cement kilns processes.

Process	Variables	Data source
Vehicle use	Share of decaBDE emitted to the atmosphere during the use of the vehicle	Emission factors of decaBDE in the different processes are retrieved from Xue et al. 2017, with which the shares of the flame retardant emitted to the atmosphere are calculated. To calculate the remaining shares or transfer coefficients, the plastic flows of these processes resulting from the plastics static MFA, are used.
	Share of decaBDE that is plastics of exported vehicles	
	Share of decaBDE that is in plastics of EOL vehicles	
	Share of decaBDE that is in plastics of EOL vehicles that go via unofficial channels	
Car dismantling	Share of decaBDE that is in plastics present in dismantled cars that are sent to Shredding	
	Share of decaBDE that is in plastics of car parts that are removed and sent to Mechanical recycling	
	Share of decaBDE that is in plastics that are sold as second hand	
Mechanical recycling	Share of decaBDE that is emitted to the atmosphere during the mechanical recycling process	
	Share of decaBDE in mechanically recycled plastic	
Shredding	Share of decaBDE that is emitted to the atmosphere during the shredding process	
Incineration	Share of decaBDE that is emitted to the atmosphere during the incineration process	
Blast furnace	Share of decaBDE that is emitted to the atmosphere during the blast furnace process	

Fluidized bed cement kilns	Share of decaBDE that is emitted to the atmosphere during the blast furnace process	

A detailed account of the calculation steps to reach the final fixed flow and transfer coefficient values can be found in Appendix B and a graphical representation of the final quantification of the flows can be found in section 3.1.

2.2. Layered Dynamic Material Flow Analysis

This section covers the concept of a dynamic MFA (DMFA) (subsection 2.2.1) and the reason why the one in this research is defined as a layered DMFA (subsection 2.2.2). Then, the scenarios that are explored in this study are described (subsection 2.2.3), and the flame retardant alternative selection is presented (subsection 2.2.4). Lastly, the goal and system definition step, and inventory and modelling step of the DMFA, for both plastics and flame retardants, are carried out in subsections 2.2.5 and 2.2.6.

2.2.1. Dynamic Material Flow Analysis

DMFAs introduce the time variable to an MFA. By doing this, they have the capacity to capture time-dependent elements, such as changes in the development of in-use stock and the associated postconsumer flows (Buchner et al., 2015). This capacity is very useful to be able to quantify the in-use stock of decaBDE over time and to make forecasts of future stocks and flows according to specific scenarios. Given the restrictions of this chemical in newly produced passenger vehicles the toxicity concerns associated to it, it is especially of significance to estimate the emissions of the flame retardant to the environment over time. A DMFA can also help to quantify how much of the substance remains in the in-use stock, and until when. In this study, several scenarios are considered with the aim of comparing the results and gaining insights into what might be the advantages and trade-offs of different decisions.

To build a DFMA of plastics and flame retardants in passenger vehicles, more information is required in addition to that needed to build a static MFA – such as the life span of the passenger vehicles, the inflow or stock of passenger vehicles over time, the transfer coefficients of the processes over time according to different scenarios, etc.

The steps of building a dynamic MFA are the same as those for a static MFA. The first step, drafting the boundaries and formulating the goal of the system, is covered in subsection 2.2.5. The second step, which is concerned with the quantification of the flows and stocks of the system is elaborated on in subsection 2.2.6 and the results are finally presented in section 3.2.

2.2.2. Layered: How?

A DMFA is created for both plastics and flame retardants. However, similarly to the layered static MFA, these are interlinked. The results obtained from the plastics dynamic MFA are useful to quantify the flows of the flame retardant MFAs. By doing this, the dynamic MFA has two layers: plastics, and flame retardants.

The DMFA has three layers: the product layer (passenger vehicles), the material layer (plastics), and the additive layer (DecaBDE and TPP). Like in the static MFA, the material level depends on the product level, and the additive level depends on the material level. By creating this layered DMFA, forecasts about future stocks and flows of flame retardants, for which data is scarce, can be made. -

2.2.3. Scenarios

The Dutch Cabinet aims to have achieved a fully CE by 2050. Fundamentally, a CE is one that uses the minimum necessary amount of material resources and produces the least amount of waste. The Dutch government recognizes that this can be achieved through the implementation of the following main circular strategies: narrowing the loop through the rejection, use intensification or use reduction of certain products (R1 and R2), slowing the loop through product reuse, repair, or refurbishment (R3 and R4), and closing the loop through product recycling (R5). These strategies are represented in the R-ladder in Figure 2.3. In general, the circular strategies that are higher on the ladder make use of the least material resources, as well as processing steps, and are therefore thought to result in the least environmental impacts (Hanemaaijer et al., 2021). Additionally, in the transition to a CE, it is important to ensure that the material flows do not contain toxic substances; recycling of such materials can result in detrimental environmental and human health impacts. To alleviate this, the toxic flows should be substituted by non-toxic or less toxic ones.

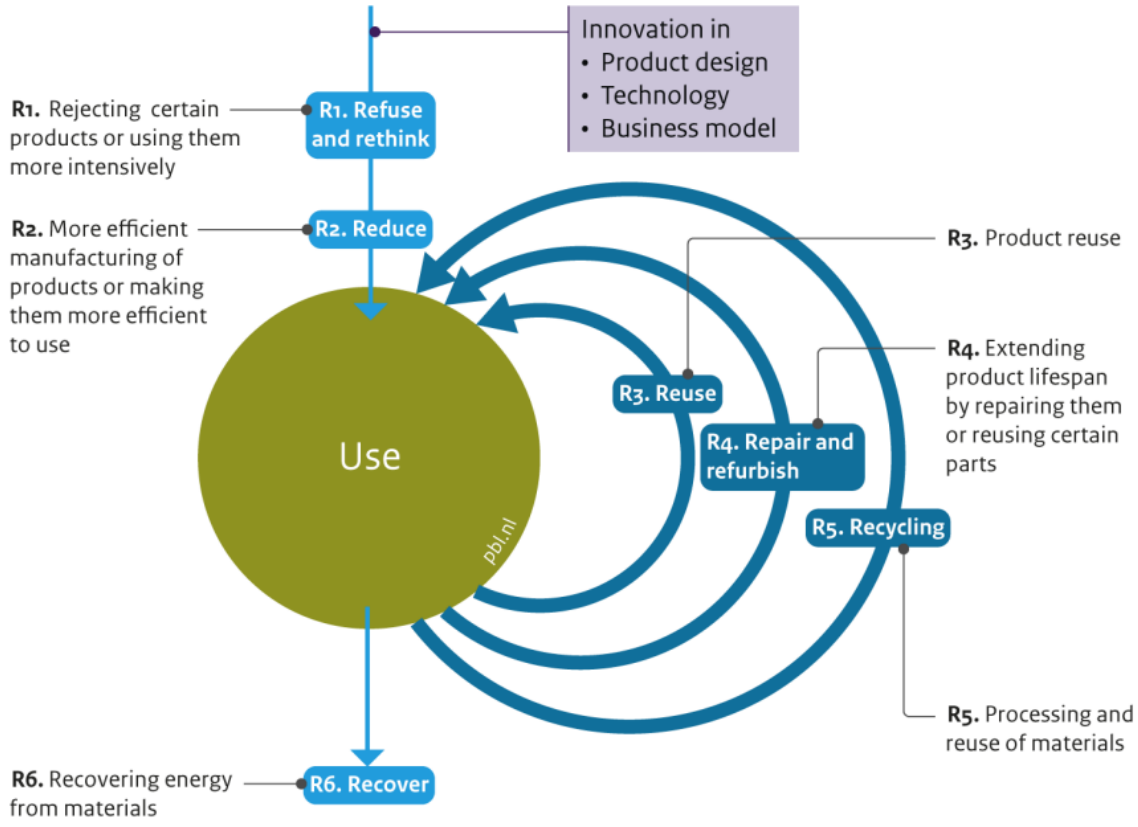

Figure 2.3. R ladder with circularity strategies (PBL, 2021).

A DMFA can give insights into the consequences that implementing different circular strategies could have over time by analyzing several scenarios. In this study, the aim is to compare the impacts

between scenarios that have different approaches to the handling of EOL flows of passenger vehicles in The Netherlands. In each scenario, a different circular strategy is used. The reuse strategy (R3) is considered to take place within the vehicle use phase, limited by the lifespan of the vehicle, and therefore is not further investigated in the scenarios. Also, the repair and refurbish strategy (R4) is left out of the scenarios due to the technical impossibility of reusing old plastic car parts in younger ones. The focus is then primarily on strategies that are considered relevant to plastics in passenger vehicles: refuse and rethink (R1), recycling (R5), and substitution of toxic flows.

Taking said strategies into account, the following six scenarios are elaborated:

– *Reference*

  In this scenario, no major changes occur regarding the shares of plastics that are recycled and incinerated. The passenger vehicle fleet evolves according to what is expected, as well as the share of plastics in vehicles and flame retardants in passenger vehicles' plastics. DecaBDE is gradually substituted by an alternative, non-banned, FR. As the share of decaBDE in passenger vehicle plastics decreases, the share of the alternative FR increases.

– *Recycling*

  In the Recycling scenario, the share of mechanically recycled plastics increases gradually over time, attempting to close the plastic loop (R5 in Figure 2.3).

– *Incineration*

  In contrast to the previous one, in this scenario the share of incinerated plastics increases over time (and the share of recycled plastics decreases).

– *Less Cars*

  In the Less Cars scenario, the car fleet does not increase, despite the growth in population and prosperity, under the assumption that less cars are used per capita due to sustainability awareness and increase in public transport use. This results in a reduction of the loop (R1 in Figure 2.3).

– *No DecaBDE Ban*

  In this scenario the decaBDE share in vehicle plastics over time remains the same, assuming that there are no restrictions on decaBDE use. This shows what the environmental impacts might have been had there not been a ban on decaBDE.

– *No Substitution*

  Lastly, mainly created for an illustrative purpose, the No Replacement scenario considers that no substitute to decaBDE is implemented, despite its decreasing use.

2.2.4. Selection of flame retardant alternative: Triphenyl Phosphate

Since the listing of PBDEs as POPs in the Stockholm Convention, a transition to organophosphorus FRs has taken place, which are believed to be less detrimental for human health and the environment (Wong et al., 2018). In this this research, one representative flame retardant alternative is included

in the dynamic MFA with the objective of gaining some insights into the health and environmental trade-offs of using a decaBDE FR versus an organophosphorus based one.

The European Commission funded research project "Life Cycle Assessment of Environment-Compatible Flame Retardants: Prototypical Case Study" (ENFIRO) sought to determine the most suitable and viable alternatives for various BFRs in electric and electronic equipment applications (https://www.enfiro.eu/). As suitable commercially available organophosphorus alternatives to decaBDE, three main additive FRs were identified: resorcinol bis(diphenyl) phosphate, bisphenol A diphenyl phosphate, and Triphenyl Phosphate (TPP). It is likely that all three (and more) FRs were used to replace decaBDE in the past years, as the replacement choice depends on case-by-case requirements, and it is not a "one size fits all" situation. In this study, however, only one of these is included in the analysis: TPP, whose presence has been widely reported in passenger vehicles in recent years (Christia et al., 2018; Fabiańska et al., 2019; Tran et al., 2020). This choice was delimited by the data availability. Unlike the other two organophosphorus FRs, there is (minimum) TPP data on concentration in passenger vehicle polymers, emission factors (relevant to build the dynamic MFA), and characterization factors (to be elaborated on in the life cycle impact assessment in section 2.3).

2.2.5. Goal and system definition

The goal of this layered dynamic material flow analysis is to answer sub-questions 3 and 4:

3. *What would the major plastic material flows of passenger vehicles look like under different circular economy scenarios in The Netherlands?*
4. *How does the flame retardant in plastics of passenger vehicles evolve under different circular economy scenarios in The Netherlands?*

The materials under study are plastics in passenger vehicles, decaBDE in plastics of passenger vehicles, and TPP in plastics of passenger vehicles. TPP is added to the dynamic analysis to have a clearer picture of the impacts of the different scenarios. As the share of decaBDE in plastics decreases over time, the share of TPP increases. By including it to the analysis, it is ensured that the impact associated to the flame retardant replacement are considered. If it were left out, it is likely that there would be impacts left unaccounted for. Moreover, it allows for an environmental performance evaluation between decaBDE and TPP, to potentially analyze how much better the replacement really is.

The system's spatial boundaries are the same as in the layered static material flow analysis. However, the level of detail of the system is somewhat simplified. Moreover, while the vehicle production process is still not a part of the system, the emission factors of this process are considered when accounting for total emissions of a given scenario. This is especially relevant to be able to compare the emissions between scenarios that have a higher recycling share than others – if material needs to be newly produced to satisfy the demand, then this comes with the emissions of the production process.

While with the static MFA the goal was to obtain a detailed depiction of the current system, the focus of the dynamic MFA is to look at the development of the main flows and stocks over time. The simplification does not cause big changes in the quantification of the flows and saves modelling time.

The temporal boundaries are set to the 1980 – 2050 period. A graphical representation of the system can be found in Figure 2.4.

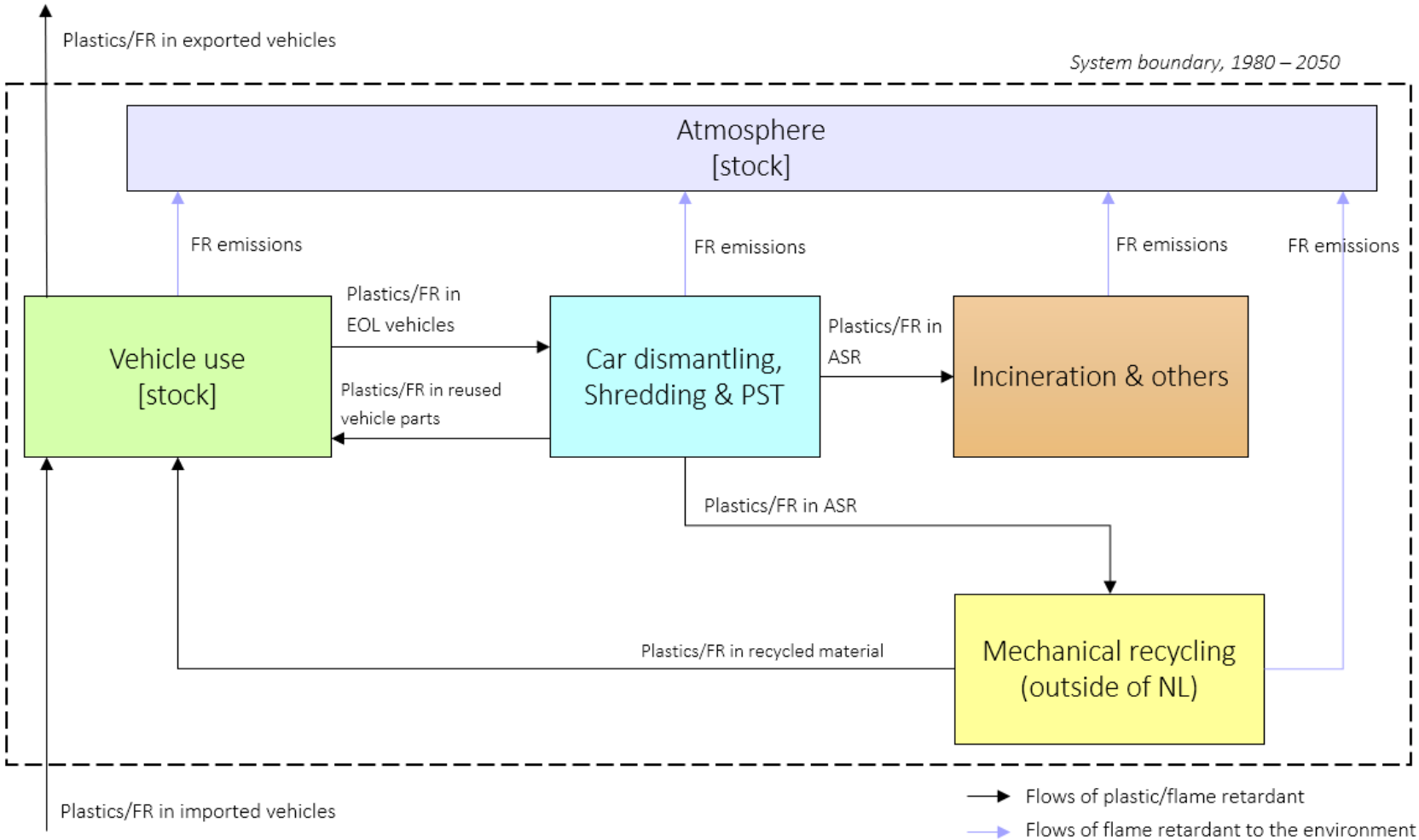

Figure 2.4. Defined system for plastics, decaBDE and TPP in the time-period 1980 – 2050 (needs update to add flow from mechanical recycling to incineration and put atmosphere outside the system boundaries).

2.2.6. Inventory and modelling

In this step, the material flows of plastics, decaBDE and TPP from are quantified from 1980 through 2050 for the six scenarios by means of the layered DMFA. The model is created in Python, a free open-source programming language that can easily handle large datasets. It is partly created using the Open Dynamic Material Systems Model (ODYM) Python module (Pauliuk, 2017/2020). Specifically, the compute_s_c_inflow_driven and compute_stock_driven_model functions, which can calculate the stocks and outflows (in the case of an inflow-driven model), or inflows and outflows (in the case of a stock-driven model), of the Vehicle Use process, given the necessary data inputs (see blue boxes in Figure 2.5). Then, the results from these calculations are weaved into the plastics and additive systems to calculate the values of the remaining flows. A link to the GitHub repository containing the model, including the calculations and data used, is available in Appendix D.

