Doctoral Thesis

Management of plastic wastes in Austria: analysis of the status quo and environmental improvement potentials

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Dissertation

Bewirtschaftung von Kunststoffabfällen in Österreich: Analyse des Ist-Zustandes und Ermittlung von Optimierungspotentiale aus ökologischer Sicht

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Abstract

The production and consumption of plastics has seen an exceptional growth over the past decades. Simultaneously, concerns around the need for sound management of plastic wastes to prevent adverse effects on environmental and human health and resource use have emerged. This thesis aims to develop a thorough understanding of the societal flows and waste management of plastics to evaluate the potential of a national waste management system to mitigate the environmental problems caused by plastic wastes, using Austria as a case study.

In the first step, the general structure of the plastics economy in Austria is analysed for 2010 to identify the major processes handling plastics and to quantify the flows connecting these processes. This shows that the primary production of polymers amounted to 1100 kt/a, whereas the total plastics consumption in Austria equalled 1300 kt/a. Packaging was the most important application and was responsible for about half of the post-consumer waste generation, while the building & construction sector was responsible for by far the largest stock increase. Two thirds of plastic wastes were incinerated with energy recovery, while the remaining waste was recycled mechanically and chemically.

The second step analyses the plastic flows and its composition in the Austrian waste management system for 2013 in more detail, for the sector that produces the largest amount of plastic wastes, packaging. This waste stream amounted to 300 kt/a, half of which was composed of large and small films while one third consisted of small hollow bodies, including PET bottles. The polymer composition was consequently dominated by LDPE (46 %), PET (19 %) and PP (14 %). 34 % of the waste was sent to mechanical recycling, which achieved the current recycling target but leaves large improvements needed to reach the recently increased targets of 50 % and 55 % by 2025 and 2030, respectively.

In the third step, the waste management system for plastic packaging is investigated from an environmental performance perspective by using the material flow model developed in step two as a basis for a life cycle assessment. This shows larger benefits than burdens for the overall waste management system for 15 out of the 16 investigated impact categories, with the largest contribution to the benefits for most impact categories coming from mechanical recycling. Furthermore, the effect of changes in the recycling rate on the environmental performance is explored as well using three alternative scenarios. For ten out of the 16 impact categories, the more material is mechanically recycled, the higher the overall net benefits are. However, half of the impact categories show a decreasing marginal benefit or absolute decrease in the net

benefit, suggesting that depending on the impact category, the optimal recycling rate may lie below 100 %. Furthermore, the response of the different impact categories varies widely, so no one optimal recycling rate exists across all impact categories. The effects of increasing the recycling rate on the environmental performance of the waste management system should thus be investigated in detail to create a sound basis for proposing recycling targets leading to an environmentally optimal outcome.

Kurzfassung

Die Produktion sowie der Konsum von Kunststoffen hat in den letzten Jahrzehnten massiv zugenommen und damit einhergehend auch die anfallende Menge an Kunststoffabfällen, die nachteilige Auswirkungen auf die Umwelt und menschliche Gesundheit verursachen können wenn diese nicht ordnungsgemäß behandelt werden. Die gegenständliche Arbeit zielt darauf ab ein umfassendes Verständnis der gesamten Kunststoffflüsse am Beispiel Österreichs zu entwickeln und explizit für die Abfallwirtschaft Optimierungspotentiale zu evaluieren um Umweltbelastungen durch Kunststoffabfälle zu verringern.

Als erster Schritt wurde die Struktur der österreichischen Kunststoffwirtschaft für das Bezugsjahr 2010 analysiert, um die wichtigsten Prozesse zu identifizieren und entsprechende Kunststoffflüsse zwischen den Prozessen zu quantifizieren. Diese Analyse zeigte, dass sich die Primärproduktion von Polymeren im Jahr 2010 auf rund 1100 kt/a belief, während der gesamte Konsum von Kunststoffen in Österreich 1300 kt/a betrug. Verpackungen stellten dabei die wichtigste Anwendung von Kunststoffen dar und waren für etwa die Hälfte aller in Österreich anfallenden post-Consumer Kunststoffabfälle verantwortlich, wohingegen der Bausektor den mit Abstand größten Lagerzuwachs von Kunststoffen verzeichnete. Zwei Drittel der Kunststoffabfälle wurden einer energetischen Verwertung zugeführt, während die verbleibenden Abfälle werkstofflich oder rohstofflich verwertet wurden.

Als zweiter Schritt wurden für den Verpackungssektor, jenen Sektor welcher die meisten Kunststoffabfälle produziert, die Kunststoffflüsse und deren Zusammensetzung im österreichischen Abfallwirtschaftssystem für das Bezugsjahr 2013 im Detail analysiert. Dieser Abfallstrom belief sich auf 300 kt/a, welcher sich zur Hälfte aus großen und kleinen Folien zusammensetzte und zu einem Drittel aus kleinen Hohlkörpern (inkl. PET-Flaschen) bestand. Die Polymerzusammensetzung bestand demnach größtenteils aus LPDE mit 46 %, PET mit 19 % und PP mit 14 %. Insgesamt wurden 34 % der Kunststoffverpackungsabfälle einer stofflichen Verwertung zugeführt, wodurch das aktuelle Recyclingziel von 22.5 % zwar erreicht werden konnte, aber großer Verbesserungsbedarf besteht, um die für 2025 und 2030 erhöhten Ziele von 50 % bzw. 55 % zu erreichen.

Als dritter Schritt wurde das aktuelle Abfallwirtschaftssystem für Kunststoffverpackungen aus ökologischer Sicht untersucht. Dazu wurde basierend auf der im zweiten Schritt entwickelten Materialflussanalyse eine Ökobilanz erstellt. Die Ergebnisse der Analyse zeigten in 15 von 16 untersuchten Wirkungskategorien größere Nutzen als Belastungen durch die gegenwärtige Abfallbewirtschaftung, wobei der größte Nut-

zenbeitrag in den meisten Wirkungskategorien durch die stoffliche Verwertung erzielt wird. Darüber hinaus wurde anhand drei verschiedener Szenarien der Effekt von veränderten Recyclingquoten auf die Umweltauswirkungen untersucht. Für 10 von 16 Wirkungskategorien konnte gezeigt werden, dass der Netto-Nutzen höher wird, je mehr Material werkstofflich verwertet wird. Allerdings zeigten die Hälfte der Wirkungskategorien einen sinkenden Grenznutzen oder eine absolute Verminderung des Netto-Nutzens, wodurch sich schließen lässt, dass abhängig von der Wirkungskategorie die optimale Recyclingquote unter 100 % liegt. Aufgrund der Tatsache, dass eine Steigerung der Recyclingrate die einzelnen Wirkungskategorien unterschiedlich beeinflusst, ergeben sich je nach Wirkungskategorie unterschiedliche Optima für die Recyclingquote. Um zukünftig eine solide Grundlage für den Vorschlag von Recyclingzielen zu schaffen, sollten die Effekte von erhöhten Recyclingquoten auf Umweltauswirkungen des Abfallwirtschaftssystems genauer untersucht werden.

Contents

List of Figures

List of Tables

Abbreviations

ABS acrylonitrile butadiene styrene

BCW bulky and commercial waste
BRF brominated flame retardant

CR collection rate

EC European Commission
EPS expanded polystyrene
ET ecotoxicity
EU European Union

FE freshwater eutrophication

GW global warming

HDPE high-density polyethylene
HTc human toxicity, cancer effects
HTnc human toxicity, non-cancer effects

IC impact category
IR ionising radiation

LCA life cycle assessment
LCI life cycle inventory
LCIA life cycle impact assessment
LDPE low-density polyethylene
LLDPE linear low-density polyethylene
LU land use

ME marine eutrophication
MFA material flow analysis
MP mixed-polymer
MSW municipal solid waste

OD ozone depletion

PE person equivalent
PET polyethylene terephthalate
PM particulate matter
PO polyolefin
POF photochemical ozone formation
PP polypropylene
PS polystyrene
PVC polyvinyl chloride

RDf resource depletion, fossil
RDm resource depletion, minerals and metals
RDw resource depletion, water
RR recycling rate
RW residual waste

SCW separately collected waste
SFA substance flow analysis
SP single-polymer
SR sorting rate

TA terrestrial acidification
TC transfer coefficient
TE terrestrial eutrophication

UK United Kingdom
USA United States of America

VCS value-corrected substitution

WMS waste management system
WtE waste-to-energy

Published articles and contributions

This thesis is based on the three journal articles listed below, which can be found in the Appendices of this work.

