TOWARDS AN ENVIRONMENTALLY SOUND METRIC TO EVALUATE POST-CONSUMER PLASTIC PACKAGING WASTE RECYCLING

An analysis of the technical, economic, and environmental performances of post-consumer plastic packaging waste recycling to evaluate the use of value-based and mass-based metrics in waste management systems

Author:

Jaime García Gutiérrez

Main supervisors:

Dr. Francesco Di Maio
Dr. Mingming Hu

External supervisors:

Dr. Bin Hu
Dr. Eleonora Foschi

ABSTRACT

In the recent years, the circular economy (CE) has emerged as an alternative to current linear economic models with prospects of achieving the decoupling of environmental impacts from economic growth. To this end, CE models focus on the permanence of products, materials, and value in the economy, eliminating as much waste as possible. In waste management policy making, traditional indicators, such as recycling rates, are still the norm as demonstrated by the latest binding goals for plastic packaging recycling set by the EU in the Packaging Waste or the Single Use Plastics Directives. However, these have been criticized for overlooking qualitative aspects that are fundamental in the CE. Consequently, several indicators have been proposed as an alternative to monitor this new economic paradigm. Among these are value-based metrics. Economic value contains information about both the quality and the quantity of the specific material or product. In addition, value can be altered using economic instruments, such as taxes, to align value with other relevant environmental or social interests.

The validity of using value-based metrics to evaluate the performance of waste management systems was analysed. Several scenarios for possible configurations of the Dutch post-consumer plastic packaging waste (PCPPW) management network were defined and assessed in terms of technical, economic, and environmental performance. Two technical metrics, intermediate recycling rates (iRR) and recycling rates (RR), and one value-based metric, the circular economy index (CEI), were compared to the environmental and economic performance of the scenarios. The CEI showed a better alignment with all the environmental impact categories than the mass-based metrics. Most importantly, the value-based metric proved capable of capturing the significance of the quality of the recycled plastics in the displacement of primary raw materials, thus fostering high quality recycling over downcycling. However, no correlation was found between the economic performance and the presented indicators. There are multiple business models that achieve good economic performances with diverse environmental, technical, or value-recovery performances. This suggests that current policies are unable to successfully align economic and environmental strategies. More research should be put in defining holistic policies that promote environmental and economically sustainable practices.

TABLE OF CONTENTS

1. INTRODUCTION

In the 1960's, Vance Packard coined the term 'The Throwaway Age', to define the mass commercialisation and consumption behaviour that was spreading into the American life-style (Packard & McKibben, 1963). More than half a century later, this take-make-dispose economic model has become the paradigm of most modern urban societies, creating large environmental pressures on both ends of the economic system (Tisserant et al., 2017). In the last two decades, the global rate of waste generation has more than doubled and the growing tendency is expected to be maintained in the coming years (Hoornweg & Bhada-Tata, 2012; Kaza, Yao, Bhada-Tata, & Woerden, 2018). At the same time, the extraction and consumption of resources has been steadily increasing, reaching levels that cannot be sustained (Steffen et al., 2015; UNEP, 2012).

This context has inspired the proposition of an alternative economic model that disputes the current linear paradigm, the circular economy (CE). CE has emerged as a substitute of the linear economic model with prospects of achieving *decoupling* of economic growth and environmental impacts (Ghisellini, Cialani, & Ulgiati, 2016). The CE model aims to increase the permanence of materials and products within the economic system, focusing on value retention, and eliminating as much waste as possible (EC, 2015). To this end, waste management systems (WMS) in the CE must evolve in order to perform a renewed role for the economy (Lee et al., 2017; Tisserant et al., 2017). Traditional waste treatment systems focused on the safe treatment and disposal of the societies waste streams (Brunner & Ma, 2009). The aim within the framework of a CE model is to minimize these waste streams and to maximise value recovery from the materials and products that have been ultimately discarded after reuse or remanufacture options have been exhausted. To this effect, waste collection, treatment and management are to be optimized for material and value recovery.

A material whose significance has been steadily increasing in the waste management systems over the last decades is plastic (EC, 2018a). Their short lifetime applications, such as packaging, and the potential pollution of natural environments make them an even more relevant stream (Van Eygen, Laner, & Fellner, 2018). The statistics reveal that the management of PCPPW is developing towards more recycling, recently overtaking the landfilling option within the European community (PlasticsEurope, 2018). Yet, this recycled fraction is often downcycled, i.e. reprocessed into low quality applications (Rigamonti, Niero, Haupt, Grosso, & Judl, 2018; Villanueva & Eder, 2014). In a more global perspective, there is also still room for improvement since almost 80% of all the plastic ever produced is estimated to remain untreated in the natural ecosystems or piled up in landfills (Geyer, Jambeck, & Law, 2017). In the Netherlands, 52% of plastic packaging was recycled in 2018 (Afvalfonds Verpakkingen, 2018). However, this is mostly achieved via mixed plastic recycling, instead of single-polymer recycling, limiting the applications and the market value (CPB, 2017).

In this line, the EU has set binding goals for material recovery to the Member States. By 2030, 55% of the plastic packaging waste ought to be recycled (EC, 2018d). Despite being challenging targets on paper, these fail to incorporate the fundamental principles of the CE. The term *recycling* does not distinguish between high and low quality reuse of materials (Haupt, Vadenbo, & Hellweg, 2017). This is a recurrent problem with mass-based indicators used in the waste management sector, with direct implications in the value of the materials and the primary material displacement achieved with these metrics (Koffler & Florin, 2013; Vadenbo, Hellweg, & Astrup, 2017). The latter is particularly important when assessing the environmental performance of a recycling system (Huysman et al., 2015). Conclusively, conventional mass-based metrics, such as collection and recycling rates, have been proved unsuccessful in reflecting the environmental and economic elements of the CE (Haupt, Waser, Würmli, & Hellweg, 2018; Van Eygen et al., 2018).

These limitations explain the development of many alternative indicators for the CE (Saidani, Yannou, Leroy, Cluzel, & Kendall, 2019). According to Moraga et al. (2019), there are five main preservation strategies in the CE, namely, the preservation of functions, products, components, materials, or embodied energy. Indicators for the CE can be correlated to these strategies, with metrics in the waste management sector falling into the materials or embodied energy preservation strategies. In general, most CE indicators expand the traditional quantitative assessments with some sort of qualitative component (Moraga et al., 2019). For instance, Huysman et al. (2017) put their focus on environmental performance and define the circular economy performance indicator (CPI) that can be used to determine the environmentally optimal treatment route for a waste stream attending to its quality. Roithner & Rechberger (2020), on the other hand, incorporate quality considerations, measured in terms of compositional purity of the recyclates, into the conventional recycling rates in their indicator, the recycling effectiveness (RE). The Circular Economy Index (CEI), defined by Di Maio & Rem (2015), focuses on the value recovered from waste streams. The quality of the recovered materials is manifested in their monetary value. Each of these examples of indicators evaluate one dimension of the CE, namely, the environmental performance, the technical performance, and the economic performance. There is a need for multiple indicators to completely understand a complex system such as the CE, demonstrated in the monitoring framework proposed by the EU (EC, 2018b). However, there is a trade-off between the coverage achieved by a set of indicators, and the simplicity and interpretability of a single indicator (Neuhoff, Cooper, Laing, Lester, & Rysanek, 2009). In addition, when indicators are used as policy targets, they should ideally devise a trajectory for a desirable future (Neuhoff et al., 2009). This is not the case with the use of mass-based rates in waste management policy. Recycling rates are easy to calculate and interpret. However, they lack a life cycle thinking approach, limiting the metric's scope to purely technical cycles and disregarding social, environmental, or economic components (Moraga et al., 2019). The validity of using mass-based rates as policy targets is determined by the alignment between the technical index and the higher policy goals. However, this has been proved unsuccessful (Gradus, Nillesen, Dijkgraaf, & van Koppen, 2017; Haupt, Waser, et al., 2018; Van Eygen et al., 2018; Zink & Geyer, 2017). In conclusion, recycling rates are not fitting policy targets since ideal waste management solutions should not aim to optimise recycling rates at the expense of any of these other policy targets. The European Commission has recently changed the methodology for measuring recycling rates (EC, 2019a), but the European Council still calls upon the improvement of the CE monitoring frameworks towards full life cycle indicators (The Council of European Union, 2018).

Table I. Examples of indicators proposed for the circular economy.

Indicator	Abbreviation	Definition	Focus
Circular Economy Performance Indicator	CPI	"ratio of the actual environmental benefit ⟨…⟩ over the ideal environmental benefit according to quality" (Huysman et al., 2017)	Environmental impacts
Recycling Effectiveness	RE	"RE describes how effective the observed recycling process could separate and concentrate its recycling input - in a quantitative and qualitative way" (Roithner & Rechberger, 2020)	Quality (in terms of mass)
Circular Economy Index	CEI	"ratio of the material value produced by the recycler (market value) by the material value entering the recycling facility" (Di Maio & Rem, 2015)	Value recovery

This study aims to determine if there are more appropriate metrics to evaluate waste management systems. More precisely, to determine if value-based indicators are better aligned with economic and environmental policies than current mass-based metrics. Value-based indicators have been claimed to be able to tackle the inefficiencies of mass-based indicators (Di Maio & Rem, 2015; Di Maio, Rem, Baldé, & Polder, 2017). Ideally, value would be weighted according to social, environmental, and economic impacts. In practice, economic value is proposed as a reasonable intermediate solution (Di Maio et al., 2017). Economic value contains information about both the quality and the quantity of the specific material or product. Thus, value-based indicators favor higher quantities and qualities of the recycled streams, promoting innovation towards higher value retention (Linder, Sarasini, & van Loon, 2017). This is expected to lead to better environmental performance as well, since high quality secondary raw materials have the potential to displace equivalent primary raw material consumption compared to lower quality streams where no effective displacement occurs (Eriksen, Damgaard, Boldrin, & Astrup, 2019).

To meet the research objectives, this research will analyse the management of post-consumer plastic packaging waste (PCPPW) in the Netherlands. Recycling networks are being adapted to reach the latest European targets. In the Netherlands, for instance, instead of increasing the share of separately collected materials, the post-separation of plastics is being proposed as an alternative to recover these valuable materials from the residual stream and increase the recycling performance (Gradus et al., 2017; Rotterdam Circulair, 2019). However, the environmental and value impacts of these adaptations are not completely understood. In the pursuit of a better waste management system, recycling and recovery rates may be being mistaken as the ultimate goal to reach instead of being considered as means to enhance social, environmental, and economic performances.

This study will address these uncertainties. Evaluating the technical, economic, and environmental performance of the recycling network of the three main collection methods of PCPPW in the Netherlands, two separate collection schemes, kerbside collection (KS) and drop-off collection (DO), and the post-separation (PS) from the residual waste (CBS, 2019b).

2. MATERIALS & METHODS

2.1. POST-CONSUMER PLASTIC PACKAGING WASTE (PCPPW)

2.1.1. General concepts of PCPPW management

Five main stages are identified in a PCPPW management system, namely, collection, sorting, reprocessing, incineration, and landfill (Ragaert, Delva, & Van Geem, 2017; Thoden Van Velzen et al., 2013). While collection is a mandatory step in any waste management system, waste can then be directed to recycling (sorting and reprocessing), energy recovery (incineration), or disposal (landfill), or to a combination of these three. According to the waste hierarchy, the EU recommends prioritising recycling over energy recovery and landfill (EC, 2008). Thus, sound waste management systems should commonly implement recycling operations as the first management route, while the residues from the recycling processes is sent to energy recovery. Finally, the ashes produced from the incineration step are landfilled. Each waste management stage is defined in more detail below.

2.1.1.a. Collection

2.1.1.a.i. Collection schemes

There are several collection methods for plastic packaging waste. These are commonly categorized looking at two main factors, the source and the service (Rodrigues, Martinho, & Pires, 2016). Plastic waste can be separated at the source or collected in the residual fraction. For the European Commission, separate collection stands for "the collection where a waste stream is kept separately by type and nature so as to facilitate a specific treatment" (EC, 2008, p. 10). These separately collected streams can be aggregated in commingled waste streams, often the case of plastics, metals and beverage cartons (PMD, in Dutch), or individually separated, a widespread practice for paper and glass streams within the EU (Seyring, Dollhofer, Weißenbacher, Herczeg, & David, 2015). The type of waste collection service can be divided in two overarching schemes, either the waste is collected at the source, or the waste generator has to bring it to a drop-off point (Rodrigues et al., 2016). The first collection schemes are commonly referred to as kerbside or door-to-door collection systems. The latter represent a more varied range of collection scenarios, encompassing neighbourhood bringing points (hereon referred to as drop-off collection), civic amenity sites (CAS), deposit refund collection, among others (Mwanza, Mbohwa, & Telukdarie, 2018; Rodrigues et al., 2016). Finally, in schemes with no source separation, plastics that are recovered from the residual stream are mechanically separated in central sorting facilities. These are referred to as post-separation schemes (Cimpan, Maul, Jansen, Pretz, & Wenzel, 2015; Thoden Van Velzen et al., 2013).

In this study, the focus will be directed to the commingled collection of plastic in the PMD stream in kerbside, drop-off, and post-separation systems since they are the most developed collection methods in The Netherlands (CBS, 2019b).

2.1.1.a.ii. Collection performance

Collection performance is a behavioural metric. It measures the attitudes of individual households in certain collection system which varies between collection schemes. The collection or capture rates are calculated at the EU level as the percentage of materials that are separately collected, mainly via kerbside or drop-off collection, with respect to the total waste generated (Seyring et al., 2015). In the Netherlands, this definition also includes the waste recovered via post-separation (Ministerie van IenW, 2019).

The performance of the collection system depends on the number of households that are participating in the separate collection scheme, i.e. how many residents are separating their waste, and how accurate they are in the separation process. Thoden van Velzen, Brouwer, & Feil (2019) called these two factors participation rate (PR) and selection rate (SR), respectively. The yield and composition of the separately collected stream, the collection rate (CR), is defined by multiplying these two factors, see Eq. 1.

$$CR = PR*SR \qquad (1)$$

Participation rate (PR)

The level of participation in the collection system varies greatly between different collection systems and among collection systems as well. In general, kerbside collection systems yield higher participation levels than drop-off collection systems (Gallardo, Bovea, Colomer, & Prades, 2012; Thoden van Velzen et al., 2019; Thoden Van Velzen et al., 2013). These discrepancies have been frequently associated to the inconveniences that drop-off collection creates to the consumer in the form of larger distances to the point of waste disposal. Moreover, the distance to the collection point are also negatively correlated to the collection performance among drop-off collection systems (Gallardo, Bovea, Colomer, Prades, & Carlos, 2010; Struk, 2017). In general, the further the collection point is, the lower is the participation.

Selection rate (SR)

The selection efficiency of the citizens participating in the separate collection scheme will determine, to a great extent, the level of impurities found in the separated fraction. The other main cause of impurities is packaging design (Eriksen & Astrup, 2019). Selection rates are generally higher than participation rates, making the latter the biggest limiting factor for achieving higher collection rates (Gallardo et al., 2010). In addition, selection rates have been found to be rather independent from the type of collection system, and their respective participation rates (Thoden van Velzen et al., 2019). This means that once citizens are willing to participate in a separate collection scheme, their performance is relatively consistent. Consequently, the selection rates for kerbside and drop-off collection scenarios in this study are equivalent.

Selection rates are better defined at the individual packaging level, e.g. PET plastic bottles or PE films. These can later be aggregated into higher levels of classification, such as the material level, e.g. plastics. In addition to the desired fractions, to determine the quality of the separately collected stream, it is interesting to define the selection rate of undesired elements. Impurities in the separate collection systems are organic materials, paper and cardboard, glass, or textiles, but also laminated film packages, e.g. chips bags. The optimal selection behaviour will maximise the selection rate of the targeted packaging (SR_{TP}) and minimise the selection rate of unwanted impurities (SR_{imp}). Non-plastic packages represent a special case. While they are not directly targeted by the collection portfolio, these are often sorted and included into the recycled goods (Brouwer et al., 2019). Thus, the selection rate of these non-targeted plastics (SR_{NTP}) is treated separately.

2.1.1.b. Sorting

The next step in the recycling system is sorting. Here, the separately collected materials from KS and DO collection systems, and the residual waste directed to material recovery in PS schemes are sent to a material recovery facility (MRF). The sorted materials are baled before being transported to the reprocessing stage (Ragaert et al., 2017).

2.1.1.b.i. Material Recovery Facilities (MRF's)

MRF's are used to improve the quality of the collected waste stream (Rigamonti et al., 2014). In an MRF, the waste materials are sorted into a variety of products with higher purity and value than the original stream. MRF can be designed to produce a range of sorted outputs (Ardolino, Berto, & Arena, 2017; Eriksen et al., 2019), and to accept different material inputs (Combs, 2012; Pressley, Levis, Damgaard, Barlaz, & DeCarolis, 2015).

In a standard MRF recovering plastics from a separate collection system, a bag opener cuts the container bags and the contents are fed into a trommel sieve where the waste is sorted according to size. Commonly, large (>220mm) and small (<20-50mm) objects are discarded into the residual stream. The intermediate fractions are directed to an air classifier that separates light (e.g. paper, plastic foils) from the heavy objects (e.g. rigid plastics, metals, beverage cartons). Next, ferrous metals are removed from the waste stream using an overhead magnet, beverage cartons are sorted out with an optical separator (NIR), and non-ferrous metals, mostly aluminium, are recovered using an eddy current separator. The remaining stream is mostly composed of plastics. These undergo a ballistic separation that separates heavier, three-dimensional objects from lighter, flat ones, which are discarded. The rigid plastics are further classified into the sorted polymer products. Using NIR optical sorting, these are categorised into the different polymers, and directed to a manual sorting station for quality control and recovery of any valuable plastic that was wrongly classified. Occasionally, PET is further sorted into colours using colour recognition sorting technologies (Cimpan, Maul, Wenzel, & Pretz, 2016; Ragaert et al., 2017). A diagram of the MRF is presented in Figure I.

The MRF's used in mixed waste collection schemes have a similar configuration. However, the higher levels of contamination on mixed-waste separation schemes require that these MRFs incorporate additional separation equipment at the beginning of the plant. According to different studies (Cimpan et al., 2015; Pressley et al., 2015; Rigamonti et al., 2014), this primarily consists of an extra trommel to separate organics and bulky materials from the recyclable stream.

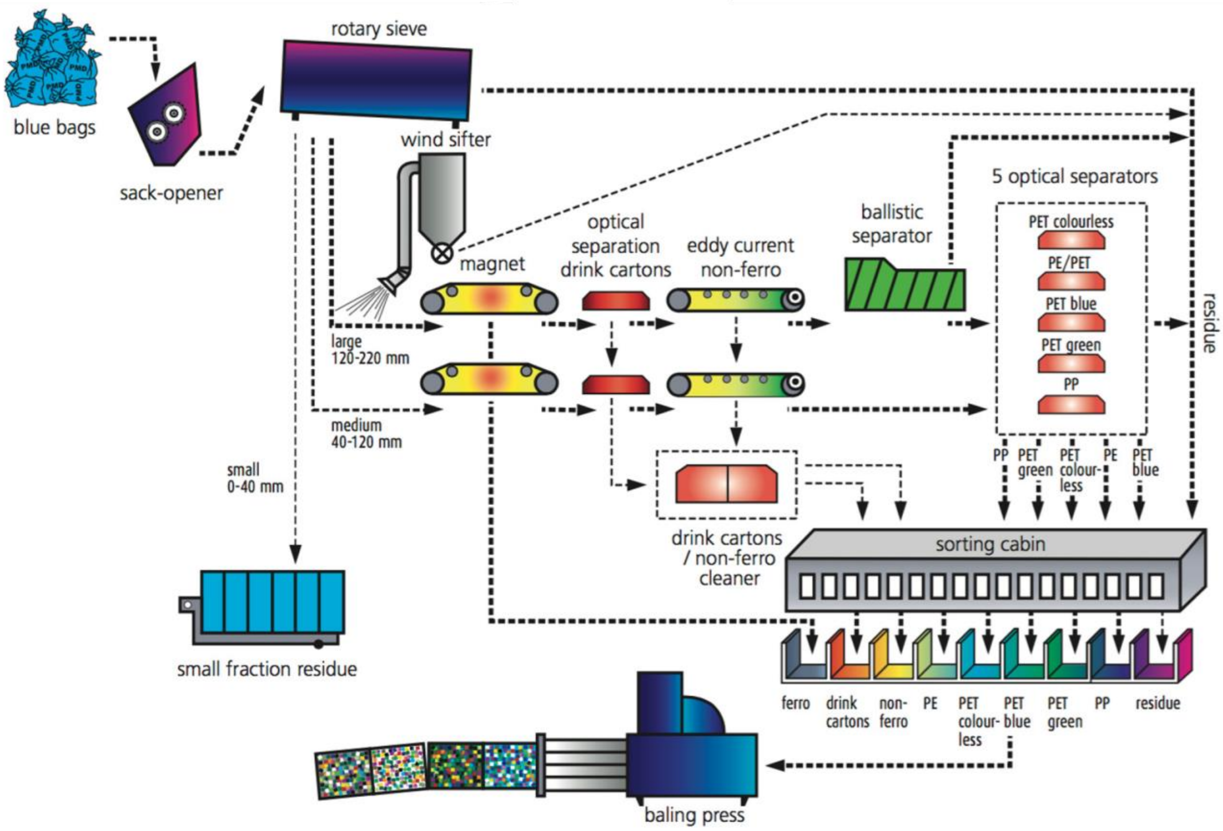

Figure I. Process diagram of a typical MRF of PMD. Retrieved from Ragaert et el. (2017)

2.1.1.b.ii. Sorting performance

The performance of an MRF is defined by the efficiency to sort the targeted materials into their intended outputs. This can be expressed in terms of the recovery and grade (Nijkerk & Dalmijn, 2001). Recovery defines the amount of materials in the inputs stream that end up in the intended output. Recovery is commonly defined as sorting efficiency in the literature (Cimpan et al., 2016; Faraca & Astrup, 2019). This value indicates the amount of materials recovered from the waste stream but does not say anything about the quality. To express the polymeric purity of that material output, the concept of grade is used. Grade is determined by the percentage of targeted materials that are present in a material output.

2.1.1.b.iii.Intermediate recycling rates (iRR)

In 2019, the EC approved a new methodology to measure recycling rates (EC, 2019a). This new definition is explained in section 2.1.1.c.iii. Under the previous methodology, however, recycling rates could be measured at the output of the MRF (Haupt et al., 2017). Hereon, these are referred to as intermediate recycling rates (iRR).

2.1.1.c. Reprocessing

The sorted fractions from the MRF's are sent to the reprocessing stage. Here, the sorted materials are converted into recycled pellets in reprocessing plants (RP's). Two main phases are identified during reprocessing: washing and compounding. In the first phase, sorted plastic waste is shredded, washed, and dried into clean flakes. Later, these flakes are compounded into pellets. This is the final step before introducing the secondary raw materials again into the manufacturing cycle. There are two main reprocessing technologies, mechanical recycling, and feedstock recycling. However, only mechanical recycling will be considered in the following assessment. It has been selected for being a well-known technology, and an accepted solution in the plastic recycling industry (Ragaert et al., 2017).

2.1.1.c.i. Reprocessing plants (RP's)

In a RP, the bales of sorted packaging plastics are transformed into valuable outputs that can be used in manufacturing processes. RP can be a separated facility receiving the outputs of an MRF, but it can also be part of an integrated facility that combines sorting and reprocessing (Ragaert et al., 2017).

A standard RP involves the grinding, washing, drying, and extrusion of the plastic inputs into recycled pellets (Faraca, Martinez-Sanchez, & Astrup, 2019; Mastellone, 2020). First, the sorted materials are grinded into flake sized particles using a shredder machine. Second, these flakes are washed to eliminate dirt attached to the polymers and further separated according to density using sink-float separation techniques. The washed particles are then sent to a mechanical drier to remove unwanted moisture. Finally, during the extrusion stage, the dried plastic flakes are melted and regranulated into pellets (Ragaert et al., 2017).

2.1.1.c.ii. Reprocessing performance

The performance of the reprocessing stage is again measured in terms of recovery and grade. The recovery of materials measures the amount of the targeted materials entering the reprocessing process that end up in the intended product. This is referred to as reprocessing efficiency or reprocessing yield (Brouwer et al., 2018; Faraca et al., 2019). The grade of the outputs determines the quality of the secondary raw materials and will influence the market applications and displacement of primary raw materials (Rigamonti et al., 2018).

2.1.1.c.iii. Recycling rates (RR)

Here, we will use recycling rates (RR) as stated by the EC in their latest methodology. Under the latest definition, recycling rates should be measured at the *calculation point* and should "include non-targeted materials only to the extent that their presence is permissible for the specific recycling operation" (EC, 2019a, p. 68). The *calculation point*, in the case of plastics, is the point in the recycling chain when the materials are "separated by polymers that does not undergo further processing before entering pelletisation, extrusion, or moulding operations" (EC, 2019a, p. 71).

2.1.1.d. Incineration

Municipal solid waste incineration is a common type of thermal treatment applied in waste management systems around Europe (Gradus et al., 2017). The inputs to municipal solid waste incinerators (MSWI's) range from the direct treatment of municipal solid waste, to the residues from sorting or reprocessing steps (Sabbas et al., 2003). The incineration of waste has some advantages and disadvantages. On the positive side, it reduces the volume and weight of waste and, consequently, reduces the demand for landfills. It also enables the recovery of energy, while eliminating pathogens or organic pollutants. On the negative end, the incineration process generates significant combustion emissions, along with other solid residues such as the residues from the air pollution treatment, or fly and bottom ashes (Islas, Manzini, Masera, & Vargas, 2018; Sabbas et al., 2003).

2.1.1.d.i. Municipal solid waste incinerators (MSWI's)

There are several technologies to recover energy from waste. According to the type of technology, there are three predominant alternatives, namely, incineration, gasification, or pyrolysis, with incineration being the leading solution for the thermal treatment of waste. There are three main types of incinerators, grate incinerators, rotary kilns, and fluidised bed incinerators (Bosmans, Vanderreydt, Geysen, & Helsen, 2013).

The incineration process is simply the direct combustion of the combustible materials in the input waste. The different technologies primarily change the air distribution and are optimised for particular types of waste, but the incineration process is essentially the same (Bosmans et al., 2013). The combustion process generates big amounts of heat in the form of high temperature flue gases. If this heat energy is recovered, the technology is called Waste-to-Energy (WtE) (Bosmans et al., 2013). WtE plants are classified depending on the type of energy generation, such as power steam, district heating, combined heat and power (CHP), or electricity production (Haupt, Kägi, & Hellweg, 2018a).

Finally, the flue gases produced during combustion and solid residues are a source of environmental pollution. These have to be properly treated and disposed of to reduce the environmental impacts of the incineration process (Bosmans et al., 2013; Sabbas et al., 2003).

2.1.1.d.ii. Incineration performance

The performance of incineration is defined here in terms of the capacity of recovering energy from the combusted materials. This is defined as the percentage of the energy contained in the waste input, their lower heating value (LHV), that is recovered for useful applications. Common CHP plants produce electricity and heat at the same time with efficiencies ranging from 12-21% for electricity generation and 12-50% for heat production, depending on the energy needs (Haupt, Kägi, et al., 2018a).