An overview of the link between the dynamic MFA of plastics and the dynamic MFA of FRs is shown in Figure 2.5. Here, the basic data requirements are also shown.

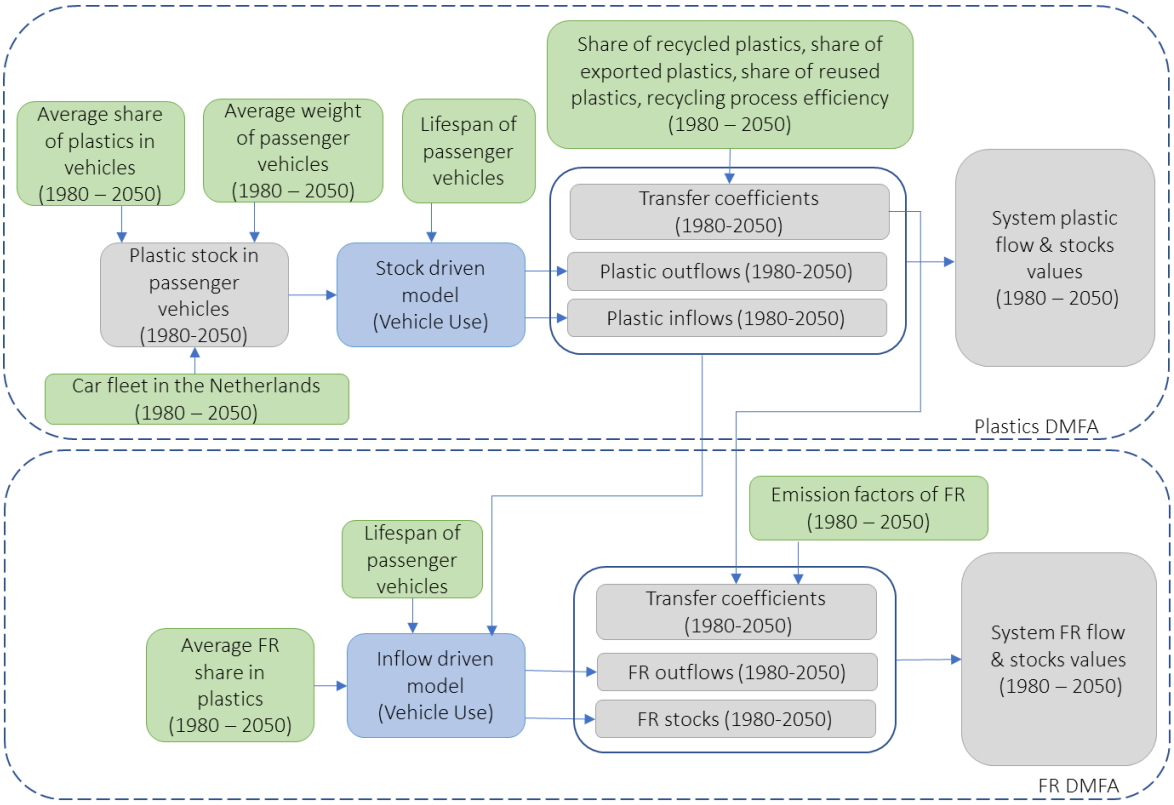

Figure 2.5. Overview of the required data and calculation steps to determine the plastic and FR system flows from 1980 through 2050. Green boxes indicate required data, light grey boxes indicate calculated values.

Depending on each scenario and its assumptions, the data used is different. However, there are some data that is the same for all scenarios. An overview of the required data, assumptions, and data sources for the plastics DMFA can be found in Table 2.5.

Table 2.5. Data requirements, assumptions, and sources for the plastics DMFA, for all scenarios.

Data requirement	Assumptions	Data source
Average content of plastics per passenger vehicle from 1980 until 2050 in NL	Same value for all six scenarios. To calculate this, data of the average plastic content in passenger vehicles and data of the average weight of passenger vehicles is collected for the years 1980, 2000, 2020, and 2050, and they are multiplied by each other to obtain the average content of plastics. A linear evolution is assumed between the years for which data is available.	(Becque & Sharp, 2020; Kearney, n.d.) and personal communication with Joint Research Centre experts.
Car fleet in NL from 1980 until 2050	Scenarios: Reference, Recycling, Incineration, No DecaBDE Ban, and No Substitution. For the period 1980 – 2019, historical data used. Then, in 2030 and in 2050, the car fleet values according to the WLO Hoog scenario created by PBL are used. A linear interpolation is used to fill in the years for which there is no WLO Hoog value available.	(CBS, 2019; CPB/PBL, 2015).
	Scenario: Less Cars. For the period 1980 – 2019, historical data used. Then, assuming a decrease in number of cars per capita, the car fleet values according to the WLO Laag scenario created by PBL are used. A linear interpolation is used to fill in the missing years.	
Lifespan of passenger vehicles in NL	Same value for all six scenarios. The average lifespan of a passenger vehicle is set to be 18.1 years and it is assumed to be constant over time. Additionally, it is assumed that the lifespan is distributed according to a Weibull distribution.	(CLO, 2019; Held et al., 2021)
Share of mechanically recycled plastics in NL	Scenarios: Reference, Less Cars, No DecaBDE Ban, No Substitution. In 2012, the first Post-Shredder Technology factory was installed in the Netherlands, which allowed the mechanical recycling of Automotive Shredder Residue plastics (ASR) for the first time. The share (or transfer coefficient) of mechanically recycled plastics was set to that obtained in the static MFA for 2019, and kept constant until 2050, assuming no major changes.	(Leslie et al., 2016)
	Scenario: Recycling.	

	The share of mechanically recycled plastics is the same as in the previously mentioned scenarios until 2012. From 2013 onwards, the share of mechanically recycled plastics is assumed to increase gradually, reaching 1 in 2050.	
	Scenario: Incineration. The share of mechanically recycled plastics is the same as in the previously mentioned scenarios until 2012. From 2013 onwards, the share of mechanically recycled is assumed to decrease to 0 and the share of incinerated plastics is assumed to increase, reaching 1 in 2050.	
Share of plastics in exported vehicles	Same value for all six scenarios. The share of exported plastics is assumed to remain constant from 1980 until 2050, adopting the same value as in the static MFA of plastics.	n.a.
Share of plastics in reused car parts	Same value for all six scenarios. From 1980 until 2012, the share of reused plastics is assumed to be the same as that obtained in the plastics static MFA. From 2012 onwards, it is assumed that plastic parts are no longer dismantled from cars for the purpose mechanical recycling, and that all plastics in the car go through the shredding and PST process.	n.a.
Efficiency of the mechanical recycling process	Same value for all six scenarios. Assumed to remain constant over time, from 1980 until 2050, adopting the value obtained in the static plastics MFA.	n.a.

As shown in Figure 2.5, the flame retardants' DMFAs build on the transfer coefficients of the plastics' DMFA and its resulting plastic inflow. As seen in the figure, additional data is required to run the model: the average content of flame retardant in plastics from 1980 until 2050, and the emission factors of the flame retardant in the various processes from 1980 until 2050. These data is presented in Table 2.6.

Table 2.6. Additional data requirements, assumptions, and sources for the flame retardants DMFA, for all scenarios.

Data requirement	Assumptions	Data source
Average content of flame retardant in passenger vehicle plastics in the Netherlands from 1980 until 2050	DecaBDE layer Specific data for the concentration of decaBDE in passenger vehicle plastics is sparse and scarce. While there is data for it for 2013, used in the static MFA, there is no Dutch-specific data for any other years. Therefore, in the dynamic MFA, values calculated by the UNEP are used – which are an estimate of the approximate content of decaBDE in ASR plastics over time based on data from various European countries and Japan. The concentrations are available for "before 1996", "before 1999", "before 2018", and "after 2018". A linear evolution between these values is assumed.	(UNEP, 2021)
	TPP layer Due to their more recent applications, there is even less data regarding TPP content in plastics of consumer products, let alone passenger vehicle plastics. Therefore, to estimate the concentration of TPP, a substitution ratio of TPP to decaBDE is created based on scientific literature. This ratio represents the amount of TPP that is needed to replace decaBDE for the plastic to retain equal flame retardancy properties. The logic is that as decaBDE is phased out, it is substituted by TPP, times the substitution ratio. Depending on the plastic, the concentration of TPP ratio can range from 1 to 2. Therefore, a ratio of 1.5 is assumed.	(EPA, 2014)
Emission factors	DecaBDE and TPP The scarcity of data associated to flame retardants also extends to the emission factors of these substances. Emission factors were found for both these flame retardants pertaining processes related electric and electronic equipment products (previously used in the static MFA). It is assumed that these remain constant over time.	(Xue et al., 2017)

To calculate the final transfer coefficients of the flame retardant system, the emission factors of each flame retardant and the transfer coefficients of the plastic system are used. The emission factor is simply included as the transfer coefficient of flame retardant being emitted to the atmosphere, and

the remaining share of flame retardant is distributed according to the plastic dynamic MFA transfer coefficients.

A more detailed description of the inputs to the layered DMFA can be found in Appendix C and the final data used, as well as the layered dynamic model can be found in the publicly accessible GitHub repository linked in Appendix D.

2.3. Life Cycle Impact Assessment

Life Cycle Impact Assessments (LCIAs) enable the conversion of life cycle inventory's information on elementary flows of different processes into environmental impact scores (Rosenbaum et al., 2018). In other words, an LCIA is able to translate emissions and resource extraction flows into a limited number of impact scores (Hauschild & Huijbregts, 2015). The outcome of an LCIA is an environmental profile, where a score list with the different environmental effects (e.g., climate change, soil acidification, etc.) is provided. With this, it is possible to evaluate which (phase of a) product contributes the most to different environmental effects (National Institute for Public Health and the Environment, 2016).

The translation of emissions into environmental impacts is done at the hands of Characterization Factors (CFs), which express the environmental impact per unit of emission. These exist at the midpoint and endpoint level. While the midpoint level is concerned with single environmental problems (e.g., water use, climate change), the endpoint level shows the aggregated environmental impacts in three main categories: human health, ecosystem quality, and resource scarcity. These ways of quantifying environmental impacts complement each other: the midpoint characterization is more closely related to the elementary flow and has a relatively low uncertainty, while the endpoint characterization supplies a better understanding of the environmental relevance of the flows but has a higher uncertainty (Hauschild & Huijbregts, 2015).

There are several LCIA methods available, which differ to a certain extent on CFs and impact categories (e.g., IPCC 2013, ILCD 2011, ReCiPe 2008). To conduct this study, ReCiPe 2016 is used. An overview of the impact categories considered by this method is presented in Figure 2.6.

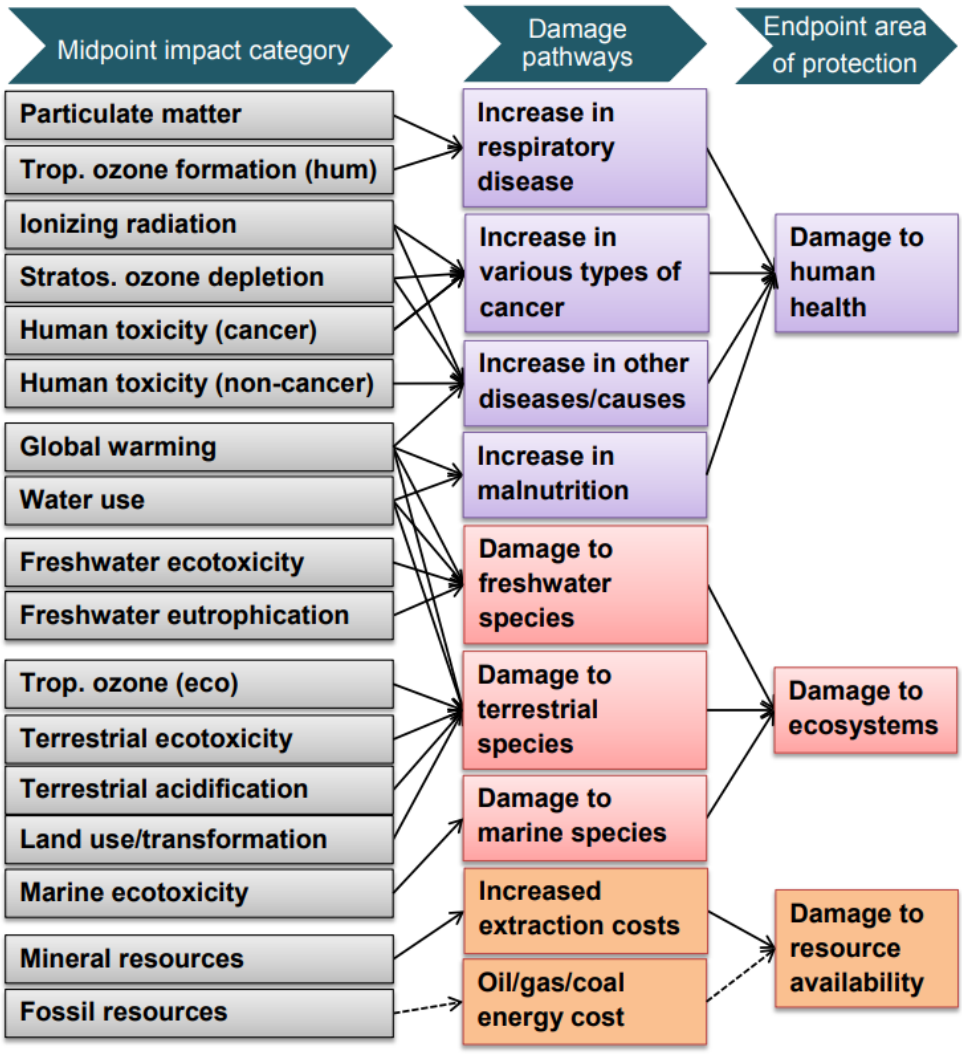

Figure 2.6. Overview of the impact categories covered by the ReCiPe 2016 method and their relation to the three endpoint levels. Figure obtained from National Institute for Public Health and the Environment, 2016. The midpoint impact categories used in this study are Human toxicity (cancer), Human toxicity (non-cancer), Terrestrial ecotoxicity, Freshwater ecotoxicity, Marine ecotoxicity, and Global warming.

To carry out the LCIA, the resulting emission flows of decaBDE and TPP, as well as the FR incineration flows, are obtained from the layered DMFA are compiled and multiplied by the corresponding CFs (Figure 2.7). These calculations are carried out in the Python; a link to the Python model can be found in Appendix D.

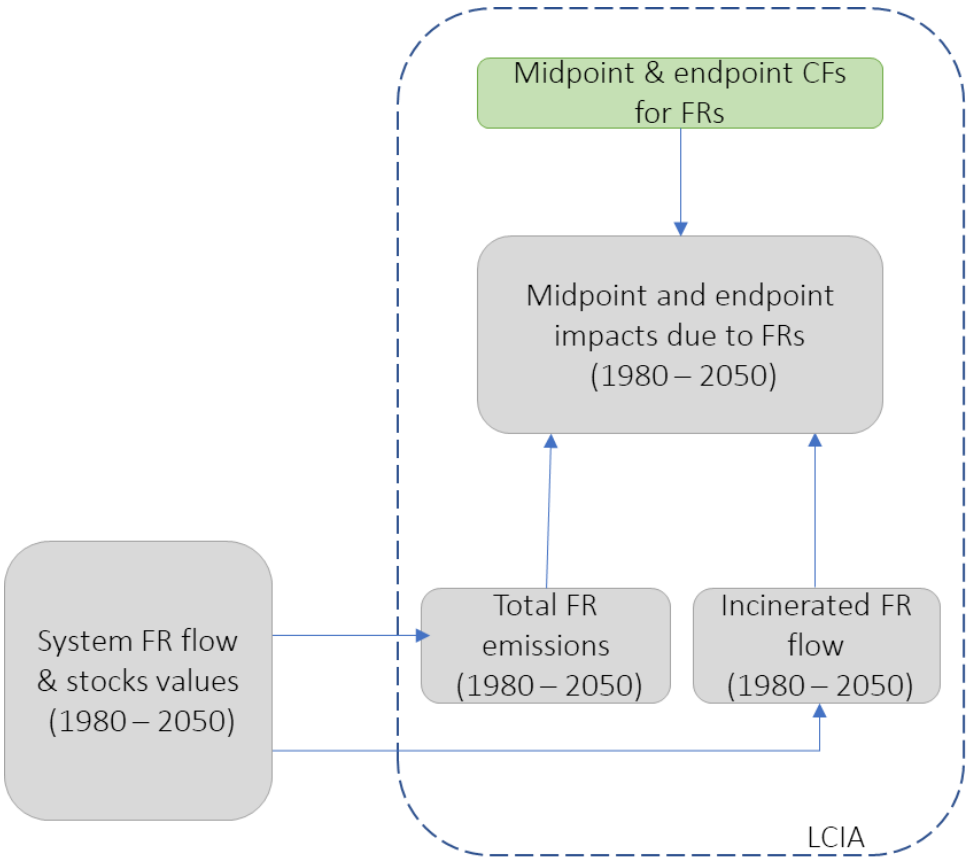

Figure 2.7. An overview of the steps required to get to the midpoint and endpoint impact results, linking the LCIA method with the flame retardant flows resulting from the layered dynamic MFA.