Paper I **Van Eygen, E.**, Feketitsch, J., Laner, D., Rechberger, H., Fellner, J. Comprehensive analysis and quantification of national plastic flows: The case of Austria. *Resources, Conservation and Recycling*, 117:183-194, **2017**, doi: 10.1016/j.resconrec.2016.10.017.

I have contributed to the data collection, model calculation, interpretation of the results and have written the paper.

Paper II **Van Eygen, E.**, Laner, D., Fellner, J. Circular economy of plastic packaging: Current practice and perspectives in Austria. *Waste Management*, 72:55-64, **2018**, doi: 10.1016/j.wasman.2017.11.040.

I have contributed to the design, data collection, model calculation, interpretation of the results and have written the paper.

Paper III **Van Eygen, E.**, Laner, D., Fellner, J. Integrating high-resolution material flow data into the environmental assessment of waste management system scenarios – the case of plastic packaging in Austria. *Revision submitted to Environmental Science & Technology.*

I have contributed to the design, data collection, model calculation, interpretation of the results and have written the paper.

Chapter 1

Introduction

1.1 The rise of plastics consumption

The production and consumption of plastics has seen an extraordinary increase, ever since the first truly synthetic polymer was developed in 1907 and the beginning of mass production in the 1940s and 1950s.[1] They have become one of the most used man-made materials globally, and have seen a stronger growth than any other material over the past 60 years (see Figure 1.1), with an average annual growth rate of 5 % since the 1980s. A global production of 335 million tonnes was reached in 2016, of which about 50 % is produced in Asia, 19 % in Europe and 18 % in North-America.[2] This strong increase in production and consumption is due to many key properties this versatile class of materials possesses: they are inexpensive, lightweight, strong, moldable, biologically and chemically inert, and very durable.[1;3]

Plastics can be classified into thermoplasts and thermosets. The latter are set, cured or hardened into a permanent shape, whereas the former can be repeatedly softened by heat and reshaped, thus enabling mechanical recycling.[4;5] Another classification, considering cost, production volume and performance, can be made between commodity, engineering and high-performance plastics. Commodity plastics, which among others include polyolefins (POs) (e.g. polyethylene and polypropylene (PP)), polystyrene (PS) and polyvinyl chloride (PVC), are produced in high volumes, are cheap and therefore find widespread use in a variety of applications. Engineering plastics (e.g. polyethylene terephthalate (PET) and acrylonitrile butadiene styrene (ABS)) have improved mechanical properties and are therefore used in more specialised applications, whereas high-performance plastics meet even higher requirements and are usually specialised for a single specific property.[4;6]

Many of the properties of the polymers can be modified and enhanced by the inclusion of a multitude of additives. Some additives influence the mechanical properties, such as plasticisers, fillers and blowing agents, whereas others, such as antioxidants and flame retardants, change the chemical performance. Furthermore, colourants

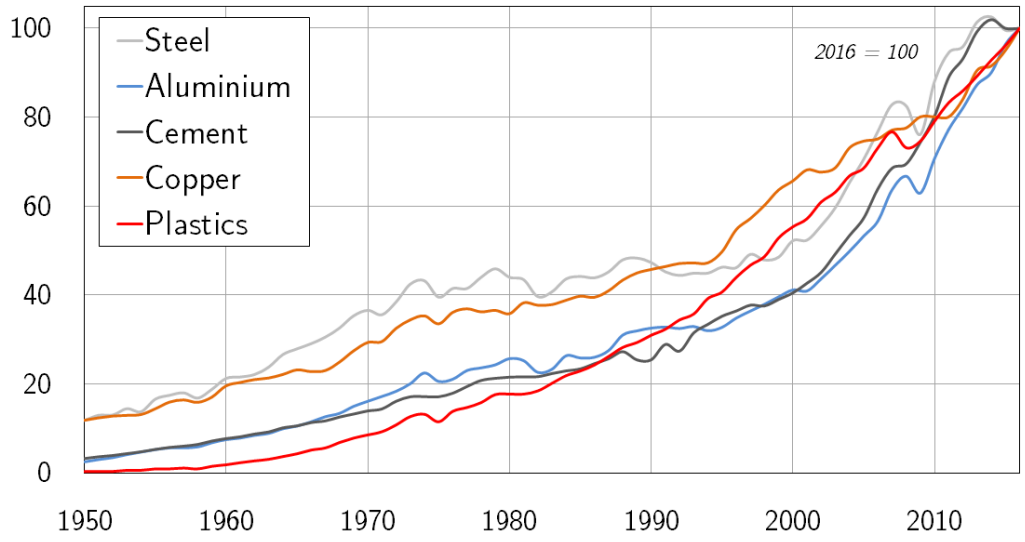

Figure 1.1: Primary production of some major man-made materials, normalised to 2016. Data from PlasticsEurope[2]; World Steel[7]; International Aluminium Institute[8]; USGS[9].

alter the visual appearance and antistatics improve the surface properties.[6;10] The mass fraction of additives in plastics range from 0 % in some polyethylene-films for food packaging to as high as 90 % for some magnetic adhesives,[6] with an average estimated to be around 7[11] to 10 %[6]. With respect to the polymers, according to Murphy[10] about 73 % of additives by volume are used in PVC compounds, 10 % in POs, 5 % in styrenics and the remaining 12 % in other polymers.

This versatility due to the combinations of many different polymers with various additives, leading to over 80 000 different compounds or formulations,[3] means that plastics form a highly complex resource leading to very diverse material flows into society. They have thus become ubiquitous and find applications in virtually every part of human society, with a demand primarily concentrated in the packaging, building & construction and automotive sectors,[2] as shown in Figure 1.2 for Europe. Plastics are thus applied in products with both very short and very long lifetimes, leading to differing challenges for end-of-life management.

1.2 Environmental aspects

The application of plastics leads to many environmental benefits. Energy savings are achieved in the transportation sector thanks to the low weight and in the building sector due to improved insulation.[3;12] The energy consumption during production is much lower compared to metals, on a volume basis,[13] which, in combination with the high strength-to-weight ratio, allows for lower energy and material usage for a

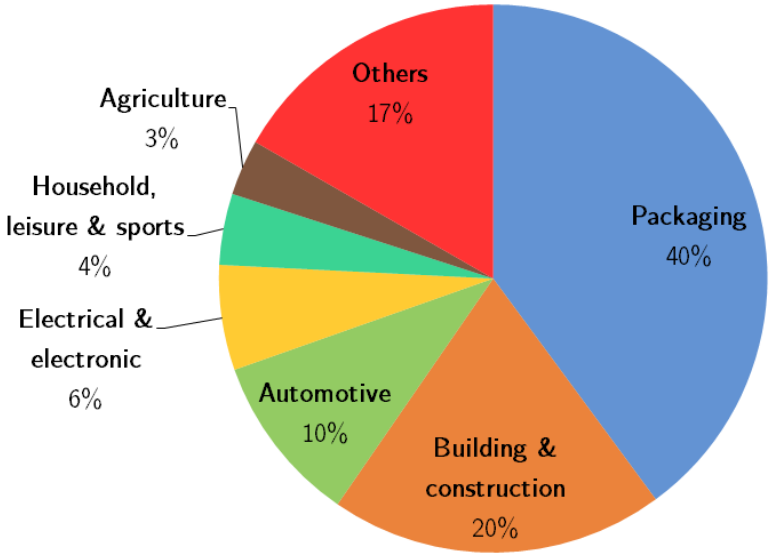

Figure 1.2: Main market sectors regarding plastics converter demand for Europe in 2016. Data from PlasticsEurope[2].