Table II. List of definitions

Term	Abbreviation	Definition
Post-consumer plastic packaging waste	PCPPW	Plastic products disposed by households that are "used for the containment, protection, handling, delivery and presentation of goods, from raw materials to processed goods, from the producer to the user or the consumer" (European Parliament and Council, 1994).
Municipal solid waste	MSW	"Mixed waste and separately collected waste from households ⟨…⟩ or from other sources, where such waste is similar in nature and composition to waste from household" (EC, 2018c, p. 121)
Separate collection	SC	The type of collection system "where a waste stream is kept separately by type and nature so as to facilitate a specific treatment" (EC, 2008, p. 10)
Kerbside collection	KS	The type of collection system where the collection is made close to the waste source and where residents are responsible for bringing their waste in previously allocated containers (bags, bins, etc.) to the kerbside on the designated collection day (Rodrigues et al., 2016).
Drop-off collection	DO	The type of collection system where the collection is made in designated containers and where residents are responsible for bringing their waste to these locations (Rodrigues et al., 2016).
Source separation efficiency / Capture rate	CR	Describes the technical performance of the separate collection scheme, i.e. how much and how accurately the targeted waste fraction is being separated. It depends on the participation rate and selection rate.
Participation rate	PR	Percentage of the households included in a separate collection scheme that are actively engaged in waste separation (Thoden van Velzen et al., 2019).
Selection rate	SR	The collection behaviour (accuracy of waste separation with regards to targeted and non-targeted materials) of the participating households (Thoden van Velzen et al., 2019).
Post-separation	PS	The type of collection system where there is no source separation and plastics are recovered from the residual stream are mechanically separated in central sorting facilities

Material recovery facility	MRF	Installations that receive waste streams containing recyclable materials and separate them into different categories concentrating the valuable materials, commonly using a combination of automated sorting technologies, such as optical sensors or magnets, and some level of manual sorting and quality control (Cimpan et al., 2015)
Reprocessing plant	RP	Installations that receive sorted plastics from MRF and converts them into recycled pellets or flakes to be used again in manufacturing processes. These facilities are sometimes specialised in certain polymer types. Reprocessing plants using mechanical recycling technologies commonly include grinding, washing, drying, and extrusion (Villanueva & Eder, 2014).
Municipal solid waste incinerator	MSWI	Installation that commonly receives MSW and where this is combusted, releasing thermal energy and reducing the size and volume of the input waste. The thermal energy can be recovered in a process that is referred to as Waste-to-Energy (Bosmans et al., 2013).
Recovery	-	Percentage of certain material present in the input stream that ends up in the desired output (Nijkerk & Dalmijn, 2001)
Grade	-	Percentage of the material output that are targeted materials. Also referred as purity of the material (Nijkerk & Dalmijn, 2001).

2.1.2. PCPPW in the Netherlands

Plastic packaging waste represents 12% by weight of the municipal solid waste generated in the Netherlands (Brouwer et al., 2019; CBS, 2019a). This amounts to 350 kilotons (kton[1]) of PCPPW that has to be managed every year in the Netherlands, at a rate of approximately 20.4 kg per inhabitant. Table III presents the waste composition of the plastic waste fraction. The compositional data was derived from Brouwer et al. (2019). The distribution on the different types of polymers and packaging fractions was defined in line with other studies (Eriksen et al., 2019; Rigamonti et al., 2014). This classification covers the most abundant polymers and fractions found in the Dutch PCPPW.

Table III. Composition of the plastic waste fraction in the Netherlands

		Composition of plastic waste (%)							
		PET	**PE**	**PP**	**PS**	**EPS**	**PVC**	**mix**	total
PCPPW	**bottle**	7%	5%	1%	0%				14%
	film	0%	21%	5%	0%		0%	6%	32%
	rigid	13%	3%	10%	3%	0%	1%	7%	38%
	laminated							4%	4%
non PCPPW		0%	3%	3%	1%		2%	2%	12%
total		21%	32%	20%	4%	0%	2%	20%	100%

In the third National Waste Management plan (LAP3, abbreviated in Dutch), the Dutch government aims to increase the separation of materials up to 75% (Ministerie van IenW, 2019). LAP3, however, indicates that these targets can be achieved using either source separation or post-separation schemes as far as safety during the collection chain and quality of the recovered materials are not compromised. In addition, according to the legislation, the implemented system should provide additional environmental benefits while being socially and economically acceptable. The most common solution for the separate collection of packaging plastics within Dutch municipalities is the kerbside collection of commingled PMD (CBS, 2019b). However, more than 60% (215 kton) of the PCPPW generated in the Netherlands is not being separately collected and, from this residual stream, only 19% (41 kton) is subjected to post-separation of plastics (Brouwer et al., 2019),. This leads to a net recycling rate of approximately 29%. A small amount of materials ends up as impurities in other sorted products

[1] Though not being standard, 'kton' is used in this study as the abbreviation of kilotons

such as beverage cartons or metals. The rest is incinerated (CBS, 2019b; Gradus et al., 2017). Figure II depicts the mass flows of the Dutch PCPPW management system.

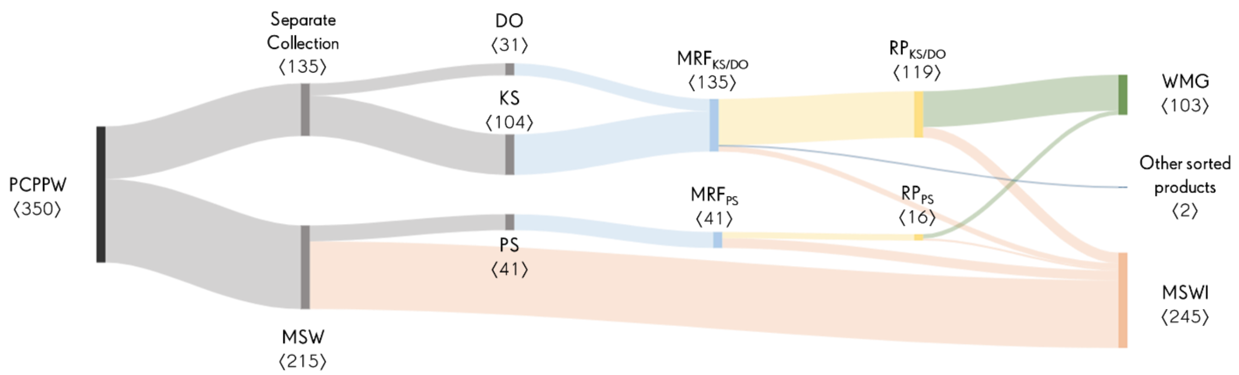

Figure II. Sankey diagram showing the mass flows of the Dutch PCPPW management system. In brackets, the mass in kilotons. Adapted from Brouwer et al. (2019)
MSW = municipal solid waste. DO = drop-off. KS = kerbside. PS = post-separation. MRF = material recovery facility. RP = reprocessing plant. WMG = washed milled goods. MSWI = municipal solid waste incineration. Other sorted products includes beverage cartons, metals, etc.

As seen in Figure II, all the waste that is not recycled in the NL is sent to incineration. There are 13 incineration plants in the NL (Rijkswaterstaat, 2018). Most of them are CHP plants that generate electricity and heat for district or industrial heating. Four of these are post-separation plants as well, including AVR, that just recently started separating plastic from the residual stream in Rotterdam (AVR, 2020).

The extended producer responsibility (EPR) scheme in the Netherlands is managed by Afvalfonds Verpakkigen. EPR schemes charge a tariff to producers to cover the costs of collection and recycling of the plastic packaging products that they put into the market and will eventually become waste. In the NL, there are two types of tariffs as of 2020, a normal tariff of 600 €/ton of plastic waste, and a reduced tariff of 340 €/ton for companies that use plastics with positive market values after sorting (Afvalfonds Verpakkingen, 2019). Thus, based on the yearly generation of PCPPW in NL, the costs of managing the complete Dutch PCPPW recycling network are estimated to be between 120 and 210 million € per year.

The performance of the Dutch recycling network is presented in Table IV. These performance indicators are a combination of mass-based indicators (CR, iRR, and RR) and value-based indicators (cost index, CI, and circular economy indicator, CEI). These are merely introduced here to enable the future validation of the model, sections 3.1.1 and 3.2.1, but a detailed description will be done in the following sections. The mass-based indicators are calculated from the mass flow analysis in Figure II, while economic indicators are estimated from the Dutch EPR tariffs. The CEI is not calculated since no data was found on the revenues obtained by the Dutch plastic recycling network.

Table IV. Performance indicators of the Dutch recycling network.
[a] the capture rate includes the materials sent to post-separation.

	mass-based indicators			value-based indicators	
	CR [a] (%)	iRR (%)	RR (%)	CI (€/ton)	CEI (%)
NL	50%	39%	29%	340 - 600	-

2.2. Model

The plastic waste recovery chain has been divided into four stages, three successive steps focused on material recovery, collection (S1), sorting (S2), and reprocessing (S3); and a fourth stage, intertwined with the previous ones, aimed at energy recovery, municipal solid waste incineration (MSWI, S4). The residues from the incineration process are sent to landfill, but this stage was considered out of the scope of this analysis.

This framework is used to generate plausible scenarios for PCPPW management systems, see section 2.2.2. The performance of the systems is analysed in terms of mass-based, value-based, and environmental performance. First, the scenario is evaluated with a mass flow analysis (MFA). MFA is an analytical method that examines the flows and stocks of a certain element thought the economic system (Brunner & Rechberger, 2016). In this study, the MFA has been outlined starting from the material composition of the original waste stream and applying transfer coefficients to the subsequent stages (see Tables II-V).

The MFA also acts as the foundation to determine economic and environmental performances. Economic and environmental models are used to estimate costs, value, or environmental impacts from the MFA. These are further explained in the following sections. A basic representation of the model is presented in Figure III.

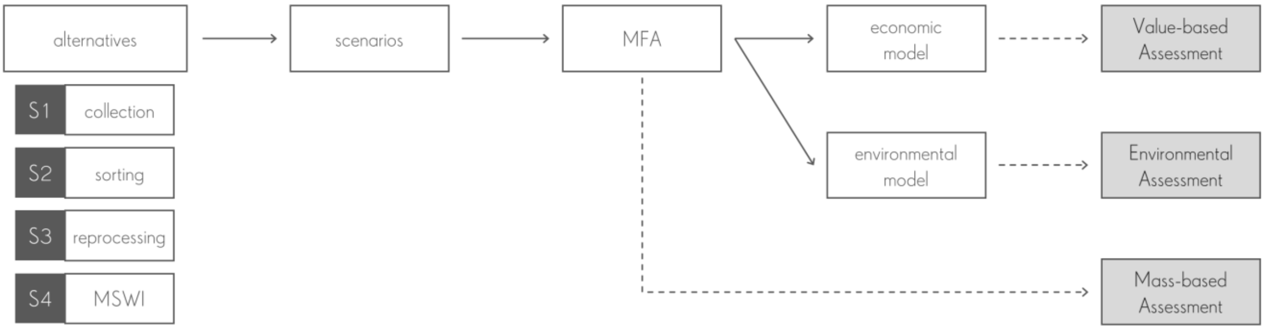

Figure III. Modelling approach

2.2.1. Goal & Scope Definition

In this study, the three collection systems are analysed in terms of technical performance (mass), economic performance (costs and value), and environmental performance (environmental impacts). Hence, the goal, scope, functional unit, and the boundaries of the analysed system must be consistent during the three different assessments.

The goal of the study is to analyse and compare the technical, economic, and environmental performance of three different collection routes (kerbside, drop-off, and post-separation) for post-consumer plastic packaging waste (PCPPW). The geographical scope is the Netherlands. The assessment will rely on current technical, economic, and environmental data of waste management practices. With a life cycle approach, the analysis will cover all the relevant stages of the waste management chain, i.e. the source separation, collection, sorting, and reprocessing of valuable material, and the incineration with energy recovery of the residual streams.

The functional unit (FU), that will be the basis for the comparison of the alternative collection schemes, is defined as the management of 1 ton of PCPPW in the Netherlands.

2.2.2. Scenario generation

Three main routes are defined for the collection of plastic packaging, namely, kerbside collection (KS), drop-off collection (DO), and post-separation (PS). For each of these three alternatives, several scenarios of plastic recovery are determined based on the combination of performances of two steps of the waste management system: the performance of the collection stage (S1) and the performance of the sorting stage (S2). Only one reprocessing (S3) and incineration (S4) alternatives were assumed in this study. Each scenario is generated selecting a collection system (S1.1), a source separation efficiency (S1.2), and a type of MRF (S2). S1.2 is defined by selecting a participation level (S1.2.a) and a selection level (S1.2.b). Besides the selection rate for PS systems that presents only one choice, there are three possible options for each of the parameters. The possible configurations and system boundaries are illustrated in Figure IV. In total, there are 63 possible combinations. The complete list of scenarios can be found in the Supplementary Information.

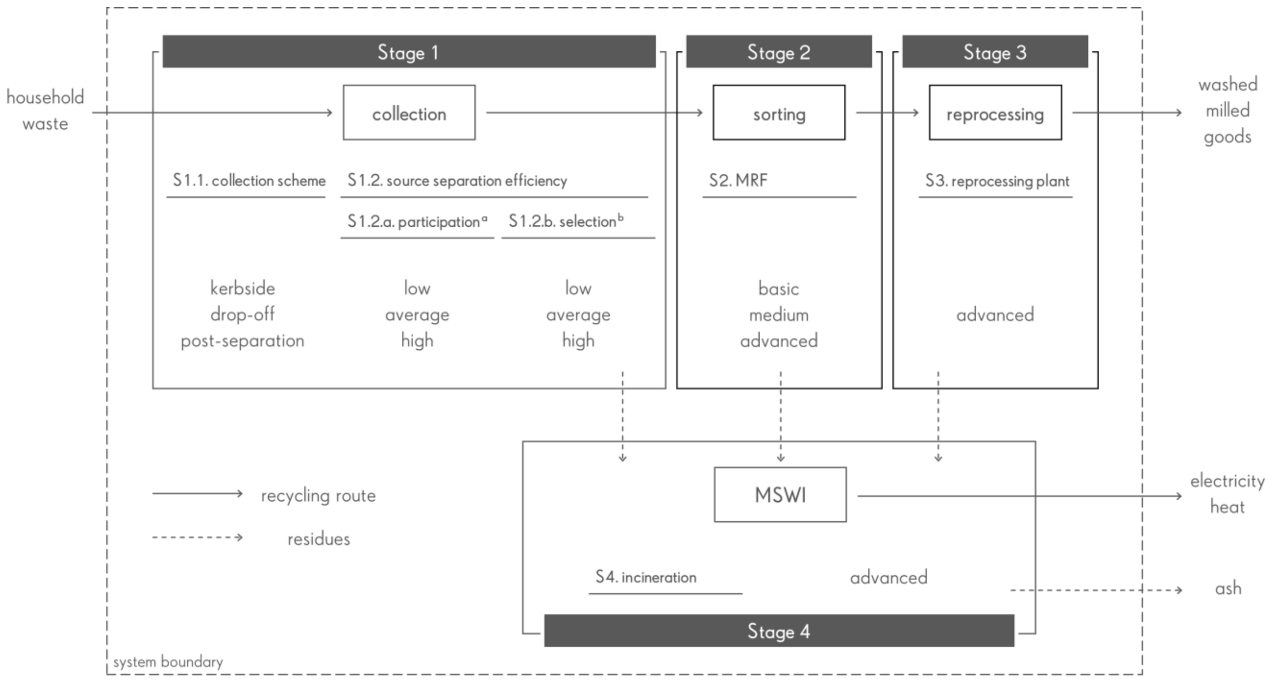

Figure IV. System boundaries and possible scenario configurations. For each scenario, a combination of collection scheme, participation rate, selection rate, and MRF type is selected. A single reprocessing and MSWI option was modelled.
[a] in post-separation, participation rate is used to model the amount of residual waste diverted to mechanical recovery
[b] in post-separation, there is only one selection rate

2.3. Mass-based Assessment

2.3.1. Mass based data related to PCPPW management systems

S1. Collection

S1.1. collection scheme

The plastic fraction can be collected according to three collection schemes (KS, DO and PS). These scenarios will be further detailed during the economic assessment.

Kerbside collection (KS)

In KS collection, plastic is collected commingled with metals and beverage cartons. The residents simply deposit a bag (provided by the municipality) with their separated PMD waste outside their households, in the street side. Fortnightly, on the predetermined collection day, the inhabitants take the PMD bag out and the bags are picked up by the collection trucks.

Drop-off collection (DO)

The DO collection system requires the consumer to bring their separated plastic into a drop-off underground container located in the street. Once the container is full, the collection trucks collect the waste from the underground containers. The frequency of collection will depend on each scenario.

Post-separation (PS)

PS scenarios are based on the kerbside collection of plastics. In this case, however, the plastics are not separated and are collected with the residual fraction. Every week, the collection truck drives to collect the residual waste from the households. On the designated collection day, the residents will take the grey bin to the street to be collected by the waste collection trucks.

S1.2. source separation efficiency

The source separation efficiency is divided into participation and selection rate, see Eq. 1.

S1.2.a. participation rate

The participation rates used in this study are mainly based on the findings of Thoden van Velzen et al. (2019) on the collection behaviour of plastic packaging waste in several Dutch municipalities. The values are presented in Table V. The low and high participation rate values are selected from the literature and the average participation is the average of these two values. In the case of post-separation, this value is used to represent the residual waste fraction that is subject to the mechanical recovery of plastics. These levels of material directed to post-separation exceed the values currently seen in the Netherlands (22%, see section 2.1.2), but are in line with scenarios presented by Thoden van Velzen et al. (2013).

Table V. Participation rate of the alternative PMD collection systems.
[a] in post-separation, participation rate models the amount of residual waste that is directed to mechanical recovery

sorting scheme	low	average	high
KS	31%	60%	89%
DO	28%	45%	61%
PS [a]	50%	75%	100%

S1.2.b. selection rate

Selection rates at the individual packaging level were adapted from Thoden van Velzen et al. (2019) to achieve aggregated SR_{TP} performances of 75%, 83%, and 89%. The high-level selection rates are presented in Table VI, more detailed information about the performance on the individual packaging fraction can be found in the Supplementary Information. The concept of selection rate, by definition, does not apply to a post-separation collection scheme. In this collection system, all the PCPPW is assumed to be collected into the residual bin. Factors that could diminish the amount of waste collected, such as littering (it is estimated that 7 to 25 kton of plastics and plastic packaging are littered in the Netherlands each year (Broekaart, Afdeling, & En, 2017)) are not included due to lack of reliable data.

Table VI. Selection rate of the alternative PMD collection systems.
[a] in post-separation, selection rate is assumed to be 100% (the low, average, and high classification does not apply)

Sorting scheme	low			average			high		
	SR_{TP}	SR_{NTP}	SR_{imp}	SR_{TP}	SR_{NTP}	SR_{imp}	SR_{TP}	SR_{NTP}	SR_{imp}
KS	75%	48%	5%	83%	43%	4%	89%	41%	3%
DO	75%	48%	5%	83%	43%	4%	89%	41%	3%
PS [a]	-	-	-	-	-	-	-	-	-

S2. Sorting

The separately collected material in KS and DO collection systems and the fraction of residual waste directed to recovery in PS schemes are sent to an MRF. The remaining waste fraction is assumed to be sent to municipal solid waste incineration (MSWI). This will be further detailed in S4.

Two different MRF are defined in this study. One type is used in the separate collection schemes (MRF_{SC}), and the other is designed for the recovery of plastic from residual waste (MRF_{PS}). The higher amount of impurities fed in the PS collection system is translated into a lower sorting efficiency of the MRF_{PS}. Finally, depending on the level of processing technology, sorting plants have lower or higher performances (Cimpan et al., 2016). Three levels of MRF performance are defined for each sorting facility in this assessment, basic, medium, and advanced. The categories reflect an increasing sorting performance as well as an increasing technological complexity. This will have implications in the economic and environmental assessment, that will be discussed later. The basic MRF produces PET, PE, and mixed plastic products, the medium MRF produces PET, PE, PP, film, and mix plastics, and the advanced MRF sorts the input material into PET, PET trays, PE, PP, PS, film, and mixed plastics. In addition, all the MRF separate beverage cartons and metals into sorting products as well. These products are baled and sold to the reprocessing plants. The remaining material are considered sorting residues and are sent into incineration.

The recovery efficiencies of the sorting plants are based on Brouwer et al. (2019) and adjusted to the different MRF categories. Table VII details the efficiency of the MRF's in sorting the targeted packaging material into the desired output. For example, in Table VII, this means that 68% of the PET bottles entering the basic MRF_{SC} from a separate collection system (KS or DO) will end up in the PET product. The sorting efficiencies of the unintended products are also considered in the assessment to determine the levels of impurities found in each sorting product. More information on the sorting efficiencies and the sorting fates is detailed in the Supplementary Information.

Table VII. Efficiency (%) of sorting targeted fraction into desired products of the different MRF. Empty values (-) mean that the type of MRF does not produce that certain output.
[a] BC = beverage cartons

		Sorting efficiency (%)					
		MRF_{SC}			MRF_{PS}		
sorted fraction	ideal sorting fate	basic	medium	advanced	basic	medium	advanced
PET bottles	PET	68%	79%	90%	57%	66%	75%
PET rigid	PET tray	-	-	80%	-	-	45%
PE bottles	PE	63%	76%	90%	42%	50%	64%
PP bottles	PP	-	67%	90%	-	31%	41%
PS	PS	-	-	70%	-	-	35%
PE film	film	-	70%	95%	-	41%	54%
PET, PE, PP, rigid	mix	70%	40%	5%	71%	49%	5%
BC [a]	BC [a]	81%	85%	90%	26%	28%	30%
metals	metals	77%	83%	90%	77%	83%	90%

S3. Reprocessing

The separate valuable outputs of the MRF enter the reprocessing stage, where the sorted fractions are grinded, washed, dried, and extruded into recycled pellets. This is the final step before introducing the recycled material again into the manufacturing cycle. Reprocessing residues are once again considered to be incinerated in an MSWI plant.

Up until reprocessing, the recycling system operates at the packaging level. Both inhabitants and sorting equipment classify the packaging waste, but the integrity of the packaging is not deliberately affected. Plastic packaging is often a multi-material product. The body of PET plastic bottles, for instance, is made from PET, but the cap and sleeves are commonly made from HDPE and PP, respectively. In the reprocessing stage, however, plastic packaging is grinded into flakes made of a single polymer and the focus shifts from the packaging level to the material level. The composition of the shredded flakes at the material level is calculated using the material composition of each individual packaging from Brouwer et al. (2018), see Supplementary Information.

To simplify, only one reprocessing technology has been defined in this study. This corresponds to an advanced mechanical reprocessing system as defined by Faraca et al. (2019). It is assumed that the sorted materials are sent to a similar RP. The technical performance of the RP is equivalent, but due to higher contamination in PS collection systems, the costs of reprocessing are not the same, see section 2.4.1 Thus, two RP are defined, $RP_{KS/DO}$ and RP_{PS}. The technical yield of the last stage of the recycling process was taken from Brouwer et al. (2018), and are based on the efficiency of density separation technologies, see Table VIII. The floating fraction from the PET, PET trays, and PS, and the sinking fraction of the PE, PP, film, and mixed plastics, are recovered and turned into recycled pellets.

Table VIII. Efficiency of the density separation step of the reprocessing stage (both $RP_{KS/DO}$ and RP_{PS}).
[a] p&c = paper and cardboard

	Separation efficiency (%)											
fraction	PET	PE	PP	PS	PVC	other plastic	metal	organic	textile	p&c [a]	glass	rest
sinking	99%	1%	2%	83%	80%	50%	100%	50%	50%	5%	100%	
floating	1%	99%	98%	17%	20%	50%						
waste								50%	50%	95%		100%

S4. MSWI

The materials that do not end up being recycled, i.e. the residual waste fraction of the collection stage, and the residues from the sorting and reprocessing, are incinerated in a municipal solid waste incinerator (MSWI). This is conformant with current practices in the Netherlands, see section 2.1.2. Plastic is only a fraction of the materials that feeds into the MSWI but, in agreement with the functional unit, only the plastic contained in the incinerated fraction is considered in the assessment.

2.3.2. Assessment Criteria

This assessment focuses on the quantities of plastics that flows through the different stages. The technical performance is the most fundamental analysis of the recycling system and it is most commonly used in policy-making to define targets, e.g. recycling targets (EC, 2018c)

Three indexes are used to compare the mass-based performance of the different scenarios. These are associated with each of the main stages of the recycling chain, and define the ratio of materials over the total waste generated that are collected separately (collection rate, CR), the materials that have been sorted to reprocessing (intermediate recycling rate, iRR), and the materials that are recycled in useful applications (recycling rate, RR). A graphical representation can be found in Figure V. In the case of post-separation systems, where collection rate does not apply, the amount of materials diverted from incineration or landfill into mechanical recovery (diversion rate, DR) will be used instead, flow e in Figure V. The precise definition of these metrics is presented below (units in brackets):

$$CR = \frac{\text{mass of separately collected PCPPW}}{\text{mass of total PCPPW}} \quad \langle\%\rangle$$

$$iRR = \frac{\text{mass of PCPPW sorted to reprocessing}}{\text{mass of total PCPPW}} \quad \langle\%\rangle$$

$$RR = \frac{\text{mass of secondary raw material}}{\text{mass of total PCPPW}} \quad \langle\%\rangle$$

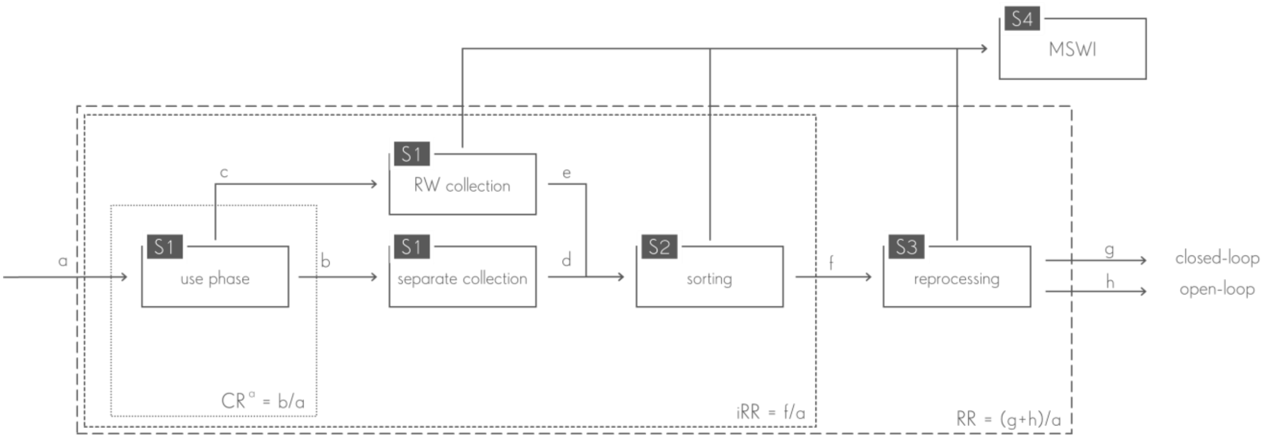

Figure V. Graphical representation of the material flows found in a generic waste management system. The dashed lines define the boundaries for the collection, rate (CR) intermediate recycling rate (iRR), and recycling rate (RR). RW=residual waste. MSWI=municipal solid waste incineration.
[a] in post-separation, collection rate is substituted by diversion rate (DR). DR = e/a.

Complementing these metrics, the streams are classified in four quality levels attending to their material composition. The quality of recycled polymers depends on their mechanical and chemical composition, content of hazardous substances, odour, strength, etc. (Eriksen et al., 2019; Villanueva & Eder, 2014), and will determine the applications for the secondary raw materials. Acknowledging these limitations, in this study, quality is determined based solely on polymeric purity of the material streams. DKR specifications (2020) and Eriksen et al. (2019) criteria were used to define quality levels of the MRF outputs. Due to lack of data on the specifications of the washed milled goods (WMG), their quality levels are defined according to the MRF quality level. It is assumed that the reprocessing of the MRF outputs increases the purity of the WMG but does not affect the quality levels. In other words, a high quality MRF output will be reprocessed into high quality WMG. In Table IX, the minimum polymeric purity is detailed for the three quality levels: high, medium, and low. The outputs that do not reach these minimum standards are considered not recyclable and will be downgraded and used in open-loop applications. These minimum purity standards are defined attending to maximum levels of impurities that are detailed in the Supplementary Information.