The CFs for decaBDE and TPP relevant to this study can be found in Table 2.7 and Table 2.8, respectively. It is relevant to note that decaBDE and TPP do not share any impact category at the midpoint level, nor at the endpoint level. The lack of CFs of decaBDE freshwater, marine water, and terrestrial ecotoxicity, and the lack of CFs of TPP for human toxicity, can be attributed to lack of reliable toxicity data (Jonkers et al., 2016).

Table 2.7. Mid-point and end-point CFs to Air for decaBDE, as per ReCiPe 2016.

Midpoint impact category	Destination	Midpoint CF (kg 1,4-DCB/kg)	Endpoint CF (DALY/kg)
Human carcinogenic toxicity	Air	0.486	1.61E-06
Human non-carcinogenic toxicity	Air	493	1.12E-04

Table 2.8. Mid-point and end-point CFs to Air for TPP as per ReCiPe 2016.

Midpoint impact category	Destination	Midpoint CF (kg 1,4-DCB/kg)	Endpoint CF (species*yr/kg)
Terrestrial ecotoxicity	Air	38.9	4.43E-10
Freshwater ecotoxicity	Air	0.212	1.47E-10
Marine ecotoxicity	Air	1.09	1.14E-10

To calculate the impact category results, the CFs of the midpoint and endpoint levels are multiplied by the (sum of) emission flow values for each FR. For example, the calculation of the human carcinogenic toxicity midpoint category results goes as follows:

$$Human\ carcinogenic\ toxicity\ potential\ of\ decaBDE = Midpoint\ CF(Human\ Carcinogenic\ Toxicity, Air)_{decaBDE} \times total\ emissions\ of\ decaBDE\ in\ 2010$$

The same logic applies to the rest of the midpoint toxicity categories.

Regarding the Global Warming midpoint category, a different approach is taken. As no CFs were available, an alternative method is implemented to account for potential carbon dioxide emissions. It is assumed that all carbon atoms of the FRs end up as carbon dioxide in the atmosphere. To do this, the molar mass, and the chemical formulas of the FRs and carbon dioxide are needed (Table 2.9), as well as the values of the decaBDE and TPP quantities headed to the incineration process each year (obtained in the layered dynamic MFA).

Table 2.9. Chemical formula and molar mass of decaBDE, TPP, and carbon dioxide.

Substance	Chemical formula	Molar mass (g/mol)
DecaBDE	$C_{12}Br_{10}O$	959.17
TPP	$C_{18}H_{15}O_4P$	326.28
Carbon dioxide	CO_2	44

As the midpoint CF of carbon dioxide is 1, the global warming potential impacts can be calculated directly. For instance, to calculate the Global Warming potential impacts of decaBDE for a given year, the following calculation is carried out:

$$Global\ warming\ potential\ impacts\ of\ decaBDE\ in\ 2010\ (kg\ of\ CO2) =$$
$$kg\ of\ decaBDE\ headed\ to\ incineration\ in\ 2010 \times \frac{12\ mol \times 0.044 \frac{kg}{mol}}{1\ mol \times 0.959 \frac{kg}{mol}}$$

Table 2.10. Midpoint and endpoint CFs for carbon dioxide (ReCiPe 2016).

Impact category	Destination	Midpoint CF (kg CO2 eq / kg)	Endpoint CF (DALY / kg)	Endpoint CF (species.yr/kg)
Global warming, human health	Air	1	0.000000928	n.a.
Global warming, terrestrial ecosystems	Air	1	n.a.	2.8E-09
Global warming, freshwater ecosystems	Air	1	n.a.	7.65E-14

Using the endpoints provided in

Table 2.10, the damage to human health and to ecosystems can be calculated.

To calculate the damage to human health for a given year:

$$\begin{aligned} &Damage\ to\ human\ health\ in\ 2010 \\ &= Total\ FR\ emissions\ to\ atmosphere\ in\ 2010 \\ &\times [Endpoint\ CF\ (Human\ carcinogenic\ toxicity, Air)_{decaBDE} \\ &+ Endpoint\ CF(Human\ non\ carcinogenic\ toxicity, Air)_{decaBDE}] \\ &+ Global\ warming\ potential\ impacts\ of\ FR\ in\ 2010 \\ &\times Endpoint\ CF\ (Global\ warming, human\ health, Air)_{CO2} \end{aligned}$$

And, lastly, to calculate the damage to ecosystems for a given year:

$$\begin{aligned} &Damage\ to\ ecosystems\ in\ 2010 \\ &= Total\ emissions\ to\ atmosphere\ in\ 2010 \\ &\times [Endpoint\ CF\ (Terrestrial\ ecotoxicity, Air)_{TPP} \\ &+ Endpoint\ CF\ (Freshwater\ ecotoxicity, Air)_{TPP} \\ &+\ Endpoint\ CF\ (Marine\ ecotoxicity, Air)_{TPP}] \\ &+ Global\ warming\ potential\ impacts\ in\ 2010 \\ &\times [Endpoint\ CF\ (Global\ warming, Terrestrial\ ecosystems)_{CO2} \\ &+ Endpoint\ CF\ (Global\ warming, Freshwater\ ecosystems)_{CO2}] \end{aligned}$$

3. Results

This chapter includes the main results obtained in this study. First, the main outcomes from the static MFA of plastics and decaBDE are covered in section 3.1. Subsequently, in section 3.2, the main results from the layered DMFA are covered. Lastly, the midpoint and endpoint impact of the different scenarios from the dynamic MFA are presented in section 3.3.

3.1. Layered Static Material Flow Analysis

3.1.1.1. Going back to the sub-questions

The layered static MFA aims to answer the two following research sub-questions:

1. *What do the major plastic material flows of passenger vehicles currently look like in The Netherlands?*
2. *What do major flame retardant flows in plastics of passenger vehicles currently look like, and how do they behave throughout the different processes?*

The questions can be partly answered by looking at the quantification of plastics in passenger vehicles in the Netherlands in 2019 (Figure 3.1) and the quantification of decaBDE in passenger vehicle plastics in the Netherlands in 2019 (Figure 3.2), respectively. These give an idea of what the flows of plastics and decaBDE look like in a static state.

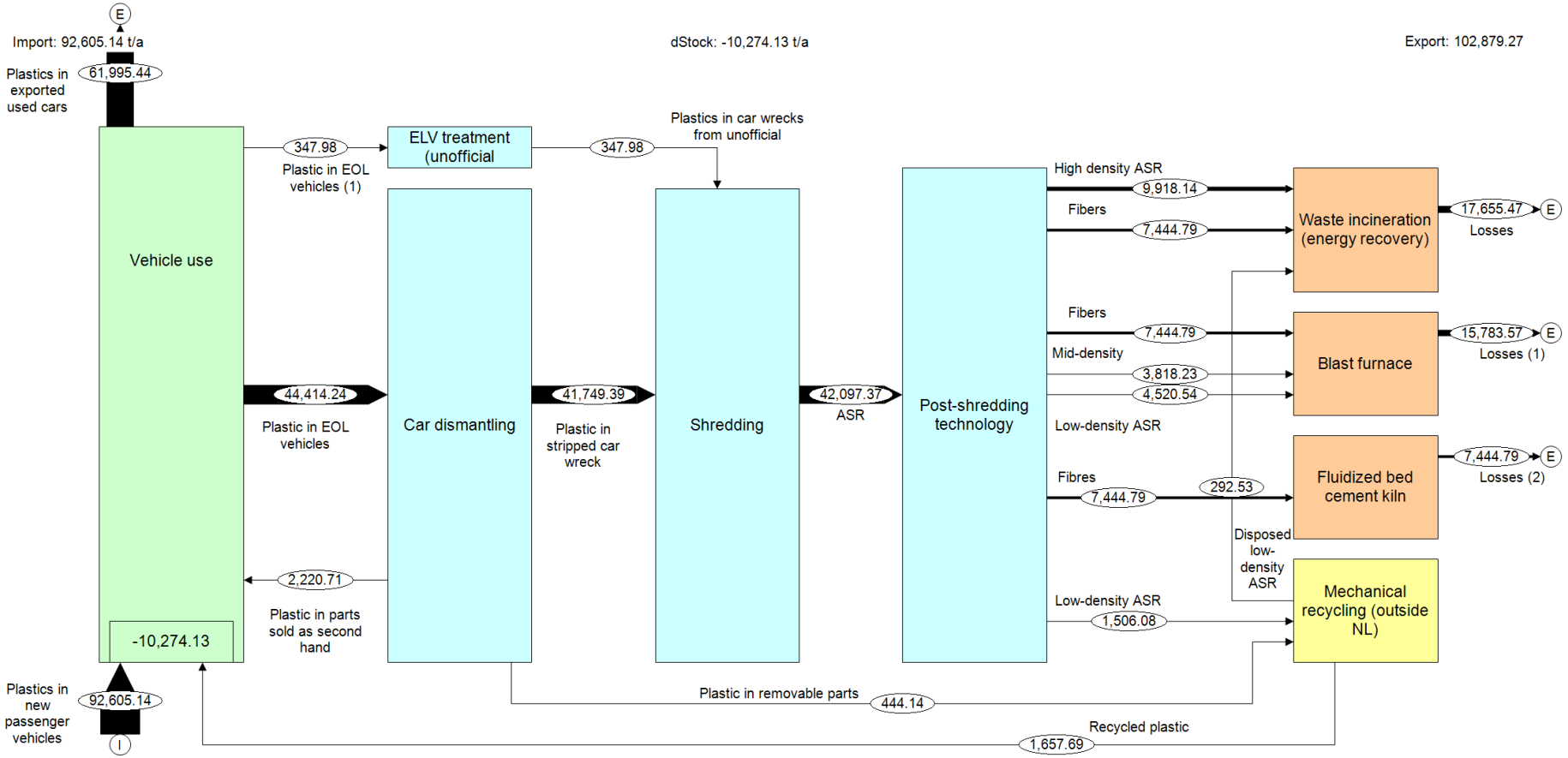

Figure 3.1. Quantification of plastics in passenger vehicles in The Netherlands in 2019 (unit: tons/year).

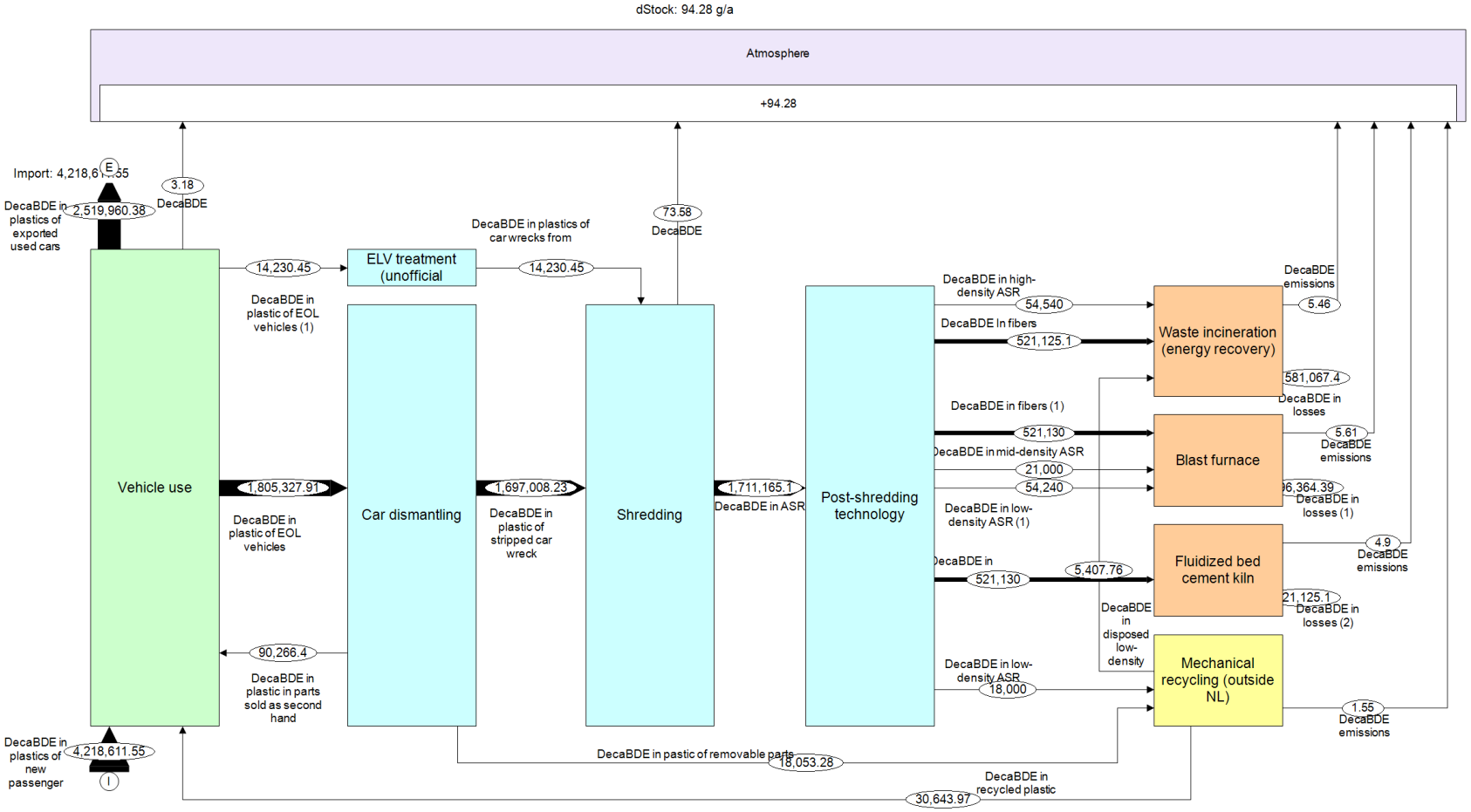

Figure 3.2. Quantification of decaBDE flows in plastics of passenger vehicles in The Netherlands in 2019 (unit: grams/year).

First, it can be observed that the share of plastics and decaBDE in exported cars is significantly high – higher than the share of cars that remain in the country to be dismantled. This is because the number of vehicles being exported from the Netherlands each year is higher than the number of vehicles being sent to be dismantled. Despite having similar characteristics as the cars being dismantled in The Netherlands, most of these exported vehicles are sent to other countries in East Europe, South America, and Africa, where they continue to be used for a few more years. This has sparked some controversy as these used vehicles cause detrimental effects in the air quality of their destinations (Human Environment and Transport Inspectorate, 2020).

When looking at the end-of-life flows, the quantity of mechanically recycled plastics is quite low compared to the share of plastics that are sent to incineration, blast furnaces, and used as fuel in fluidized cement kilns (Figure 3.3). This value of decaBDE present in plastics that get mechanically recycled is even smaller (Figure 3.4). This is because currently, mainly low-density ASR plastics are sent to recycling, which represent the lowest share of the type of plastics that exit the Post-Shredder Technology process. The reason this flow is suitable for recycling is precisely due to its low decaBDE content – recycling plastics with a high content of decaBDE is not allowed due to restrictions put in place by the EU POP Regulation.

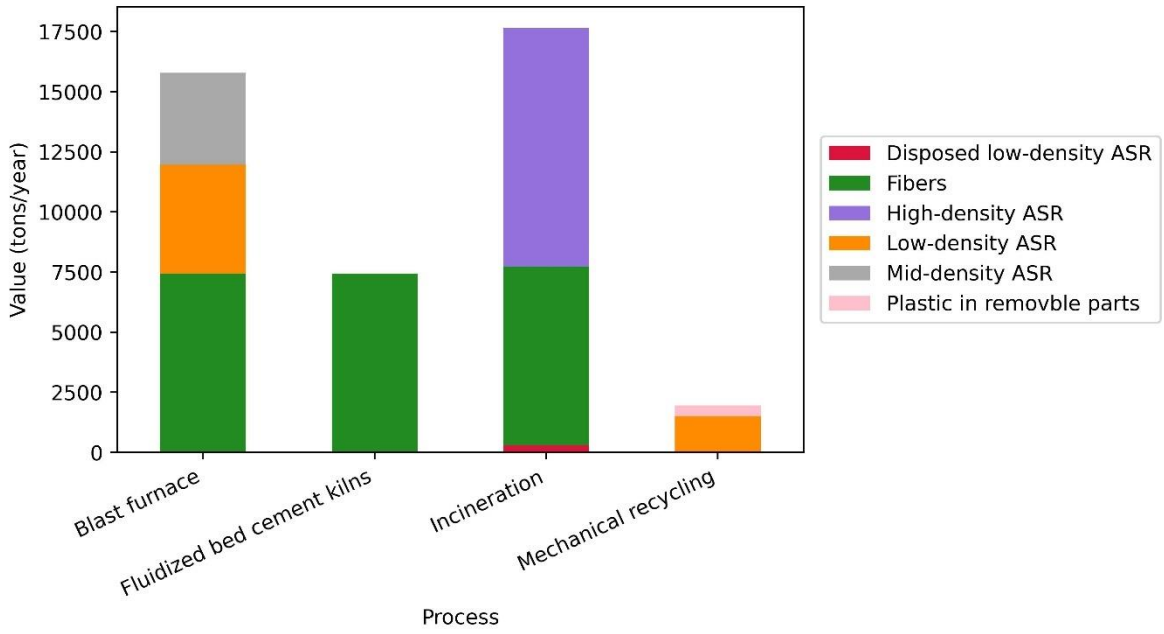

Figure 3.3. Stacked bar plot of the main destinations of ASR plastics after exiting the Post-Technology Shredder process. Each bar represents total amount of plastics destined to each process. Within each bar, distinctions are made between the types of plastics flows that are destined to the process.