given performance. Many applications where plastics are indispensable, such as improved food packaging reducing food waste or renewable energy solutions, contribute to a reduction of environmental impacts as well.[3;12]

However, the widespread use and specific properties are causes for a number of environmental concerns as well. From a resource use perspective, plastics are synthesised almost exclusively from fossil feedstocks, consuming about 4-5 % of annual petroleum production for conversion into plastics and an additional 3-4 % as energy during the production process.[3;14] The share of bio-based polymer production capacity currently only makes up about 2 % of global plastics production, just growing with a rate similar to the overall polymer capacity growth rate.[15]

After use, plastic wastes have become an increasingly problematic waste flow, especially when discarded improperly, as one of the main properties that makes plastics so useful, its durability, also means that they do not degrade at a meaningful rate and therefore accumulate in the natural environment.[3] This has gone so far as to suggest plastics as a geological indicator for the Anthropocene, a proposed geological epoch of time characterised by human domination over many geological processes.[16] These plastics in the environment generally arise in two forms, namely macroplastics and microplastics, although there is no universally adopted size-boundary between the two, with ranges from 1 to 10 mm between authors. Primary microplastics are manufactured purposefully, whereas secondary microplastics are the result of the

breakdown of larger particles due to exposure to sunlight, abrasion and microbial activity.[3;17]

On land, plastics litter is widely distributed but its mobility is lower than in aquatic environments, thus causing correlations with areas of high use and disposal such as agriculture, landfill sites, roads, and decommissioned pipe and cable networks.[16] However, the quantities, fate and effects of terrestrial plastic debris have not been studied widely.[18–20] The main focus has been on the transportation of terrestrial debris to marine environments through rivers and coastlines, as it has been recognised that the vast majority of marine debris originates from land.[16;21;22] Plastics thus mainly accumulate in marine habitats, where they cause a multitude of environmental problems. Macroplastics cause physical hazards for organisms due to e.g. entanglement, ingestion and smothering, as well as promote hitch-hiking of invasive species.[23] Microplastics are even more prone to ingestion, and pose additional risks due to leaching of monomers, oligomers and additives from the plastic itself or due to extraneous pollutants adsorbed to the particles, causing potential biological consequences and accumulation through the food chain.[17;24]

The proper collection and management of plastic wastes is thus crucial to prevent improper disposal and leakage to the environment, both from a resource and from an environmental and human health protection perspective.

1.3 Current practice in waste plastics management

The recycling of plastics can be generally grouped into four categories.[14;25] Primary and secondary recycling both refer to mechanical recycling, with the former producing re-granulate for products with equivalent properties (i.e. closed-loop recycling) and the latter for products with lower requirements (i.e. open-loop recycling). Tertiary recycling, also known as chemical or feedstock recycling, recovers the petrochemical constituents, be it the monomers, other basic chemicals or fuels, or acts as a reducing agent. Quaternary recycling only recovers the energy content of the plastics through incineration, although this process is strictly speaking not to be seen as recycling, as the material is fully destroyed and lost. In general, these four categories progressively require a decreasing input purity, but result in lower material recovery yields.[14;25;26] The final option is disposal, either in a sanitary landfill where due to the inert nature of plastics the material will generally be contained, or in open landfills or dumps, where the wastes are more prone to eventually end up in the terrestrial or marine environment.

Currently, mechanical recycling is the only widely adopted recycling technology (when excluding energy recovery from the term “recycling”), as chemical recycling is not commonly implemented on an industrial scale due to economic considerations.[26]

Geyer et al.[11] estimated that globally in 2014 18 % of nonfiber waste plastics were recycled and 24 % incinerated, whereas the rest was discarded into landfills or the natural environment. For Europe in 2016, the recycling (31 %) and incineration rates (42 %) were higher, but large differences exist between member states, with some reporting landfill rates around 80 %.[2] For packaging specifically, it was estimated that globally in 2013 32 % were leaked to the environment, 40 % landfilled, 14 % incinerated, and just 8 % used for open-loop and 2 % for closed-loop recycling.[27] Large geographical differences thus exist with respect to plastic waste management practice, but it should also be pointed out that these estimates are quite uncertain due to a general lack of data.

1.4 European waste (plastics) policy

To tackle the environmental concerns regarding plastic wastes mentioned in Section 1.2, several policies were enacted by the European Union (EU). For waste management in general, the waste framework directive[28;29] lays down the general principles and outlines the waste hierarchy, whereby the following priority order applies in waste prevention and management legislation:

- prevention
- preparing for re-use
- recycling
- other recovery, e.g. energy recovery
- disposal.

However, this generalised hierarchy is not to be used a priori, as it is mentioned that the options delivering the best overall environmental outcome should be encouraged, which may require departing from the waste hierarchy for specific waste streams if this is justified by life cycle assessment (LCA) studies.

To bring this hierarchy into practice, recyclable materials are banned from being landfilled by 2030 and the overall amount of municipal solid waste (MSW) sent to landfills is to be reduced to 10 % or less.[30] Some member states, including Austria, have gone further already and have banned the landfilling of waste with an organic carbon content over 5 % altogether.[31;32] Furthermore, minimum targets for preparation for re-use and recycling are specified for MSW amounting to 50 % by weight by 2020,[28] which increases every five years up to 65 % by 2035.[29] More specifically for individual waste streams, separate collection and recovery targets were adopted for end-of-life vehicles and electronic waste, although these targets apply to the products as a whole and not to specific materials.[33;34] For packaging waste however, recycling targets were defined for the waste stream as a whole as well as for five constituting materials.[35;36] For plastics, this target amounted to 22.5 % by 2008,[35], which was recently increased substantially to 50 % by 2025 and 55 % by 2030, counting the mass entering the final recycling facility.[36] This increase was

part of a broader initiative by the European Commission (EC) to improve resource efficiency and reduce resource dependency[37] through the adoption of an action plan towards a more circular economy.[38] In this action plan, plastics are one of the five priority areas, and the EC thus communicated a strategy for plastics in a circular economy[39]. Here, measures are proposed for achieving a circular plastics economy and all players in the plastic value chain are encouraged to take action as well.

Chapter 2

Objectives and thesis structure

From the issues described in Chapter 1, it is clear that the waste management sector has a large role to play in mitigating the environmental problems caused by plastic wastes. To evaluate the potential of a waste management system (WMS) in doing so, this thesis aims to evaluate a country-level WMS, using Austria as a case study.

This goal is tackled in three main steps. First, the general structure of the plastics economy in Austria is analysed. This is done by establishing the Austrian plastics budget for the full life cycle of plastics, starting from primary production over manufacturing, use and waste management. The major processes handling plastics are thus identified and all flows connecting these processes are quantified, thus enabling a thorough understanding of production, demand and waste generation of plastic materials in Austria. This subsequently serves as a basis to identify opportunities for increasing overall plastics resource efficiency.

Second, the focus is shifted to the waste management stage, where the flows of one particular waste stream (packaging) are analysed in more detail, now including data on its composition regarding polymers and product types. This waste stream was chosen as it makes up the largest share of the post-consumer wastes of all sectors and is often regarded as the most problematic due to its short lifespan, large littering potential and thus high visibility to the public. It therefore receives the most attention from policy makers, usually through imposing recycling targets. Therefore, to accurately assess the performance and improvement potential of a WMS regarding these targets, detailed knowledge on the waste flows is needed.

Third, the assessment is moved beyond the investigation of mass flows to include the environmental performance as well. To this end, the material flow model developed in step two is used as a basis for an environmental assessment analysing the status quo of plastic packaging waste management in Austria. Subsequently, the effect of changes in recycling rates, as demanded by the policy target mentioned above, is explored by assessing plastic packaging waste management in a landfill disposal-

dominated WMS and in a situation where the imposed increased recycling target is met. The environmental performance of increasing recycling targets as a policy instrument is thus investigated.

These three steps can be translated into the following research questions:

(i) What are the most important processes and flows in the Austrian plastics budget?

(ii) What is the resource potential of plastic wastes on a national level?

(iii) What is the composition of waste plastic packaging entering the WMS regarding polymers and product types, and how are these polymers and product types routed through the system?