Table IX. Quality levels based on the minimum polymeric purity of the product for the outputs of the sorting stage (MRF). Empty values (-) mean that the type of output cannot achieve certain quality levels.

		minimum purity (%)						
	quality	PET	PET trays	PE	PP	PS	film	mix
MRF output	high	92%	88%	88%	92%	91%	92%	-
	medium	86%	82%	80%	76%	88%	90%	90%
	low	80%	78%	64%	66%	81%	84%	84%

2.4. Economic Assessment

2.4.1. Economic data related to PCPPW management systems

S1. Collection

Collection costs are modelled based on Groot et al. (2014). There are nine collection scenarios for the separate collection schemes and three for PS. This adds up to 21 total configurations of participation rate, selection rate, and collection scheme. The model estimates the costs of each collection scenario based on four categories: vehicle costs, labour costs, and bags and container costs. All cost estimations are calculated for the Netherlands, considering the total number of households, distances, and plastic waste generated, and then allocated to each scenario according to the functional unit. The essential details of the calculations are presented here, for the complete description of the collection costs see the Supplementary Information.

a. Vehicle costs

Vehicle costs are subdivided into operational and investment costs. The operational costs consist of the costs of fuel and maintenance, insurance, and tax costs. The latter are yearly fixed costs associated to each vehicle. Total fuel costs, on the other hand, vary depending on the collection routes. Collection routes are modelled depending on the collection scheme. These are a function of the distance travelled or the time spent in each of the three main collection activities (driving between stops, idling during a stop, and hauling), the fuel consumption during each of these activities, and the fuel price. KS and PS collection systems are both based on the regular collection of waste from the kerbside of each household. The total collection distance depends on the number of stops required to collect the plastic waste, which is determined multiplying the number of households by the frequency of collection, and the distance between stops (see next section). In KS collection the separate fraction is collected fortnightly, while residual waste is collected on a weekly basis (CBS, 2020). In the case of DO collection, the frequency of collection is a variable parameter that depends on the capacity of the underground containers. Investment costs are derived from the needs of the collection routes. The number of vehicles needed to fulfil the collection requirements is determined by dividing the time required for collection by the time a vehicle can be used per year (3000 h/year with a 20% downtime due to maintenance or inefficiencies of the collection scheme). KS and PS collection trucks are loaded manually, while DO trucks are loaded automatically. The manual loading makes KS and PS trucks cheaper to buy and operate, see Table XI. Depreciation period of the vehicles is 5 years (Cimpan et al., 2016; Groot et al., 2014).

Relation between participation and distance between collection stops

Participation on drop-off collection systems has been negatively correlated to the distance to the deposit point (Gallardo et al., 2010; Struk, 2017). When the distance to the drop-off point decreases, the participation rate increases. In the case of KS and PS, higher participation implies that more waste is collected per stop, which means that the collection truck gets full faster, and more trips are needed to the transfer station. Increased levels of participation in KS collection systems have been correlated to different levels of urbanism (Thoden Van Velzen et al., 2013). Rural communities with fewer high-rise buildings and larger distances between collection stops exhibit higher participation levels. These correlations between participation and distances between containers, or urbanity levels are translated into the model. In KS and PS collection systems it only affects the distance between collection stops, while in DO collection systems it influences the number of containers. The variables used in the cost calculations can be found in Table X and in more detail in the Supplementary Information.

Table X. Distance between stops depending on the collection system and the level of participation.
[a] in KS collection, higher participation levels are associated with less urbanized areas and longer distances between collection stops.
[b] in DO collection, higher participation levels are associated with smaller distances to drop-off points and higher container density (container/m^2)

	distance between stops (km)		
participation	KS[a]	DO[b]	PS
low	0.15	2.5	0.175
average	0.175	1.5	0.175
high	0.2	1.0	0.175

b. Labour costs

The costs of personnel during the collection stage that are accounted in this model are the wages of drivers and loaders. No assumptions were made with regards to administration costs, management costs, etc. In KS and PS collection schemes, the collection trucks stop in front of the houses and the loaders put the waste bags into the truck. This operation requires one driver and two persons to load the waste into the vehicle. DO collection systems use a mechanised loading system. This collection system is less labour intensive since it only requires a driver to operate the truck. The salary is set to be 30.000 €/year for the drivers and in 25.000 €/year for loaders for a total of 165 working hours per year (Groot et al., 2014).

c. Bag and container costs

Kerbside collection uses a plastic bag that is provided by the municipality and thus is included in the system's costs (Groot et al., 2014). The cost of a plastic bag is 0.055 €. Drop-off underground containers have a capacity of 5 m^3 (approximately 750 kg[2]) with an investment cost of 10.300 € and 60 € of maintenance per year (Rodrigues, 2016). The PS scheme uses 240l containers, at a cost of 58 €/container. No maintenance costs are allocated to these containers. Depreciation period of the containers is 10 years (Bertanza, Ziliani, & Menoni, 2018; Groot et al., 2014)

d. Value of the collected materials

The value of the collected materials is determined by the gate fee that the MRF charges to process the incoming waste. These gate fees vary highly between countries, materials, performance of the MRF, or contamination of the waste stream (Cimpan et al., 2016; Nolan ITU, 2004; Villanueva & Eder, 2014). In this assessment, the gate fee is calculated based on the net costs of the sorting facility, see next section.

[2] Density of the PMD waste in a container is estimated to be approximately 150 kg/m^3 (WRAP, 2010)

Table XI. List of parameters used in the calculation of collection costs

parameter	KS	DO	PS	units
vehicle				
cost of a vehicle	206000	250000	206000	€
salvage cost of a vehicle	30900	37500	30900	€
depreciation (vehicle)	5	5	5	year
insurance	2500	2500	2500	€/year
tax	1000	1000	1000	€/year
maintenance of the vehicle	3000	4000	3000	€/year
availability of vehicle	80%	80%	80%	%
use of vehicle	3000	3000	3000	h/year
average speed (hauling)	60	60	60	km/h
average distance (to hauling)	18	18	18	km
average speed (driving)	25	40	15	km/h
households per kerbside point	10	0	10	-
average time per stop	0.014	0.5	0.069	h/stop
average truck load per collection round	1800	750	7200	kg/round
fuel consumption (driving)	0.33	0.25	0.4	l/km
fuel consumption (idling)	4	4	4	l/h
fuel consumption (hauling)	0.25	0.25	0.33	l/km
fuel price	1.35	1.35	1.35	€/l
labour				
drivers per vehicle	1	1	1	-
loaders per vehicle	2	0	2	-
working hours (driver)	1650	1650	1650	h/year
working hours (loader)	1650	1650	1650	h/year
driver wage	30000	30000	30000	€/year
loader wage	25000	25000	25000	€/year
container and bags				
capacity (underground container)	0	750	0	kg
capacity (container 240l)	0	0	50	kg
cost of underground container	0	10300	0	€
cost of container 240l	0	0	58	€
cost of a bag	0.055	0	0	€
maintenance of container	0	60	0	€/year
depreciation (container)	0	10	10	year
other				
interest rate of the investment	5%	5%	5%	%
frequency of collection	26	depends on the scenario	52	times/year

S2. Sorting

The economic performance of an MRF is determined by the difference between the costs of investment and operation of the sorting plant, and the revenues obtained by the sale of sorted materials. This usually entails net costs for the MRF, so, in order to balance the business model, MRF's charge a gate fee at the reception of waste from the collection companies (Cimpan et al., 2016). In this study, this gate fee is calculated to simply cover the extra costs and achieve a net zero cost. Gate fees vary between collection systems and MRF's, meaning that the value of the collected materials is not only dependent on the system of collection, but also on the next stage on the recycling chain.

a. MRF costs

The three MRF scales (basic, medium, and advanced) use different sorting technologies and configurations, and produce different outputs. In addition, MRF_{PS} use one extra trommel at the beginning of the plant that incurs in additional sorting costs. The capital and operational costs are retrieved from Cimpan et al. (2016). The total average costs of operating an MRF_{SC} are 112, 89, and 97 €/ton for the basic, medium, and advanced configurations, respectively. For the MRF_{PS}, they are 129, 106, and 113 €/ton, respectively. More details about the cost distribution and the assumptions of each configuration can be found in the Supplementary Information.

.

b. Value of MRF outputs

Revenues from the MRF come from selling the sorted products to reprocessing facilities, and by charging the gate fee to collection companies for receiving their waste. The value of the outputs depends on the quality and the market for a certain recycled material (Villanueva & Eder, 2014). Prices of sorted bales are defined for each product and each of the four categories of plastic qualities defined in the mass-assessment. The values are selected from high and low values found in the literature (Cimpan et al., 2016; Mastellone, 2020; plasticker, 2020). They are presented in Table XII. The revenues from selling metallic output of the MRF is an important aspect of the business model and were included in the calculations along with beverage cartons (BC) and the disposal of residues.

Table XII. Market value of the MRF outputs according to quality levels. BC = beverage cartons
[a] BC, metal, and residues only have one quality level. It is included in the not recyclable quality level for calculation purposes.

	Market value (€/ton)									
quality	PET	PET trays	PE	PP	PS	film	mix	BC [a]	metal [a]	residues [a]
high	190	120	240	224	120	190	-	-	-	-
average	155	110	215	162	110	120	-	-	-	-
low	120	100	190	100	100	50	0	-	-	-
not recyclable	0	0	0	0	0	0	-30	0	200	-50

S3. Reprocessing

The modelling of the reprocessing plant is similar to the sorting plant. In this case, however, reprocessing plants do not charge a gate fee for receiving the material. Instead, they buy the baled goods from the MRF.

Two reprocessing plants were modelled, $RP_{KS/DO}$ and RP_{PS}. As seen in section , the technologies used are the same, but the higher contamination of the PS stream incurs in higher washing costs (Thoden Van Velzen et al., 2013). The annual costs of operating the plant, accounting for investment and operational expenditures, are estimated to be 312 and 327 €/ton, for the $RP_{KS/DO}$ and RP_{PS}, respectively, based on Faraca et al. (2019) and consultations with experts[3]. The value of the recycled materials is once again dependent on the quality of the reprocessed outputs. The market value of the washed milled goods is estimated from high and low values found in the literature (Faraca et al., 2019; plasticker, 2020).

[3] Dr. Norbert Fraunholcz, Founder and Managing Director of Recycling Avenue BV, personal communication, June 14, 2020.

Table XIII. Market value of the WMG according to quality levels

	Market value (€/ton)						
quality	PET	PET trays	PE	PP	PS	film	mix
high	1000	675	880	870	890	720	-
average	825	513	730	720	750	625	-
low	650	350	580	570	610	530	200
not recyclable	250	250	250	250	250	250	100

S4. MSWI

The non-recycled materials are incinerated with energy recovery. Electricity and heat are produced from the incineration of the plastic residues and sold to the market. The efficiency of the WtE process is 17% and 27% for electricity and heat generation, respectively, based on the average performance of Dutch WtE facilities (Rijkswaterstaat, 2018). The lower heating value of the plastic stream is assumed to be 25 MJ/kg (Burnley & Coleman, 2018). The selling price of the electricity produced by the MSWI is 0.025 €/kWh and the price of heat is 0.005 €/MJ (Faraca et al., 2019). Finally, the investment and operational costs of running the MSWI are estimated to be 155 €/ton of material burned, based in Faraca et al. (2019). This includes both the capital costs of building the MSWI plant and the operational costs of running the incineration process, such as electricity, personnel, or carbon taxes, among others. More details can be found in the Supplementary Information.

2.4.2. Assessment Criteria

Economic sustainability and value retention are fundamental components of the circular economy (EC, 2018a; Milios, 2018). Thus, economic criteria are also commonly used to compare the performance of waste management scenarios. In this study, two economic-based indicators are used to analyse the waste management systems, the first one focuses on the net costs of treatment per functional unit (cost index, CI), and the second one on the value recovered (circular economy index, CEI). The CEI is directly taken from Di Maio & Rem (2015). The formulas for the calculation are detailed below:

$$CI = \frac{\text{net cost of treatment of 1 ton of plastic waste}}{\text{1 ton of plastic waste}} \quad \left({}^{€}/_{\text{ton}} \right)$$

$$CEI = \frac{\text{value of secondary raw material}}{\text{virgin value of materials in the waste stream}} \quad (\%)$$

The virgin value of the materials in the waste stream is 1332.4 €/ton based on polymer prices in the European market (PIE, 2020; plasticker, 2020).

The MFA serves as the basis for the economic evaluation. The cost of collection, sorting, reprocessing, and incineration are scaled based on material throughput, and are linearly adapted to material flows of each scenario For instance, scenarios with low participation require less expenses allocated to separate collection and more to residual waste collection. This means that if 40% of the plastic waste is collected separately and the rest is collected in the residual waste, for every FU, 400 kg will be collected at the cost of separate collection and will continue to the MRF, and 600 kg will be collected at the residual waste collection costs and will go directly to incineration. The same applies to the recovered value where the quality and quantity of the WMG will determine their market value.

2.5. ENVIRONMENTAL ASSESSMENT

2.5.1. Environmental data related to PCPPW management systems

The main source of life cycle inventory data is the ecoinvent database 3.4. (Wernet et al., 2016). The system boundaries include the recycling chain from the collection to reprocessing of recycled pellets. Foreground processes have been adapted from the database to meet the characteristics of the product system of analysis. European or Dutch markets were selected for the relevant background processes, such as transportation, electricity or heat generation. The production of primary materials was selected from the global market, since it is assumed that recycled plastics can substitute virgin materials from any origin.

The consumption of electricity, heat, fuel, and water of the different scenario depends on the specific configuration and material flows. The use of resources of machinery and vehicles is scaled based on material throughput. This way, the resource needs can be linearly distributed based on the MFA. Table XIV shows the relevant resource consumption data. More information is available in the Supplementary Information.

Table XIV. Resource consumption per collection scheme used in the environmental assessment

stage	resource	level	KS	DO	PS	unit
collection	diesel	collection truck ⟨driving⟩	0.33	0.25	0.4	l/km
		collection truck ⟨idling⟩	4	4	4	l/h
		collection truck ⟨hauling⟩	0.25	0.25	0.33	l/km
sorting	electricity	basic	102.4	102.4	150.7	kWh/ton
		medium	89.8	89.8	138.1	kWh/ton
		advanced	96.5	96.5	144.8	kWh/ton
	diesel	basic	3.7	3.7	3.7	l/ton
		medium	2.2	2.2	2.2	l/ton
		advanced	2.2	2.2	2.2	l/ton
reprocessing	electricity	-	197	197	197	kWh/ton
	heat	-	432	432	432	MJ/ton
	water	-	9	9	9	m^3/ton

Plastic bags are not included in the economic costs of DO and PS systems, but it is assumed that they are still used for the storage and transport of waste in these collection systems. In PS, one bag is used per collection. Since these same bags are also used to collect MSW, the number of bags used for PCPPW collection is allocated attending to the waste composition. In DO collection systems, inhabitants bring their waste to the drop-off container when the bag is full. Thus, the number of bags is calculated dividing the total waste by the capacity of a plastic bag, which is assumed to be 3kg of PCPPW per bag[4].

2.5.2. Substitution

The displacement of primary production from recovered resources is commonly used in environmental studies (Vadenbo et al., 2017). It is acknowledged that defining the effective displacement of primary plastic production and electricity and heat generation attends to complex market dynamics, beyond simple functional equivalence (Zink & Geyer, 2017). In this study, the substitution factors are taken from Haupt et al. (2018a). The researchers use the framework of Vadenbo et al. (2017) to determine the substitution potential of different quality grades of recycled polymers as a function of technical, institutional, or user-perception aspects. High quality recyclates are capable of displacing virgin plastic production, while lower grades are substitutes of wood products or used as aggregates in the production of concrete. The substitution factors and materials substituted are detailed in Table XV.

[4] Assuming the density of the PMD to be 150 kg/m^3 (WRAP, 2010) and using 20l bags.

Table XV. Substitution factors of the materials and energy recovered based on quality levels

product	quality	material / energy substituted	substitution factor	comments
PET	high	PET bottle grade	1	bottle produced from recycled PET
	medium	PET bottle grade	0.7	thickness increases in recycled products
	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
PET trays	high	PET amorphous	0.95	thickness increases in recycled products
	medium	PET amorphous	0.7	thickness increases in recycled products
	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
PE	high	PE	0.9	thickness increases in recycled products
	medium	PE	0.7	thickness increases in recycled products
	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
PP	high	PP	0.9	thickness increases in recycled products
	medium	PP	0.7	thickness increases in recycled products
	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
PS	high	PS	0.9	thickness increases in recycled products
	medium	PS	0.7	thickness increases in recycled products
	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
film	high	PE film	0.9	thickness increases in recycled products
	medium	PE film	0.7	thickness increases in recycled products
	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
mix	low	wood	0.05	1 kg substitutes 0.05 EURO pallets (22kg)
	not recyclable	concrete	0.001	1 kg substitutes 0.001 m3 concrete
electricity		electricity, low voltage, NL	1	Dutch electricity mix
heat		heat, district heating, RER	1	European district heating network

2.5.3. Assessment Criteria

The environmental impact assessment used in this study follows the guidelines of the ILCD (EC-JRC-IES, 2010). The selection of impact categories has to be consistent with the goal of the study, covering all the relevant environmental impacts for the analysed system. In this environmental assessment, the ReCiPe characterisation method is used since it has been developed in collaboration with the Dutch National Institute for Public Health and the Environment (RIVM) and it is often used in LCA's (Huijbregts et al., 2016). The included impact categories are climate change (CC), fossil depletion (FD), particulate matter formation (PMF), human toxicity (HT), photochemical oxidant formation (POF), and terrestrial acidification (TA). The analysis is focused on impact categories that are relevant to current Dutch policies such as climate change and the air quality (PMF). The remaining categories are selected due to their relevance to in the waste management sector and in line with other studies (Faraca et al., 2019; Rigamonti et al., 2014; Thoden Van Velzen et al., 2013). The remaining categories are selected due to their relevance to in the waste management sector and in line with other studies (Faraca et al., 2019; Rigamonti et al., 2014; Thoden Van Velzen et al., 2013).

3. Results

3.1. Mass-based Assessment

The performances of the different collection routes are shown in Table XVI. Collection rates are highly defined by the level of participation rate. Within collection schemes, the biggest differences in CR are found between low, average, and high participation rates. The maximum recycling rates are achieved in KS collection schemes, with a highest of 47% recycled material. Post-separation schemes have high diversion rates, but the higher contamination of the collected materials dramatically reduces the amount of materials that are recovered in the subsequent stages. The RR achieved from DO and PS systems are in the same order of magnitude with a maximum of 32% and 35% respectively. The importance of the point of measurement becomes apparent when comparing iRR and RR. PS systems achieve higher iRR on average than DO collection systems, but, again, the higher levels of contaminations and attached moisture and dirt of this collection alternative is reflected in the RR's.

Table XVI. Results of the average mass-performance of the different collection methods.
[a] in post-separation, collection rate represents diversion rate

	CR		iRR		RR	
CODE	average	std	average	std	average	std
KS	46%	*19%*	41%	*17%*	27%	*12%*
DO	34%	*11%*	31%	*10%*	20%	*8%*
PS [a]	75%	*22%*	29%	*10%*	19%	*9%*

Incorporating the quality perspective adds an extra dimension to the mass-based indicators. In general, most of the materials that are considered recycled are being downcycled or used in low-quality applications, see Figure VI. The sorting method is the most influential factor in the quality of the recovered materials. Basic MRF's yield low quality recyclates and in lower quantities, while advanced MRF's recover more materials and with higher purity. No quality-quantity trade-off was found between the different sorting options. The higher number of outputs of the advanced MRF have some influence in this result, however, the main explanation is the role of technological progress. Advanced sorting technologies can detect the target material more accurately and yield higher qualities and quantities of sorted materials at the same time (Nijkerk & Dalmijn, 2001).

Selection rates are less determinant than participation rates in the technical performance of the waste management systems. An increase in selection rates reduces the number of contaminants in the collected stream and increases the amount of targeted materials that is recovered. However, according to the model, the higher quality of the collected materials does not have a significant impact in the quality of the materials at the end of the recycling chain.

In PS, the higher contamination on the collection stage does not seem to affect the quality of the final products. The share of high-quality materials is the highest in this collection system. KS and DO collection systems achieve equivalent qualities of the recycled materials, but the quantities recovered in the KS collection system are higher. The low participation of DO collection systems limits the recovery potential of this collection alternative.

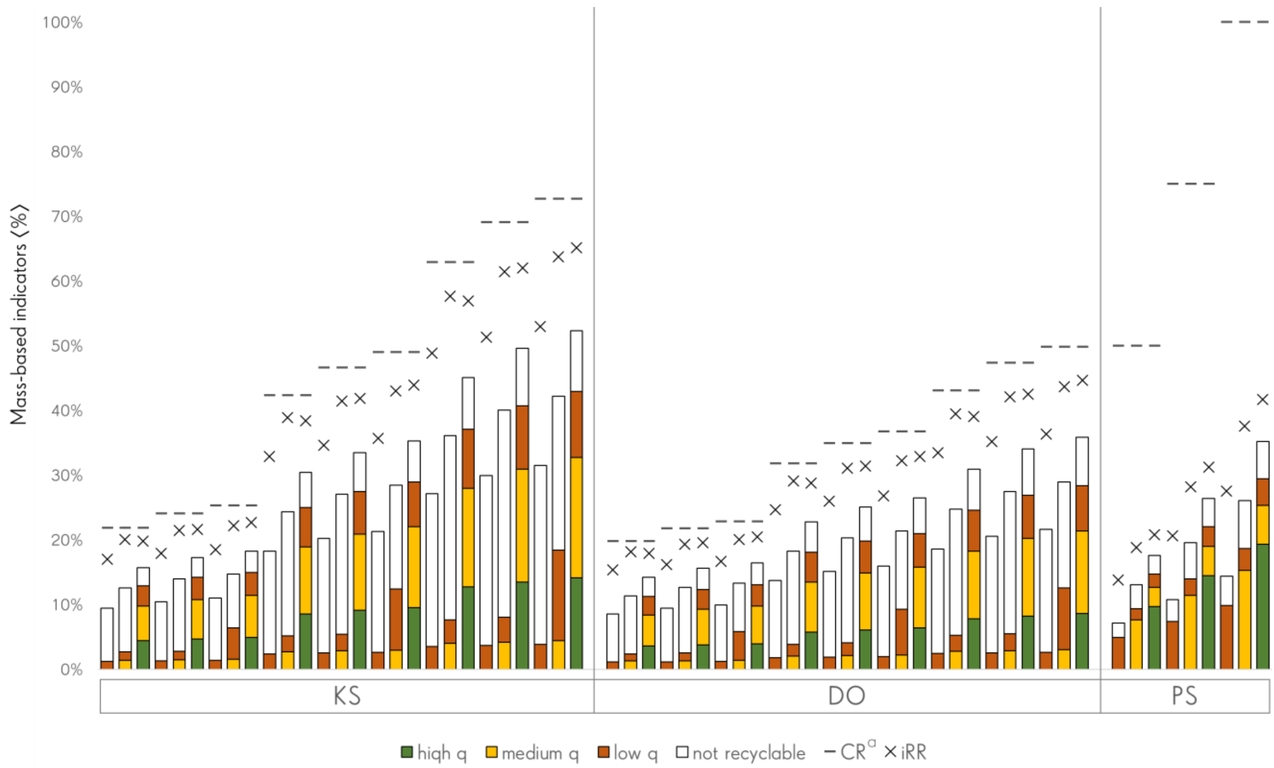

Figure VI. Mass-based performance of all scenarios. The recycling rate is divided into quality levels. The top of the bar indicates the RR value as defined in section 1.5. The order of the scenarios is given in the Supplementary Information KS = kerbside, DO = drop-off, PS = post-separation.
[a] in post-separation, collection rates is substituted by diversion rate.

3.1.1. Validation of the mass-based assessment

The model is used to calculate the mass-based performance of the Dutch network as seen in Figure II. The average values from all the scenarios presented in Table XVI are weighted to calculate the average recycling network. The mass collected in the different collection systems was kept constant, 104 kton of PCPPW are separately collected in KS collection systems, 31 kton are collected in DO collection systems, and 41 kton are sent to PS. The first validation involves the coverage of separate collection systems, i.e. the number of households that have access to a separate collection scheme. The coverage is estimated by dividing the KS and DO collected masses by the average capture rates of the systems[5]. This calculation indicates that, to achieve this level of separate collection, 88% of the waste should be covered by a separate collection system, 63% KS, and 25% DO collection systems. This means that, according to the model, all the waste that is not directly collected in PS systems, is part of a separate collection scheme. No specific data was found on the coverage of separate collection systems in NL, but according to Dutch regulations, municipalities should decide between separate collection or post-separation of PCPPW packaging waste (Ministerie van IenW, 2019). Thus, the modelling of the collection schemes is considered positively validated.

The sorting and reprocessing stages present larger differences with the Dutch baseline. The modelled MRF produce slightly less sorted products, on average, and, more importantly, the quality is also lower. This is seen in the significant reduction in WMG produced in the reprocessing stage. The sorted materials from the MRF's have more impurities and non-targeted materials that are removed during the reprocessing stage, affecting the performance of the last stage of the recycling network. While iRR only decreased from 39% to 37%, RR decreased from 29% to 25%, when comparing the Dutch baseline case to the average scenario modelling. The difference could be explained by the weight given to each of the alternative sorting option. The NL is a frontrunner

[5] KS mass collected is 104 kton, average capture rate is 46%, thus KS coverage is 104/0.46 = 226 kton or 63% of the total PCPPW. DO mass collected is 31 kton, average capture rate is 34%, thus DO coverage is 31/0.34 = 91 kton or 25% of the total PCPPW.

in recycling technologies, and it is possible that the influence given to basic MRF's is disproportionate to their actual presence in the Dutch recycling network.

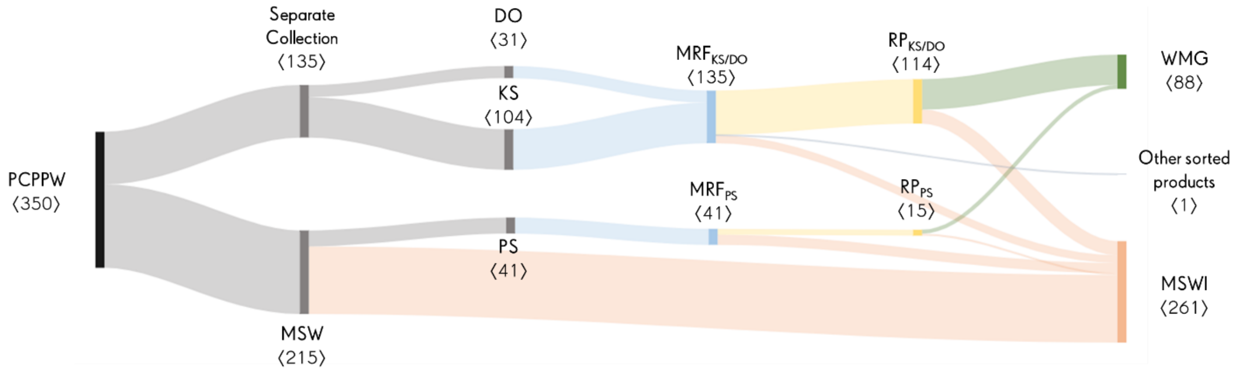

Figure VII. Sankey diagram showing the mass flows of the average of the modelled scenarios. In brackets, the mass in kilotons.
DO = drop-off. KS = kerbside. PS = post-separation. MRF = material recovery facility. RP = reprocessing plant. WMG = washed milled goods.
MSWI = municipal solid waste incineration. Other sorted products includes beverage cartons, metals, etc.