Moreover, not all low-density ASR plastics are sent to mechanical recycling – only the ones formed by the polymer Acryl Butadiene Styrene (ABS). Economic feasibility is the reason other polymers from the low-density ASR category (e.g., polypropylene and polystyrene) are not sent to recycling: there is no market for these recycled polymers at the present. Therefore, if a high decaBDE concentration

was allowed in recycled plastics, and recycling of polymers other than ABS were economically feasible, the share of recycled plastics would increase significantly.

Fibers contain the highest shares of decaBDE (Figure 3.4). This is because they are mostly composed by polyurethane foams, which are required to be flame retardants. This type of ASR plastics is not easy to recycle – and ends up moslty incinerated, in blast furnaces or used as feedstock in the fluidized bed cement kilns process.

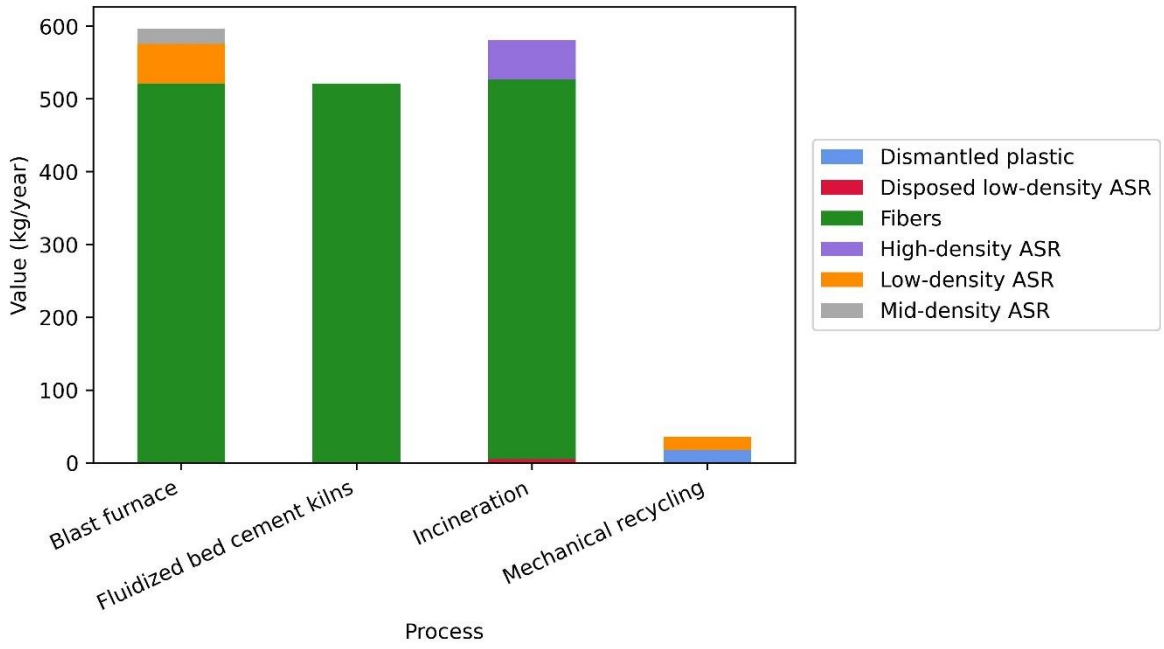

Figure 3.4. Stacked bar plot of the main destinations of decaBDE after exiting the Post-Technology Shredder process. Each bar represents total amount of plastics destined to each process. Within each bar, distinctions are made between the types of decaBDE-containing plastics flows that are destined to the process.

Concerning the decaBDE static MFA, it is relevant to look at the emissions to the atmosphere (Figure 3.5). Most emissions originate from the Shredding process. This is because Shredding is the process where the largest flow of decaBDE goes through, and the process with the highest emission factor. The Waste incineration, Blast furnace, and Fluidized bed cement kilns processes produce a similar number of emissions, whereas Vehicle Use and Mechanical recycling produce the least.

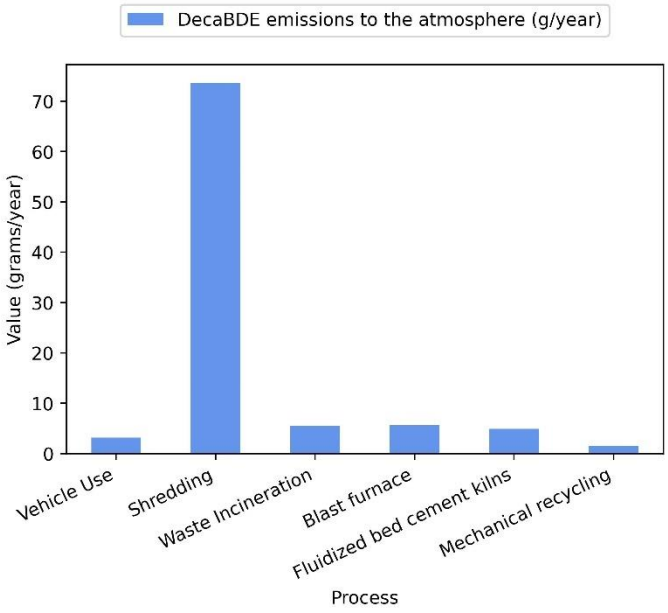

Figure 3.5. DecaBDE emissions to the atmosphere per process according to the static decaBDE MFA (unit: grams/year).

3.2. Layered Dynamic Material Flow Analysis

3.2.1.1. Answering the sub-questions

The layered dynamic MFA aimed to answer the sub-questions 3 and 4:

3. *How would the major plastic material flows of passenger vehicles evolve under different circular economy scenarios in The Netherlands?*
4. *How would the flame retardant in plastics of passenger vehicles evolve under different circular economy scenarios in The Netherlands?*

Plastics

The evolution of the stocks of plastic in the Vehicle Use process is pictured for all scenarios in Figure 3.6. However, only two curves are distinctly visible. Scenarios Reference (S_0), Recycling (S_1), Incineration (S_2), No DecaBDE Ban (S_4), and No Substitution (S_5) have the same plastics stock evolution (in grey color), while the Less Cars scenario (S_3) diverges in 2023. This is due to the vehicle fleet data used for the scenarios: while the first four scenarios mentioned assume a fleet growth according to the WLO Hoog scenarios, the Less Cars scenario assumes a lower fleet growth, according to the WLO Laag scenarios. Another variable that has an impact in the stock of plastics in Vehicle Use process is the plastic weight per passenger vehicle (determined by the average vehicle weight and the plastic share of the total weight). This variable is the same for all scenarios, and therefore has no influence on the discrepancies between the curves. However, the evolution of this variable has an influence in the shape of the curves. While the average vehicle weight of vehicles is expected to increase over time, the plastic shares in vehicles are expected to increase as well, which results in an increasing plastic weight over time. The increasing average plastic weight over time, together with the growing car fleet, are the reason behind the steep increase of the stock of plastics in the Vehicle Use phase visible in Figure 3.6.

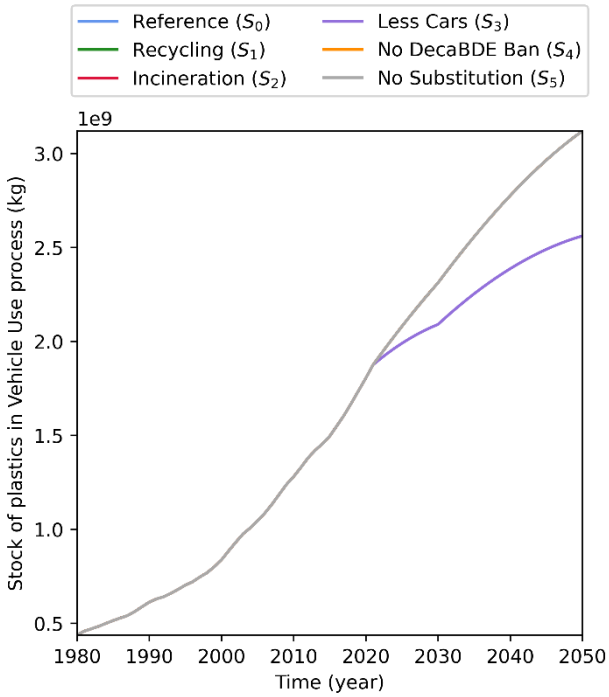

Figure 3.6. Stock of plastics in Vehicle Use process.

In Figure 3.7, the quantity of incinerated plastics per scenario is plotted over time. As expected, the quantity of plastics sent to incineration is highest in the Incineration scenario (S_2), growing steadily throughout the years. According to this scenario, by 2050, all plastics would be sent to incineration. In second place, are the Reference (S_0), No DecaBDE Ban (S_4), and No Substitution (S_5) scenarios, which have overlapping values. Despite the lower values compared to the Incineration scenario, there is relatively not a significant difference. This is explained by the share of incinerated plastics for the Reference (S_0) scenario (and the others), already being high to begin with. The Less Cars (S_3) scenario plastics to incineration flow is lower due to the lower car fleet, as expected, and the Recycling (S_1) scenario plastics to incineration flow decreases significantly from 2013 onwards, reaching zero in 2050.

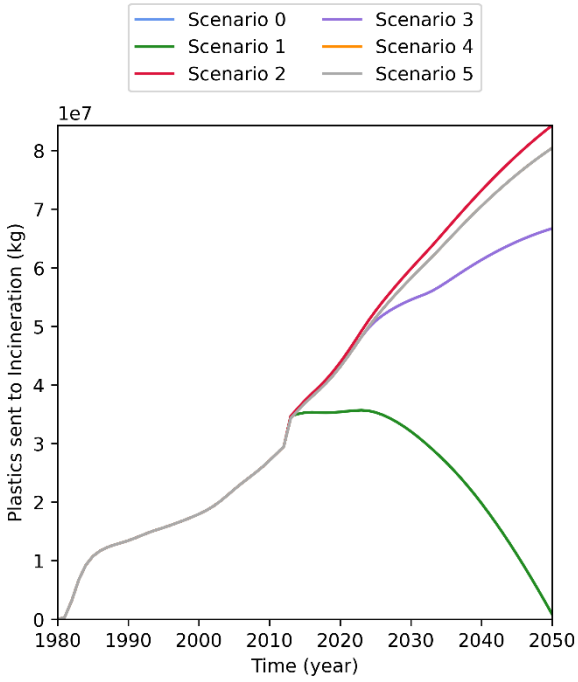

Figure 3.7. Plastics sent to Incineration, for all scenarios.

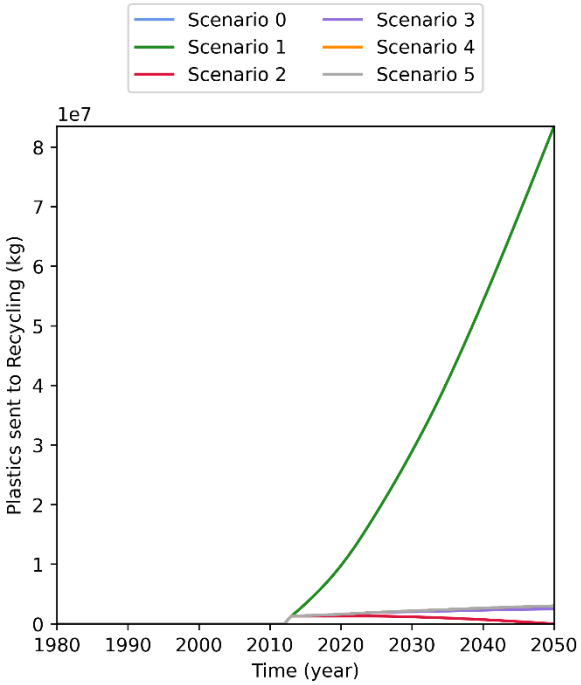

Figure 3.8. Plastics sent to Recycling, for all scenarios.

Conversely, Figure 3.8 shows the flows of plastic sent to recycling. The biggest flow belongs to the Recycling (S_1) scenario, which increases sharply from 2012, and assumes that all plastics are sent to incineration in the year 2050. The contrast with the other scenarios is evident. In all other scenarios, except for the Incineration one, the share of plastics sent to incineration remains constant over time from 2012 onwards, at only 3.5%.

Flame retardants

When it comes to flame retardants, it is also relevant to look at the stock of the substance in the Vehicle Use phase. This is shown in Figure 3.9 for decaBDE, and in for TPP in Figure 3.10.

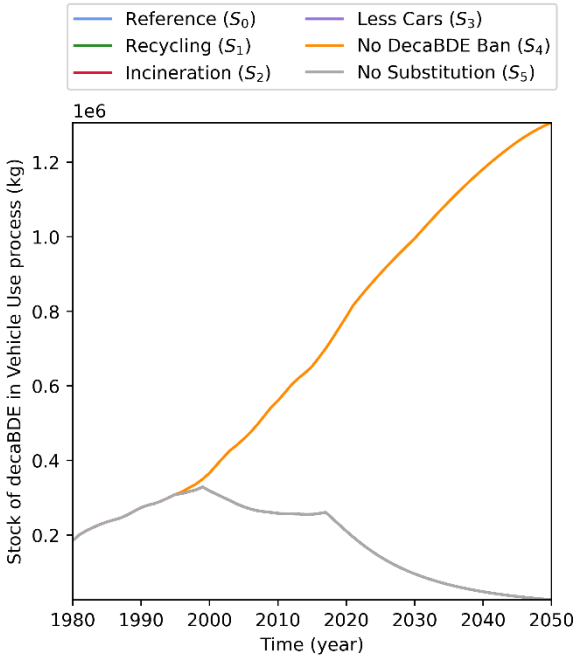

Figure 3.9. Stock of decaBDE in Vehicle Use process.

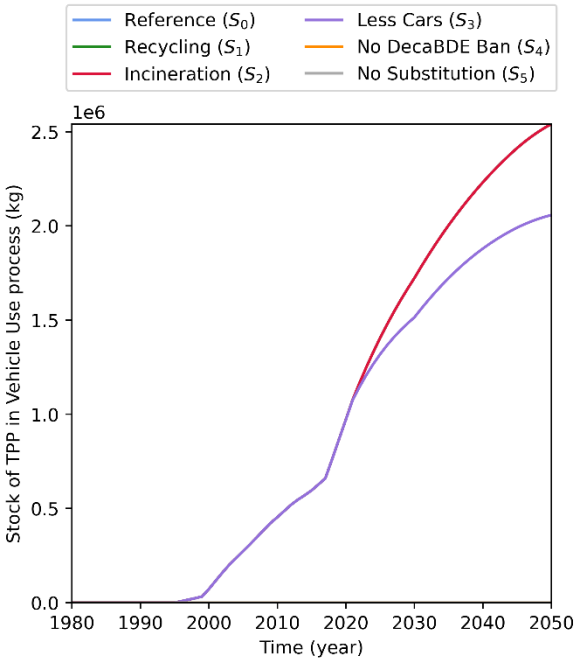

Figure 3.10. Stock of TPP in Vehicle Use process.

In Figure 3.9, the grey curve represents the phasing out of decaBDE, which begins around 2000. This is the case for all scenarios, except the No DecaBDE Ban (S_4) scenario, in which the use of decaBDE is still allowed. In S_0, S_1, S_2, S_3, and S_4, the share of decaBDE in new vehicle plastics slowly decreases over time – reaching 0 in 2018. This is not immediately visible in Figure 3.9: due to the car lifespans, decaBDE is still present in the vehicle stock until later. The spikes seen in the years 2000 and 2002 in the grey curve can be explained by the change in decaBDE concentration. The TPP stock (Figure 3.10) has a similar shape to the plastics stock, the red curve representing the TPP stock for the Reference (S_0), Recycling (S_1), and Incineration (S_2) scenarios, and the purple curve the Less Cars (S_3) scenario. In these four, the use of TPP began in the late 90's, and slowly increased over time as the use of decaBDE decreased. S_3, the Less Cars scenario, diverges from the other three in 2023 – as the car fleet data of the scenarios changes. In the case of the No DecaBDE Ban (S_4) and No Replacement (S_5) scenarios, the stock of TPP is 0 as in these scenarios TPP is not used in newly produced plastic car parts.

In Figure 3.11 and Figure 3.12, the obtained total emissions of decaBDE and TPP to the atmosphere are presented. When it comes to TPP, the emissions are directly proportional to the stock of TPP in the Vehicle Use phase. The TPP emission factor of the Vehicle Use process is the highest one from all the processes (including decaBDE EFs), which can be attributed to TPP's high volatility (Pawlowski & Schartel, 2007).

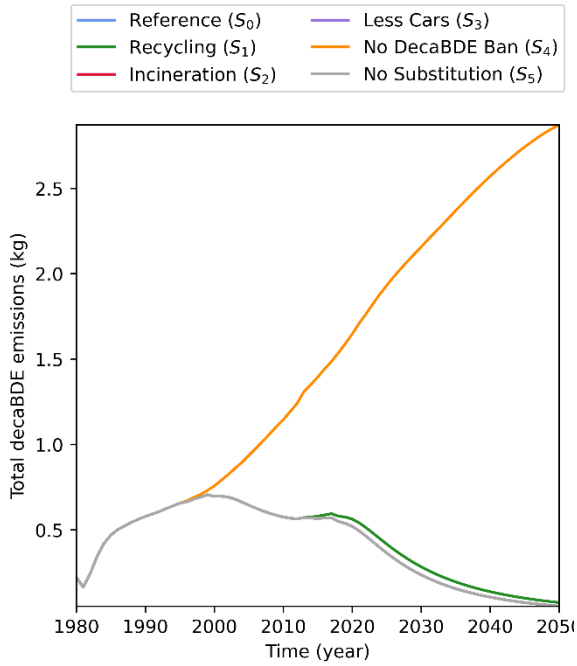

Figure 3.11. Total decaBDE emissions to the atmosphere for all scenarios.