(iv) How does the status quo of the WMS compare against the current recycling target, and where are the improvement potentials to achieve the future target?

(v) What is the environmental performance of the Austrian WMS for plastic packaging and what are the most important parameters determining this result?

(vi) How does the environmental performance of the WMS change with a varying recycling rate, and what are the implications for recycling targets as an environmental policy instrument?

This thesis builds on three papers which are added in the Appendices, corresponding to the three steps described above. Paper I deals with questions (i) and (ii), whereas Paper II looks at questions (iii) and (iv) and Paper III at questions (v) and (vi).

The further structure of this thesis is built up as follows. In Chapter 3, the main methods are briefly described and an overview of the existing literature on the application of these methods to plastics (waste management) is given. Next, Chapter 4 describes the modelling approaches, assumptions and input data for the material and environmental models used in this thesis. Finally, Chapter 5 summarises the results of the three papers while Chapter 6 discusses these results in the context of the research questions introduced above and presents an outlook for further research.

Chapter 3

Scientific background

3.1 Methods

3.1.1 Material flow analysis

Material flow analysis (MFA) is defined as "a systematic assessment of the state and changes of flows and stocks of material within a system defined in space and time".[40] An MFA thus provides comprehensive information on the sources, pathways and sinks of the material under consideration, using the law of conservation of matter to compare all inputs, stocks and outputs of a process or system. This can be done on the level of goods, looking at economic entities of matter, or on the level of substances, looking at specific chemical elements or compounds which constitute goods (also known as substance flow analysis (SFA)).[40;41]

Static MFAs present a snapshot of the material flow system, usually looking at one specific year. Time-series can also be built by carrying out multiple static MFAs to keep track of stocks and flows through time, highlight trends and evaluate the effect of policy changes (also known as material flow accounting). Finally, dynamic MFA modelling introduces time as a modelling parameter, which makes it possible to predict the future behaviour of a system.[40;41]

3.1.2 Life cycle assessment

LCA is defined as the "compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle".[42] LCAs thus quantify all relevant emissions and consumed resources along the full life cycle of a product or service, from raw materials acquisition through production, use and waste management, and assesses the resulting impacts on human and environmental health as well as resource depletion.[42;43]

The general framework of LCA consists of four phases. First, the goal and scope are defined, which includes setting the system boundary and level of detail. Here,

the functional unit is also defined, which quantitatively and qualitatively specifies the function of the product or system and provides a normalisation reference for the inventory data. Second, in the life cycle inventory (LCI) the system under study is modelled and data is collected on the inputs and outputs of each process. Third, this information is used as the input to the life cycle impact assessment (LCIA), where all elementary flows are classified and characterised into impact categories (ICs) with a common unit (e.g. kg CO_2-equivalents for global warming), which results in several indicators related to human and environmental health as well as resource depletion. The fourth and final stage then consists of interpreting the results and deriving conclusions. As an LCA study is carried out iteratively, this interpretation may point to improvement potentials of aspects of the previous steps, such as a need to improve the LCI model and collect better data on certain processes which have a large influence on the results.[42;43]

Both methods, MFA and LCA, can be integrated, as MFA can be regarded as a method to establish (parts of) the inventory for an LCA in a transparent and consistent way, especially when assessing systems rather than a single product.[40] In the case of the assessment of a WMS for example, an MFA can provide information on the mass and composition of the waste flows and transfer coefficients (TCs) in the various treatment processes.

3.1.3 Uncertainty in MFA and LCA

Both MFAs and LCAs, being models which are simplified versions of the real-world system, are inherently uncertain. These uncertainties stem from various sources, such as statistical variation, subjective judgement, linguistic imprecision, variability in space and time, inherent randomness, expert disagreement, and model approximations.[44;45]

In general, data uncertainties are grouped into aleatory (or stochastic) uncertainty, which results from inherent randomness and cannot be reduced, and epistemic uncertainty, which arises because of incomplete knowledge and can be minimised through further investigation. However, both types are usually not distinguished from each other and are treated together in MFA and LCA modelling.[46;47] Especially in the context of LCA, uncertainties can be subdivided into three categories according to the source. First, parameter uncertainties are introduced by input data being inherently variable, difficult to measure, or unavailable altogether. Second, scenario uncertainties are related to normative choices in constructing scenarios and variability in their characteristics. Third, model uncertainties arise from the structure and mathematical relationships within the model.[45;48;49]

For the assessment of the parameter uncertainty in LCA, the characterisation of the uncertainty of input data is a common difficulty, as sample data for a statistical

analysis are usually not available.[50] In this case, the pedigree matrix approach can be used,[51] as implemented the LCI database ecoivent,[52] where basic uncertainties express the varying nature of the uncertainty of different types of data (e.g. data on CO_2 emissions are generally less uncertain than data on emissions of heavy metals). Furthermore, an additional uncertainty is added to express the quality of the data regarding the actual quantity of interest, which is estimated using data quality indicators evaluating the data source based on its reliability, completeness and temporal, geographical and further technological correlations.[52] By default, this procedure then results in a standard deviation of a lognormal distribution, although this can be converted to other distributions as well.[53] Once uncertain input data are available, through statistical analyses or the pedigree matrix approach, these can be propagated analytically or using sampling-based methods such as Monte Carlo simulation.[54] To account for scenario and model uncertainties, a scenario analysis can be carried out, where different choices or model formulations are tested individually and the effects of these changes are evaluated.[45;47]

Uncertainty analyses in MFA studies are done more rarely, and usually deal with parameter uncertainties only,[46] although some studies have explored the influence of model structure and scenario assumptions on the results (see e.g. refs[55–57]). Regarding the parameter uncertainty, especially for regional MFAs, statistical data are usually not available. Rather, isolated values may be present or no value at all may be available, and one has to resort to estimations based on up- or downscaling from related systems or expert's judgement. For these types of input data, statistical methods for the evaluation of the uncertainty are not applicable, instead the uncertainty can be characterised using data quality assessments, which is done analogously as in LCA as described above.[58;59] Specifically, Laner et al.[58] modified the pedigree matrix approach used in LCA to characterise the uncertainty of input data for MFA. The quality of the input data is again assessed semi-quantitatively, after which the resulting scores are used to quantitatively estimate the uncertainty values for each input data point as coefficients of variation of a normal distribution.[58] These input uncertainties can subsequently be propagated, for example in the software STAN, where data reconciliation is used to solve inconsistent datasets.[60]

3.2 Literature overview

3.2.1 MFAs of plastics and waste plastics management

The regional (anthropogenic) flows of many elements have received quite a lot of attention over the recent years for developing knowledge on resource efficiency aspects such as extraction rates, types of uses, losses and recycling rates. However, this knowledge was mainly built up on an elemental level for metals, for which hundreds of

studies were published, and non-elemental compounds such as plastics have not been investigated to the same extent.[61] Public data from governments or industry on the chemical sector in general is much less available compared to other industries such as steel or aluminium,[62] and therefore only limited research on the societal flows of individual polymers or plastics in general has been performed. The main publications describing the flows and/or stocks of plastics on a national or supranational level are shortly described in this section.

Two studies have investigated the Austrian plastics budget. Fehringer and Brunner[63] determined the flows and stocks of total plastics as well as specifically for polyethylene and PVC for 1994, and compared these flows with five scenarios regarding the waste management phase. Bogucka and Brunner[64] built on and updated this budget for 2004, and compared these results for Austria with the situation in Poland. Other publications on static plastic MFAs include Kawecki et al.[65], who looked at the flows of 7 polymers in Europe and Switzerland in 2014, and Tukker et al.[66], who focused on the national flows of a single polymer, PVC, in Sweden for 1994. Besides these studies using more classical MFA methods, three studies used methods related to input-output analysis for the quantification of national plastic flows: Duchin and Lange[67] for the United States of America (USA) in 1987, Joosten et al.[68] for the Netherlands in 1990, and Nakamura et al.[69] for PVC in Japan in 2000.