Table XVII. Comparison of the performance indicators for the Dutch recycling network and for the average of the modelled scenarios.

	mass-based indicators		
	CR [a] (%)	iRR (%)	RR (%)
NL	50%	39%	29%
Model (average)	50%	37%	25%

3.2. Value-based Assessment

Table XVIII shows the results for the performance indicators selected in the economic assessment. KS collection scenarios have higher costs in collection (S1), sorting (S2), and reprocessing (S3) than DO collection systems because more materials are collected separately and recovered in this system. For the same reasons, incineration costs (S4) are higher in DO collection since more plastics are sent to energy recovery in these scenarios. The high costs of the extra separation for the PS system are determinant in the overall costs of the collection scenario. The value recovered from selling the recyclates varies significantly among scenarios, see standard deviation in Table XVIII and Figure IX. The value recovered through incineration is an important contributor to the CEI with a steady 4% through the different collection schemes. The total value recovered in the system is the sum of S3 and S4. S1 and S2 represent intermediate prices of the collected or sorted materials. The CEI in S1 represents the value for the collected materials. This is negative because MRF's charge a gate fee to receive the collected products from the collection companies.

Table XVIII. Results of the average economic performance for the different collection schemes.
[a] in S4, CEI is used to represent the revenues generated by selling the electricity and heat (not accounted in the formal definition)

	Total Cost (€/ton)									
collection scheme	S1		S2		S3		S4		total	
	Av.	*std.*	Av.	*std.*	Av.	*std.*	Av.	*std.*	Av.	*std.*
KS	147	*9*	91	*38*	122	*51*	129	*22*	489	*60*
DO	123	*10*	68	*22*	91	*29*	141	*13*	422	*48*
PS	97	*0*	174	*53*	87	*30*	142	*16*	501	*67*

collection scheme	Cost Index (€/ton)									
	S1		S2		S3		S4		total	
	Av.	*std.*	Av.	*std.*	Av.	*std.*	Av.	*std.*	Av.	*std.*
KS	222	*34*	72	*38*	20	*51*	81	*22*	333	*60*
DO	179	*33*	54	*22*	16	*29*	89	*13*	291	*48*
PS	250	*49*	149	*53*	-12	*30*	90	*16*	346	*67*

collection scheme	Circular Economy Index (%)									
	S1		S2		S3		S4 [a]		total	
	Av.	*std.*	Av.	*std.*	Av.	*std.*	Av.	*std.*	Av.	*std.*
KS	-5%	*3%*	1%	*2%*	8%	*6%*	4%	*1%*	12%	*6%*
DO	-4%	*2%*	1%	*1%*	6%	*6%*	4%	*0%*	10%	*4%*
PS	-11%	*4%*	2%	*1%*	7%	*6%*	4%	*0%*	11%	*3%*

The detailed collection costs of the separately collected fraction can be seen in Figure VIII (a). The cost of KS collection decrease with increasing separation rates due to economies of scale and because it is assumed that a collection service has to be provided even if the citizens do not participate. The collection routes will still be equivalent regardless the participation levels. On the other hand, DO collection costs increase slightly with participation rates because more containers are installed to achieve the higher participation levels. The marginal costs of installing extra containers are higher than the benefits in terms of collection efficiency obtained from the higher collection rates. The overall costs of collection a ton of plastic waste, the costs per FU seen in Table XVIII, are the sum of the cost of collecting the separated plastic in the separate collection system and the cost of collecting the non-separated plastic in the residual waste collection.

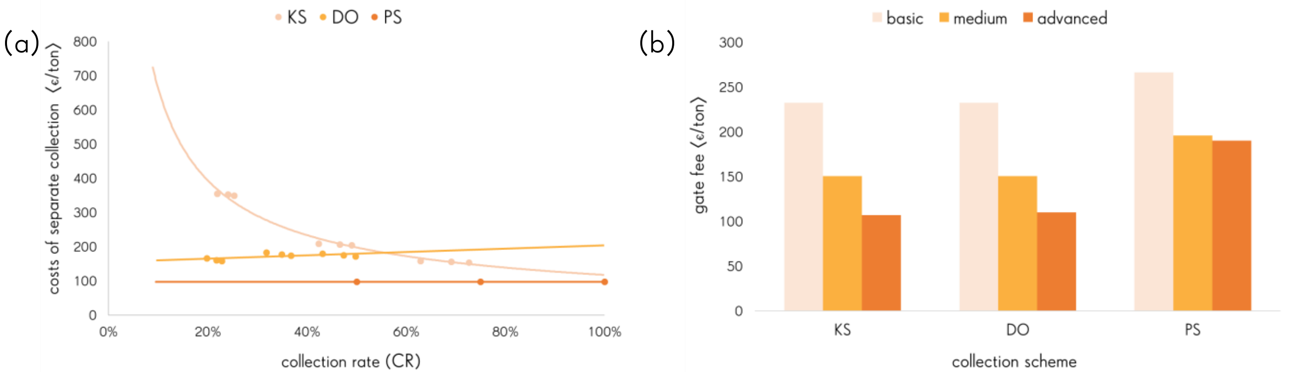

Figure VIII. (a) Cost of collection of the different collection system depending on the collection rate. (b) Gate fee per ton of plastic depending on the collection system and the MRF technology.

In Figure VIII (b), the gate fees are scaled to the functional unit of 1 ton of plastic. This makes the gate fees on this analysis higher than the ones reported in other studies where metals, beverage cartons and impurities are accounted for in the calculation of the sorting costs (Cimpan et al., 2016; Nolan ITU, 2004). There is no difference in the sorting costs of kerbside and drop-off collection systems, which is manifested in the almost identical gate fees. The lowest share of plastics found in post-separation systems, significantly increases the costs of sorting per ton of plastic. In addition, post-separation schemes produce more sorting residues and the disposal of these residues has a negative market value, which incurs in extra costs.

When analysing the scenarios in more detail, see Figure IX, it is noticed that net costs of the system (costs minus revenues, marked with an 'x' in the figure) follow a clear pattern. This is linked to the three technological levels of the MRF's. Advanced technologies, with higher recovery performances, recover more materials and of higher quality that end up being sold at higher prices. The overall costs of treating 1 ton of plastics remains relatively constant in the advanced scenarios, since, when more materials are recovered, the higher costs of sorting and reprocessing are compensated with the higher revenue obtained by selling those materials. This is not the case for basic and medium scenarios. In general, the net costs per ton of material recovered become smaller with more technological development. This allows for different business approaches to optimise the economic performance of the waste treatment.

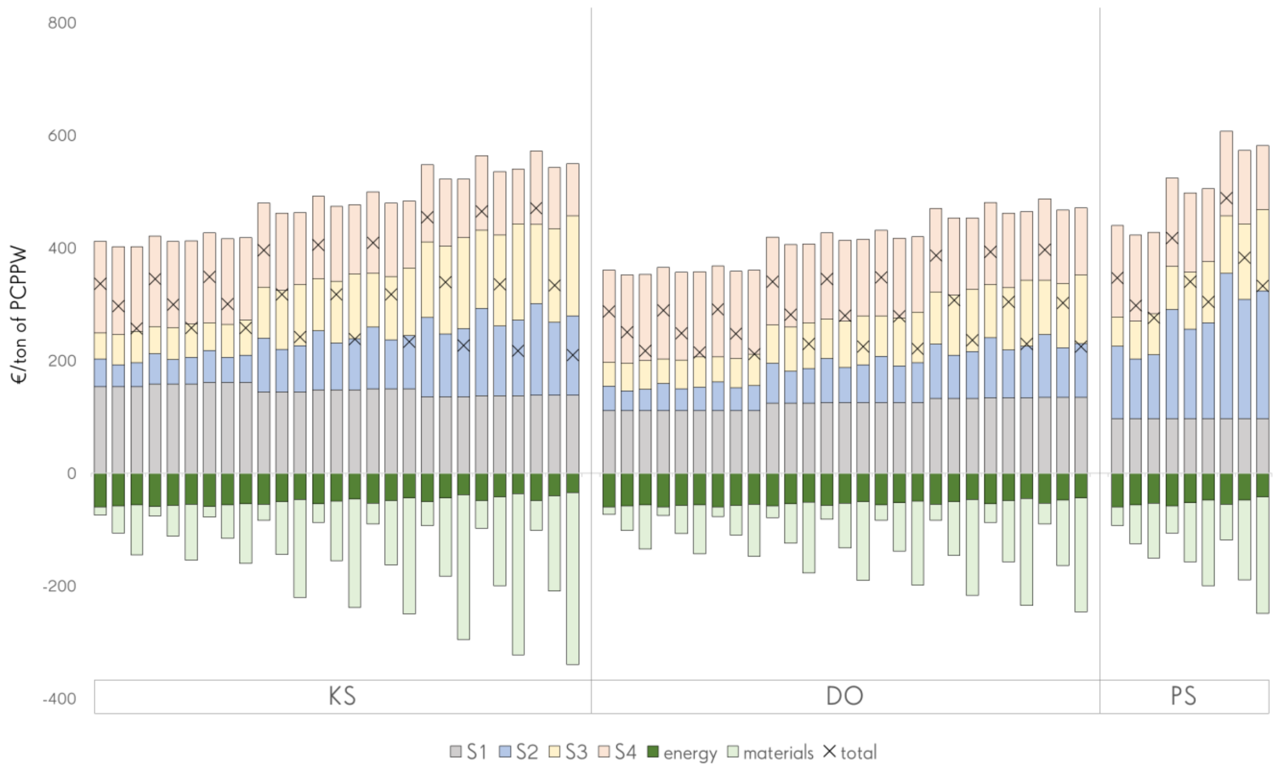

Figure IX. Economic-based performance of all scenarios. The negative values represent the revenues, which are divided into material and energy. The top of the orange bar indicates the total costs. The black cross (x) indicates the CI. The order of the scenarios is given in the Supplementary Information. KS = kerbside, DO = drop-off, PS = post-separation.

3.2.1. Validation of the value-based assessment

The model is used to calculate the value-based performance of the Dutch network as seen in Figure II. The average economic values from all the scenarios presented in Table XVIII are weighted according to the MFA of the modelled average recycling network from Figure VII.

The costs of managing the recycling network are slightly lower in the model when compared to the estimated range based on EPR tariffs. The disparity could be explained by the administrative, monitoring, or other hidden costs, such as the prevention of littering, in the recycling network that are covered by EPR tariffs (Afvalfonds Verpakkingen, 2019) but are not included in the model.

Table XIX. Comparison of the value-based performance indicators for the Dutch recycling network and for the average of the modelled scenarios.

	value-based indicators	
	CI (€/ton)	CEI (%)
NL	340 - 600	-
Model (average)	324	7%

3.3. ENVIRONMENTAL ASSESSMENT

The results for the environmental assessment are shown in Figure X for the different collection systems. MSWI is the dominant contributor in most of the impact categories in terms of both the negative impacts, from the incineration process, and the positive impacts as well, associated with the substitution of other forms of electricity and heat generation. On the other hand, most of the impacts of substitution of raw material are related to high quality materials. Their ability to displace more primary raw material production is translated into a better environmental performance of the system, see Supplementary Information. The impacts of the recycling chain are relatively small compared to the environmental burdens of incineration. The impacts of the collection stage (S1) are higher in the KS collection systems, and lower in PS. The opposite is true for the environmental impacts of the sorting stage (S2), the burdens of sorting the materials in a PS scheme are the highest from a life cycle perspective. due to the additional sorting steps.

The differences in the performance at the level of the collection system are not so significant. On average, PS systems have the lowest environmental impacts in FD, PMF, POF, and TA impact categories. The best performance in terms of CC and HT are achieved with KS collection. However, there is great variability on the environmental profiles of the different scenarios within a collection scheme (see Supplementary Information) which makes the comparisons between collection systems less significant. Nonetheless, the analysis indicates that to achieve the best environmental performance in any impact category it is fundamental to increase the amount of plastics that are incinerated and produce high quality secondary raw materials at the same time. This can be achieved with various collection schemes, but the KS separate collection scheme exhibits the biggest potential to optimise this environmental performance when participation and selection levels are high, and advanced sorting technologies are used. However, if MSWI is substituted by low quality recycling, this can have negative effects in the environmental performance of the waste management system. Interestingly, this implies that for all the impact categories but climate change, the worst performance is achieved precisely by a KS collection system with high source separation, but a basic MRF.

Climate change and human toxicity are the only impact categories with no average net savings. According to the model, while some scenarios achieve net savings in terms of HT, there is no scenario that yields a positive environmental outcome in terms of CC. On average, one ton of CO_2-eq is emitted to manage a ton of PCPPW, with 325 kg and 1301 kg being the best and worst scenarios, respectively. In these scenarios, the share of CO_2-eq emissions savings associated with material displacement varies significantly. Material substitution contributed with only 2% (minimum of all scenarios) of the avoided emissions in the worst performing scenario, while it accounted for 52% (maximum of all scenarios) for the benchmark configuration. This accentuates the importance of material substitution to achieve better environmental performances in waste management configurations.

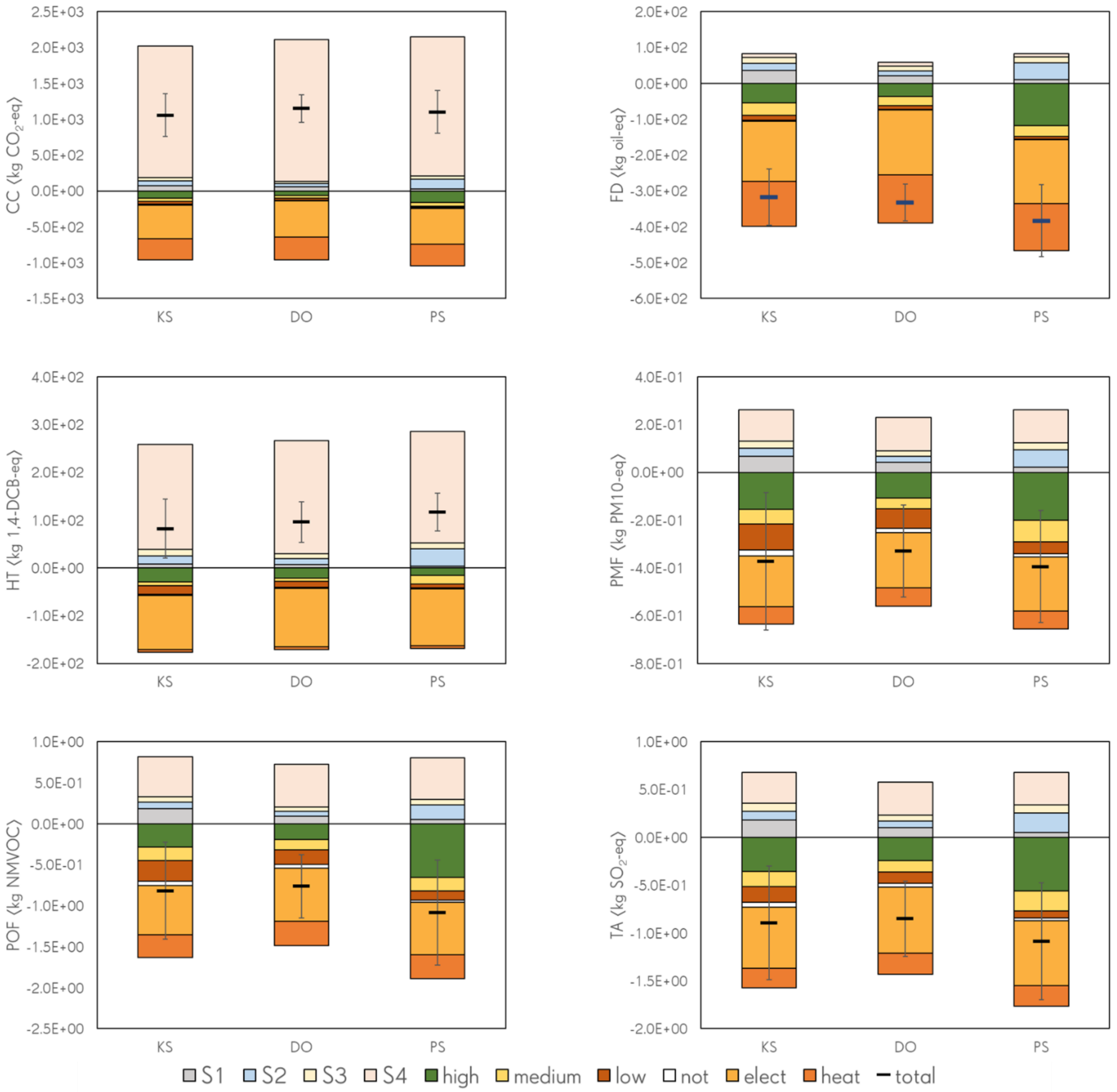

Figure X. Result for the average environmental impacts of the different collection systems. The error bars represent the variability in the performance of the different scenarios.
CC = climate change. FD = fossil depletion. HT = human toxicity. PMF = particulate matter formation. POF = photochemical oxidant formation. TA = terrestrial acidification.

3.4. Correlation between indicators

3.4.1. Correlation between environmental impacts and performance indicators

In Figure XI, the results for each scenario in terms of environmental impact are plotted against the iRR, RR and CEI of the scenario for each impact category. The figure shows that CEI seems to be more correlated to the environmental performance of the system for all studied categories. In particular, when the value recovered from the waste stream is higher, the environmental impacts decrease. The influence of the effective primary materials displacement achieved by high quality materials proves to be more represented in value-based indicators than in mass-based indicators. Moreover, the farther the point of measurement from the remanufacturing stage, the more uncertainty on the final performance of the system. This can be noticed in the gradually higher correlations found between iRR, RR, and CEI with the environmental performances.

Finally, for two of the most relevant categories in the study, such as climate change and particulate matter formation, the indicator shows the highest correlations (R^2>0.9). This supports the idea that the use of CEI as a metric to assess the performance of waste management systems is more aligned with current political interest.

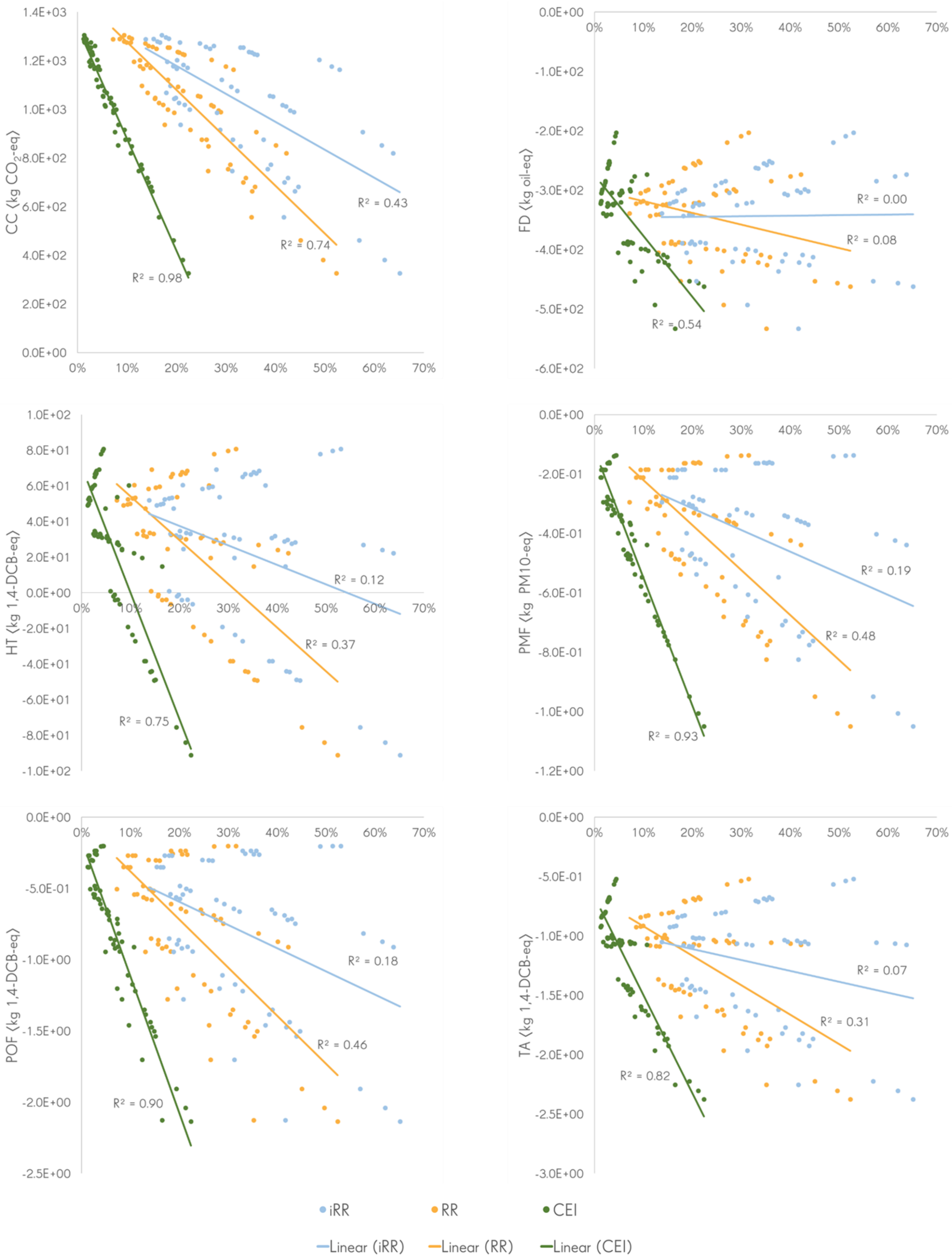

Figure XI. Correlation between environmental impacts and intermediate recycling rates (iRR, blue), recycling rates (RR, yellow), and circular economy index (CEI, green). Each point represents the performance of an scenario with the environmental impacts in the y-axis and the iRR, RR, and CEI in the x-axis. The lines represent the linear trends of the data points.
CC = climate change. FD = fossil depletion. HT = human toxicity. PMF = particulate matter formation. POF = photochemical oxidant formation. TA = terrestrial acidification.

3.4.2. Correlation between costs and performance indicators

Interestingly, there is a weak correlation between the CEI and the total costs and net costs (CI) of the systems, see Figure XII. Both iRR and RR seem to be more aligned with the total costs of the system than the value recovery index. However, these correlations are still relatively weak.

On the other hand, there is no apparent correlation between the indicators and the net costs of the waste management chain. The main factor that explains these weak correlations is the three different technological levels in the sorting stage. These incur in three distinct dynamics between costs and CEI. On the one hand, basic MRF's produce more low-quality products, with low impact in the CEI. The costs of using a basic MRF increase rapidly with small increases on the CEI. At the same time, the revenues obtained from these low-quality materials are not enough to cover the increased processing costs, so the net costs of this technological route also increase with higher recovery rates. On the other hand, advanced MRF's produce higher quality outputs, with high market value and high impact in the CEI. The total costs of using an advanced MRF increase more slowly with higher CEI. At the same time, the revenues obtained from the high-quality recyclates are more or less equal to the costs, which is translated to a rather stable net costs with increasing material and value recovery. More specifically, in the scenarios with advanced MRF's, if the collection method is KS, net costs slightly decrease, while in DO and PS systems, they slightly increase. Finally, the dynamics of medium MRF's are in between the basic and advanced levels.

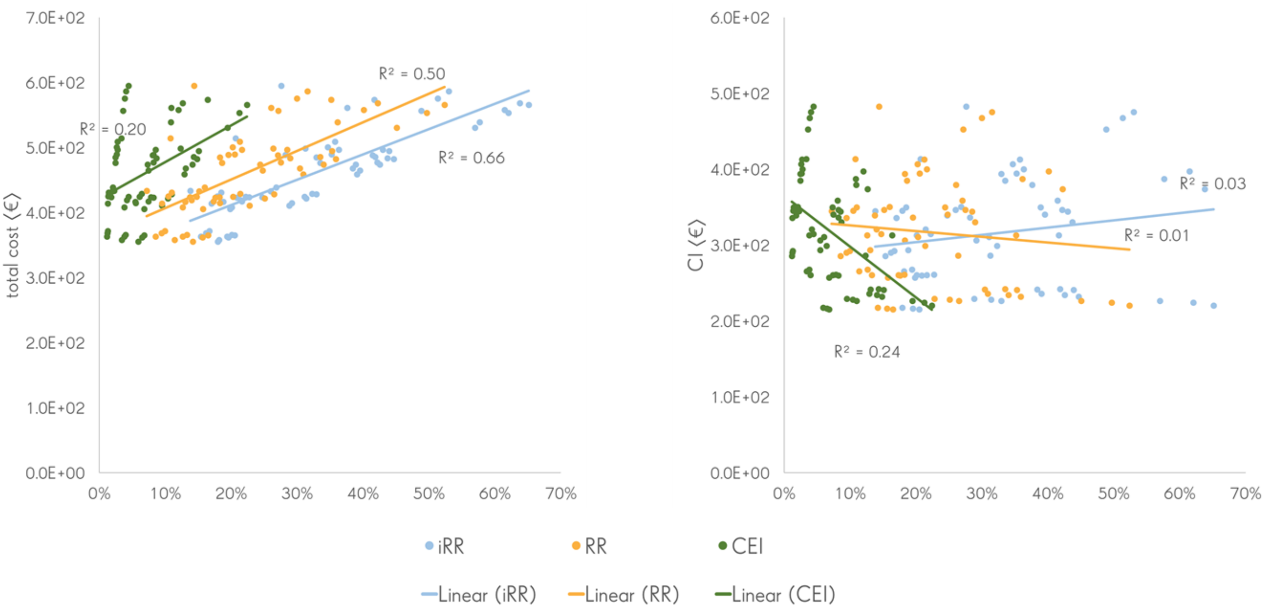

Figure XII. Correlation between costs and intermediate recycling rates (iRR, blue), recycling rates (RR, yellow), and circular economy index (CEI, green). Each point represents the performance of an scenario with the total cost (left) or the cost indicator (CI, right) in the y-axis and the iRR, RR, and CEI in the x-axis. The lines represent the linear trends of the data points.
CC = climate change. FD = fossil depletion. HT = human toxicity. PMF = particulate matter formation. POF = photochemical oxidant formation. TA = terrestrial acidification.

3.5. Sensitivity Analysis

The sensitivity analysis evaluates how critical are the assumptions made during the different assessment stages in the final results. Ultimately, this will affect the conclusions that can be drawn from these results. To this end, changes in the process data or methods used are intentionally introduced to detect and assess alterations in the results. Here, the sensitivity analysis focuses on the assumption made with regards to energy and material substitution, and the efficiency of MRF's.

The original electricity mix from the Netherlands is 50% natural gas, 2% oil, 29% coal, and 12% of renewables, including wind, solar, and biomass, and 7% of nuclear, non-renewable waste fraction, and other energy sources (CBS, 2017). Considering the current developments towards the decarbonisation of the energy sector, such as the European Green Deal (EC, 2019b), the first sensitivity analysis studies the effects of changing the electricity mix with a 100% renewable energy mix, with 30% biomass, 45% wind, and 25% solar energy. The effects of the change of the electricity mix on the environmental performance vary among the different impact categories. The performance on climate change and fossil depletion worsens by 34% and 43%, respectively, because the substitution of primary energy production by electricity produced in the MSWI is no longer environmentally beneficial. HT improves substantially, 193%, and the rest of the impact categories present some improvement as well, around 2-15%. There are slight variations between the different collection systems, but the conclusions remain the same, overall, the variation on the electricity mix shifts the problem towards the impacts of waste management treatments in climate change and the use of fossil fuels, while stressing the importance of material displacement and material recovery over energy recovery to achieve sound environmental performances. In other words, there seem to be better strategies for energy generation, but not so many for material production.