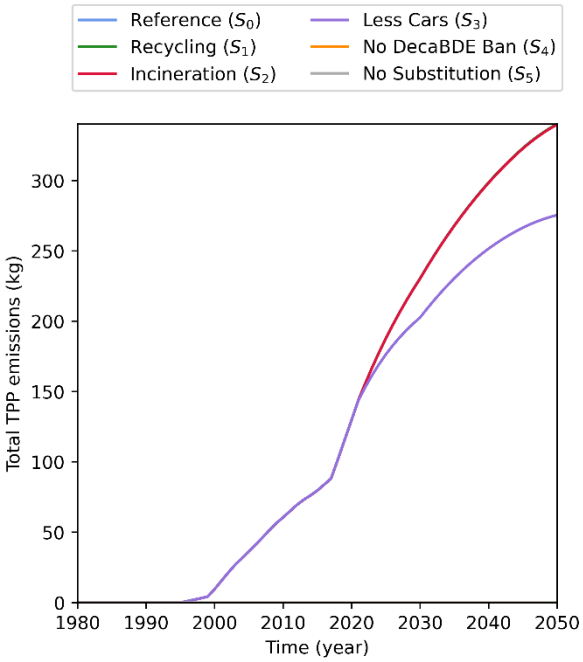

Figure 3.12. Total TPP emissions to the atmosphere for all scenarios.

The big discrepancy between total TPP and decaBDE emissions may be traced back to two main factors: TPP's high volatility, evidenced in its comparatively higher emission factors, and the increasing replacement of decaBDE with TPP. The first factor, however, seems to have the biggest weight, as in no point in time, for any scenario, were ever decaBDE emissions higher than TPP emissions, even though more quantities of decaBDE per vehicle were used in earlier years.

In the case of decaBDE (Figure 3.11), it is apparent that from 2013 onwards, the Recycling scenario (S_1) has slightly higher emissions than the Reference (S_0), Incineration (S2), and No Substitution (S_5) scenarios. This is because the emission factor of decaBDE the recycling process is higher than that of the Incineration process – which results in more decaBDE emissions to the atmosphere. Additionally, according to these results, the No DecaBDE Ban (S_4) scenario is by far the one with the highest decaBDE emissions to the Atmosphere, from the year 2000 onwards. This shows how much emissions are saved by implementing the ban.

Figure 3.13 and Figure 3.14 show the decaBDE emissions to the atmosphere per process, for the Reference (S_0) scenario, and the Recycling (S_1) scenario, respectively. The difference between these two figures is evident: the curve of emissions from the recycling process. As suspected, starting from 2012, the decaBDE emissions to the atmosphere originating from the recycling process in the Recycling scenario change with respect to the Reference scenario. This change in emissions is ultimately what makes the Recycling scenario distinguish itself from the others in Figure 3.11

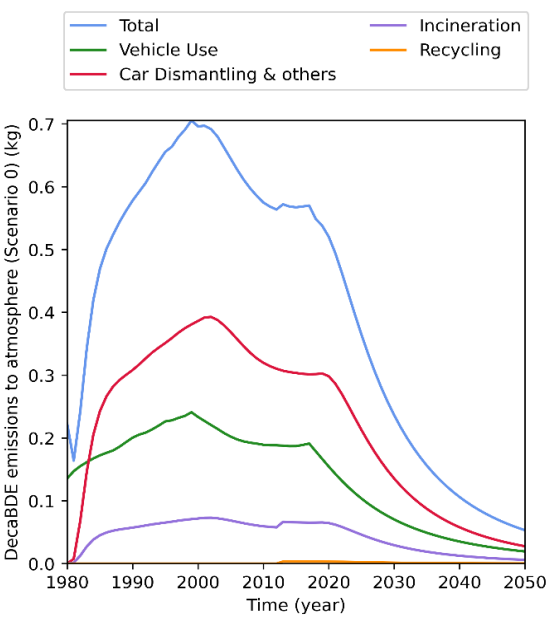

Figure 3.13. DecaBDE emissions to atmosphere per process for the Reference scenario (S0)

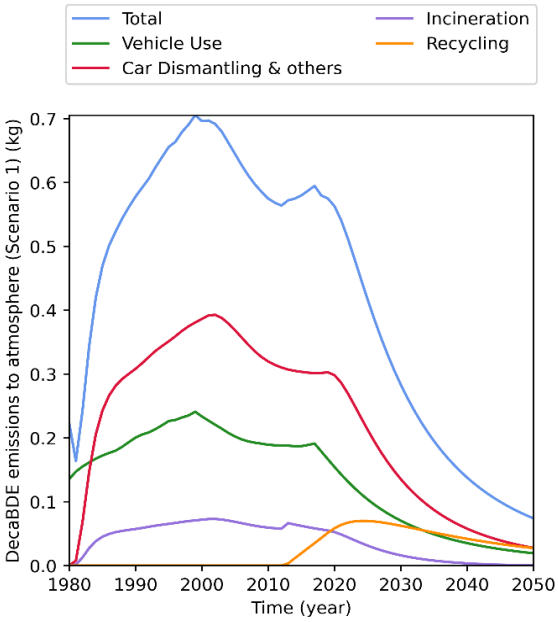

Figure 3.14. DecaBDE emissions to atmosphere per process for the Recycling scenario (S1).

A similar analysis can be done for TPP emissions (Figure 3.16 and Figure 3.15). In this case, there is no difference between the scenarios. This is because the TPP emitted during the recycling proces is neglible compared to the TPP emitted during the Vehicle Use process, according to the emission factors used in this study. This is confirmed by comparing the values in Table 3.1.

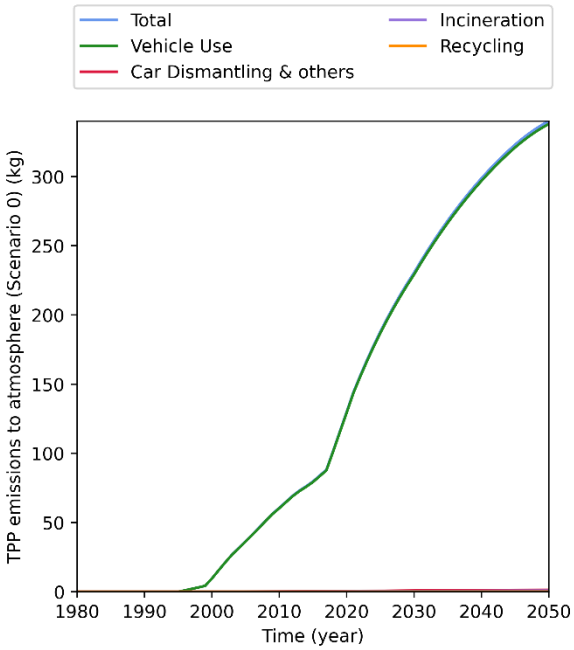

Figure 3.16. TPP emissions to atmosphere per process for the Reference scenario (S0).

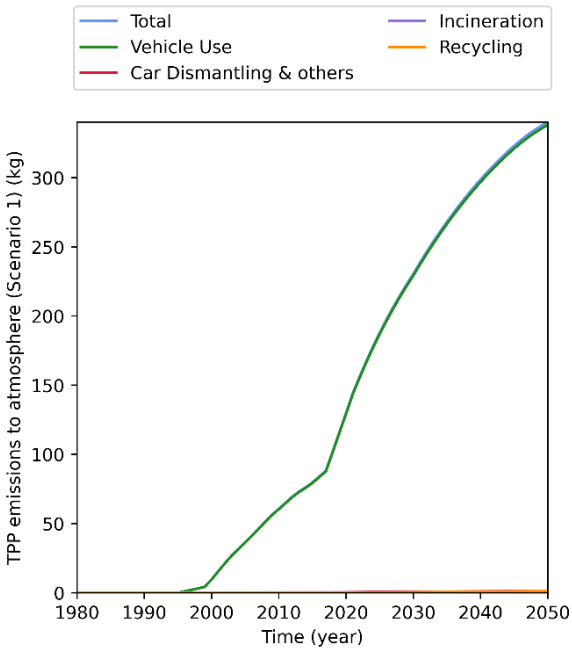

Figure 3.15. TPP emissions to atmosphere per process for the Recycling scenario (S1).

Table 3.1. Total emissions and TPP emissions for four scenarios for the year 2050.

Variable	Reference scenario (S0)	Recycling scenario (S1)	Incineration scenario (S2)	Sustainable mobility scenario (S3)
Total TPP emissions (kg)	444.6	444.7	444.1	382.6
TPP emissions from the Vehicle Use process (kg)	441.9	441.9	441.9	380.3

3.3. Life Cycle Impact Assessment

The LCIA aims to answer the fifth and final sub-question of this study:

5. *How can the flame retardant flow values be translated into health and climate impacts?*

To answer this question, environmental impacts were analyzed at the midpoint and endpoint level.

3.3.1. Midpoint level results

The human carcinogenic toxicity, human non-carcinogenic toxicity, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, and global warming impacts are shown in Figure 3.17, Figure 3.18, Figure 3.19, Figure 3.20, Figure 3.21, and Figure 3.22. When it comes to the toxicity impacts, the shape of the curves follows the shape of the emissions. In the case of the human-related toxicity impacts, the graphs follow the shape of the decaBDE total emissions to the atmosphere, and in the case of the ecosystem-related toxicity impacts, the graphs follow the shape of the TPP total emissions to the atmosphere. This is due to the fact that toxicity-related CFs were only available for decaBDE and ecosystem-related CFs were only available for TPP.

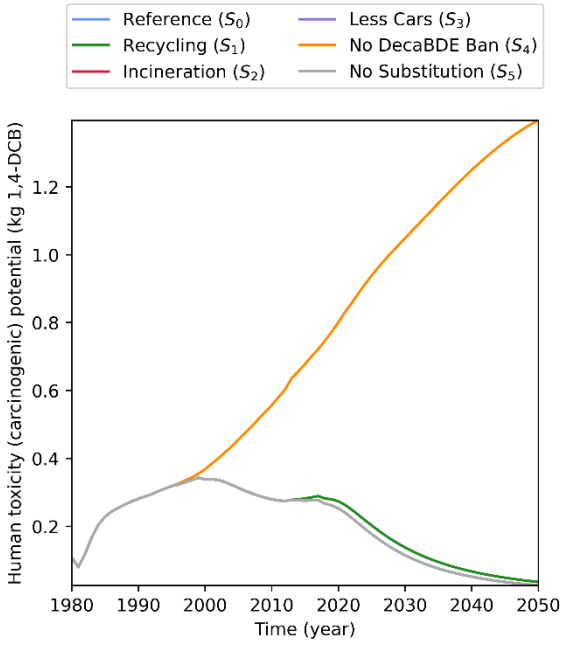

Figure 3.17. Human toxicity carcinogenic potential impacts.

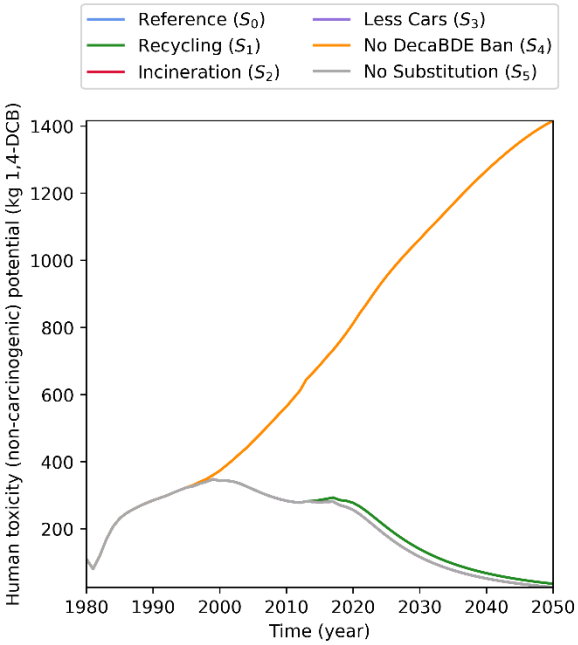

Figure 3.18. Human toxicity non carcinogenic potential impacts.

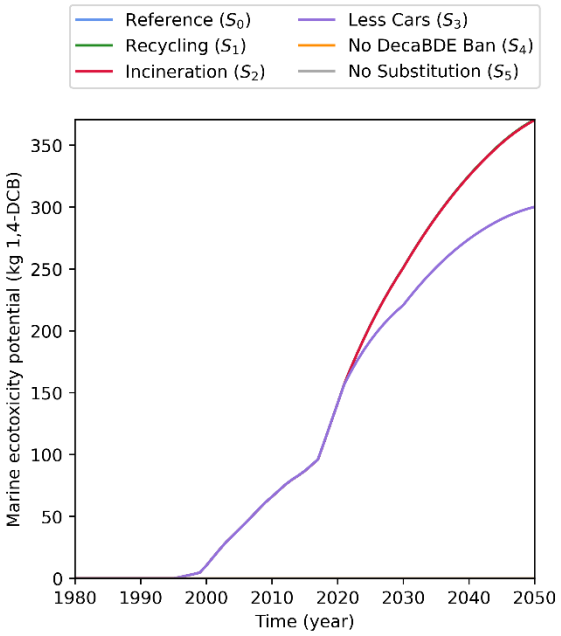

Figure 3.19. Marine ecotoxicity potential impacts.

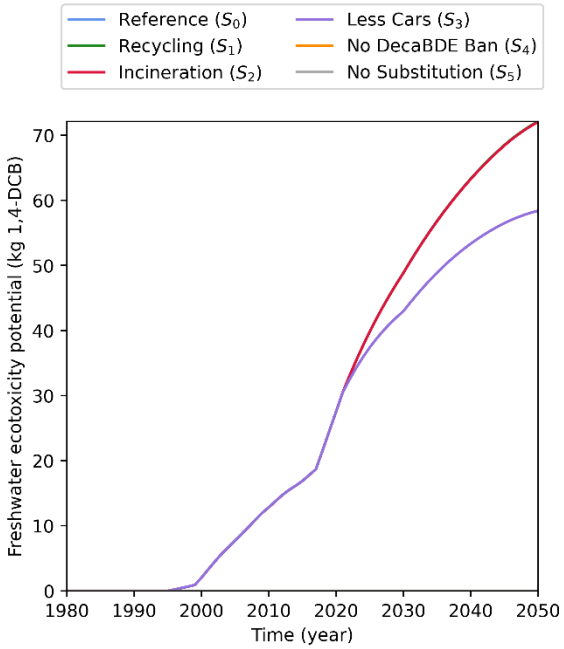

Figure 3.20. Freshwater ecotoxicity potential impacts.

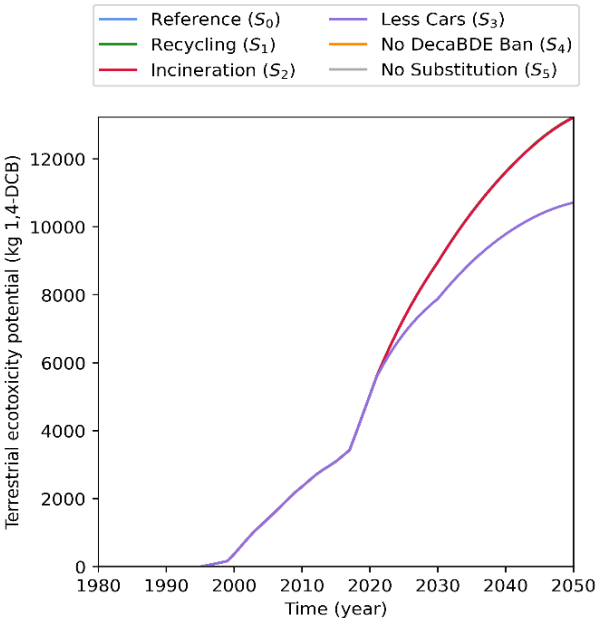

Figure 3.21. Terrestrial ecotoxicity potential impacts.

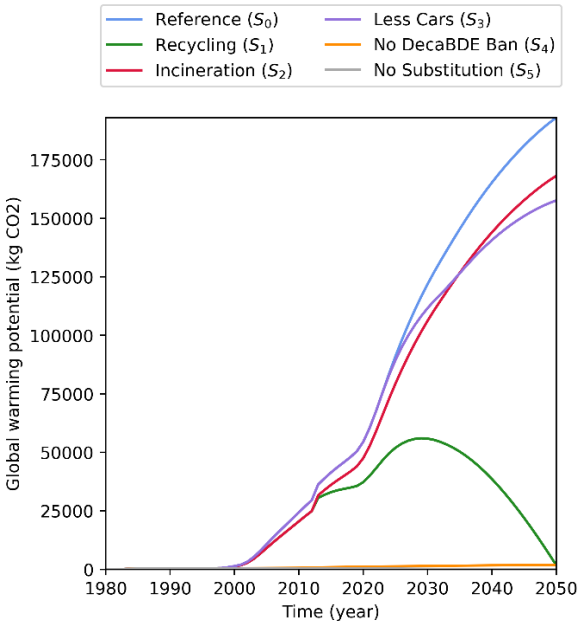

Figure 3.22. Global warming potential impacts.