Other than these static MFAs, several studies have used time-series and dynamic MFA to look at the evolution of plastic flows in the past and into the future. For the total plastic flows, Geyer et al.[11] estimated the production, use and fate on the global level of all plastics ever produced up to 2015. These total plastic flows were investigated for Germany for 1976 to 1995[70] and India from 1960 to 2000[71], which were then used to forecast future plastic flows to 2025 and 2030 respectively. Finally, for single polymers, Kuczenski and Geyer[72] investigated PET in the USA from 1996 to 2007, Kleijn et al.[73] used the situation of PVC in Sweden[66] to examine the delaying mechanisms of stocks with respect to waste production, Ciacci et al.[74] looked at the PVC flows in the EU from 1960 to 2012, while Zhou et al.[75] estimated PVC waste generation in China using primary production statistics and lifetime distributions.

Although most studies mentioned above do include the waste management stage, this is usually not done in great detail. MFA studies focusing specifically on a plastics WMS are not numerous, including Haupt et al.[76] who quantified the flows of PET bottle wastes in Switzerland as part of an MFA of the full Swiss WMS. Other MFA studies for plastic wastes have been carried out in connection with a subsequent LCA study, and are discussed in the next section. Finally, two studies were published concurrently with Paper II. Brouwer et al.[77] investigated the Dutch plastic packag-

ing WMS for household waste on the level of product types and polymers, whereas Dahlbo et al.[78] estimated the recycling potential of household plastic wastes in Finland.

The studies that were published before Paper I and Paper II and quantify the national or supranational flows of all plastics thus present results that are 10 to 25 years old, which is especially relevant for such a rapidly growing sector. Furthermore, most studies do not determine the plastic flows for all life cycle stages, and do not distinguish between different consumption sectors. The waste flows, which are usually the most uncertain ones due to a general lack of data,[59] are estimated in most studies by using lifetime distributions of the consumed goods, and none of the studies used bottom-up data. These waste flows were moreover not investigated on a quantitative and qualitative basis. These research gaps are addressed by Paper I and Paper II.

3.2.2 LCAs of waste plastics management

The EU waste framework directive[29], as mentioned in Section 1.4, allows deviating from the waste hierarchy if justified by a superior environmental performance determined by an LCA study. LCAs have thus proven to be a valuable decision-support tool for the environmental assessment of WMSs, as they expand the scope beyond the system itself by including the impacts and benefits of the surrounding systems, and thus provide a holistic perspective.[79;80] A comprehensive review on LCA studies of WMSs has been provided by Laurent et al.[50;80], whereas Lazarevic et al.[81] discuss the results of ten selected studies on plastic wastes. Both these reviews conclude that, although a general tendency favouring recycling over waste-to-energy (WtE) and landfill disposal is clear, no definite conclusion on which technology performs better in general can be made due to uncertainties regarding the study assumptions and local conditions.

A number of LCA studies on the waste management of plastic (packaging) have thus been carried out. However, many of these focus on the plant or technology level (see e.g. refs[82–93]), often looking at only one polymer (see e.g. refs[94–96]), where generally two or more treatment options treating a certain waste amount (usually one tonne) are assessed side by side and their environmental performances are compared. These studies provide valuable understanding on the environmental impacts and benefits of these single processes, but do not allow assessing their performance when integrated into a WMS. To add this angle, other studies have taken a more systemic approach, looking at the full WMS of plastics specifically (see e.g. refs[97–100]), or as a part of packaging waste (see e.g. refs[101;102]) or MSW as a whole (see e.g. refs[103–108]). Detailed information on the actual waste flows is usually lacking though, especially in the studies assessing the complete MSW stream. Region-specific knowledge on the structure of the WMS and the amount and composition of the

plastic waste, in terms of polymers as well as elements, is thus usually not considered or reported in detail, although this has a considerable influence on the result.[50;109]

This limitation can be addressed by the integration of MFA and LCA. Detailed information regarding the availability, composition and flows of the waste through the WMS can thus be connected to input-dependent data on the environmental performance of each of the treatment processes, which allows for evaluating all the treatment options simultaneously to assess the system as a whole and to identify trade-offs (e.g. between material and energetic recovery) within the WMS. Two studies have taken this approach for the assessment of a regional WMS for MSW. Turner et al.[110] assessed the climate performance of the existing management of waste collected in Cardiff, Wales, and compared this with three alternative scenarios. With respect to the plastic wastes, the composition was determined by 11 fractions comprising six polymers, characterised by three elemental composition datasets, and the mechanical recycling of waste plastics was modelled using three distinct processes. Furthermore, Haupt et al.[111] investigated the WMS in Switzerland, compared the current situation with five alternative scenarios, and regarding the plastic part of the waste stream, took three polymers into account for which the recycling is modelled using three datasets in two versions each (one domestic and one European).

Owing to the fact that plastics are regarded by the EC as one of the key priority areas with specific challenges due to its properties (see Section 1.4),[38] a holistic environmental assessment separate from the overall MSW stream may be warranted or even required, enabling an even more detailed assessment of the plastics packaging WMS and its waste composition throughout the system. Furthermore, the previously mentioned studies mainly constructed various scenarios comparing a range of different policy options (e.g. decreased landfilling, increased incineration or increased recycling). The relationship between the environmental performance and recycling rate has not been explicitly explored though, despite the fact that waste management policies are often based on this metric as a target, as is the case for plastic packaging (see Section 1.4). It should thus be investigated if such policies improve the environmental performance of the WMS across a wide range of ICs, or if potential burden shifting is introduced by increasing the recycling rate. Finally, none of the mentioned studies included a comprehensive uncertainty analysis to assess the robustness of the results and identify the most important parameters. These research gaps are considered in Paper III.

Finally, for the environmental modelling of WMSs, one of the main methodological aspects, especially for plastic wastes, is the substitution factor.[112] This factor governs the amount of primary resources (i.e. virgin plastics) which is assumed to be substituted by the secondary resource (i.e. the re-granulate) produced by the recycling processes. Many studies use a substitution factor of one, often without

justification.[90;92;95;97;102;108] However, Schmidt and Strömberg[113] suggest that a quality loss of 10-20 % should be assumed, a supposition which has been adopted by other authors.[82;83;88;91;106] A further approach to estimate the quality loss and market uptake of recycled materials is value-corrected substitution (VCS).[112;114–116] This method uses the ratio of the market prices of the secondary to primary material, and has been used in the assessment of plastics recycling systems before.[107;117;118]

Chapter 4

Material and environmental modelling

4.1 National Austrian plastic flows

A static MFA model was set up to investigate the flows of plastics in Austria for the reference year 2010, using the software STAN.[60] All plastic types were taken into account, including rubbers, as well as the additives incorporated into the plastic material. The sum of these materials is further on referred to as plastics. An overview of the system boundaries for the modelling of the Austrian plastics budget is provided in Figure 4.1. This model is subdivided in three main stages: production, consumption and waste management.

In the production stage, the primary polymers are produced (in the case of Austria, these include polyolefins, polystyrenes and resins) and moulded into semi-finished and final products. Imports and exports of primary polymers and semi-finished products are taken into account as well. The final products are then traded in the consumption stage, after which they are used in 11 consumption sectors. These consumption sectors, with examples for each of the product types, are presented in Table 4.1. After use, the products arise as waste flows from the respective consumption sectors or are partially exported as used goods (for the transport and electronics sectors), while the outflow of the non-plastics sector is modelled as an export flow, as it is not relevant for the waste management sector due to the dissipative nature of its applications. In the waste management stage, six treatment categories are used: reuse, mechanical recycling, chemical recycling (as a reduction agent in the steel industry), WtE, industrial incineration in the cement industry, and landfill.