With regards to material displacement, the substitution factors of the recycled plastics are reduced by 10% to analyse how a lower material displacement of the closed-loop applications (high and medium quality) affects the environmental profile of the systems. Lower substitution factors increase the environmental impacts of the system for all the impact categories. However, this increase only represents 2-4% additional impacts overall. Scenarios with advanced MRF exhibit a bigger increase in environmental impacts, while basic MRF's remain mostly unaltered.

Finally, the last assumption that is subjected to a sensitivity analysis is the performance of the MRF_{PS}. Some studies indicate that PS systems are incapable of producing high-quality recyclates due to the high amount of impurities (Eriksen et al., 2019; Rigamonti et al., 2014). The model is adapted so the amount of impurities in the sorted products is increased by 50% while the sorting efficiency of the targeted materials and, consequently, the recycling rates are maintained. Under this configuration, PS collection systems do not produce high-quality materials anymore. This significantly increases the environmental impacts for all the impacts categories in ranges from 9-29%. Achieving lower quality recyclates also affects the economics of the system. The revenues from selling the secondary raw materials decrease by 15-30%, incurring in an increase in the overall costs of the system of 5%.

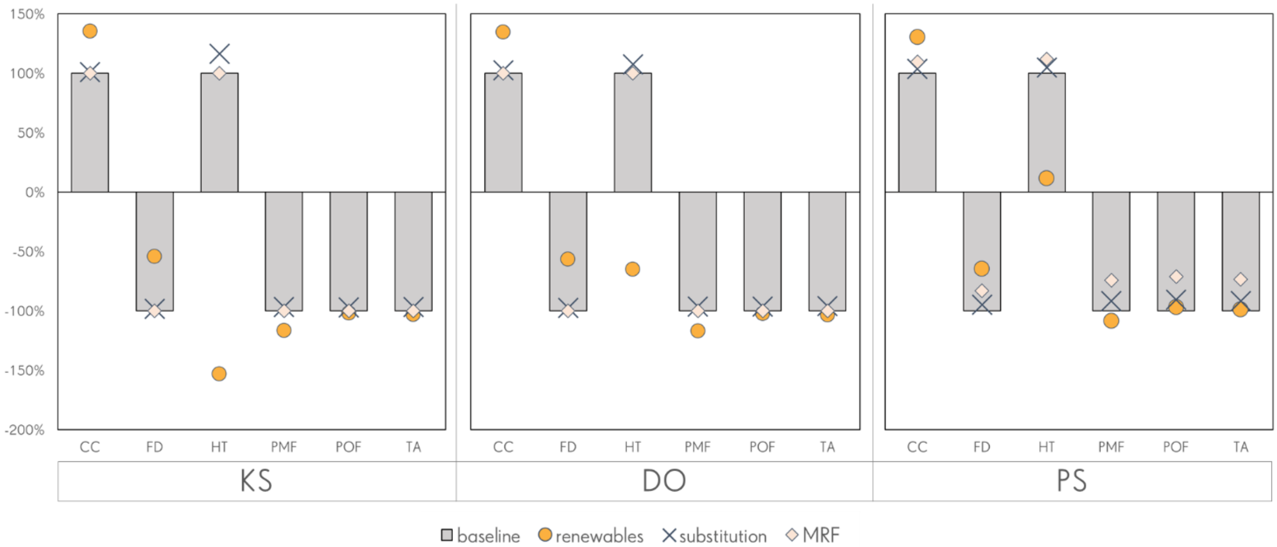

Figure XIII. Sensitivity analysis of the environmental impacts. The average value of the baseline scenario from Figure X is set to plus or minus 100%, depending if there are negative impacts or net savings in the impact category. The variation in the indicator is shown as a percentage.

3.5.1. Sensitivity analysis on the correlation

The correlations studied in section 3.4.1 are relatively maintained during these sensitivity analyses, with no significant variations on the correlation coefficients. This is interpreted as a robust alignment between CEI and environmental performance. The results can be seen in Table XX. The correlations from section 3.4.2 remain virtually constant and are, thus, not shown here. Only the sensitivity analysis on the MRF_{PS} affects the economic performance of the scenarios. These results can be found in the Supplementary Information

.

Table XX. Sensitivity analysis of the correlation between environmental impacts and CEI (top), RR (middle) and iRR (bottom of the table).

		impact category					
indicator	**sensitivity analysis**	CC	FD	HT	PMF	POF	TA
CEI	baseline	0.99	0.47	0.72	0.90	0.88	0.77
	renewables	0.97	0.78	0.48	0.90	0.89	0.85
	substitution	0.97	0.47	0.88	0.91	0.89	0.84
	MRF	0.95	0.48	0.85	0.88	0.88	0.82
RR	baseline	0.80	0.13	0.40	0.55	0.54	0.38
	renewables	0.90	0.43	0.21	0.57	0.56	0.50
	substitution	0.83	0.13	0.64	0.59	0.56	0.49
	MRF	0.81	0.16	0.62	0.59	0.58	0.50
iRR	baseline	0.45	0.01	0.09	0.24	0.26	0.12
	renewables	0.60	0.17	0.01	0.23	0.24	0.18
	substitution	0.46	0.00	0.26	0.24	0.23	0.17
	MRF	0.43	0.00	0.25	0.22	0.21	0.15

4. Discussion

4.1. Limitations of the model

The waste management scenarios presented in this study aim to reflect some of the treatment alternatives for PCPPW in the Netherlands and the effects of boundary conditions, for instance, the effects of moving the system to another country, are not analysed. This limits the scope to the Dutch waste management systems. It is expected that local conditions, such as the regional electricity mix, policies, wages, or urbanism, will play an important role in defining the performance of waste collection schemes.

Markets have been simplified. It is assumed that all the materials recovered by the MRF's are sold to reprocessing plants and that the WMG's produced by the latter are subsequently reintroduced into different markets depending purely on their quality levels. However, quality is not the only driver of the market of secondary plastics, for instance, the demand for recyclates is highly intertwined with the current oil prices. Except for recycled PET, the market for recycled plastics shrinks when the price of virgin plastics goes down (Milios et al., 2018). Another limitation on the market dynamics applies to the type of collection system. Companies can be wary of the purity of materials recovered from post-separation, limiting their potential applications. This was analysed to some extent in the sensitivity analysis and proved to have a significant impact in the performance of PS schemes.

Finally, the modelling was developed using literature data and consulting experts; however, it has not been applied to any real-world case study. Aggregated data on the performance of collection schemes, MRF, reprocessing plants, or incineration plants were used in this study. In addition, reprocessing and incineration stages were simplified to only one option, disregarding other alternatives such as feedstock recycling or the use of plastic waste as fuel in clinker production (Haupt, Kägi, & Hellweg, 2018b; Ragaert et al., 2017). That level of detail was considered acceptable for this analysis; however, it is recommended to apply the methodology to specific waste management configurations to validate the results.

4.2. Limitations of value-based metrics

Simplicity has made recycling rates a successful indicator. Value-based indicators are a more complex and dynamic metric, that relies on fluctuating markets and it can be directly affected by policy instruments, such as taxes. While this complexity makes value-based indicators more complete, it hinders their interpretability, and can have negatives effects in its implementation.

The ideal value-based indicator should be able to reflect current social, environmental, and economic policies. Unfortunately, there is no parameter that perfectly captures these relations. The use of economic value recovered seems to be a good approximation to reflect environmental performance in the studied system, however, it shows that the business aspects are not completely aligned. The effectiveness of money as a proxy for value is limited to the implementation of policy instruments, such as carbon or landfilling taxes, that translate the political interest into monetary terms. As seen once again in this model, these taxes have been historically inefficient (Andrew, 2008) and, in addition, they are not globally implemented (Cramton, MacKay, Ockenfels, & Stoft, 2018).

Finally, optimal indicators and policies should not incite misreporting. Hogg et al. (2017) indicate that with current waste management indicators, producers and EPR organisations alike, are encouraged to under-report the amounts of products that are put into the market to reduce paying fees for the first, and to inflate recycling rates for the latter. This issue is not fully addressed by value-indicators.

4.3. Implications for the circular economy

The EU is committed to transition towards a CE. According to the *European Strategy for Plastics* and the *Single Use Plastics Directive*, all plastic packaging placed on the EU market should be reusable or recyclable and that PET bottles should contain at least 30% recycled plastics by 2030 (EC, 2018a, EC, 2019c). In this context, this research suggests that not any type of recycling is valid, that achieving high quality recycling is fundamental to determine the best environmental performance of a recycling system. In line with CE principles, high quality recycling focuses on optimising value retention and achieves higher displacement of primary raw material consumption. However, the downcycling of PCPPW into low quality applications, such as plastic fibres, pipes, or bags, is common practice and actively promoted (Hahladakis & Iacovidou, 2018). These solutions achieve positive performances in traditional waste management and circularity metrics. However, the model indicates that increasing circularity purely based in mass-based criteria leads the system towards a suboptimal use of resources and advocates for focusing on high quality recycling. Moreover, the performance of these systems is traditionally analysed using relative indicators, such as RR, CEI, or CI, which can be misleading. For instance, recycling or value recovery rates can be optimised while the rate of waste generation also increases. This depicts a situation where the performance of the system has improved in relative terms, but the absolute performance has not. To tackle this problem, a combination of absolute and relative performance indicators is suggested.

The economic assessment reveals that different business models can be used to optimise the economic performance of the system. However, these vary significantly in terms of technical, value-recovery, and environmental performances. The recycling industry, and any industry in general, will not evolve towards more environmentally sound solutions unless it is backed by favourable economic models. To promote an effective CE, there is a need for robust policies to align economic and environmental performances, such as the creation of robust and independent markets for secondary material, the implementation of effective carbon taxes, or the promotion of high-quality recycling. In addition, there are potential negative effects arising from the increased circularity, such as the circular economy rebound effect (Zink & Geyer, 2017). Life cycle thinking strategies should be defined to mitigate these potential drawbacks.

Value-based indicators are one type of indicator that, with the raise of the CE, have being proposed to measure the performance of this economic model. However, as Roger Levett (1998) accurately indicates "the search for better indicators is not only a technical problem. It is also a stimulus to more thinking about precisely what it is that we value". This study analysed the role of value-based indicators in aligning technical, economic, and environmental performances. These are pillars of the CE and, all or some of them, are commonly addressed by most CE indicators (Saidani et al., 2019). However, other *values* are consistently overlooked, such as social, cultural, or intrinsic values (Kuhlman & Farrington, 2010). These flaws have been identified in other widely used metrics, such as GDP (Schumacher, 1973). Incorporating value into new performance indicators should strive to integrate these overlooked social principles and morals and inspire societies to question what is the desirable future that these metrics will optimise. In the case of the CE and considering some of the adverse effects of plastic products (Eriksen et al., 2014; Wright, 2017), this could start by wondering what kind of materials should be used in this future and if plastics have space there at all.

5. CONCLUSION

The validity of using value-based metrics to evaluate the performance of waste management systems was analysed. Several scenarios for possible configurations of the Dutch PCPPW management network were defined and assessed in terms of technical, economic, and environmental performance. Two technical metrics, intermediate recycling rates (iRR) and recycling rates (RR), and one value-based metric, the circular economy index (CEI), were compared to the environmental and economic performance of the scenarios. The CEI showed a better alignment with all the environmental impact categories than the mass-based metrics. Most importantly, the value-based metric proved capable of capturing the significance of the quality of the recycled plastics in the displacement of primary raw materials, thus fostering high quality recycling over downcycling. However, no correlation was found between the economic performance and the presented indicators. There are multiple business models that achieve good economic performances with diverse environmental, technical, or value-recovery performances. This suggests that current policies are unable to successfully align economic and environmental strategies. More research should be put in defining holistic policies that promote environmental and economically sustainable practices.

6. BIBLIOGRAPHY

Afvalfonds Verpakkingen. (2018). Recycling results. Retrieved May 14, 2020, from https://afvalfondsverpakkingen.nl/en/packaging-waste-fund

Afvalfonds Verpakkingen. (2019). Tarieven. Retrieved June 16, 2020, from https://afvalfondsverpakkingen.nl/verpakkingen/alle-tarieven

Andrew, B. (2008). Market failure, government failure and externalities in climate change mitigation: The case for a carbon tax. In *Public Administration and Development* (Vol. 28, pp. 393–401). John Wiley & Sons, Ltd. https://doi.org/10.1002/pad.517

Ardolino, F., Berto, C., & Arena, U. (2017). Environmental performances of different configurations of a material recovery facility in a life cycle perspective. *Waste Management*, *68*, 662–676. https://doi.org/10.1016/j.wasman.2017.05.039

AVR. (2020). Plastic separation from residual waste bag | AVR. Too good to waste. Retrieved June 7, 2020, from https://www.avr.nl/en/plastic-separation-from-residual-waste-bag

Bertanza, G., Ziliani, E., & Menoni, L. (2018). Techno-economic performance indicators of municipal solid waste collection strategies. *Waste Management*, *74*, 86–97. https://doi.org/10.1016/j.wasman.2018.01.009

Bosmans, A., Vanderreydt, I., Geysen, D., & Helsen, L. (2013). The crucial role of Waste-to-Energy technologies in enhanced landfill mining: A technology review. *Journal of Cleaner Production*, *55*, 10–23. https://doi.org/10.1016/j.jclepro.2012.05.032

Broekaart, J., Afdeling, H., & En, A. (2017). Op naar een schoon nederland.

Brouwer, M., Picuno, C., Thoden van Velzen, E. U., Kuchta, K., De Meester, S., & Ragaert, K. (2019). The impact of collection portfolio expansion on key performance indicators of the Dutch recycling system for Post-Consumer Plastic Packaging Waste, a comparison between 2014 and 2017. *Waste Management*, *100*, 112–121. https://doi.org/10.1016/j.wasman.2019.09.012

Brouwer, M., Thoden van Velzen, E. U., Augustinus, A., Soethoudt, H., De Meester, S., & Ragaert, K. (2018). Predictive model for the Dutch post-consumer plastic packaging recycling system and implications for the circular economy. *Waste Management*, *71*, 62–85. https://doi.org/10.1016/j.wasman.2017.10.034

Brunner, P. H., & Ma, H. W. (2009). Substance flow analysis an indispensable tool for goal-oriented waste management. *Journal of Industrial Ecology*, *13*(1), 11–14. https://doi.org/10.1111/j.1530-9290.2008.00083.x

Brunner, P. H., & Rechberger, H. (2016). *Practical handbook of material flow analysis*. *Practical Handbook of Material Flow Analysis* (Vol. 22). https://doi.org/10.1007/bf02979426

Burnley, S., & Coleman, T. (2018). The environmental and financial benefits of recovering plastics from residual municipal waste before energy recovery. *Waste Management*, *79*, 79–86. https://doi.org/10.1016/j.wasman.2018.07.034

CBS. (2017). Energy balance sheet; supply, transformation and consumption. Retrieved May 24, 2020, from https://opendata.cbs.nl/statline/#/CBS/en/dataset/83140ENG/table?ts=1590246652094

CBS. (2019a). CBS StatLine - Huishoudelijk afval per gemeente per inwoner. Retrieved February 4, 2020, from https://opendata.cbs.nl/statline/#/CBS/nl/dataset/83452NED/table?ts=1580735152291

CBS. (2019b). Gemeentelijke afvalstoffen; hoeveelheden. Retrieved April 29, 2020, from https://opendata.cbs.nl/statline/#/CBS/nl/dataset/83558NED/table?ts=1588175647718

CBS. (2020). Personal Communication. The Hague.

Cimpan, C., Maul, A., Jansen, M., Pretz, T., & Wenzel, H. (2015). Central sorting and recovery of MSW recyclable materials: A review of technological state-of-the-art, cases, practice and implications for materials recycling. *Journal of Environmental Management*. https://doi.org/10.1016/j.jenvman.2015.03.025

Cimpan, C., Maul, A., Wenzel, H., & Pretz, T. (2016). Techno-economic assessment of central sorting at material recovery facilities - The case of lightweight packaging waste. *Journal of Cleaner Production*, *112*, 4387–4397. https://doi.org/10.1016/j.jclepro.2015.09.011

Combs, A. R. (2012). *Life Cycle Analysis of Recycling Facilities in a Carbon Constrained World*. North Carolina State University. https://doi.org/10.1192/bjp.111.479.1009-a

CPB. (2017). *De circulaire economie van kunststof: van grondstoffen tot afval*. Retrieved from www.cpb.nl

Cramton, P., MacKay, D. J., Ockenfels, A., & Stoft, S. (2018). Global Carbon Pricing. In *Global Carbon Pricing*. https://doi.org/10.7551/mitpress/10914.003.0008

Di Maio, F., & Rem, P. C. (2015). A Robust Indicator for Promoting Circular Economy through Recycling. *Journal of Environmental Protection*, *06*(10), 1095–1104. https://doi.org/10.4236/jep.2015.610096

Di Maio, F., Rem, P. C., Baldé, K., & Polder, M. (2017). Measuring resource efficiency and circular economy: A market value approach. *Resources, Conservation and Recycling*, *122*, 163–171. https://doi.org/10.1016/j.resconrec.2017.02.009

DKR. (2020). Specifications. Retrieved April 30, 2020, from https://www.gruener-punkt.de/en/downloads.html

Eriksen, M. K., & Astrup, T. F. (2019). Characterisation of source-separated, rigid plastic waste and evaluation of recycling initiatives: Effects of product design and source-separation system. *Waste Management*, *87*, 161–172. https://doi.org/10.1016/j.wasman.2019.02.006

Eriksen, M. K., Damgaard, A., Boldrin, A., & Astrup, T. F. (2019). Quality Assessment and Circularity Potential of Recovery Systems for Household Plastic Waste. *Journal of Industrial Ecology*. https://doi.org/10.1111/jiec.12822

Eriksen, M., Lebreton, L. C. M., Carson, H. S., Thiel, M., Moore, C. J., Borerro, J. C., … Reisser, J. (2014). Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. *PLoS ONE*, *9*(12), e111913. https://doi.org/10.1371/journal.pone.0111913

EC. European Commission. (2008). Directive 2008/98/EC of the European Parliament and of the Council of 19 November 2008 on waste and repealing certain directives (Waste framework) http://ec.europa.eu/environment/waste/framework/ (accessed 08.20.2016). *Official Journal of the European Union*, 3–30. https://doi.org/2008/98/EC.; 32008L0098

EC. European Commission. (2015). Closing the loop - An EU action plan for the Circular Economy. Brussels. https://doi.org/10.1017/CBO9781107415324.004

EC. European Commission. (2018a). A European Strategy for Plastics in a Circular Economy. https://doi.org/10.1021/acs.est.7b02368

EC. European Commission. (2018b). Circular economy: closing the loop. Monitoring framework for the circular economy. https://doi.org/10.2779/83272

EC. European Commission. (2018c). Directive (EU) 2018/851 of the European Parliament and of the Council of 30 May 2018 amending Directive 2008/98/EC on waste (Text with EEA relevance). *Official Journal of the European Union*, (L150), 109–140. Retrieved from https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32018L0851

EC. European Commission. (2018d). DIRECTIVE (EU) 2018/852 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 30 May 2018 amending Directive 94/62/EC on packaging and packaging waste (Text with EEA relevance). *Official Journal of the European Union*, *150*, 141–154. Retrieved from https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32018L0852

EC. European Commission. (2019a). COMMISSION IMPLEMENTING DECISION (EU) 2019/ 1004 - of 7 June 2019 - laying down rules for the calculation, verification and reporting of data on waste in accordance with Directive 2008/ 98/ EC of the European Parliament and of the Council and repealing Co. *Official Journal of the European Union*, *163*, 66–100.

EC. European Commission. (2019b). *COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE EUROPEAN COUNCIL, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS The European Green Deal*. Retrieved from https://sustainabledevelopment.un.org/post2015/transformingourworld

EC. European Commission. (2019c). DIRECTIVE (EU) 2019/904 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL of 5 June 2019 on the reduction of the impact of certain plastic products on the environment. *Official Journal of the European Union*, *155*(June), 1–19. Retrieved from https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32019L0904&from=EN

EC-JRC-IES. European Commission - Joint Research Centre - Institute for Environment and Sustainability. (2010). *International Reference Life Cycle Data System (ILCD) Handbook - General guide for Life Cycle Assessment - Detailed guidance* (First Edit). Luxembourg: Publications Office of the European Union. https://doi.org/10.2788/38479

European Parliament and Council. (1994). DIRECTIVE 94/62/EC of 20 December 1994 on packaging and packaging waste. *Official Journal of the European Communities*, *365*, 10–23.

Faraca, G., & Astrup, T. (2019). Plastic waste from recycling centres: Characterisation and evaluation of plastic recyclability. *Waste Management*, *95*, 388–398. https://doi.org/10.1016/j.wasman.2019.06.038

Faraca, G., Martinez-Sanchez, V., & Astrup, T. F. (2019). Environmental life cycle cost assessment: Recycling of

hard plastic waste collected at Danish recycling centres. *Resources, Conservation and Recycling*, *143*, 299–309. https://doi.org/10.1016/j.resconrec.2019.01.014
Gallardo, A., Bovea, M. D., Colomer, F. J., & Prades, M. (2012). Analysis of collection systems for sorted household waste in Spain. *Waste Management*, *32*(9), 1623–1633. https://doi.org/10.1016/j.wasman.2012.04.006
Gallardo, A., Bovea, M. D., Colomer, F. J., Prades, M., & Carlos, M. (2010). Comparison of different collection systems for sorted household waste in Spain. *Waste Management*, *30*(12), 2430–2439. https://doi.org/10.1016/j.wasman.2010.05.026
Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, use, and fate of all plastics ever made - Supplementary Information. *Science Advances*, (July), 25–29. https://doi.org/10.1126/sciadv.1700782
Ghisellini, P., Cialani, C., & Ulgiati, S. (2016). A review on circular economy: The expected transition to a balanced interplay of environmental and economic systems. *Journal of Cleaner Production*, *114*, 11–32. https://doi.org/10.1016/j.jclepro.2015.09.007
Gradus, R. H. J. M., Nillesen, P. H. L., Dijkgraaf, E., & van Koppen, R. J. (2017). A Cost-effectiveness Analysis for Incineration or Recycling of Dutch Household Plastic Waste. *Ecological Economics*, *135*, 22–28. https://doi.org/10.1016/j.ecolecon.2016.12.021
Groot, J., Bing, X., Bos-Brouwers, H., & Bloemhof-Ruwaard, J. (2014). A comprehensive waste collection cost model applied to post-consumer plastic packaging waste. *Resources, Conservation and Recycling*, *85*, 79–87. https://doi.org/10.1016/j.resconrec.2013.10.019
Hahladakis, J. N., & Iacovidou, E. (2018). Closing the loop on plastic packaging materials: What is quality and how does it affect their circularity? *Science of the Total Environment*, *630*, 1394–1400. https://doi.org/10.1016/j.scitotenv.2018.02.330
Haupt, M., Kägi, T., & Hellweg, S. (2018a). Life cycle inventories of waste management processes. *Data in Brief*, *19*, 1441–1457. https://doi.org/10.1016/j.dib.2018.05.067
Haupt, M., Kägi, T., & Hellweg, S. (2018b). Modular life cycle assessment of municipal solid waste management. *Waste Management*, *79*, 815–827. https://doi.org/10.1016/j.wasman.2018.03.035
Haupt, M., Vadenbo, C., & Hellweg, S. (2017). Do We Have the Right Performance Indicators for the Circular Economy?: Insight into the Swiss Waste Management System. *Journal of Industrial Ecology*, *21*(3), 615–627. https://doi.org/10.1111/jiec.12506
Haupt, M., Waser, E., Würmli, J. C., & Hellweg, S. (2018). Is there an environmentally optimal separate collection rate? *Waste Management*, *77*, 220–224. https://doi.org/10.1016/j.wasman.2018.03.050
Hogg, D., Elliott, T., Corbin, M., Hilton, M., Tsiarta, C., Hudson, J., … Kazlauskaitė, L. (2017). *Study on Waste Statistics-A comprehensive review of gaps and weaknesses and key priority areas for improvement in the EU waste statistics*. Retrieved from http://ec.europa.eu/environment/waste/pdf/Eunomia_study_on_waste_statistics.pdf
Hoornweg, D., & Bhada-Tata, P. (2012). *What a Waste: A Global Review of Solid Waste Management*. *World Bank*. https://doi.org/10.1111/febs.13058
Huijbregts, M. A. J., Steinmann, Z. J. N., Elshout, P. M. F., Stam, G., Verones, F., Vieira, M. D. M., … van Zelm, R. (2016). *ReCiPe 2016 A harmonized life cycle impact assessment method at midpoint and endpoint level Report I: Characterization*.
Huysman, S., De Schaepmeester, J., Ragaert, K., Dewulf, J., & De Meester, S. (2017). Performance indicators for a circular economy: A case study on post-industrial plastic waste. *Resources, Conservation and Recycling*, *120*, 46–54. https://doi.org/10.1016/j.resconrec.2017.01.013
Huysman, S., Debaveye, S., Schaubroeck, T., Meester, S. De, Ardente, F., Mathieux, F., & Dewulf, J. (2015). The recyclability benefit rate of closed-loop and open-loop systems: A case study on plastic recycling in Flanders. *Resources, Conservation and Recycling*, *101*, 53–60. https://doi.org/10.1016/j.resconrec.2015.05.014
Iriarte, A., Gabarrell, X., & Rieradevall, J. (2009). LCA of selective waste collection systems in dense urban areas. *Waste Management*, *29*(2), 903–914. https://doi.org/10.1016/j.wasman.2008.06.002
Islas, J., Manzini, F., Masera, O., & Vargas, V. (2018). Solid biomass to heat and power. In *The Role of Bioenergy in the Emerging Bioeconomy: Resources, Technologies, Sustainability and Policy* (pp. 145–177). Elsevier. https://doi.org/10.1016/B978-0-12-813056-8.00004-2
Kaza, S., Yao, L., Bhada-Tata, P., & Woerden, F. van. (2018). *What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050*. Washington DC. https://doi.org/10.1596/978-1-4648 -1329-0
Kenton, W. (2019). Equivalent Annual Cost - EAC Definition. Retrieved June 4, 2020, from

https://www.investopedia.com/terms/e/eac.asp
Koffler, C., & Florin, J. (2013). Tackling the downcycling issue - A revised approach to value-corrected substitution in life cycle assessment of aluminum (VCS 2.0). *Sustainability (Switzerland)*, *5*(11), 4546–4560. https://doi.org/10.3390/su5114546
Kuhlman, T., & Farrington, J. (2010). What is sustainability? *Sustainability*, *2*(11), 3436–3448. https://doi.org/10.3390/su2113436
Lee, P., Sims, E., Bertham, O., Symington, H., Bell, N., Pfaltzgraff, L., … Benke, J. (2017). *Towards a circular economy – Waste management in the EU*. https://doi.org/10.2861/978568
Levett, R. (1998). Sustainability indicators - Integrating quality of life and environmental protection. *Journal of the Royal Statistical Society. Series A: Statistics in Society*, *161*(3), 291–302. https://doi.org/10.1111/1467-985X.00109
Linder, M., Sarasini, S., & van Loon, P. (2017). A Metric for Quantifying Product-Level Circularity. *Journal of Industrial Ecology*, *21*(3), 545–558. https://doi.org/10.1111/jiec.12552
Mastellone, M. L. (2020). Technical description and performance evaluation of different packaging plastic waste management's systems in a circular economy perspective. *Science of the Total Environment*, *718*. https://doi.org/10.1016/j.scitotenv.2020.137233
Milios, L. Advancing to a Circular Economy: three essential ingredients for a comprehensive policy mix, 13 Sustainability Science § (2018). Springer Japan. https://doi.org/10.1007/s11625-017-0502-9
Milios, L., Holm Christensen, L., McKinnon, D., Christensen, C., Rasch, M. K., & Hallstrøm Eriksen, M. (2018). Plastic recycling in the Nordics: A value chain market analysis. *Waste Management*, *76*, 180–189. https://doi.org/10.1016/j.wasman.2018.03.034
Ministerie van IenW. Deel B.3. Afvalscheiding. Landelijk afvalbeheerplan 2017 – 2029 (eerste wijziging) | LAP3 (2019).
Moraga, G., Huysveld, S., Mathieux, F., Blengini, G. A., Alaerts, L., Van Acker, K., … Dewulf, J. (2019). Circular economy indicators: What do they measure? *Resources, Conservation and Recycling*, *146*, 452–461. https://doi.org/10.1016/j.resconrec.2019.03.045
Mwanza, B. G., Mbohwa, C., & Telukdarie, A. (2018). The Influence of Waste Collection Systems on Resource Recovery: A Review. In *Procedia Manufacturing* (Vol. 21, pp. 846–853). https://doi.org/10.1016/j.promfg.2018.02.192
Neuhoff, K., Cooper, S., Laing, T., Lester, S., & Rysanek, A. (2009). *Indicator Choices and Tradeoffs: Facilitating the Success of International Climate Policies and Projects*. Retrieved from https://www.jstor.org/stable/resrep15568.6%0D
Nijkerk, A. A., & Dalmijn, W. L. (2001). *Handbook of recycling techniques*. Den Haag.
Nolan ITU. (2004). *Getting More from our Recycling Systems. Assessment of domestic waste and recycing systems. Final Report*. *Sustainability Programs Division*.
Packard, V., & McKibben, B. (1963). *The Waste Makers*. Penguin Books.
PIE. (2020). Polymer prices – Plastics Information Europe. Retrieved May 3, 2020, from https://pieweb.plasteurope.com/default.aspx?pageid=200
plasticker. (2020). Market Report Plastics. Retrieved May 1, 2020, from https://plasticker.de/preise/marktbericht_en.php
PlasticsEurope. (2018). *Plastics – The facts 2018*. https://doi.org/10.1016/j.marpolbul.2013.01.015
Pressley, P. N., Levis, J. W., Damgaard, A., Barlaz, M. A., & DeCarolis, J. F. (2015). Analysis of material recovery facilities for use in life-cycle assessment. *Waste Management*, *35*, 307–317. https://doi.org/10.1016/j.wasman.2014.09.012
Ragaert, K., Delva, L., & Van Geem, K. (2017). Mechanical and chemical recycling of solid plastic waste. *Waste Management*. https://doi.org/10.1016/j.wasman.2017.07.044
Rigamonti, L., Grosso, M., Møller, J., Martinez Sanchez, V., Magnani, S., & Christensen, T. H. (2014). Environmental evaluation of plastic waste management scenarios. *Resources, Conservation and Recycling*, *85*, 42–53. https://doi.org/10.1016/j.resconrec.2013.12.012
Rigamonti, L., Niero, M., Haupt, M., Grosso, M., & Judl, J. (2018). Recycling processes and quality of secondary materials: Food for thought for waste-management-oriented life cycle assessment studies. *Waste Management*, *76*, 261–265. https://doi.org/10.1016/j.wasman.2018.03.001
Rijkswaterstaat. (2018). *Afvalverwerking in Nederland, gegevens 2017*.
Rodrigues, S. (2016). *Classificação e Benchmarking de Sistemas de Recolha de Resíduos Urbanos*.