In Figure 3.22, the global warming impacts are shown. These are the kilograms of CO2 assumed to be released to the atmosphere once the flame retardant is incinerated. In the No DecaBDE Ban (S_4) and No Substitution (S_5) scenarios, where decaBDE is the sole flame retardant being used in plastic vehicles, the global warming impacts are very close to 0. This is mainly because of the chemical formula of the substance – less CO2 is formed per kg of decaBDE compared to TPP. As expected, the Incineration (S_2) scenario is the one with the highest CO2 emissions, as it is the scenario in which the flows of plastics (and TPP and decaBDE) that are a sent to incineration are the largest. Moreover, it is possible to see that the global warming impacts in the Recycling (S_1) scenario start decreasing around 2030 – this is the turning point where more plastics (and decaBDE and TPP) are directed to recycling rather than incineration.

To conclude, when it comes to the human toxicity-related impact categories the No DecaBDE Ban (S_4) scenario is the worst performing one, and Reference (S_0), Incineration (S_2), and Less Cars (S_3) and No Substitution (S_5) the best ones. When it comes to ecosystem-related impact categories, the Reference (S_0), Recycling (S_1), and Incineration (S_2) scenarios score the highest, and No DecaBDE Ban (S_4) and No Substitution (S_5) the lowest. Lastly, when it comes to the global warming category, the Incineration (S_2) scenario scores the highest impacts, while the No Substitution (S_5) scenario the lowest. Therefore, there is no scenario that is the clear best performer in all categories.

3.3.2. End-point results

The endpoint results are hopefully able to give a clear picture of the environmental performance of each scenario. The total damage to human health and damage to ecosystems results over time are shown in Figure 3.23 and Figure 3.24. The results have a similar shape to that of the global warming impacts. This suggests that the global warming impacts have a heavier influence in the damage to human health and damage to ecosystems results than the toxicity related impacts. Because of this, the Incineration (S_2) scenario is the one with the highest damage to both human health and ecosystems.

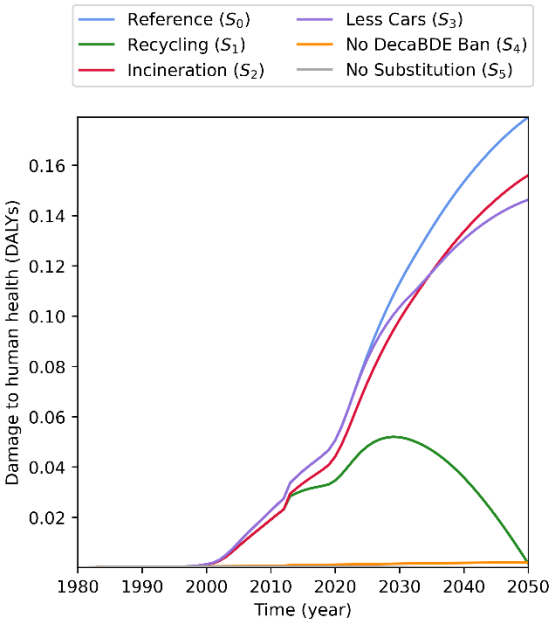

Figure 3.23. Damage to human health per year for all scenarios.

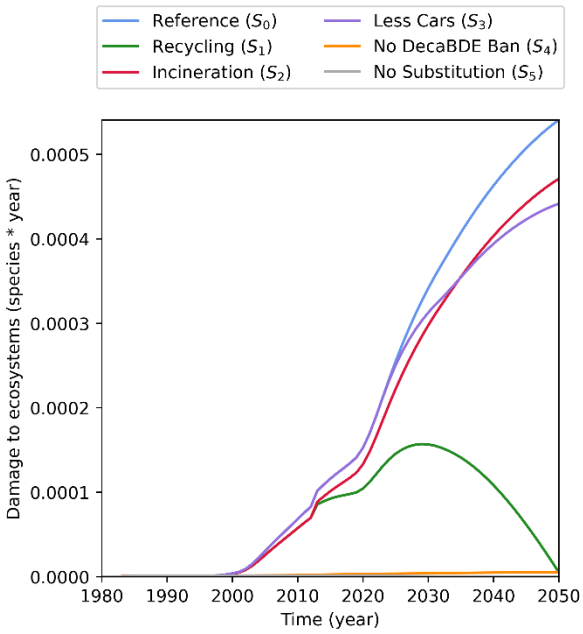

Figure 3.24. Damage to ecosystems per year for all scenarios.

In Figure 3.25 and Figure 3.26 the cumulated damages to human health and damage to ecosystems from 1980 until 2050 are presented. These are consistent with previous findings – and they indicate that the Incineration (S_2) scenario is the one with the most damage to both human health and ecosystems, closely followed by the Reference (S_0) scenario. The best performing scenarios are the No Substitution (S_5) scenario, closely followed by the No DecaBDE Ban (S_4) scenario.

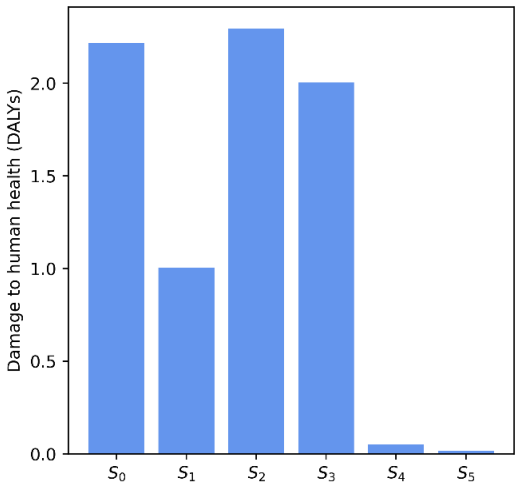

Figure 3.25. Damage to human health per scenario, cumulated form 1980 until 2050.

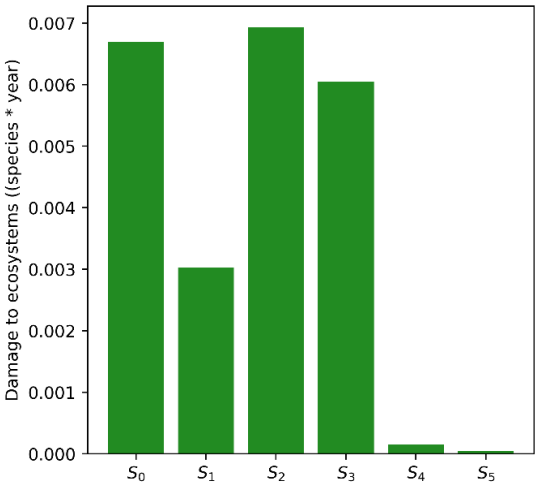

Figure 3.26. Damage to ecosystems per scenario, cumulated form 1980 until 2050.

By excluding the global warming-related impacts from damage to human health and damage to ecosystems, additional insights are gained. This way, it is possible to see the damage to human health due to human toxicity-related impacts, and the damage to ecosystems due to ecosystem-toxicity impacts. In Figure 3.27 and Figure 3.28, the resulting damage to human health and damage to ecosystems from 1980 until 2050, excluding global warming-related impacts, are presented.

Concerning human health, the No DecaBDE Ban (S_4) scenario is responsible for the most damages. The No DecaBDE Ban (S_4) scenario is characterized by its constant use of decaBDE, which is the substance responsible for the effects in human health due to its human toxicity potential. The rest of the scenarios have similar levels of damage to human health. Though almost imperceptible, the Recycling (S_1) scenario comes second.

Concerning ecosystems, the first three scenarios, Reference (S_0), Recycling (S_1), and Incineration (S_2), followed by the Less Cars (S_3) scenario, are the ones causing the biggest ecosystem toxicity-related damages. This is consistent with the ecosystem toxicity findings described earlier, in Figure 3.19, Figure 3.20, and Figure 3.21. The impacts to the ecosystem can be attributed to the use of TPP in vehicle plastics. Also, the damages to ecosystem in the No DecaBDE Ban (S_4) and No Substitution (S_5) scenarios is 0, as no TPP is used, which is responsible for the impacts in the ecosystem.

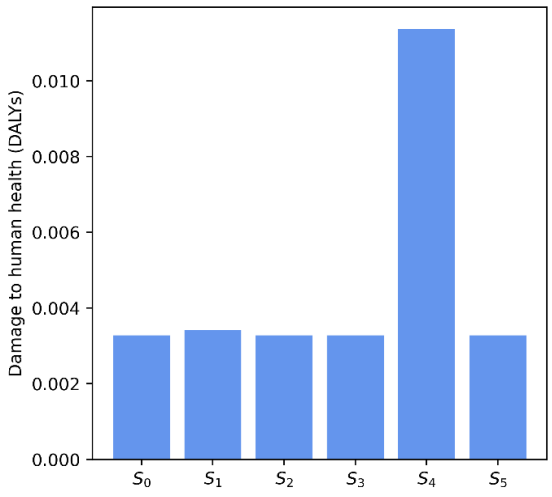

Figure 3.27. Cumulated damage to human health (1980-2050), excluding Global Warming-related impacts.

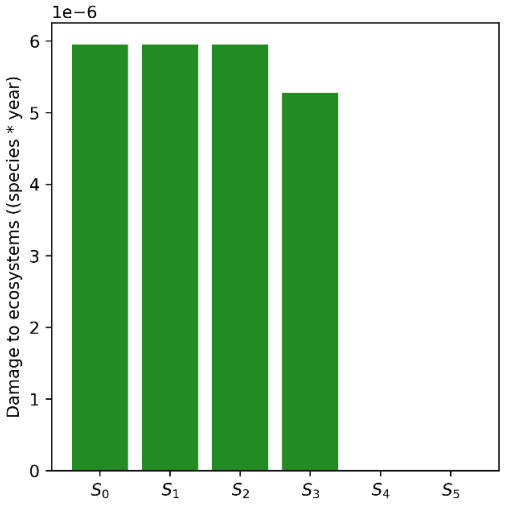

Figure 3.28. Cumulated damage to ecosystems (1980-2050), excluding Global Warming-related impacts.

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4. Discussion

This chapter discusses the methodology and main findings of the study. Method limitations and choices are covered in section 4.1 and the main results are discussed in section 4.2. Finally, in section 4.3, the study is placed into a wider context.

4.1. Discussion on methodology

4.1.1. Method limitations

The static MFA model does not include time as a variable. Therefore, the flow delay between the Vehicle Use process and Car dismantling and exports is not represented. The static model also assumed that the distribution of the flows occurs instantly (within 2019). Hence, the magnitude of the EOL flow values do not actually reflect the values of 2019, but the ones in 18 years (lifespan of a car) presuming that the process transfer coefficients remain the same. The static nature of the model also has an effect in the number of decaBDE emissions. Due to its static quality, the model is not able to account for the decaBDE emissions resulting from the plastic vehicle stock. This results in an underrepresentation of the emissions originating from this process. Such limitations are addressed in the layered dynamic MFA.

When conducting the LCIA, the focus was on a select number of categories: human toxicity, (carcinogenic), human toxicity (non-carcinogenic), freshwater ecotoxicity, terrestrial ecotoxicity, marine ecotoxicity, and global warming. While these give a good overview of the overall environmental performance, it is not a complete representation of it. Therefore, there is an underrepresentation of the environmental performance of the scenarios.

4.1.2. Methodological and modelling choices

In the static and dynamic layered MFAs, the vehicle production process was kept out of both the decaBDE and plastics system boundaries. Also, emissions associated to flame retardant production are excluded, and it is possible that said process is an important source of emissions. To counteract this, the vehicle production decaBDE and TPP emissions were considered to calculate the total emissions in the dynamic MFA model. However, their inclusion did not significantly change the results.

To define the system of the dynamic MFA, a simplification of the system resulting from the layered static MFA was used. This simplification comes at the cost of possibly overseeing emissions of flame retardant in some processes and losing detail on the metabolism of the system. For example, three end-of-life processes (blast furnace, fluidized bed cement kilns, and incineration) were joined into

one, but the emission factor of incineration was used for the conjoined process. Additionally, in the layered dynamic MFA, it is assumed that all plastics that are recycled are used again in new passenger vehicles, replacing the production of new plastics and flame retardants. This is an important assumption, as this situation might not precisely reflect the reality.

Lastly, in the LCIA, it is assumed that all carbon atoms in the flame retardants form carbon dioxide when the flame retardant goes through the incineration process. This assumption was chosen because there were no characterization factors for the global warming impact category for the flame retardants and should be kept in mind when interpreting the results of this study.

4.1.3. Data choices

To quantify the decaBDE in the plastics exiting the Post-Shredder Technology process in the static MFA, concentrations of decaBDE in these plastic flows was used. However, these concentrations were based on sample measurements, adding a certain level of uncertainty to the study.

Moreover, due to lack of data for emission factors of decaBDE for the Blast furnace and Fluidized bed cement kilns processes, the emission factor of the waste incineration process was used as a proxy. This has an influence in the resulting number of emissions and may not reflect an accurate representation of reality. Ideally, process-specific emission factors for these processes should be used. The data for emission factors for TPP was especially scarce. Where no emission factor was available, the same value as decaBDE was used. Additionally, it was assumed that the emission factors remain constant over time – while it is likely that they change according to improvements in technology when it comes to industrial processes.

Additionally, the substitution ratio by which decaBDE is replaced by TPP as decaBDE is gradually phased out was based on an approximation of singular cases presented in scientific literature. In Appendix E, a sensitivity analysis is carried out which evaluates the end point results after assuming a variation in the values of this substitution ratio.

4.1.4. Selection of decaBDE alternative

The choice of TPP as the decaBDE alternative was based, among other things, on the sufficient data availability to carry out this study. However, there are other commercially available existing alternatives to decaBDE that may have had less health and climate impacts than TPP. Therefore, this selection has a significant influence in the results.

4.2. Discussion on results

4.2.1. Territorial-based versus consumption-based approach

The scope of this study was limited to the geographic region of the Netherlands – a territorial approach was taken. Consequently, only the material flows, processes and emissions occurring physically in the Dutch territory were considered (except for the flame retardant emissions associated to the vehicle production process). By adopting this approach, some relevant flows that are indirectly associated to the Dutch consumption of passenger vehicles are excluded from the scope. These emission flows include the flame retardant emissions occurring during flame retardant production

for plastics that end up being added to Dutch passenger vehicles. It is sensible to think that a significant amount of flame retardant emissions would be associated to this process. Ultimately, a consumption-based approach would be able to uncover a more detailed account of what the climate and health impacts according to different circular strategies really are. Limiting the system scope to the Dutch territory likely results in underestimated impacts.

Additionally, other emissions that are left out from this study's system are the flame retardant emissions originating from the use, car dismantling processes, and end-of-life treatment processes of passenger vehicles exported from the Netherlands. Used Dutch cars are exported to low-income countries in East Europe and Africa, where their lifespan is elongated due to economic reasons (Held et al., 2021; Human Environment and Transport Inspectorate, 2020). Despite the foreign final consumption of these vehicles, it is questionable whether the flame retardant emissions associated to the dismantling and end-of-life treatment of the passenger vehicles should be allocated solely to the importing country. Especially given the magnitude of passenger vehicles exported from the Netherlands: almost half of all vehicles exit the use phase each year.

The debate between the usage of territorial-based versus consumption-based emission footprints is not new, especially when it comes to greenhouse gases (Knight & Schor, 2014; Peters, 2008). To achieve a CE in the Netherlands, changes are not only needed in the Netherlands, but worldwide. International cooperation is needed. And specifically, to achieve a fully CE of flame retardants in plastics of Dutch passenger vehicles, all processes and flows involved should be considered.

4.2.2. TPP as not an ideal, but worse, substitute to decaBDE

Over the past decades, there has been a cycle of banning hazardous flame retardants, resulting in implementation of substitutes that end up being found harmful years later (de Boer & Stapleton, 2019; Stapleton et al., 2012). One of the main findings from this study's results is that substituting decaBDE with the seemingly safer option TPP does not result in desirable outcomes. This discrepancy could be attributed to the methods used to compare the health and environmental performance of these substances. Usually, these are compared by means of environmental assessments (which deal with ecotoxicity and environmental fate data) and health assessments (which deal with toxicological data) (Stuer-Lauridsen et al., 2007). However, in this study, a partial life cycle impact assessment was carried out, which includes a global warming impact category among other toxicity related categories. This global warming impact category is an influence in the overall damage to human and ecosystem health. Due to lack of characterization factors of the substances for this category, an assumption was made to calculate its impacts. It was assumed that during the incineration process, all carbon atoms of the flame retardant would transform into carbon dioxide. Interestingly, the results showed that most of the human and ecosystem health impacts originated from the global warming impact category, while the toxicity and ecotoxicity impact categories had comparatively little influence. Thus, it might be possible that the inclusion of the global warming component is one of the reasons behind the inconsistency, making the use of TPP responsible for most climate and health impacts in this study. Furthermore, even if the global warming impacts were excluded from this study, using TPP would still not be decidedly better than using decaBDE due to the existing ecotoxicity impacts associated to it.

These results should be complemented with environmental risk assessments, including exposure and hazard assessments, as they are able to capture elements that a life cycle impact assessment cannot.

4.3. The big picture

Using the methodology presented in this study (a layered DMFA followed by an LCIA) is useful for the transition to a sustainable and healthy circular economy. Especially, when it comes to products that contain data-scarce chemicals which make the product difficult to handle at their end-of-life stages.

The results from this study show the different benefits and drawbacks of implementing different policies. Also, this study is able to also evaluate the substitution to seemingly less harmful chemical alternatives. This makes the methodology suitable to implement in comparable products, such as those in the electric and electronic equipment sector and the textiles sector, which can also contain hazardous chemicals

.