For details on the input data and its sources, the reader is referred to the Main article and the Supplementary data of Paper I. As these data sources are very diverse and have varying levels of quality, it was deemed necessary to characterise the quality of

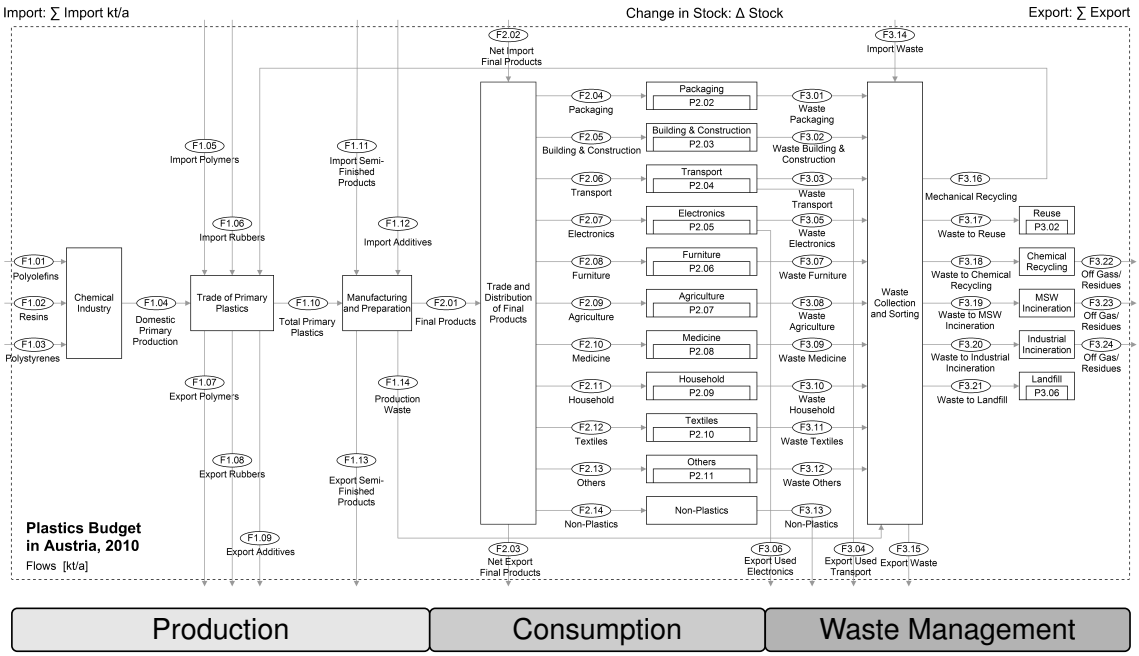

Figure 4.1: Model overview and system boundaries of the Austrian plastics budget. From Figure 1 of Paper I.

these data sources in describing the desired quantitative information, acknowledge data gaps and validate the results. This was done through performing a uniform and transparent quantitative uncertainty description with the approach proposed by Laner et al.[58] and introduced in Section 3.1.3. It has to be noted though that this approach introduces two major aspects of subjectivity. First, the indicator scores are assigned on more or less stringent criteria, and second, the quantitative uncertainty values for the indicator scores are estimated. With respect to the first aspect, although most evaluation criteria do not leave much room for interpretation, others are not that unambiguous, and therefore rely on the experience and tacit knowledge of the modeller. For the second aspect, although the underlying mathematical functions allow the transparent and consistent quantitative uncertainty characterisation, the actual definition of these functions remains up to the modeller, and no empirical data are usually available for this choice, so uncertainty estimates may differ from one MFA study to another. Therefore, although the approach builds on reproducible and internally consistent uncertainty estimates, comparisons of these estimates should be done cautiously.[55;58]

The pedigree matrix with the data quality indicators and evaluation criteria, as well as the corresponding quantitative uncertainties, are displayed in Tables 2 and 3 in the Main article of Paper I, whereas the resulting uncertainties for each of the data points are presented in the Supplementary data of Paper I.

Table 4.1: Eleven consumption sectors considered in the Austrian plastics budget and product examples. From Table 1 of Paper I.

Consumption sector	Product examples
Packaging	Foils, bottles, carrier bags
Building & construction	Pipes, floor coverings, window profiles
Transport	Cars, trucks, bicycles
Electronics	Household appliances, cables, batteries
Furniture	Chairs, mattresses, tables
Agriculture	Silage foils, seed trays, mulch films
Medicine	Syringes, infusion bags, disposable gloves
Household	Kitchen utilities, cutlery, storage containers
Textiles	Clothing, furniture covers, yarns
Others	Toys, sporting equipment, carpets
Non-plastics	Lacquers, adhesives, colouring pigments

4.2 Plastic packaging flows in the Austrian waste management system

Similarly as for the national plastic flows, a static MFA was calculated in STAN[60] to describe the waste flows of plastic packaging in the Austrian WMS, now with 2013 as the reference year. An overview of the model including its system boundaries is given in Figure 4.2, which displays a comprehensive overview of the possible flows, although not all waste flows and treatment options were in fact present in 2013. These system boundaries are drawn to include all plastic packaging products from becoming waste in Austria until the final treatment step is concluded. In contrast to the full plastics budget, where all polymers were summed together, here each flow was quantified regarding seven product types: PET bottles, small (<5 L) and large (≥ 5 L) hollow bodies, small ($<1.5\,m^2$) and large ($\geq 1.5\,m^2$) films, large expanded polystyrene (EPS) (≥ 0.1 kg), and other products. Products made out of material composites, such as food and drink cartons, were not considered. Moreover, each of the waste flows was quantified on a polymer level as well, including eight polymers: low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), high-density polyethylene (HDPE), PP, PS, EPS, PET, and PVC. These account for 99 % of all plastics used in packaging in Europe.[2]

Plastic packaging waste is either collected in the separately collected waste (SCW), residual waste (RW) or bulky and commercial waste (BCW). The SCW is subsequently sorted based on polymer, product type and/or colour, and sent to mechanical recycling, either food-grade (PET bottles only), single-polymer or mixed-polymer recycling. The sorting residues are mainly sent to industrial incineration in the ce-

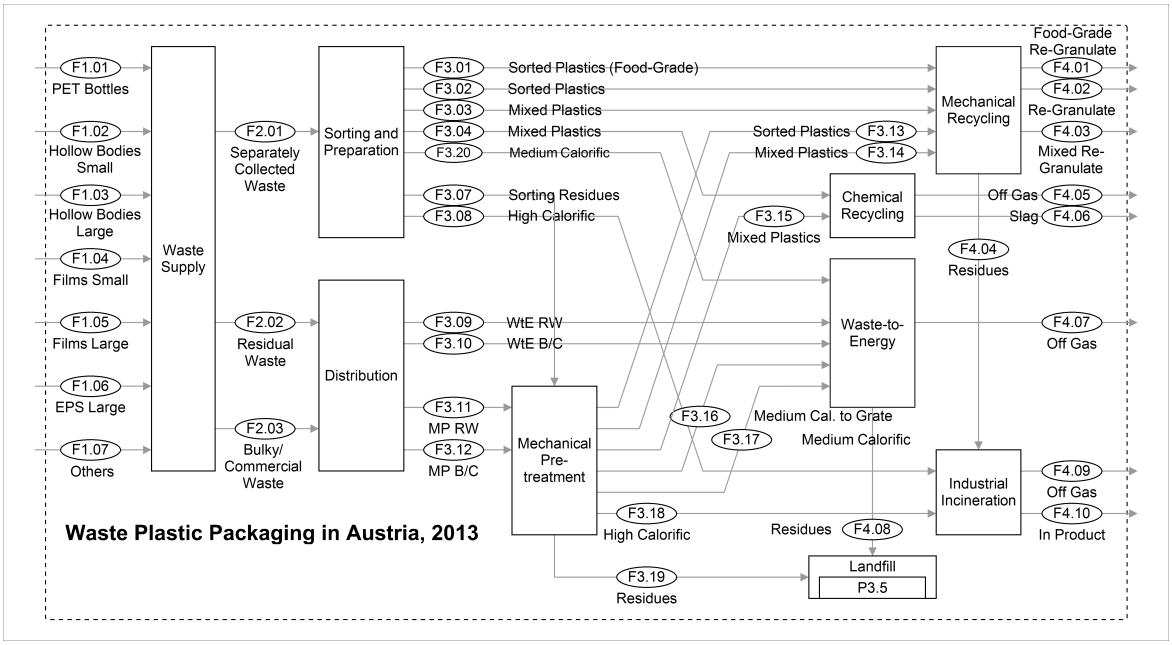

Figure 4.2: Model overview and system boundaries for the Austrian waste management system of plastic packaging. From Figure 1 of Paper II.

ment industry. The RW and BCW are largely incinerated directly in WtE facilities, although parts of these waste streams first pass a mechanical pre-treatment step. The input data and its sources are again detailed comprehensively in the Main article and the Supplementary data of Paper II. For the uncertainty characterisation of these data, the same method as for the national plastic flows was used.