https://doi.org/10.1021/jf1041382
Rodrigues, S., Martinho, G., & Pires, A. (2016). Waste collection systems. Part A: A taxonomy. *Journal of Cleaner Production*, *113*, 374–387. https://doi.org/10.1016/j.jclepro.2015.09.143
Roithner, C., & Rechberger, H. (2020). Implementing the dimension of quality into the conventional quantitative definition of recycling rates. *Waste Management*, *105*, 586–593. https://doi.org/10.1016/j.wasman.2020.02.034
Rotterdam Circulair. (2019). *From trash to treasure. Rotterdam Circularity Programme 2019 - 2023*.
Sabbas, T., Polettini, A., Pomi, R., Astrup, T., Hjelmar, O., Mostbauer, P., … Lechner, P. (2003). Management of municipal solid waste incineration residues. *Waste Management*, *23*(1), 61–88. https://doi.org/10.1016/S0956-053X(02)00161-7
Saidani, M., Yannou, B., Leroy, Y., Cluzel, F., & Kendall, A. (2019, January 10). A taxonomy of circular economy indicators. *Journal of Cleaner Production*. Elsevier Ltd. https://doi.org/10.1016/j.jclepro.2018.10.014
Schumacher, E. F. (1973). *Small is Beautiful: a study of economics as if people mattered*. Random House. https://doi.org/10.1111/j.1759-5436.1975.mp7001008.x
Seyring, N., Dollhofer, M., Weißenbacher, J., Herczeg, M., & David, M. (2015). *Assessment of separate collection schemes in the 28 capitals of the EU*. Retrieved from http://ec.europa.eu/environment/waste/studies/pdf/Separate collection_Final Report.pdf
Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., … Sörlin, S. (2015). Planetary boundaries: Guiding human development on a changing planet. *Science*, *347*(6223). https://doi.org/10.1126/science.1259855
Struk, M. (2017). Distance and incentives matter: The separation of recyclable municipal waste. *Resources, Conservation and Recycling*, *122*, 155–162. https://doi.org/10.1016/j.resconrec.2017.01.023
SUEZ. (2020). KSI - SUEZ in the Netherlands. Retrieved June 7, 2020, from https://www.suez.nl/nl-nl/locaties/rotterdam/ksi
The Council of European Union. (2018). Delivering on the EU Action Plan for the Circular Economy - Council conclusions. *Official Journal of the European Communities*. Retrieved from http://data.consilium.europa.eu/doc/document/ST-10447-2018-INIT/en/pdf
Thoden van Velzen, E. U., Brouwer, M. T., & Feil, A. (2019). Collection behaviour of lightweight packaging waste by individual households and implications for the analysis of collection schemes. *Waste Management*, *89*, 284–293. https://doi.org/10.1016/j.wasman.2019.04.021
Thoden Van Velzen, U., Groot, J., Hilke, B.-B., Groot, J., Bing, X., Jansen, M., & Luijsterburg, B. (2013). *Scenarios study on post-consumer plastic packaging waste recycling*. Retrieved from www.wur.nl
Tisserant, A., Pauliuk, S., Merciai, S., Schmidt, J., Fry, J., Wood, R., & Tukker, A. (2017). Solid Waste and the Circular Economy: A Global Analysis of Waste Treatment and Waste Footprints. *Journal of Industrial Ecology*, *21*(3), 628–640. https://doi.org/10.1111/jiec.12562
UNEP. (2012). *Responsible Resource Management for a Sustainable World: Findings from the International Resource Panel*. Retrieved from www.unep.org/resourcepanel
Vadenbo, C., Hellweg, S., & Astrup, T. F. (2017). Let's Be Clear(er) about Substitution: A Reporting Framework to Account for Product Displacement in Life Cycle Assessment. *Journal of Industrial Ecology*, *21*(5), 1078–1089. https://doi.org/10.1111/jiec.12519
Van Eygen, E., Laner, D., & Fellner, J. (2018). Integrating High-Resolution Material Flow Data into the Environmental Assessment of Waste Management System Scenarios: The Case of Plastic Packaging in Austria. *Environmental Science and Technology*, *52*(19), 10934–10945. https://doi.org/10.1021/acs.est.8b04233
Villanueva, A., & Eder, P. (2014). *End-of-waste criteria for waste plastic for conversion. Technical proposals. Publications Office of the European Union*. https://doi.org/10.2791/13033
Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): overview and methodology. *International Journal of Life Cycle Assessment*, *21*(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8
WRAP. (2010). *Material bulk densities*. Retrieved from www.wrap.org.uk
Wright, S. (2017). Plastic and Human Health: A Micro Issue? https://doi.org/10.1021/acs.est.7b00423
Zink, T., & Geyer, R. (2017). Circular Economy Rebound. *Journal of Industrial Ecology*, *21*(3), 593–602. https://doi.org/10.1111/jiec.12545

SUPPLEMENTARY INFORMATION

1. LIST OF SCENARIOS

		collection		MRF	reprocessing	MSWI
CODE	sorting scheme	participation	selection	type	type	type
A-1-1	KS	low	low	basic	-	-
A-1-2	KS	low	low	medium	-	-
A-1-3	KS	low	low	advanced	-	-
A-2-1	KS	low	average	basic	-	-
A-2-2	KS	low	average	medium	-	-
A-2-3	KS	low	average	advanced	-	-
A-3-1	KS	low	high	basic	-	-
A-3-2	KS	low	high	medium	-	-
A-3-3	KS	low	high	advanced	-	-
A-4-1	KS	average	low	basic	-	-
A-4-2	KS	average	low	medium	-	-
A-4-3	KS	average	low	advanced	-	-
A-5-1	KS	average	average	basic	-	-
A-5-2	KS	average	average	medium	-	-
A-5-3	KS	average	average	advanced	-	-
A-6-1	KS	average	high	basic	-	-
A-6-2	KS	average	high	medium	-	-
A-6-3	KS	average	high	advanced	-	-
A-7-1	KS	high	low	basic	-	-
A-7-2	KS	high	low	medium	-	-
A-7-3	KS	high	low	advanced	-	-
A-8-1	KS	high	average	basic	-	-
A-8-2	KS	high	average	medium	-	-
A-8-3	KS	high	average	advanced	-	-
A-9-1	KS	high	high	basic	-	-
A-9-2	KS	high	high	medium	-	-
A-9-3	KS	high	high	advanced	-	-
B-1-1	DO	low	low	basic	-	-
B-1-2	DO	low	low	medium	-	-
B-1-3	DO	low	low	advanced	-	-
B-2-1	DO	low	average	basic	-	-
B-2-2	DO	low	average	medium	-	-
B-2-3	DO	low	average	advanced	-	-
B-3-1	DO	low	high	basic	-	-
B-3-2	DO	low	high	medium	-	-
B-3-3	DO	low	high	advanced	-	-
B-4-1	DO	average	low	basic	-	-
B-4-2	DO	average	low	medium	-	-
B-4-3	DO	average	low	advanced	-	-
B-5-1	DO	average	average	basic	-	-
B-5-2	DO	average	average	medium	-	-
B-5-3	DO	average	average	advanced	-	-
B-6-1	DO	average	high	basic	-	-
B-6-2	DO	average	high	medium	-	-
B-6-3	DO	average	high	advanced	-	-

B-7-1	DO	high	low	basic	-	-
B-7-2	DO	high	low	medium	-	-
B-7-3	DO	high	low	advanced	-	-
B-8-1	DO	high	average	basic	-	-
B-8-2	DO	high	average	medium	-	-
B-8-3	DO	high	average	advanced	-	-
B-9-1	DO	high	high	basic	-	-
B-9-2	DO	high	high	medium	-	-
B-9-3	DO	high	high	advanced	-	-
C-1-1	PS	-	low	basic	-	-
C-1-2	PS	-	low	medium	-	-
C-1-3	PS	-	low	advanced	-	-
C-2-1	PS	-	average	basic	-	-
C-2-2	PS	-	average	medium	-	-
C-2-3	PS	-	average	advanced	-	-
C-3-1	PS	-	high	basic	-	-
C-3-2	PS	-	high	medium	-	-
C-3-3	PS	-	high	advanced	-	-

2. S1. Collection

2.1. Technical assessment

2.1.1. Selection rates

The detailed selection rates of the separate collection schemes are presented in Table XXI.

Table XXI. Selection rates at the packaging level for the separate collection systems (KS and DO).
B = bottle. F = film. R = rigid. L = laminated. NP = non-packaging. BC = beverage cartons. p&c = paper and cardboard. LAMD = land and attached moisture and dirt. TP = targeted plastic, NTP = non-targeted plastic, imp = impurities

CODE	material	fraction	targeted by collection portfolio	**low**	**average**	**high**
PET-B	PET	B	TP	86%	91%	95%
PET-F	PET	F	TP	53%	63%	68%
PET-R	PET	R	TP	88%	93%	98%
PE-B	PE	B	TP	88%	93%	98%
PE-F	PE	F	TP	71%	84%	90%
PE-R	PE	R	TP	41%	48%	52%
PP-B	PP	B	TP	65%	76%	82%
PP-F	PP	F	TP	75%	88%	94%
PP-R	PP	R	TP	82%	86%	90%
PS-B	PS	B	TP			
PS-F	PS	F	TP	68%	80%	86%
PS-R	PS	R	TP	76%	89%	96%
EPS-R	EPS	R	TP	14%	17%	16%
PVC-F	PVC	F	TP	70%	82%	88%
PVC-R	PVC	R	TP	47%	55%	59%
mix-F	mix	F	TP	52%	61%	65%
mix-R	mix	R	TP	66%	77%	83%
mix-L	mix	L	NTP	61%	72%	69%
PET-NP	PET	NP	NTP	15%	14%	13%
PE-NP	PE	NP	NTP	66%	60%	57%

PP-NP	PP	NP	NTP	42%	38%	36%
PS-NP	PS	NP	NTP	40%	36%	34%
PVC-NP	PVC	NP	NTP	45%	41%	39%
mix-NP	mix	NP	NTP	41%	37%	35%
other-NP	other	NP	NTP	0%	0%	0%
BC	BC	NP	imp	89%	94%	99%
metal	metal	NP	imp	55%	58%	61%
organic	organic	NP	imp	3%	2%	2%
textile	textile	NP	imp	3%	2%	2%
p&c	p&c	NP	imp	12%	8%	6%
glass	glass	NP	imp	9%	6%	5%
LAMD	LAMD	NP	imp	9%	6%	5%

2.2. ECONOMIC ASSESSMENT

2.2.1. Price of plastics

The price of virgin plastics in the waste stream is estimated from polymer prices in the European market (PIE, 2020; plasticker, 2020). Considering the composition and 1332.4 €/ton based on

		price (€/ton)						
		PET	PE	PP	PS	EPS	PVC	mix
PCPPW	bottle	1170	1330	1360	1780	-	-	-
	film	1170	1300	1380	1780	1470	1390	1320
	rigid	1170	1330	1360	1780	1470	1430	1450
	laminated	1170	1300	1380	1780	1470	1390	1380
non PCPPW		1170	1390	1390	1680	1410	1390	1370

2.2.2. Collection costs parameters

Table XXII. List of parameters used in the calculation of collection costs

parameter		KS	DO	PS	units
vehicle					
cost of a vehicle	C_{veh_inv}	206000	250000	206000	€
salvage cost of a vehicle	C_{veh_sal}	30900	37500	30900	€
depreciation (vehicle)	dep_{veh}	5	5	5	year
insurance	C_{ins}	2500	2500	2500	€/year
tax	C_{tax}	1000	1000	1000	€/year
maintenance of the vehicle	M_{veh}	3000	4000	3000	€/year
availability of vehicle	av_{veh}	80%	80%	80%	%
use of vehicle	T_{veh}	3000	3000	3000	h/year
average speed (hauling)	v_h	60	60	60	km/h
average distance (to hauling)	d_h	18	18	18	km
average speed (driving)	v_d	25	40	15	km/h
households per kerbside point	n_{hh_stop}	10	0	10	-
average time per stop	t_{stop}	0.014	0.5	0.069	h/stop
average truck load per collection round	$truck_{load}$	1800	750	7200	kg/round
fuel consumption (driving)	FC_d	0.33	0.25	0.4	l/km

fuel consumption (idling)	FC_i	4	4	4	l/h
fuel consumption (hauling)	FC_h	0.25	0.25	0.33	l/km
fuel price	P_f	1.35	1.35	1.35	€/l
labour					
drivers per vehicle	n_{driver_veh}	1	1	1	-
loaders per vehicle	n_{loader_veh}	2	0	2	-
working hours (driver)	t_{driver}	1650	1650	1650	h/year
working hours (loader)	t_{loader}	1650	1650	1650	h/year
driver wage	w_{driver}	30000	30000	30000	€/year
loader wage	w_{loader}	25000	25000	25000	€/year
container and bags					
capacity (underground container)	CAP_{DO}	0	750	0	kg
capacity (container 240l)	CAP_{PS}	0	0	50	kg
cost of underground container	P_{cont_DO}	0	10300	0	€
cost of container 240l	P_{cont_PS}	0	0	58	€
cost of a bag	P_{bag}	0.055	0	0	€
maintenance of container	M_{cont}	0	60	0	€/year
depreciation (container)	dep_{cont}	0	10	10	year
other					
interest rate of the investment	r	5%	5%	5%	%
frequency of collection	$freq_{col}$	26	depends on the scenario	52	times/year

2.2.3. Collection costs formulas

The formulas used in the calculations of the collection costs are presented below. These are directly taken from Groot et al. (2014), except for the calculation of the annuity factor for the capital costs that was retrieved from Kenton (2019). The emissions costs defined by Groot et al. (2014) are assumed to be included in the price of fuel and are not calculated separately.

The total costs (T) are subdivided into vehicle costs (V), labour costs (L), and containers and bags costs (C&B).

$$T = V + L + C\&B$$

2.2.3.a. Vehicle costs

The vehicle costs are further subdivided into variable (V_v) and fixed costs (V_f)

$$V = V_v + V_f$$

Vehicle variable costs

Variable costs are associated with the use of the vehicle. The two expenses that directly depend on the vehicle use are the cost of fuel (F) and the costs of vehicle maintenance (M_{veh}).

$$V_v = F + M_{veh}$$

During collection, the vehicle can be driving between collection stops, idling during the loading of the waste at a collection stop, or hauling to unload the truck in the MRF. The sum of the fuel costs during these three activities, namely, driving (F_d), idling (F_i), and hauling (F_h), determine the total costs of fuel.

$$F = F_d + F_i + F_h$$

Each of the fuel costs depend on the total consumption of fuel and the price of fuel (P_f). The fuel consumed during driving depends on the fuel consumption while driving (FC_d) times the distance travelled while driving (D_d). The fuel consumed during idling depends on the fuel consumption while idling (FC_i) times the idling time (T_i). The fuel consumed during hauling depends on the fuel consumption while hauling (FC_h) times the distance travelled while hauling (D_h).

$$F_d = FC_d * D_d * P_f$$
$$F_i = FC_i * T_i * P_f$$
$$F_h = FC_h * D_h * P_f$$

The distance travelled while driving is calculated multiplying the number of stops (n_{stop}) minus one, times the distance between stops (d_d). The time spent idling equals to the number of stops times the time spent idling per stop (t_{stop}). The distance travelled while hauling is two times the distance to hauling (d_h) times the number of loads (n_{loads})

$$D_d = (n_{stop} - 1) * d_d$$
$$Ti = n_{stop} * t_{stop}$$
$$D_h = 2 * d_h * n_{loads}$$

The number of stops is calculated in two different ways depending on the collection system. For kerbside and post-separation collection systems it is the number of households (n_{hh}) times the frequency of collection ($freq_{col}$) divided by the number of households per stop (n_{hh_stop}). For drop-off collection it is assumed that the truck has to haul to the unloading station every time it collects a container. In other words, the number of stops of drop-off systems is the same as the number of loads. The number of loads is estimated from the total amount of waste (W) and the average truck load ($truck_{load}$).

$$n_{stop} = (n_{hh} * freq_{col}) / n_{hh\_stop}, \text{ for KS and PS}$$
$$n_{stop} = n_{loads}, \text{ for DO}$$
$$n_{loads} = W / truck_{load}$$

Table XXIII. Summary of the formulas used in the calculation of the variable costs of the vehicle

	formula
cost of vehicle variable	$V_v = F + M_{veh}$
cost of fuel	$F = F_d + F_i + F_h$
cost driving	$F_d = FC_d * D_d * P_f$
distance driving	$D_d = (n_{stop} - 1) * d_d$
number of stops	$n_{stop} = (n_{hh} * freq_{col}) / n_{hh\_stop}$, for KS and PS
	$n_{stop} = n_{loads}$, for DO
cost idling	$F_i = FC_i * T_i * P_f$
time idling	$Ti = n_{stop} * t_{stop}$
cost hauling	$F_h = FC_h * D_h * P_f$
distance hauling	$D_h = 2 * d_h * n_{loads}$

number of loads	$n_{loads} = W / truck_{load}$
cost of vehicle maintenance	M_{veh}

Vehicle fixed costs

The fixed costs of vehicles depend on the number of vehicles (n_{veh}) times the capital costs per vehicle (C_{veh}), the insurance costs (C_{ins}), and the vehicle taxes (C_{tax}).

$$V_f = n_{veh} * (C_{veh} + C_{ins} + C_{tax})$$

The number of vehicles depends on the time required for collection (T_{col}), the time one vehicle can be used per year (T_{veh}), and the effective availability of the vehicle (av_{veh}) due to maintenance or other reasons.

$$n_{veh} = (T_{col} / T_{veh}) * (1 / av_{veh})$$

Time of collection is the sum of time spend during the three collection activities, namely, time driving (T_d), time idling (T_i), and time hauling (T_h).

$$T_{col} = T_d + T_i + T_h$$

Time driving is calculated dividing the distance travelled while driving (D_d) by the average speed while driving (v_d). The time hauling is the division of the distance travelled while hauling (D_h) by the average seed while hauling (v_h). Time idling was defined before.

$$T_d = D_d / v_d$$
$$T_h = D_h / v_h$$

The annual capital costs of the vehicle are the total costs of investment (C_{veh_inv}) minus the salvage costs (C_{veh_sal}) of the vehicle divided by the annuity factor. The annuity factor (AF) incorporates the depreciation times (dep_{veh}) and interest rates (r) to calculate the annual capital costs of the vehicle.

$$C_{veh} = (C_{veh\_inv} - C_{veh\_sal}) / AF$$

$$AF = \frac{\left(1 - \frac{1}{(1+r)^{dep_{veh}}}\right)}{r}$$

Table XXIV. Summary of the formulas used in the calculation of the fixed costs of the vehicle

	formula
cost of vehicle fixed	$V_f = n_{veh} * (C_{veh} + C_{ins} + C_{tax})$
number of vehicles	$n_{veh} = (T_{col} / T_{veh}) * (1 / av_{veh})$
time collection	$T_{col} = T_d + T_i + T_h$
time driving	$T_d = D_d / v_d$
time idling	$Ti = n_{stop} * t_{stop}$
time hauling	$T_h = D_h / v_h$
capital cost vehicle	$C_{veh} = (C_{veh\_inv} - C_{veh\_sal}) / AF$
annuity factor	$AF = \frac{\left(1 - \frac{1}{(1+r)^{dep_{veh}}}\right)}{r}$
cost of vehicle insurance	C_{ins}

cost of vehicle taxes	C_{tax}

2.2.3.b. Labour costs

The costs of personnel are limited to the cost associated with the wages of drivers (C_{driver}) and loaders (C_{loader}) and the number of vehicles.

$$L = (C_{driver} + C_{loader}) * n_{veh}$$

The annual cost of a drivers is the product of the wage of the driver (w_{driver}) and the number of drivers per year ($n_{drivers}$). The annual cost of loaders is the multiplication of the wage of the loader (w_{loader}) and the number of loaders required per year ($n_{loaders}$). It is assumed that the number of drivers or loaders depends linearly on the required collection time and can be a non-integer. The number of drivers or loaders per year is the product of multiplying the number of drivers (n_{driver_veh}) or loaders (n_{loader_veh}) per vehicle by the T_{veh} and diving by the working hours of a driver (t_{driver}) or loader (t_{loader}) per year.

$$C_{driver} = n_{drivers} * w_{driver}$$
$$C_{loader} = n_{loaders} * w_{loader}$$
$$n_{drivers} = n_{driver\_veh} * T_{veh} / t_{driver}$$
$$n_{loaders} = n_{loader\_veh} * T_{veh} / t_{loader}$$

Table XXV. Summary of the formulas used in the calculation of the labour costs of the vehicle.

	formula
cost of labour	$L = (C_{driver} + C_{loader}) * n_{veh}$
cost of driver	$C_{driver} = n_{drivers} * w_{driver}$
number of drivers	$n_{drivers} = n_{driver\_veh} * T_{veh} / t_{driver}$
cost of loader	$C_{loader} = n_{loaders} * w_{loader}$
number of loaders	$n_{loaders} = n_{loader\_veh} * T_{veh} / t_{loader}$

2.2.3.c. Container and bags costs

The costs of containers and bags is the sum of the costs of containers (C_{cont}) and the costs of bags (C_{bags}).

$$C\&B = C_{cont} + C_{bags}$$

There are two types of containers, DO or PS containers. The costs of each container type are calculated in the same way, multiplying the number of containers (n_{cont_DO}. n_{cont_PS}) times the capital costs of a container ($C_{cont\_DO\_cap}$, $C_{cont\_PS\_cap}$) and the maintenance costs (M_{cont_DO}, M_{cont_PS}).

$$C_{cont\_DO} = n_{cont\_DO} * (C_{cont\_DO\_cap} + M_{cont\_DO})$$
$$C_{cont\_PS} = n_{cont\_PS} * (C_{cont\_PS\_cap} + M_{cont\_PS})$$

The capital costs are calculated the same way than the capital costs of the vehicle, but with no salvage costs. This means, capital costs of the container are the division of the investment costs (P_{DO}, P_{PS}) by the annuity factor (AF).

$$C_{cont\_DO\_cap} = P_{DO} / AF$$
$$C_{cont\_PS\_cap} = P_{PS} / AF$$

To estimate the costs of bags, it is assumed that one bag is used per collection. The costs of bags depend on the number of households (n_{hh}) times the frequency of collection ($freq_{col}$) times the price of each bag (P_{bag})

$$C_{bags} = n_{hh} * freq_{col} * P_{bag}$$

Table XXVI. Summary of the formulas used in the calculation of the container and bags costs of the vehicle.

	formula
cost of containers and bags	$C\&B = C_{cont} + C_{bags}$
cost of containers	$C_{cont\_DO} = n_{cont\_DO} * (C_{cont\_DO\_cap} + M_{cont\_DO})$, for DO
	$C_{cont\_PS} = n_{cont\_PS} * (C_{cont\_PS\_cap} + M_{cont\_PS})$, for PS
capital cost container	$C_{cont\_DO\_cap} = P_{DO} / AF$, for DO
	$C_{cont\_PS\_cap} = P_{PS} / AF$, for PS
annuity factor	$AF = \frac{\left(1 - \frac{1}{(1+r)^{dep_{cont}}}\right)}{r}$
cost of bags	$C_{bags} = n_{hh} * freq_{col} * P_{bag}$

2.2.4. Density of drop-off containers

The number of drop-off containers per household (container/hh) was used to estimate the distance between containers and the number of containers of each scenario. CBS (2019) defines five urbanity levels depending on the density of addresses in a certain region, see Table XXVII. Based on this criteria, the average container densities of municipalities with DO collection systems were calculated and further classified into urbanisation levels, see Table XXVIII (CBS, 2020).