5. Conclusions

The goal of this thesis is to discover what the best strategies are to achieve a sustainable and healthy circular economy for flame retardants in Dutch passenger vehicles plastics. To do this, a layered static material flow analysis, layered dynamic material flow analysis, and a life cycle impact assessment were carried out. Through these, an evaluation of the environmental performance of different scenarios is performed, bringing the best practices to light.

The layered static material flow analysis answers research sub-questions 1 and 2, which concern the current situation of plastics and decaBDE in Dutch passenger vehicles. By mapping the relevant plastic and flame retardant flows, a general understanding of the system is achieved, and the following main insights are gained:

- Virtually all Dutch passenger vehicles are imported.
- Approximately half of all passenger vehicles that reach their end-of-life in the Netherlands are exported.
- The share of plastics sent to mechanical recycling is very low (5%), most plastics are sent to incineration, blast furnace, or fluidized bed cement kilns processes.
- Due to the restrictions associated to the substance in recycled products, the share of decaBDE that is sent to mechanical recycling is even lower (3%) than the share of plastics.
- The recycling of some plastic types (polypropylene and polystyrene) is hindered by economic factors, even though it is already technically feasible to mechanically recycle them.
- The shredding process is the highest decaBDE emitting process.

Subsequently, a layered dynamic material flow analysis was developed to forecast the plastic and flame retardant flows of passenger vehicles according to various scenarios using different circular strategies over time, answering sub-questions 3 and 4. This analysis also considers an alternative flame retardant that is assumed to substitute decaBDE, namely TPP. This analysis provides the following relevant insights:

- The stock of plastics in the passenger vehicle use phase grows over time; the evolution of the stock of decaBDE and TPP depends on the scenario.
- The magnitude of TPP emissions is higher than that of decaBDE for most scenarios.
- The car dismantling & others process (which includes the shredding process) is the largest contributor of decaBDE emissions, and the vehicle use process is the largest contributor of TPP emissions.

Lastly, a life cycle impact assessment is carried out to evaluate the climate and health impacts of the different scenarios, with a focus on human toxicity impacts, ecosystem toxicity impacts, and global warming impacts. The findings of this assessment are:

- Scenarios that replace decaBDE by TPP have worse climate and health impacts than those that do not, suggesting that this substitution is not a suitable option to transition to a sustainable and healthy circular economy.
- Recycling strategies and reducing cars per capita strategies proved to be among the best performing environmental strategies over time.

Ultimately, to achieve a sustainable and healthy circular economy for flame retardants in Dutch passenger vehicle plastics, the inclusion of recycling strategies, as well as the reduction of car ownership per capita, are key elements. Additionally, the flame retardant used in the passenger vehicles has a significant influence in the health and climate impacts – and it is therefore of high importance to choose it carefully.

According to the findings of this study, TPP is not an adequate substitute candidate to decaBDE. Though data limitations complicate the research of other flame retardant alternatives, this study indicates the importance of a deeper investigation of the climate and heath impacts of more flame retardants.

6. Recommendations

Based on the findings, the discussion and conclusion of this study, four recommendations for further research are given in section Figure 6.1, and three policy recommendations are presented in section 6.2.

6.1. Recommendations for further research

- *Implementing study methodology to analyse other relevant applications*

  The methodology framework used in this thesis could be used to study other applications of similar characteristics. Specifically, products made of materials that contain hazardous plastics. These could be, for example, products from the electronic and electric equipment sector or the textiles sector.

- *Evaluating the performance of other decaBDE alternatives*

  This study focused on TPP as the flame retardant alternative to decaBDE. However, there are other alternatives in the market that might yield different results than TPP and therefore provide further insights into the flame retardant transition to a circular economy.

- *Complementing study with environmental risk assessments*

  To get a better understanding of the environmental performance of the scenarios, additional approaches to evaluate the performance could be taken, such as those involved in environmental risk assessments.

- *Transition from a territorial-based approach to a consumption-based approach*

  As previously discussed, extending this study to a consumption-based approach could provide additional insights into the environmental performance of different circular strategies. This would require expanding the system boundaries of this research to include vehicle production, flame retardant production, plastics production, and any other processes related to plastics and flame retardants that may not take place in the Netherlands. In doing so, critical international trade flows would become visible, and conclusions regarding the transition to a sustainable and healthy circular economy at a greater scale could be drawn.

6.2. Policy recommendations

- *Encouraging plastic recycling in the Dutch automotive industry*

  Increasing the share of plastics that are recycled in the automotive sector and decreasing the quantity of plastics that are incinerated through the implementation of economic incentives for manufacturers that use recycled automotive plastics. By doing this, the global warming impacts associated to flame retardant incineration would be significantly reduced, also lowering the damage to human health and ecosystems.

- *Promote public transport use over personal passenger vehicle ownership*

  A future with less cars also has less flame retardant plastics associated to it. Incentivizing public transport instead of private car ownership would be an effective way to minimize potentially harmful flame retardant flows.

- *Funding research on safer flame retardant alternatives*

  To prevent regrettable substitutions of flame retardants, further research into the health and climate impacts of novel alternatives is necessary. This includes further research on characterization factors, emission factors, environmental risk assessments, and development of new alternatives.

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Appendix A Plastic waste treatment of passenger vehicles in the Netherlands

Vehicle use in the Netherlands

Approximately 400,000 passenger vehicles are registered per year in the Netherlands. In recent years, the main sources of passenger vehicles in the Netherlands are Germany, France, and Japan (BOVAG, 2022; Eurostat, 2022). The current and past stock of passenger vehicles in the Netherlands is available at CBS Open data StateLine. Yearly stock as well as age of the stock is available from 2000. There is also detailed data available for which types of cars are being used at BOVAG Rai, as well as average car weight.

End-of-life treatment of passenger vehicles

Once a vehicle reaches its end of life, it can go in one of the following directions: export, official car dismantling route, or unofficial car dismantling route. Roughly, in recent years, slightly more than half of all ELVs are exported and slightly less than half remain in the country.

Car dismantling

If the passenger vehicle is not exported, it will most likely be treated at one of one the 301 existing car dismantling facilities in the Netherlands. There, the car wreck is first depolluted: hazardous substances and liquids, including the battery, brake fluids and fuels are removed. Then, the functional parts of the car are removed, which are posteriorly sold as spare/second-hand parts (Leslie et al., 2013). The share of plastic car parts that fall under this category can be assumed to be 15% at maximum, considering as well that only car parts of young vehicles will be removed, as they are the only ones that have the potential to be resold due to compatibility issues (Personal communication, STIBA). Up until 2012, other parts were removed for material recycling: bumpers, hubcaps, grilles, polyurethane (PUR), back lights, seatbelts, and rubber. Since the installation of the 2013 post-shredding technology in 2013, this step has been eliminated from the car dismantling process (Personal communication, ARN).

Shredding

Next, the car wreck is transported to one of the 7 shredding companies in the Netherlands. The shredding process results in a fraction of metals (75%), fraction of ASR (20%) and fluids not removed

in the car dismantling stage (5%). The metal and ASR fractions are transported to a Post-Shredder Technology company to be treated further (Personal communication between ARN and RIVM).

Post-shredder technology process

In this installation, several separation techniques are applied (grinders, magnetic and eddy current separation, air and cyclone techniques, density separation using water baths) (Leslie et al., 2013). This installation has various output streams, including plastic fractions of different densities and fibers (Personal communication between ARN and RIVM):

- lower than 1.1 tons per cubic meter, which is directed to plastic recyclers
- between 1.1 and 1.3 tons per cubic meter, which is used for incineration with energy recovery, and used as agent reductor in blast furnaces
- higher than 1.3 tons per cubic meter, which is sent to incineration energy recovery
- fibers (half of which originate from PUR material), which are sent to incineration energy recovery and used as an additive in fluidized bed cement kilns

Plastic recycling

Currently, mechanical recycling is the only type of recycling performed for plastics originating from ASR. The recycling takes place in companies outside the Netherlands: most notably, Galloo Plastics, MBA Polymers, and MGG Polymers. ABS is the main plastic currently being recycled. According to their websites, depending on the polymer type, the recycled polymers can be destined to the manufacturing of non-aesthetic car parts including arm rests and pipes, and other non-vehicle related applications, such as LED lamps, printers, vacuum cleaners, washing machines, etc (Leslie et al., 2013).

However, numbers or shares regarding the distribution of the applications per polymer type are not publicly available. Therefore, at first glance, it is not possible to estimate how much of the recycled plastic ends up in a closed-loop (by being utilized in the automotive industry) or in an open-loop (by being utilized for other applications).

Appendix B Layered Static Material Flow Analysis

Plastics Layer

Fixed flow variables

A number of plastic flows are calculated to enter as fixed variables to the MFA model. To estimate the flow of plastics entering the Dutch passenger vehicle use-phase in 2017, data for the *newly registered passenger vehicles, average plastic content in a passenger vehicle* are used. The flow of plastics present in exported ELV, plastics in ELV treated via official dismantling routes, via unofficial dismantling routes, can be calculated in the same way, where the only changing variable is the number of vehicles concerning each flow (see Table 6.1).

Table 6.1. Values of the variables used to calculate flow inputs to static MFA of plastics.

Variable	Value	Source
Number of new vehicles ***(a)***	445217	
Number of exported vehicles ***(b)***	298055	(CBS, n.d.)
Number of EOL vehicles ***(c)***	213530	(CBS, n.d.)
Number of vehicles via unofficial channels ***(d)***	1673	(CBS, n.d.)
Average plastics in new vehicles in 2019 (kg) ***(e)***	208	Personal communication, JRCC (article to be published soon)

In Table 6.2, the resulting flows to be introduced as inputs to the static MFA are presented as well as the calculation carried out to produce the result.

Table 6.2. Flow inputs to static MFA of plastics.

Flow	*Value (tons)*	*STAN reference*	*Calculation/Assumption*

Plastics in new vehicles	92605.136	F1	F1 = a * e
Plastics in EOL vehicles	44414.24	F4	F4 = b * e
Plastics in EOL vehicles (unofficial route)	347.984	F5	F5 = c * e
Plastics in exported used cars	61995.44	F13	F6 = d * e

Transfer coefficient inputs

The transfer coefficient inputs are built according to insights of relevant organizations, literature and data provided by ARN. For most processes, these are shares that translate immediately into transfer coefficients. However, in the case of the Post-Shredder Technology Installation process, the transfer coefficients were calculated based on data provided by ARN. The data consisted of the material fractions of the ASR flow at the PST factory at ARN in 2017, and the polymer composition of the plastics contained in ASR flow. Moreover, the possible usage of each fraction of the ASR flow is given (see Figure 6.1 and Table 6.3).

ASR (ARN)	Gained Fraction (ARN 2017)		Bromine Content** [ppm] 2016/17 **measured at ARN, Tiel with XRF	Possible Usage	Recycling	Recovery
Bromine < 1.000 ppm	Fibres	30,4 %	527 (median) DecaBDE: 0 – 48 ppm (2018)	Additive for cement kilns / In future extrusion of waterfront facing sheet	•	•
	Minerals	20,6 %	197 (median)	Material for civil construction purposes (Clinker filler after incineration in cement kiln)	•	
	Plastics > 1.3 g/cm³	13,5 %	2.600 (median)	Waste incineration (energy recovery)		•
	Plastics < 1.1 g/cm³	8,2 %	86 (median) DecaBDE: 0 – 12 ppm (2018)	Injection moulding / extrusion	•	
	Rubber/Wood/plastics	8,0 %	42 (median)	Waste incineration (ernergy recovery)		•
	Misc. Heavy parts	5,7 %	Not measured	Recycling / energy recovery / landfill	•	• + landfill
	Plastics 1.1-1.3 g/cm³	5,2 %	2.220 (median)	Reduction agent for blast furnace	•	
	Dust	4,6 %	489 (median)	Waste incineration (ernergy recovery)		•
	Metals	3,8 %	Not measured	Metal Recycling	•	

Figure 6.1.Material fractions of automotive shredder residue at the Post-Shredder Technology factory of ARN in 2017. (Figure created by RIVM).

Table 6.3. Polymer fractions of the plastics found in the Automotive Shredder Residue flows at the Post-Shredder Technology installation in 2017.

Polymer fractions in the plastic ASR flows at PST ARN installation	
Polymer	Density (g/cm^3)

	<1.1	1.1-1.3	>1.3
ABS	0.25	0.174	0.001
Bakeliet	0	0	0.004
HDPE	0.041	0.043	0.084
Nylon 6 (PA6)	0.004	0.127	0.515
PC	0	0.179	0.044
PE	0.003	0	0
PET/PBT	0	0	0.075
PMMA	0	0.004	0.004
PEEU	0	0.001	0.015
POM	0	0.001	0.108
PP	0.451	0	0
PP talc	0	0.384	0.085
PPEs	0	0	0.008
PS	0.099	0.085	0.002
Rest	0.151	0	0
Other plastics	0	0.003	0.047

In Table 6.4 , the fractions of high-density plastics (density higher than 1.3 g/cm^3), mid-density plastics (density between 1.1 g/cm^3 than 1.3 g/cm^3), low-density plastics (density lower than 1.3 g/cm^3) and fibres are presented without considering all other materials in the ASR flow that are not relevant to this study (minerals, rubber, wood, heavy parts, dust, metals).

Table 6.4. Fractions of materials composing the ASR flow at the ARN PST installation in 2017, with and without considering materials other than plastics and fibres.

ASR	Fraction	Calculated fraction (plastics and fibres only)

High density plastics	0.135	0.2356
Mid density plastics	0.052	0.09075
Low density plastics	0.082	0.1431
Fibers	0.304	0.5305
Others (minerals, rubber, wood, heavy parts, dust, metals)	0.4216	n.a.

To finish building the transfer coefficients, the possible destinations of the material fractions presented in Figure 6.1 are considered, as well as the polymer composition of the plastics flow presented in Table 6.3. High-density plastics are exclusively sent to an incineration process, while mid-density plastics are exclusively sent to a blast furnace process. Low-density plastics are partly sent to a mechanical recycling process and partly incinerated. From a communication between ARN and RIVM, it is learnt that only ABS polymers are currently separated for mechanical recycling. This is because separating PP and PE is not financially sound at the moment. The rest of the low-density plastics fraction is sent to incineration. Lastly, it is assumed that the fibres flow is distributed equally among the waste incineration, blast furnace, and fluidized cement kilns processes. The resulting transfer coefficients that are inputs to the plastics static MFA can be found in Table 6.5.

Table 6.5. Transfer coefficient inputs to static MFA of plastics, per process.

Process & Flows	Transfer coefficient	STAN reference	Source	Calculation/Assumption
Car dismantling		P4		
Plastic in stripped car wreck (to Shredding)	0.94	F8		F8 = 1 – F7 – F6
Plastic in removable parts (to Mechanical recycling)	0.01	F7	Communication with STIBA	
Plastic in parts sold as second hand (to Vehicle use)	0.05	F6	Communication with STIBA	
ELV treatment (unofficial channels)		P3		

Plastics in car wrecks from unofficial channels (to Shredding)	1	F15		
Shredding		P6		
ASR (to Post-shredding technology)	1	F12		
Post-shredding technology		P7		
High density plastics (to Incineration)	0.2356	F9	ASR data provided by ARN (Figure 6.1), based on results shown in Table 6.4.	
Mid density plastics (to Blast furnace)	0.0908	F10		
Low density plastics (to Blast furnace)	0.1073	F26		
Low density plastics (to Mechanical recycling)	0.0358	F11		
Fibers (to Waste incineration)	0.1768	F21		
Fibers (to Blast furnace)	0.1768	F27		
Fibers (to Fluidized bed cement kilns)	0.1768	F17		
Mechanical recycling		P5		
Recycled plastic (to Vehicle Use)	0.85	F16	(Broeren et al., 2022)	
Disposed low-density ASR (to Waste incineration)	0.15	F22		

The final flow values for plastics are presented in Table 6.6.

Table 6.6. Quantified flows for the static MFA of plastics.

Flow name	Mass flow	Mass flow (calculated)

Plastics in new passenger vehicles	92605.136 t/a	92605.136 t/a
Mid-density ASR		3818.23142272 t/a
Low-density ASR		1506.0754948096 t/a
ASR		42097.3696 t/a
Plastics in exported used cars	61995.44 t/a	61995.44 t/a
Plastics in car wrecks from unofficial channels		347.984 t/a
Recycled plastic		1657.68521058816 t/a
Fibres		7444.7935216512 t/a
Fibers		7444.7935216512 t/a
Disposed low-density ASR		292.53268422144 t/a
Losses		17655.4664836326 t/a
Losses (1)		15783.566784128 t/a
Losses (2)		7444.7935216512 t/a
Low-density ASR		4520.5418397568 t/a
Fibers		7444793.5216512 kg/a
Plastic in EOL vehicles	44414.24 t/a	44414.24 t/a
Plastic in EOL vehicles (1)	347.984 t/a	347.984 t/a
Plastic in parts sold as second hand		2220.712 t/a
Plastic in removable parts		444.1424 t/a
Plastic in stripped car wreck		41749.3856 t/a
High density ASR		9918.14027776 t/a

DecaBDE Layer

Fixed flow variables

In the case of the decaBDE static MFA, the flow calculated flow inputs are those leaving the Post-Shredder Technology process. These flows have different decaBDE concentrations and must be calculated according to these. In this study, the decaBDE concentrations are based on empirical evidence, which consists of the measurements of decaBDE taken at the ARN PST installation factory in 2013 (Table 6.7). In Table 6.8, the resulting values are given for all flows exiting the Post Shredder Technology process, after multiplying the plastic flow value by the corresponding decaBDE concentration.