4.3 Environmental assessment of plastic packaging waste management

This part of the work uses the MFA of plastic packaging waste described in the previous section, and combines these results with input-dependent inventory data. This type of modelling is achieved in the EASETECH software, which uses a reference flow which can be described with respect to the material composition, allowing tracking of single materials or elements through the system and input-dependent calculation (i.e. in relation to the waste composition) of the environmental impacts.[119]

As (part of) the goal of this LCA study is to provide guidance on the extent to which a policy target makes sense from an environmental perspective, and actions in a WMS can be expected to have effects on the background system,[79] a meso/macro-level decision-oriented context situation is considered (context situation B according to EC[43]). A consequential modelling framework is thus applied, which entails using marginal background data for processes with large-scale, structural consequences.[43]

To solve multifunctionality, system expansion by substitution is used. A generation-based functional unit is selected,[50] so the treatment of the total amount of waste plastic packaging generated in Austria in 2013 is assessed, and the zero burden assumption is applied, thus attributing the burden of the incoming waste to the producer.[79] The system boundaries of the model are shown in Figure 4.3, which include capital goods for each of the processes as they may make a significant contribution to the overall results.[120]

The status quo of waste plastic management in Austria, as described in the MFA (see Paper II), is compared to three alternative scenarios. One considers a WMS where mainly disposal is used (="Mainly disposal"), and is constructed based on how plastic packaging waste was collected, sorted and treated in Austria in 1994, which is comparable to the situation in some of the new EU member states, especially regarding the share of landfilling.[2] Two further scenarios represent a WMS which achieves the increased recycling target of 55 % (see Section 1.4):[36] one which continues the current focus on (high-quality) single-polymer recycling (="EU-target-SP", single-polymer) and one which achieves the required recycling amount through mixed-polymer recycling only (="EU-target-MP", mixed-polymer). Compared to the status quo, these three alternative scenarios only differ with respect to the routing of the waste flows through the system, and the mass and composition of the waste input, and thus the functional unit, remain constant.

To assess the robustness of the results, the parameter uncertainties were characterised and propagated through the model, whereas the scenario uncertainties were investigated in a scenario analysis. The model uncertainties, however, were outside the scope. For the parameter uncertainties, where statistical analyses or uncertain input data were not available, the pedigree matrix approach, as introduced in Section 3.1.3, was used. These input uncertainties were propagated analytically and interpreted using a global sensitivity analysis framework.[121] A detailed overview of the inventory data, including their uncertainties, for all processes displayed in Figure 4.3 is given in the Supplementary data of Paper III.

For the LCIA, a wide range of ICs was considered to include a large spectrum of environmental consequences and prevent potential burden shifting.[50] The 14 midpoint ICs and their respective LCIA characterisation models as recommended by the ILCD were evaluated, as these were found to represent best practice.[122;123] Two ICs were further subdivided though: aquatic eutrophication into freshwater and marine eutrophication, and mineral and fossil resource depletion into one covering minerals and metals and one covering fossil resources. The following ICs and LCIA methods were thus taken into account:

- Global warming (GW): IPCC[124]
- Ozone depletion (OD): WMO[125]

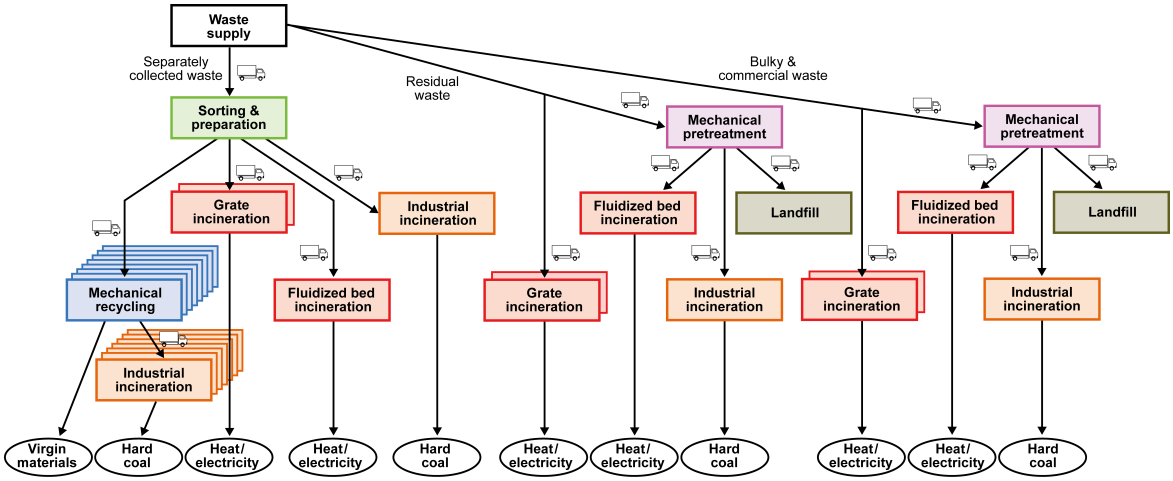

Figure 4.3: Model overview and system boundaries for the assessment of the Austrian waste management system of plastic packaging. From Figure 1 of Paper III.

- Human toxicity, cancer effects (HTc): USEtox[126]
- Human toxicity, non-cancer effects (HTnc): USEtox[126]
- Particulate matter (PM): Humbert[127]
- Ionising radiation (IR): Frischknecht et al.[128]
- Photochemical ozone formation (POF): ReCiPe[129]
- Terrestrial acidification (TA): Seppälä et al.[130]; Posch et al.[131]
- Terrestrial eutrophication (TE): Seppälä et al.[130]; Posch et al.[131]
- Freshwater eutrophication (FE): ReCiPe[132]
- Marine eutrophication (ME): ReCiPe[132]
- Ecotoxicity (ET): USEtox[126]
- Land use (LU): Milà i Canals et al.[133]
- Resource depletion, water (RDw): Frischknecht et al.[134]
- Resource depletion, minerals and metals (RDm): CML[135]
- Resource depletion, fossil (RDf): CML[135]

The characterised results were normalised to person equivalents (PEs) using the global normalisation factors for 2010 established by Laurent et al.[136].

Chapter 5

Results and discussion

In this chapter, the results from the three published papers are summarised in three sections corresponding to the three papers, and for the full results and discussions, the reader is referred to the Appendices.

As a general note, the description of the results in this thesis on the performance of the collection, sorting and recycling processes uses three rates: the collection rate (CR) (the amount separately collected divided by the total waste amount), the sorting rate (SR) (the amount sorted and sent to the mechanical recycling processes divided by the total waste amount, as adopted by the EU for the recycling targets, see Section 1.4), and the recycling rate (RR) (the amount of re-granulate produced at the mechanical recycling plant divided by the total waste amount). It has to be noted that this definition of the "recycling rate" differs from the way this term is usually defined, as it is generally used to refer to the amount in the input of the recycling plant (i.e. the SR in this thesis) rather than in the output as defined here. However, I feel that this approach is more accurate in describing what these three rates actually represent, and the explicit distinction between the three rates helps in clearly communicating the results.

When discussing the results of other studies, it is not always clear which calculation method is used, although most studies seem to use the SR. When the calculation method is clear from the text, the respective abbreviation as proposed above will be used in the discussion, whereas otherwise the term introduced by the respective authors will be reproduced.