Table XXVII. Urbanisation classes according to CBS

urbanisation	
class	address/km^2
1	>2500
2	1500-2500
3	1000-1500
4	500-1000
5	<500

Table XXVIII. Average urbanisation, household, and container densities of Dutch municipalities with DO collection systems attending to their urbanisation classes.

	urbanisation		household	container	
type of collection	class	address/km^2	hh/km^2	container/km^2	container/hh
DO	1	2939	1434	2.3	0.0006
DO	2	1925	826	1.5	0.0014
DO	3	1254	342	0.9	0.0019
DO	4	704	97	0.3	0.0030
DO	5	395	135	0.6	0.0048

This classification was used to assign container densities to the different participation levels. The Netherlands has an average urbanisation level of 2 (1994 addresses/km^2). Assuming a linear regression from the empirical data from Table XXVIII, the container density for the Netherlands is 1.6 containers/km^2 or 0.0015 containers/household, based on data from CBS. This container density was assigned to the average participation category. The number of container/hh for low participation scenarios was estimated from highly urbanised cities, such as Rotterdam or The Hague, while high participation was estimated from the third urbanity class. Attending to the number of containers/hh, the average distance between containers is calculated. It is assumed that the containers are evenly distributed among a certain area with a fixed household density and that the effective distance between containers is 1.5 times the straight-line separation between two containers to account the effects of typical urban designs. The final values are presented in Table XXIX

.

Table XXIX. Number of containers per household and average distance between containers or DO collection systems

participation	container/hh	distance between containers
low	0.0007	2.5
average	0.0015	1.5
high	0.002	1

In Figure XIV, The Hague is used as an example to show the distance between drop-off containers.

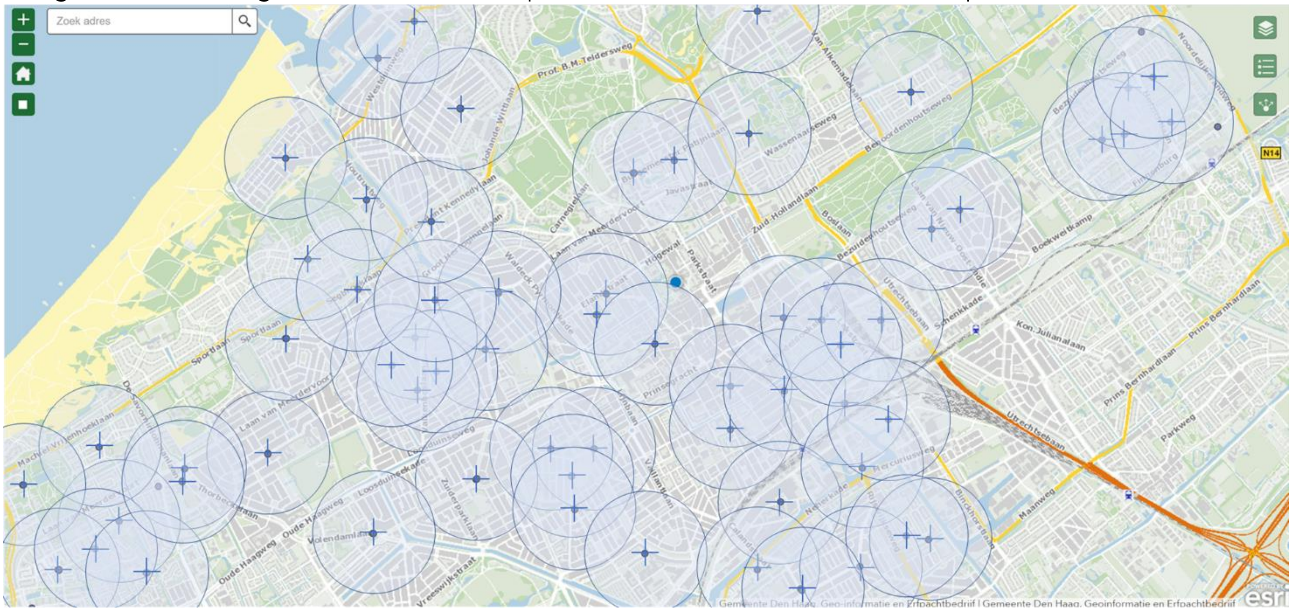

Figure XIV. Location of PMD containers in The Hague. The blue circles have a radius of 500m.

3. S2. Sorting

3.1. Configuration of the MRF's

The configuration of the three levels of MRF's is based on Cimpan et al. (2016). $MRF_{KS/DO}$ are directly derived from Cimpan et al. (2016), while for MRF_{PS} an extra trommel is installed at the beginning of the plant. The process diagrams are presented in Figures Figure XV-Figure XVII.

The plant layouts were validated with designs reported by Dutch sorting plants such as Suez'z MRF and AVR post-separation plant in Rotterdam (AVR, 2020; SUEZ, 2020).

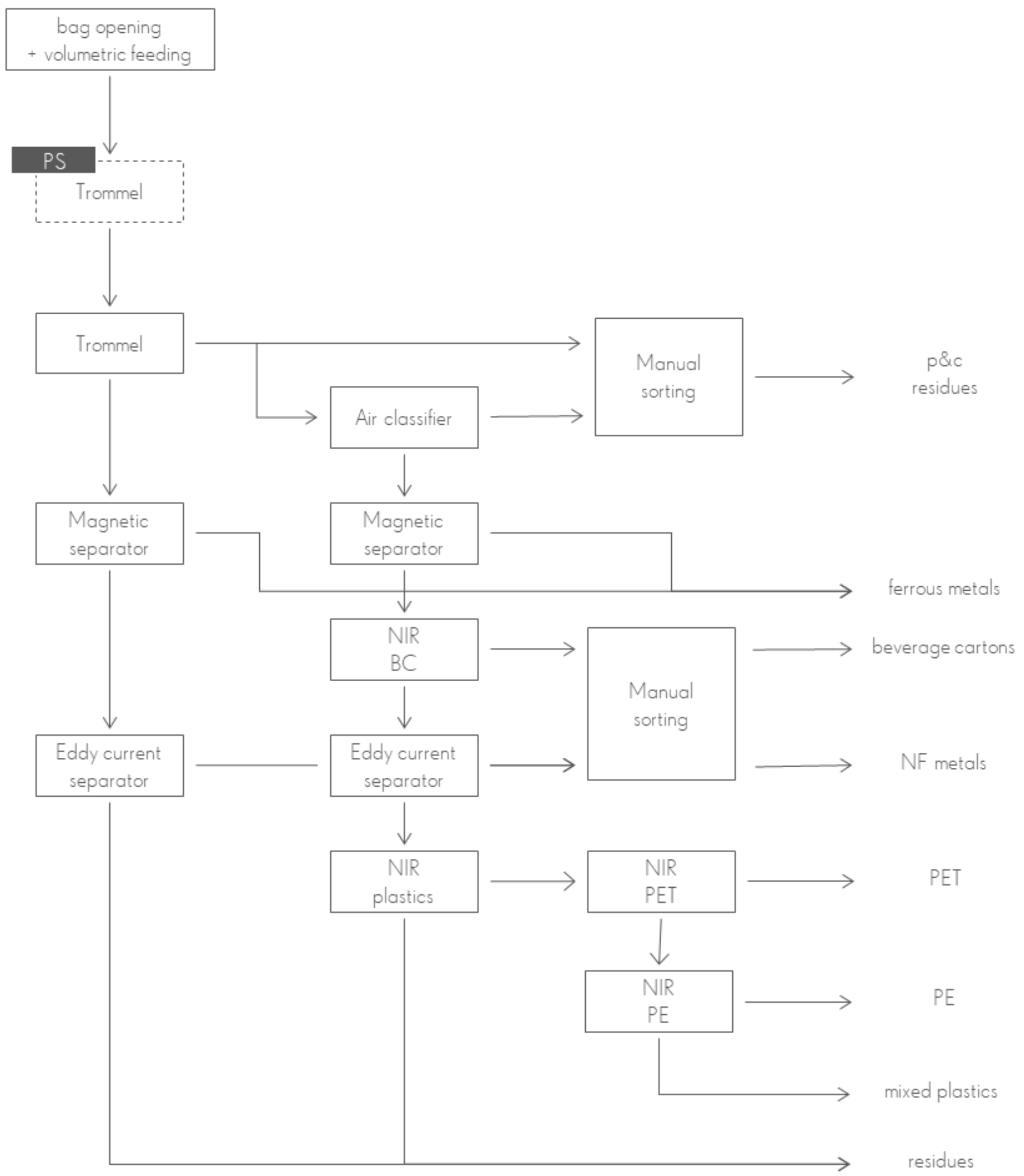

Figure XV. Process diagram of the basic MRF. In dashed line and labelled with PS is an extra trommel that is only included in the MRF_{PS}

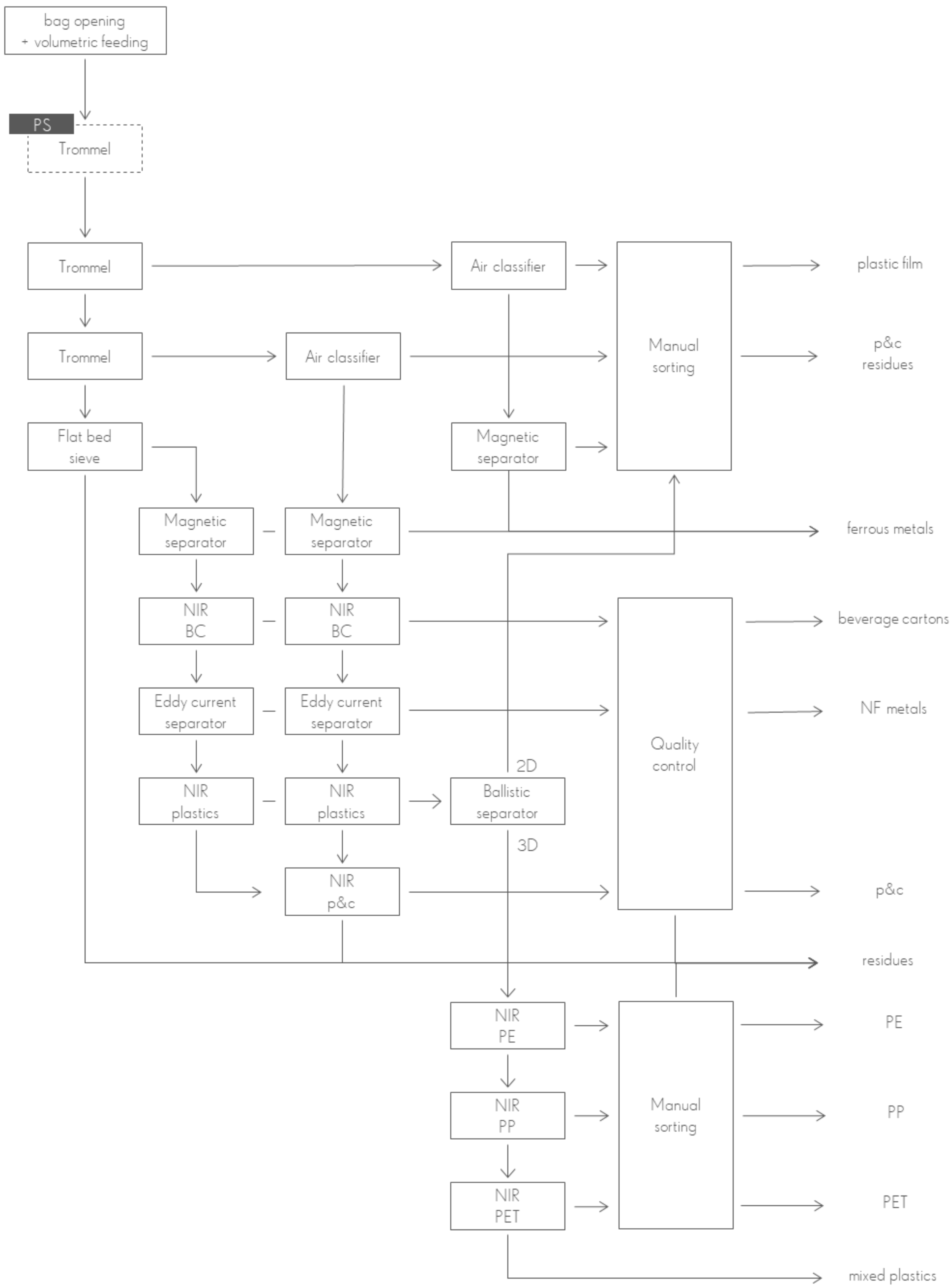

Figure XVI. Process diagram of the medium MRF. In dashed line and labelled with PS is an extra trommel that is only included in the MRF_{PS}

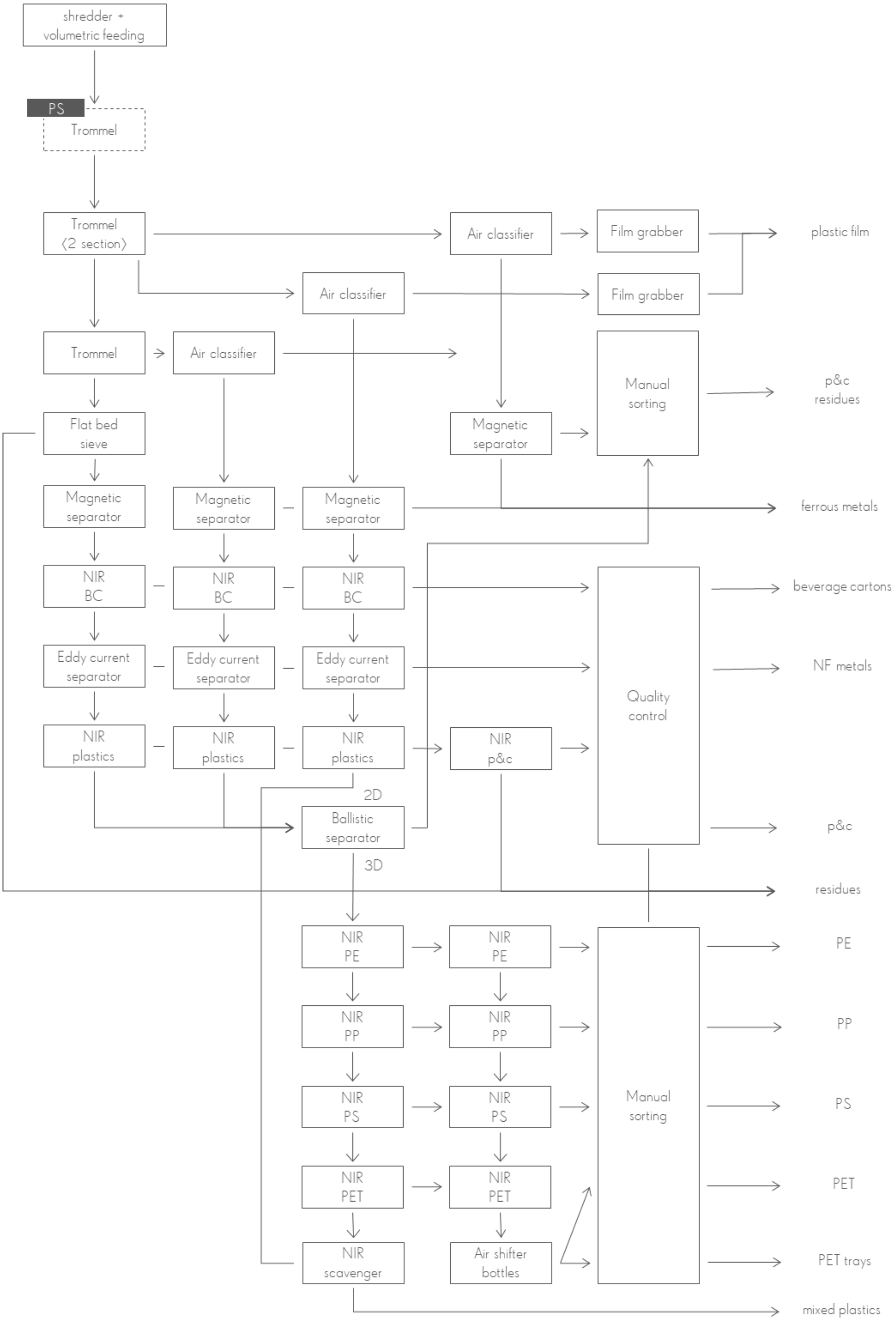

Figure XVII. Process diagram of the advanced MRF. In dashed line and labelled with PS is an extra trommel that is only included in the MRF_{PS}

3.2. TECHNICAL ASSESSMENT

3.2.1. Sorting efficiency MRF

The detailed sorting efficiencies of the MRF's at the packaging level are presented in Tables Table XXXII and Table XXXIII. The values in the tables express the percentage of certain packaging that is directed to the output, in weight.

3.2.2. Quality of the sorted materials

The criteria to define the quality of the sorted materials was taken from DKR specifications (2020) and Eriksen et al. (2019). The main targeted material for each sorted output and DKR specification used to define the high-quality class is presented in the table below.

Table XXX. Main targeted material and DKR specification of the MRF outputs.

output	main targeted plastic	specification
PET	PET-B	DKR 328-1
PET trays	PET-R	DKR 328-5
PE	PE-B, PE-R	DKR 329
PP	PP-B, PP-R	DKR 324
PS	PS-B, PS-R	DKR 331
film	F	DKR 310
mix	PET, PE, PP, and PS B and R	DKR 350

The classification into quality levels is based on highest level of impurities or accepted plastics admitted in a sorted output and are presented in Table XXXI. The results are translated to minimum purity requirement in Table IX, in the main paper.

Table XXXI. Quality levels based on the maximum polymeric contamination of the product for the outputs of the sorting stage (MRF). Empty values (-) mean that the type of output cannot achieve certain quality levels.

	high			medium			low		
output	other plastic accepted	other plastic impurities	non-plastic impurities	other plastic accepted	other plastic impurities	non-plastic impurities	other plastic accepted	other plastic impurities	non-plastic impurities
PET	10%	4%	4%	15%	7%	7%	20%	10%	10%
PET trays	20%	6%	6%	20%	10%	8%	20%	12%	10%
PE	-	7%	5%	-	10%	10%	-	18%	18%
PP	-	4%	4%	-	12%	12%	-	17%	17%
PS	1%	4%	4%	2%	5%	5%	3%	8%	8%
film	-	4%	4%	-	5%	5%	-	8%	8%
mix	-	-	-	-	5%	5%	-	8%	8%

Table XXXII. Sorting efficiency of the three levels of $MRF_{KS/DO}$ at the packaging level. The numbers represent percentages (%). B = bottle. F = film. R = rigid. L = laminated. NP = non-packaging. BC = beverage cartons. p&c = paper and cardboard. LAMD = land and attached moisture and dirt.

basic

CODE	PET	PET trays	PE	PP	PS	film	mix	BC	metal	residues
PET-B	68		0				15	0	1	16
PET-F	0						42	4		54
PET-R	2		0				59	1	1	37
PE-B			63				9	0	0	27
PE-F	1		6				47	0	1	46
PE-R	1		21				32	0	1	45
PP-B			1				71			28
PP-F	2		1				29	0	1	67
PP-R	1		1				31	0	1	65
PS-B							78		1	21
PS-F							66		3	31
PS-R	2		1				64	1	1	32
EPS-R	3						65		1	31
PVC-F							36		1	63
PVC-R			0				47	1	0	52
mix-F	0		0				42	1	1	56
mix-R	1		1				50	2	0	46
mix-L	3		10				55	1	2	29
PET-NP	20						41	1	0	38
PE-NP	0		9				23	0	1	66
PP-NP			0				25	0	0	74
PS-NP	5						68		0	27
PVC-NP	1						30	0	0	69
mix-NP	0		0				37	0	1	60
other-NP			15				23			62
BC			0				9	81	2	8
metal	0		0				8	0	77	14
organic	3		0				19	1	2	76
textile			0				21	1	0	78
p&c	0		0				31	4	3	62
glass							7	0	0	93
LAMD	6		1				6	18	4	65

medium

CODE	PET	PET trays	PE	PP	PS	film	mix	BC	metal	residues
PET-B	79		0	0		0	8	0	0	12
PET-F	0			0		13	23	2		66
PET-R	3		0	0		2	31	0	1	64
PE-B			76	1		0	5	0	0	17
PE-F	0		4	0		71	24	0	1	23
PE-R	1		28	2			17	0	0	52
PP-B			0	67			36			19
PP-F	1		0			37	17	0	1	56
PP-R	1		1	38		2	17	0	0	54
PS-B							40		1	59
PS-F				2		37	35		2	37
PS-R	1		0	0		2	32	0	1	63
EPS-R	2			1		7	33		0	56
PVC-F						3	20		1	77
PVC-R			0	0		3	25	0	0	71
mix-F	0		0	0		56	23	1	0	38
mix-R	0		1	4		2	27	1	0	65
mix-L	2		5	3		29	30	0	1	39
PET-NP	12					1	21	1	0	65
PE-NP	0		7	0		64	12	0	1	38
PP-NP			0	2		5	13	0	0	80
PS-NP	3			0		0	34		0	62
PVC-NP	0			0		1	15	0	0	82
mix-NP	0		0	2		1	19	0	1	77
other-NP			11	5			12			72
BC			0	0		0	5	85	1	8
metal	0		0	0		0	4	0	83	11
organic	1		0	2		0	10	0	1	85
textile			0	0		0	11	0	0	88
p&c	0		0	1		3	16	2	2	76
glass							4	0	0	96
LAMD	4		1	3		11	3	14	3	62

advanced

CODE	PET	PET trays	PE	PP	PS	film	mix	BC	metal	residues
PET-B	90	4	0	0		0	1	0	0	4
PET-F	0	20		0		17	3	1		59
PET-R	3	80	0	0		2	2	0	0	12
PE-B			90	1		0	1	0	0	8
PE-F	0	0	3	0		95	1	0	0	0
PE-R	0	3	35	2	1		2	0	0	57
PP-B			0	90			1			9
PP-F	0	1				49	5	0	0	44
PP-R	0	1	1	51		2	2	0	0	42
PS-B					70		1		0	28
PS-F				2		49	5		0	44
PS-R	0	4	0	0	70	2	0	0	0	22
EPS-R	1	2		1	70	7	0		0	19
PVC-F		3				3	3		0	91
PVC-R		4	0	0		3	4	0	0	89
mix-F	0	0		0		75	4	0	0	21
mix-R	0	2	0	4		2	4	0	0	87
mix-L	1	3		3		39	5	0	0	50
PET-NP	4	7				1	1	0	0	86
PE-NP	0	0	5	0	1	85	1	0	0	8
PP-NP		0	0	2		5	1	0	0	92
PS-NP	1	0		0	70	0	0		0	28
PVC-NP	0	0		0		1	1	0	0	97
mix-NP	0	0	0	2		1	1	0	0	96
other-NP			7	5			1			87
BC		0	0	0		0	0	90	1	8
metal	0	0	0	0		0	0	0	90	9
organic	1	0	0	2		0	1	0	0	96
textile		1	0	0		0	1	0	0	97
p&c	0	1	0	1		3	1	0	0	93
glass		0					0	0	0	100
LAMD	1	2		3		11	0	10	1	71

Table XXXIII. Sorting efficiency of the three levels of MRF_{PS} at the packaging level. The numbers represent percentages (%). B = bottle. F = film. R = rigid. L = laminated. NP = non-packaging. BC = beverage cartons. p&c = paper and cardboard. LAMD = land and attached moisture and dirt.

CODE	basic PET	basic PET trays	basic PE	basic PP	basic PS	basic film	basic mix	basic BC	basic metal	basic residues
PET-B	57		1				13	0	1	28
PET-F							39			61
PET-R	1		0				71	0	1	27
PE-B	0		42				13	0	0	45
PE-F	0		0				7	0	1	91
PE-R	0		10				11	0	1	77
PP-B	1		6				71			22
PP-F							2	0	1	96
PP-R	0		1				9	0	1	88
PS-B							6		1	93
PS-F									3	97
PS-R							3		1	96
EPS-R							3		1	96
PVC-F							2	1	1	95
PVC-R			1				2		0	97
mix-F							2	0	1	97
mix-R	0		0				6	0	0	93
mix-L			0				9	0	2	88
PET-NP	3						17	0	0	79
PE-NP			56				12	0	1	31
PP-NP	0		0				8	0	0	91
PS-NP							6		0	93
PVC-NP							2		0	97
mix-NP							9		1	90
other-NP			12				5			83
BC	0						0	26	2	71
metal	0		0				0	0	77	23
organic	0		0				0	0	0	98
textile							0	0	0	100
p&c	0						0	0	1	97
glass							0		0	100
LAMD	0		0				1	1	2	93

CODE	medium PET	medium PET trays	medium PE	medium PP	medium PS	medium film	medium mix	medium BC	medium metal	medium residues
PET-B	66		1	1		0	7	0	0	25
PET-F						8	21			74
PET-R	1		0	1		0	49	0	1	48
PE-B	0		50	3			7	0	0	39
PE-F	0		0	0		41	4	0	1	68
PE-R	0		13	1		0	6	0	0	79
PP-B	1		5	31			36			38
PP-F				1		14	1	0	1	88
PP-R	0		1	23		0	5	0	0	78
PS-B							3		1	96
PS-F									2	98
PS-R							1		1	98
EPS-R							1		0	98
PVC-F						9	1	1	1	88
PVC-R			1				1		0	98
mix-F						31	1	0	0	78
mix-R	0		0	0		0	3	0	0	95
mix-L			0	0		7	5	0	1	88
PET-NP	2					1	9	0	0	88
PE-NP			42	1		30	6	0	1	31
PP-NP	0		0	2		3	4	0	0	91
PS-NP						0	3		0	96
PVC-NP				0		0	1		0	98
mix-NP				0		0	4		1	95
other-NP			9				3			88
BC	0			0		0	0	28	1	70
metal	0		0	0		0	0	0	83	16
organic	0		0	0		0	0	0	0	99
textile						0	0	0	0	100
p&c	0			0		0	0	0	0	98
glass						0	0		0	100
LAMD	0		0	0		0	1	0	2	95

CODE	advanced PET	advanced PET trays	advanced PE	advanced PP	advanced PS	advanced film	advanced mix	advanced BC	advanced metal	advanced residues
PET-B	75	9	1	1		0	1	0	0	13
PET-F						10	3			87
PET-R	1	45	0	1		0	3	0	0	49
PE-B	0		59	3			1	0	0	37
PE-F	0	0	0	0		54	0	0	0	45
PE-R	0	0	16	1	1	0	1	0	0	81
PP-B	0	1	3	41			1			53
PP-F		0		1		18	0	0	0	80
PP-R	0	3	0	31		0	1	0	0	65
PS-B		3			35		0		0	62
PS-F									0	100
PS-R		0			35		0		0	65
EPS-R					35		0		0	65
PVC-F						9	0	1	0	90
PVC-R		0	1				0		0	99
mix-F		0				41	0	0	0	59
mix-R	0	0	0	0		0	1	0	0	99
mix-L		1	0	0		9	1	0	0	89
PET-NP	1	3				1	1	0	0	94
PE-NP		0	28	1	1	40	0	0	0	30
PP-NP	0	1	0	2		3	0	0	0	94
PS-NP					35	0	0		0	65
PVC-NP				0		0	0		0	100
mix-NP				0		0	0		0	99
other-NP			6				0			94
BC	0	0		0		0	0	30	1	69
metal	0	0	0	0		0	0	0	90	10
organic	0	0	0	0		0	0	0	0	100
textile		0				0	0	0	0	100
p&c	0	0		0		0	0	0	0	99
glass						0	0		0	100
LAMD	0	0	0	0		0	0	0	1	98

3.3. ECONOMIC ASSESSMENT

Economic data for the sorting costs was retrieved from Cimpan et al. (2016) and Pressley et al. (2015).