Table 6.7. Concentration of decaBDE in ASR samples at the PST installation factory of ARN in 2013 (Leslie et al. 2013; Leslie et al., 2018).

Type of plastics	Concentration of decaBDE (kg/kg) in ASR at ARN in 2013
High density plastics (c1)	0.0000055
Mid density plastics (c2)	0.0000055
Low density plastics (c3)	0.0000012
Fibers (c4)	0.00007

Table 6.8. Flow inputs to the decaBDE static MFA.

Flows (Plastics)	Value (kg of plastics)	Flows (DecaBDE)	Value (kg of decaBDE)	Calculation/Assumption
High density plastics (to Incineration)	9918140.0000	*DecaBDE in high density plastics (to Incineration)*	54.54977	h = a * c1
Mid density plastics (to Blast furnace)	3818231.0000	*DecaBDE in mid density plastics (to Blast furnace)*	21.0002705	i = b * c2
Low density plastics (to Blast furnace)	4520541.0000	*DecaBDE in low density plastics (to Blast furnace)*	54.246492	j = c * c3
Low density plastics (to Mechanical recycling)	1506075.0000	*DecaBDE in low density plastics (to Mechanical recycling)*	18.0729	k = d * c3

Fibers (to Waste incineration)	7444793.0000	*DecaBDE in fibers (to Waste incineration)*	521.13551	l = e * c4
Fibers (to Blast furnace)	7444793.0000	*DecaBDE in fibers (to Blast furnace)*	521.13551	m = f * c4
Fibers (to Fluidized bed cement kilns)	7444793.0000	*DecaBDE in fibers (to Fluidized bed cement kilns)*	521.13551	n = g * c4

Transfer coefficients

In contrast to the static MFA of plastics, the decaBDE static MFA also includes the emissions of the flame retardant to the atmosphere. To include the emissions, the emission factors are needed. The emission factors used in this study are presented in Table 6.9. It is assumed that the Blast Furnace process and the Fluidized bed cement kilns process have the same emission factors as Waste Incineration, due to lack of more specific data.

Table 6.9. Emission factors of decaBDE (Xue et al., 2017).

Life cycle phase	DecaBDE (kg/kg*year)
Use phase **(ef1)**	7.33E-07
Shredding, sorting **(ef2)**	4.30E-05
Recycling **(ef3)**	4.30E-05
Waste incineration **(ef4)**	9.40E-06

The transfer coefficients used in the decaBDE static MFA must be transformed to contemplate these emissions. To do this, the value of the emission factor is used as the transfer coefficient to the atmosphere, in all processes that have emissions to the environment. Lastly, the remaining share is divided among the remaining flows according to the transfer coefficients of the plastics static MFA. In the case of the Vehicle Use process, the transfer coefficients are calculated according to the flow input data of the plastics static MFA. The resulting transfer coefficients per process can be found in Table 6.10.

Table 6.10. Transfer coefficient inputs to the static MFA of decaBDE per process.

Process & Flows	Transfer coefficient	Process & Flow STAN reference	Calculation/Assumption
Vehicle Use		P2	

DecaBDE emissions (to Atmosphere)	0.000000733	D28	D28 = ef1
DecaBDE in plastics of exported vehicles (Export flow)	0.580711419	D13	D13 = (F13/(F13+F5+F4)*(1-D28)
DecaBDE in plastics of EOL vehicles (to Car dismantling)	0.416028281	D4	D4 = (F4/(F13+F5+F4)*(1-D28)
DecaBDE in plastics of EOL vehicles (to ELV treatment via unofficial channels)	0.003259567	D5	D5 = (F5/(F13+F5+F4)*(1-D28)
Car dismantling		P4	
DecaBDE in plastics of stripped car wreck (to Shredding)	0.94	D8	D8=F8 (Assuming no emissions occur during car dismantling – lack of data)
DecaBDE in plastic of removable parts (to Mechanical recycling)	0.01	D7	D7=F7 (Assuming no emissions occur during car dismantling – lack of data)
DecaBDE in plastics in parts sold as second hand (to Vehicle use)	0.05	D6	D6=F6 (Assuming no emissions occur during car dismantling – lack of data)
ELV treatment (unofficial channels)		P3	
DecaBDE in plastics of car wrecks from unofficial channels (to Shredding)	1	D15	
Shredding		P6	
DecaBDE emissions (to Atmosphere)	0.000043	D30	D30 = ef2

DecaBDE in ASR (to Post-shredding technology)	0.999957	D12	D12 = 1 – D30
Mechanical recycling		P5	
DecaBDE emissions (to Atmosphere)	0.000043	D34	D34 = ef3
DecaBDE in recycled plastic (to Vehicle Use)	0.84996345	D16	D16 = F16 * (1-D34)
DecaBDE in disposed low-density ASR (to Waste incineration)	0.14999355	D22	D22 = F22 * (1-D31)
Waste incineration		P9	
DecaBDE emissions (to Atmosphere)	0.0000094	D31	D31= ef4
Losses	0.9999906	D23	D23 = 1-D31
Blast furnace		P8	
DecaBDE emissions (to Atmosphere)	0.0000094	D32	D32 = ef4
Losses	0.9999906	D24	D24 = 1-D32
Fluidized bed cement kilns		P10	
DecaBDE emissions (to Atmosphere)	0.0000094	D33	D33 = ef4
Losses	0.9999906	D25	D25 = 1- D33

The final flow values for decaBDE are presented in Table 6.11.

Table 6.11. Quantified decaBDE flows.

Flow	Flow name	Mass flow	Mass flow (calculated)
D1	DecaBDE in plastics of new		4218611.549189 g/a

	passenger vehicles		
D10 (1)	DecaBDE in mid-density ASR	21000 g/a	21000 g/a
D11 (1)	DecaBDE in low-density ASR	18000 g/a	18000 g/a
D12 (1)	DecaBDE in ASR		1711165.1 g/a
D13 (1)	DecaBDE in plastics of exported used cars		2519960.37544873 g/a
D15 (1)	DecaBDE in plastics of car wrecks from unofficial channels		14230.4510082945 g/a
D16 (1)	DecaBDE in recycled plastic		30643.9694592121 g/a
D17 (1)	DecaBDE in fibres	521130 g/a	521130 g/a
D21 (1)	DecaBDE In fibers	521125.1 g/a	521125.1 g/a
D22 (1)	DecaBDE in disposed low-density ASR		5407.75931633154 g/a
D23 (1)	DecaBDE in losses		581067.397231454 g/a
D24 (1)	DecaBDE in losses (1)		596364.394122 g/a
D25 (1)	DecaBDE in losses (2)		521125.101378 g/a
D26 (1)	DecaBDE in low-density ASR (1)	54240 g/a	54240 g/a
D27 (1)	DecaBDE in fibers (1)	521130 g/a	521130 g/a

D28	DecaBDE emissions	3.18086956294802 g/a
D30	DecaBDE emissions	73.5832633803254 g/a
D31	DecaBDE emissions	5.46208487757352 g/a
D32	DecaBDE emissions	5.605878 g/a
D33	DecaBDE emissions	4.898622 g/a
D34	DecaBDE emissions	1.55029099986288 g/a
D4 (1)	DecaBDE in plastic of EOL vehicles	1805327.90665435 g/a
D5 (1)	DecaBDE in plastic of EOL vehicles (1)	14230.4510082945 g/a
D6 (1)	DecaBDE in plastic in parts sold as second hand	90266.3953327173 g/a
D7 (1)	DecaBDE in pastic of removable parts	18053.2790665435 g/a
D8 (1)	DecaBDE in plastic of stripped car wreck	1697008.23225509 g/a
D9 (1)	DecaBDE in high-density ASR	54540 g/a 54540 g/a

Appendix C Layered Dynamic Material Flow Analysis

Data collection for layered DMFA model

In this section, the data collection, and assumptions for the layered DMFA model are presented.

Average share of plastics in vehicles

The average share of plastics and average weight of plastics from 1980 through 2050 was collected from various sources (Table 6.12). In the model, a linear interpolation was carried out to fill in the years for which there was no data found.

These variables are the same for all scenarios implemented in the layered DMFA.

Table 6.12. Average vehicle weight, plastic weight and plastic share of passenger vehicles for different years.

Year	Average vehicle weight	Plastic weight	Plastic share	Source
1980	1180	106.2	0.09	(Kearney, n.d.)
2000	1100	132	0.12	Personal communication, JRCC (Current ELV) (article to be soon published)
2020	1300	208	0.16	Personal communication, JRCC JRC (Future ELV) (article to be soon published)
2050	900	300	0.33	(Becque & Sharp, 2020)

Car fleet in the Netherlands

Table 6.13. Values for car fleet in the Netherlands 1980 - 2050.

	1980 – 2019	2020 – 2050
Car fleet in The Netherlands for all scenarios except S3	Historical data (CBS, 2019)	2030: 9,1000,000 passenger vehicles 2050: 10,400,000 passenger vehicles Values taken from the WLO Hoog scenario (CPB/PBL, 2015). A linear interpolation is carried out to fill in the years for which there is no indication value available.
Car fleet in The Netherlands for S3	Historical data (CBS, 2019)	2030: 8,229,000 passenger vehicles 2050: 8,539,000 passenger vehicles Values taken from WLO Laag scenario. A linear interpolation is carried out to fill in the years for which there is no indication value available.

Average FR share in passenger vehicle plastics

DecaBDE

Over the years, different concentrations of decaBDE have been reported in passenger vehicles. The concentration per vehicle depends on the year produced, model, and polymers used in the vehicle, among other things. The concentrations used in S0, S1, S2, S3, and S5 of the model are the ones suggested by UNEP – as a result of their literature review on decaBDE concentrations in passenger vehicles, and due to lack of Netherlands-specific data over time (UNEP, 2021). S4 assumes that no ban is imposed on decaBDE – therefore the concentration of the substance in ASR plastics is assumed to remain constant over time (Table 6.14). The value of the decaBDE concentration in ASR plastics before 1996 is assumed to be the concentration of decaBDE needed to satisfy the fire safety requirements – as it decreases, it is assumed that it is replaced with another FR.

Table 6.14. DecaBDE concentrations per kg of ASR plastics used in the model.

Scenario	Time period	DecaBDE concentration in ASR plastics (kg/kg)
S0, S1, S2, S3, S5	Before 1996	0.000406
	Before 1999	0.000335
	Before 2018	0.000120
	After 2018	0
S4	Before 1996	0.000406
	Before 1999	0.000406
	Before 2018	0.000406
	After 2018	0.000406

TPP

There is scattered data regarding TPP concentration in polymers – and no data on TPP concentration in a passenger vehicle. Therefore, significant assumptions are made to determine the concentration of TPP in passenger vehicle plastics in this study. For the same polymer, decaBDE and TPP are loaded at different weights – more quantity of TPP than decaBDE is needed to comply with fire safety requirements (EPA, 2014). While decaBDE is often applied at 10% by weight, TPP is applied at 15%. Therefore, it is assumed that 1.5 kg of TPP are needed for every kg of decaBDE. And, as the concentration of decaBDE decreases, the concentration of TPP increases according to the previously mentioned 1.5 ratio. This applies for scenarios S0, S1, S2, and S3. In the case of S4 and S5, it is assumed that TPP is not used.

Table 6.15. TPP concentration per kg of ASR plastics in passenger vehicles used in the model.

Scenario	Time period	TPP concentration in ASR plastics (kg/kg)
S0, S1, S2, S3	Before 1996	0
	Before 1999	0.0001065
	Before 2018	0.000429
	After 2018	0.000812
S4, S5	Before 1996	0

	Before 1999	0
	Before 2018	0
	After 2018	0

Overview of evolution of transfer coefficients over time per scenario

The layered dynamic MFA uses transfer coefficients (like in the static MFA) – however, in the case of the dynamic MFA, these change over time, according to each scenario. The main aspects influencing the transfer coefficient changes are mentioned in Table 6.16. The resulting transfer coefficients per material and per scenario can be found in the links provided in Appendix D.

Table 6.16. Scenario assumptions affecting transfer coefficients.

Scenario	Assumptions affecting transfer coefficients
Reference (S0)	In 2012, the first PST factory was installed in the Netherlands, which allowed for the separation of plastics to send to mechanical recycling passenger vehicle plastics (Leslie et al., 2016). It is assumed that mechanical recycling from ASR plastics began in 2012. The share (or transfer coefficient) of mechanically recycled plastics was set to that obtained in the static MFA for 2019, and kept constant until 2050, assuming no major changes. Also, from 2012 onwards, it is assumed that plastic parts are no longer dismantled from cars for the purpose mechanical recycling, and that all plastics in the car go through the shredding and PST process.
Recycling (S1)	Up until 2012 – the transfer coefficients are the same as in S0. From 2013, the share of mechanically recycled plastics increases gradually, finally reaching 1 in 2050 (and the share of incinerated plastics decreases in an inversely proportional manner).
Incineration (S2)	Up until 2012 – the transfer coefficients are the same as in S0. From 2013 onwards, the share of incinerated plastics increases gradually, finally reaching 1 in 2050 (and the share of mechanically recycled plastics decreases in an inversely proportional way).
Sustainable Mobility (S3)	Same transfer coefficients as S0.
No Ban (S4)	Same transfer coefficients as S0.

No replacement (S5)	Same transfer coefficients as S0.

The input data, which includes the transfer coefficients, plastic shares, decaBDE shares, and TPP shares used in every scenario is available in a publicly accessible GitHub repository presented in Appendix D.

Appendix D Python model

The python model created for this study can be found in the following repository:

https://github.com/mila-gv/layered-dynamic-mfa

The scenario input data:

https://github.com/mila-gv/layered-dynamic-mfa/tree/main/data

The code for the layered dynamic MFA:

https://github.com/mila-gv/layered-dynamic-mfa/blob/main/dmfa/layered_dmfa.py

https://github.com/mila-gv/layered-dynamic-mfa/blob/main/dmfa/dmfa.py

The calculation of LCIA impacts:

https://github.com/mila-gv/layered-dynamic-mfa/blob/main/dmfa/impacts.py

Appendix E Sensitivity analysis of the decaBDE to TPP substitution ratio

This sensitivity analysis aims to analyse how variations of the substitution factor value influence the damage to human health and damage to ecosystems cumulated from 1980 until 2050. The main reason behind performing this analysis is the uncertainty associated to the substitution factor choice.

The model is run for three substitution factor values: 1.0, 1.5 (the one used in the main study), and 2.0. As observed in Figure 6.2 and Figure 6.3, the total damage to human health and ecosystems scales linearly according to the substitution factors and the magnitude of the TPP flow headed into incineration. This showcases once more the influence that the TPP flow headed to incineration has on the endpoint impacts. As shown in both figures, the scenarios where this flow is the lowest also have the lower overall impacts.

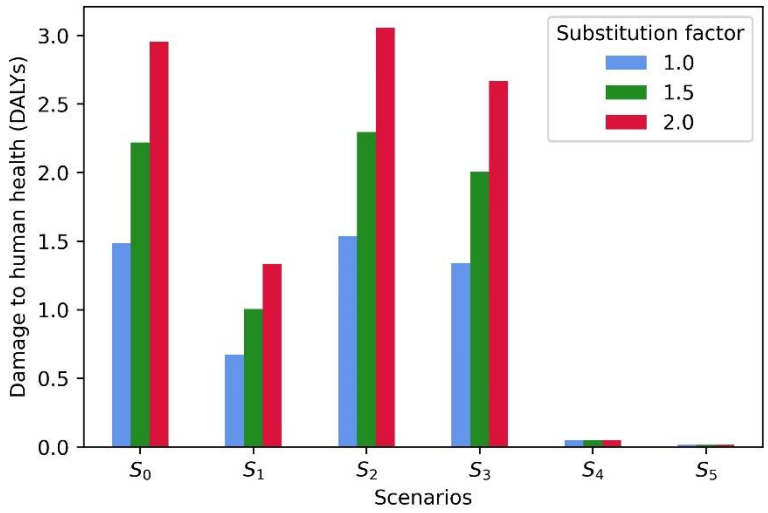

Figure 6.2. Damage to human health cumulated from 1980 until 2050 for all scenarios using a substitution factor of 1, 1.5 and 2.

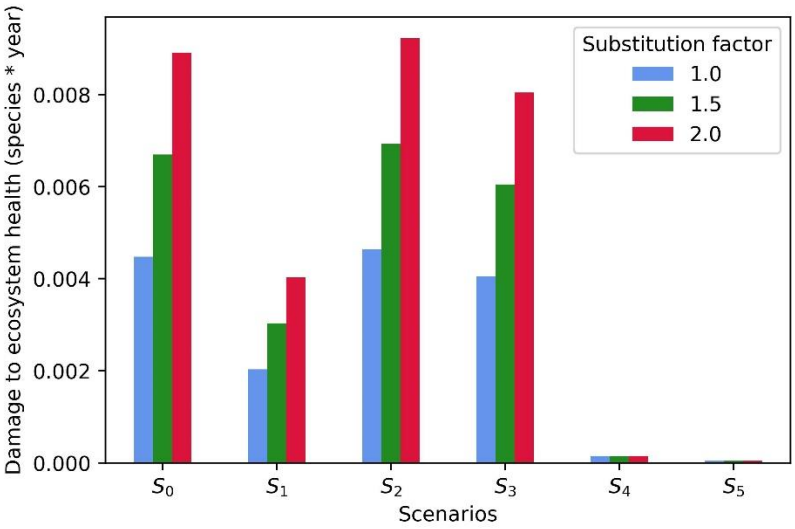

Figure 6.3. Damage to ecosystem health cumulated from 1980 until 2050 for all scenarios using a substitution factor of 1, 1.5 and 2.