4. S3. REPROCESSING

4.1. TECHNICAL ASSESSMENT

4.1.1. Material composition

Packaging objects are commonly made from multiple materials, see Figure XVIII. The average material composition of the PCPPW is presented in Table XXXIV

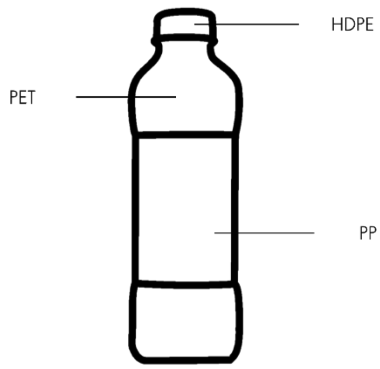

Figure XVIII. Material composition of a PET plastic bottle

Table XXXIV. Material composition of the individual packaging products. B = bottle. F = film. R = rigid. L = laminated. NP = non-packaging. BC = beverage cartons. p&c = paper and cardboard. LAMD = land and attached moisture and dirt.

	plastics						other materials					
CODE	PET	PE	PP	PS	PVC	other plastic	metal	organic	textile	p&c	glass	rest
PET-B	84%	8%	7%							1%		
PET-F	97%									3%		
PET-R	89%	1%	6%		1%			0%	0%	4%		
PE-B	1%	95%	2%	0%						2%		
PE-F		96%								3%		
PE-R	6%	89%	1%	0%	2%		1%			1%		
PP-B		10%	87%			1%				2%		
PP-F			98%							2%		
PP-R	1%	2%	94%	0%				1%	1%	1%		
PS-B			4%	96%								
PS-F				100%								
PS-R	1%	2%	1%	86%			1%			8%		
EPS-R				100%								
PVC-F		1%		1%	96%					1%		
PVC-R					99%					1%		
mix-F						100%						
mix-R						100%						
mix-L	10%	57%	20%		6%		7%					
PET-NP	100%											
PE-NP		100%										
PP-NP			100%									
PS-NP				100%								

PVC-NP				100%							
mix-NP					100%						
other-NP	100%										
BC	20%					5%			75%		
metal						100%					
organic							100%				
textile								100%			
p&c									100%		
glass										100%	
LAMD											100%

4.2. Economic assessment

Economic data for the reprocessing costs was retrieved from Faraca et al. (2019) and consultations with experts. Washing costs in the RP_{PS} plant are estimated to be higher and are translated into a higher water consumption.

Table XXXV. Reprocessing costs

parameter	$RP_{KS/DO}$	RP_{PS}	units
reprocessing plant building costs	19	19	€/ton
equipment			
shredder	16	16	€/ton
washer	16.2	16.2	€/ton
dryer	21	21	€/ton
extruder / pelletiser	120	120	€/ton
labour	45.3	45.3	€/ton
resources			€/ton
electricity	16	16	€/ton
heat	5	5	€/ton
water	10	10	€/ton
treatment			€/ton
water	37	52	€/ton
insurance	1.3	1.3	€/ton
maintenance	5.3	5.3	€/ton

5. S4. MSWI

5.1. Technical assessment

The technical data for the incineration plant was retrieved from Faraca et al. (2019), Haupt et al. (2018), Rijkswaterstaat (2018).

5.2. Economic assessment

Economic data for the incineration costs was retrieved from Faraca et al. (2019) and communication with the municipality of Rotterdam on carbon taxes (32.2 €/ton).

6. RESULTS

6.1. ENVIRONMENTAL ASSESSMENT

6.1.1. Impact categories

The explanation of the impact categories analysed in this study can be found in Table XXXVI. These are the definitions used in the ReCiPe method.

Table XXXVI. Midpoint impact categories and characterisation factors. NMVOC: Non-Methane Volatile Organic Carbon compound. 1,4-DCB: 1,4 dichlorobenzene.
[a] unit of the physical or chemical phenomenon.
[b] unit of the indicator result

Impact category	Abbr.	Indicator	Unit [a]	Characterisation Factor	Unit [b]
climate change	CC	increase in infra-red radiative forcing	W*yr/m2	global warming potential	kg CO_2 to air
fossil depletion	FD	upper heating value	MJ	fossil fuel potential	kg oil
human toxicity	HT	risk increase in cancer disease incidence	-	human toxicity potential	kg 1,4-DCB-eq to urban air
particulate matter formation	PMF	increase in PM10 population intake	kg	particulate matter formation potential	kg PM10-eq to air
photochemical oxidant formation	POF	increase in tropospheric ozone	kg	photochemical oxidant formation potential	kg NMVOC to air
terrestrial acidification	TA	proton increase in natural soils	$yr*m^2*mol/l$	terrestrial acidification potential	kg SO_2 to air

6.1.2. LCI

The life cycle inventory data used in the environmental assessment is presented here. The unit processes are dividing according to stages and the environmental aspects are calculated in a modular approach as defined by Haupt et al. (2018b).

S1. Collection

Table XXXVII. Unit process for the collection of PCPPW in the collection stage (S1)

Input	amount	units	data source
plastic waste	depends on scenario	kg	
Transport		ton-km	
collection bag		bags	
DO container		DO containers	
PS container		PS containers	

Table XXXVIII. Unit process for the collection of MSW in the collection stage (S1)

Input	amount	units	data source
MSW	depends on scenario	kg	
Transport		ton-km	
collection bag		bags	
PS container		PS containers	

Table XXXIX. Unit process for the Transport in the collection stage (S1)

Input	amount	units	data source
Transport	1	ton-km	

lorry 16-32 metric ton, EURO6 {RER}	depends on scenario	lorry	scenario
maintenance, lorry 16-32 metric ton, EURO6 {RER}	depends nb lorry	-	scenario
road	1.05E-03	m/year	ecoinvent 3.4
brake wear emissions	2.22E-05	kg	ecoinvent 3.4
tyre wear emissions	2.20E-04	kg	ecoinvent 3.4
road wear emissions	1.91E-05	kg	ecoinvent 3.4
diesel low sulphur	depends on scenario	l	scenario

Table XL. Unit process for the collection bags in the collection stage (S1)

Input	amount	units	data source
collection bag	1	bags	
extrusion production, plastic film ⟨RER⟩	0.02	kg	Haupt et al. (2018b)
Polyethylene, low density, granulate {GLO}| market for | Alloc Rec, U	0.02	kg	Haupt et al. (2018b)

Table XLI. Unit process for the DO container in the collection stage (S1)

Input	amount	units	data source
DO container	1	DO container	
production of DO container	1	DO container	
steel, low alloyed	205	kg	Iriarte et al. (2009)
concrete	1	m^3	Iriarte et al. (2009)
excavation	13	m^3	Iriarte et al. (2009)
maintenance of a DO container		-	
water	1.1	m^3	Iriarte et al. (2009)

Table XLII. Unit process for the PS container in the collection stage (S1)

Input	amount	units	data source
PS container	1	PS container	
production of DO container	1	PS container	
Injection moulding {RER}| processing	14.1	kg	Iriarte et al. (2009)
polyethylene, high density, granulate {GLO}| market for	14.1	kg	Iriarte et al. (2009)
maintenance of a PS container		-	
water	1.2	m^3	Iriarte et al. (2009)

S2. Sorting

Table XLIII. Unit process for the sorting in the sorting stage (S2)

Input	amount	units	data source
plastic waste / MSW	1	kg	
MRF	depends on scenario	MRF	scenario
electricity low voltage, NL	depends on scenario	kW	scenario
diesel low sulphur		m/year	ecoinvent 3.4
baling	1	wire/kg	
steel, low-alloyed {GLO}| market for	0.0056	kg	Haupt et al. (2018b)
wire drawing, steel {RER}| processing	0.0056	kg	Haupt et al. (2018b)

Output			
sorted plastic	depends on MRF	kg	Scenario

S3. Reprocessing

Table XLIV. Unit process for the reprocessing in the reprocessing stage (S3)

Input	amount	units	data source
sorted plastic	1	kg	
RP	depends on scenario	RP	scenario
electricity low voltage, NL	depends on scenario	kW	scenario
heat, natural gas, RER	depends on scenario	MJ	scenario
water use	1	water/kg	
water	9	m^3	Faraca et al. (2019)
water treatment	9	m^3	Faraca et al. (20199

S4. MSWI

Table XLV. Unit process for the MSWI in the incineration stage (S4)

Input	amount	units	data source
plastics to MSWI	1	kg	
MSWI	depends on scenario	MSWI	scenario
treatment of 1kg of plastic waste	depends on scenario	-	scenario
Output			
electricity	1.2	kW	
heat	6.75	MJ	

6.1.3. Detailed environmental impacts

The detailed environmental performance of the different scenarios is presented in Figures Figure XIX-Figure XXIV, for each individual impact category.

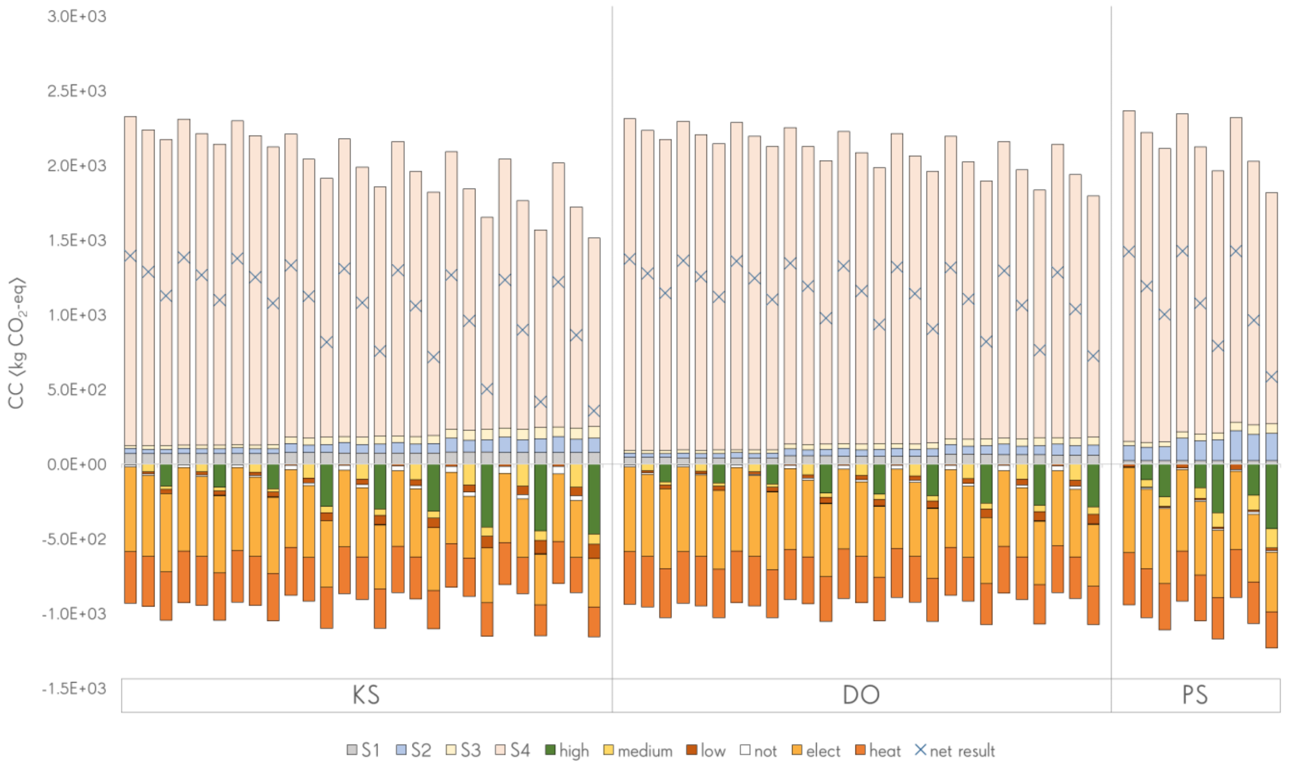

Figure XIX. Environmental performance of the modelled scenarios in terms of climate change

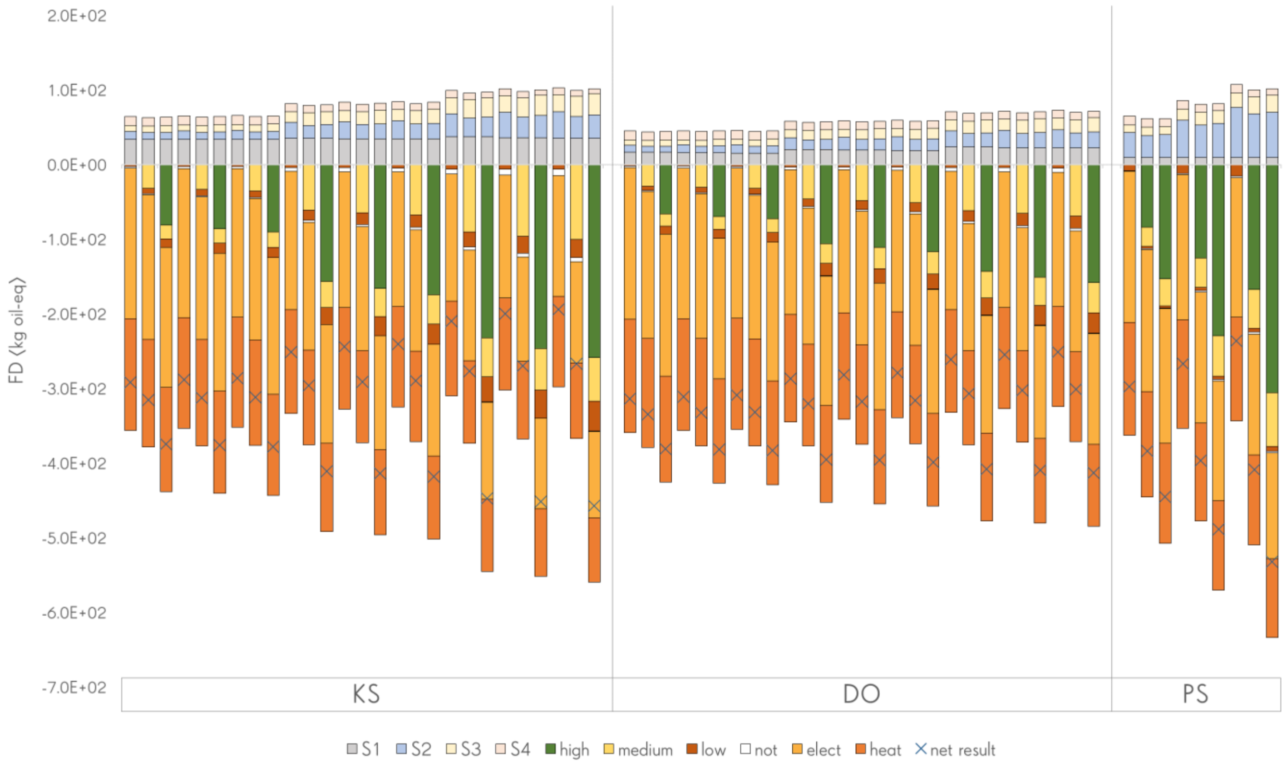

Figure XX. Environmental performance of the modelled scenarios in terms of fossil depletion

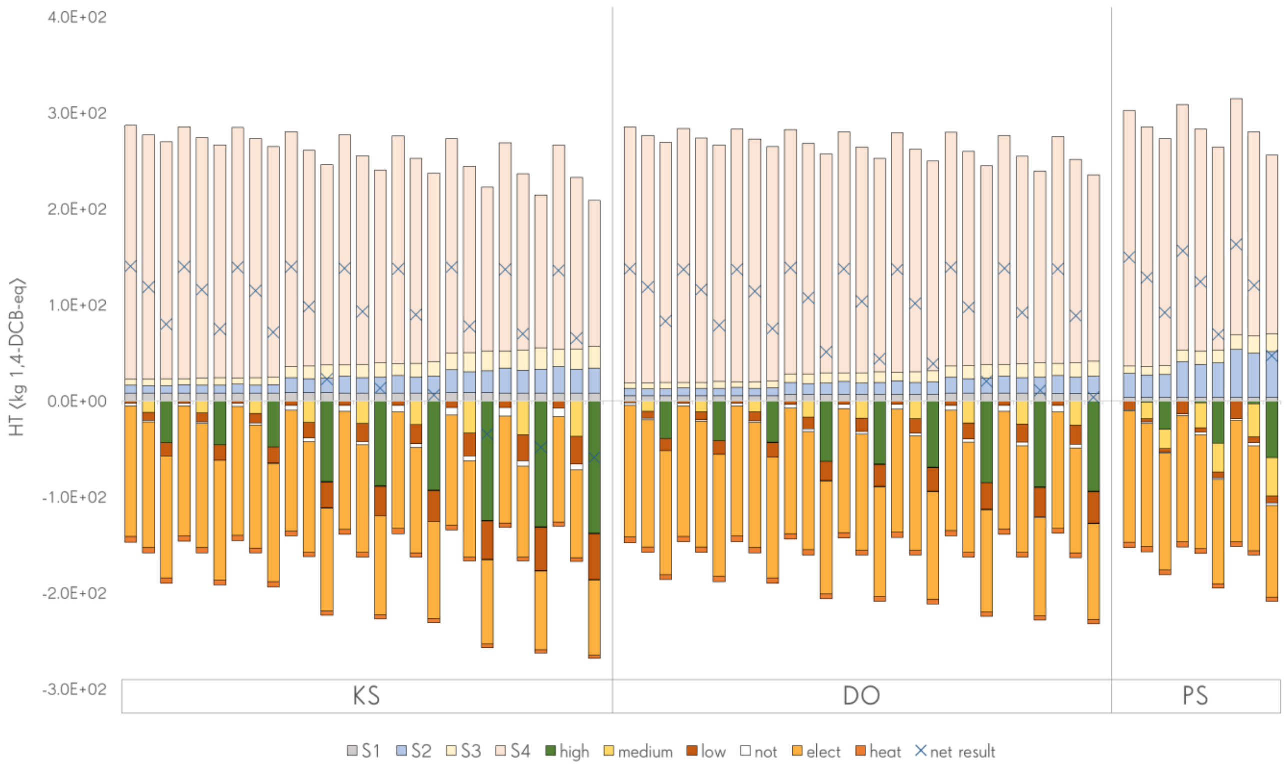

Figure XXI. Environmental performance of the modelled scenarios in terms of human toxicity

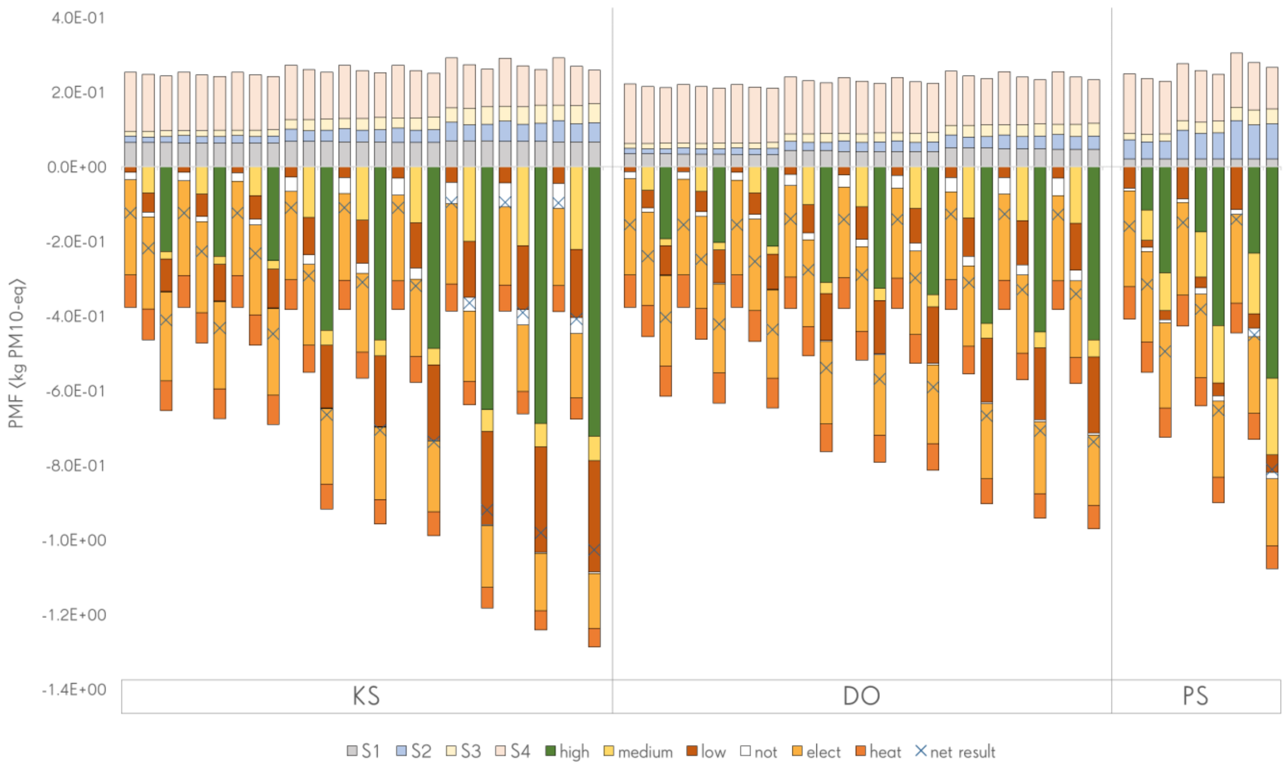

Figure XXII. Environmental performance of the modelled scenarios in terms of particular matter formation

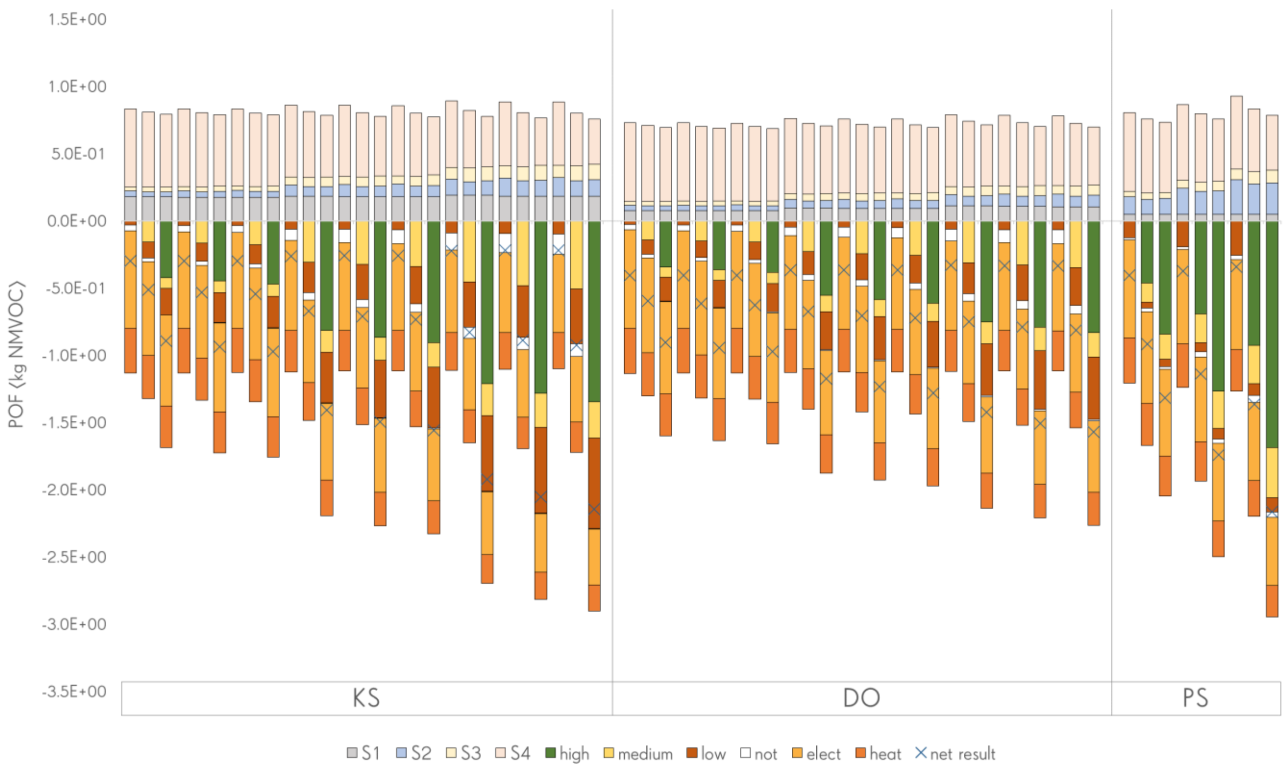

Figure XXIII. Environmental performance of the modelled scenarios in terms of photochemical oxidant formation

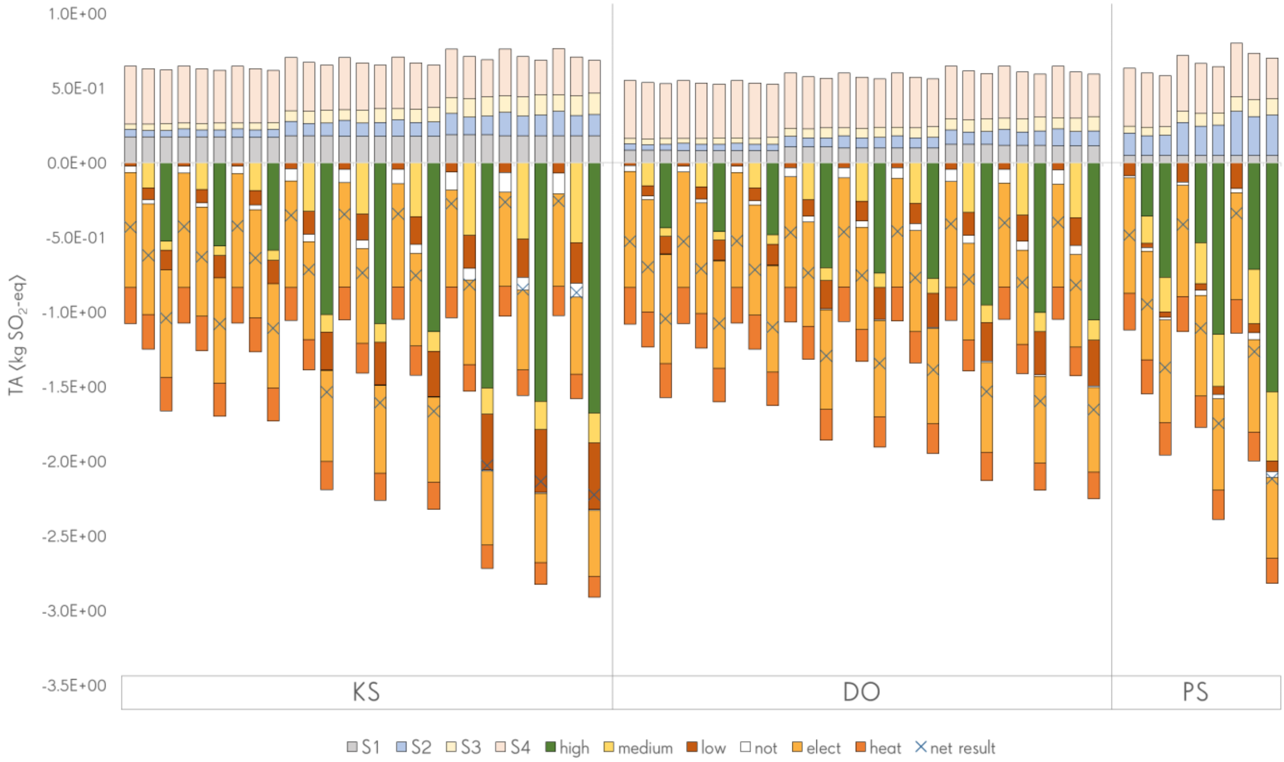

Figure XXIV. Environmental performance of the modelled scenarios in terms of terrestrial acidification