5.1 National Austrian plastic flows

The results of the Austrian plastics budget for 2010 are shown in Figure 5.1. The total domestic primary polymer production amounted to 1100 kt/a±2%, mainly producing polyolefins (79 %) followed by polystyrenes (16 %) and resins (5 %). These primary polymers were heavily traded, with large import and export flows of about

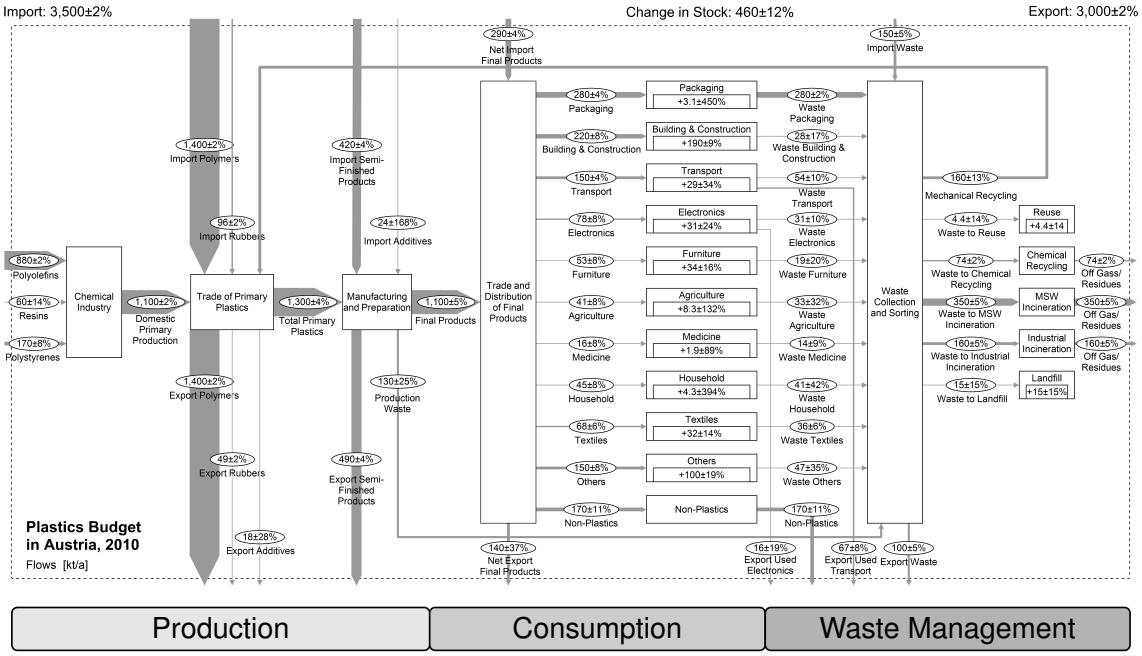

Figure 5.1: Results of the Austrian plastics budget in 2010. The flow values are given by the mean with two significant digits and the relative standard deviation. From Figure 3 of Paper I.

1400 kt/a±2% each. After further imports and exports of rubbers, additives and semi-finished goods, about 1100 kt/a±5% of domestically produced final products went into the consumption stage.

An extra net amount of 150 kt/a±34% of final products was imported, meaning that the overall plastics consumption added up to 1300 kt/a±5% (154 kg/cap·a). The sectoral distribution for plastics consumption is displayed in Figure 5.2, and was different compared to the one shown in Figure 1.2 for the whole of Europe, as more plastic types and sectors were taken into account in this work. The most important consumption sector from a mass perspective was the packaging sector, followed by building & construction, non-plastics, transport and others. Together, these five sectors represented about 80 % of total plastics consumption. The total change of the in-use stock amounted to +440 kt/a±12%, which means that the net increase of the total size of the stocks was equal to about one third of the incoming amount of consumer goods. The stocks increased in all sectors, but large differences between the sectors exist, as is evident from Figure 5.2. Sectors with products with short lifetimes, such as packaging and medicine, had negligible stock increases, whereas the largest build-up in stock took place in the building & construction sector (44 % of the total stock increase). This large stock increase means that plastics consumption is far from reaching a steady state.

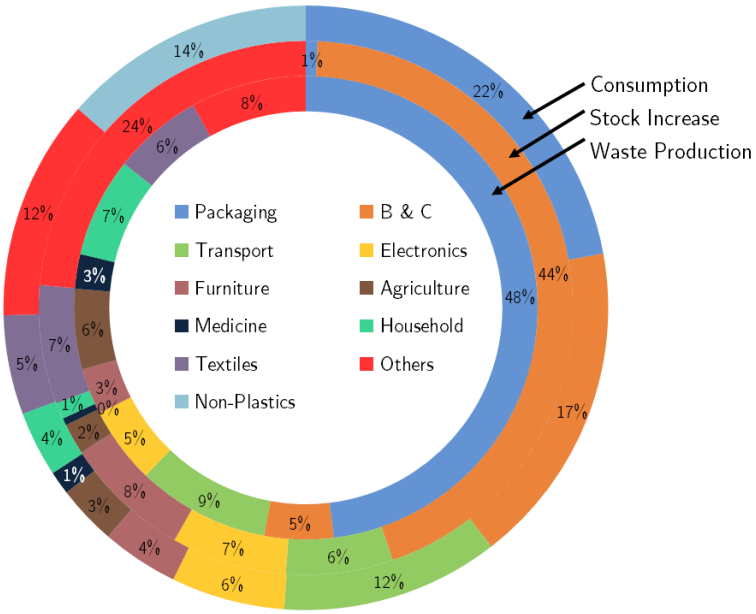

Figure 5.2: Sector shares of consumption, stock increase, and waste production, for the Austrian plastics budget of 2010.

Two sectors had important exports of used products, namely transport and electronics. This means that these end-of-life flows became or will become waste outside of Austria and are not handled by the Austrian waste management system, thus constituting a loss of potentially valuable materials. Of the total post-consumer waste, which amounted to 580 kt/a±5% (70 kg/cap·a), about half was caused by the packaging sector (see Figure 5.2). The total waste stream which needed to be treated, after inclusion of production wastes and net imported streams, was 760 kt/a±3%, or about 91 kg/cap·a. This amount was mainly incinerated with energy recovery: 46 % in WtE plants and 21 % in the cement industry. An RR of 21 % was reached, whereas 10 % was directed to the steel industry, and thus considered to be chemically recycled. Finally, minor amounts were reused (1 %, mainly textiles) and landfilled (2 %). All in all, 98 % of the wastes were utilised, be it via recycling or thermal utilisation, which is much higher than the European average, where about 42 % of waste plastics were still landfilled in 2010. [(2)]

From a resource perspective, the large stock increase and relatively low recycling rates limit the domestic resource potential of secondary plastics. This becomes clear when looking at the quantity of produced re-granulate from mechanical recycling (160 kt/a±13%), which amounts to just 12 % of the total plastics consumption (1300 kt/a±5%).

To summarise, the variety of the challenges posed by plastics from the different sectors can be exemplified by the two extreme and, from a mass perspective, the two

most important sectors: packaging and building & construction. Packaging products have very short lifetimes, which directly cause large waste streams needing proper management. These short lifetimes also mean that changes in design (e.g. increasing considerations of eco-design) have immediate repercussions on the composition of the generated waste stream. Effective cooperation along the whole value chain can therefore ensure sustainable management of this waste stream. In contrast, the products from the building & construction sector can remain in use for over a century, and the application of plastics in this sector has seen a strong increase over the past few decades.[137] Many potentially harmful additives (e.g. brominated flame retardants (BRFs) such as polybrominated diphenyl ethers and hexabromocyclododecane) were used in building products before being regulated (e.g. by the Stockholm Convention) and are still present in the current building stock as legacy substances.[138;139] Therefore, it is important to gain knowledge on the size and composition of the stocks that are currently being built up as well as on those from the past in order to appropriately deal with waste stream challenges arising in the future.

Quantitatively comparing studies on plastic flows in varying regions and time periods can be challenging, mainly due to differing scopes with respect to which materials are taken into account (e.g. looking only at thermoplastics or including non-plastic applications as in this study). With this caveat, when comparing these results with the two previous studies for Austria,[63;64] it is clear that both absolute and per capita consumption increased strongly between 1994 and 2004 (from 1100 to 1300 kt/a and 142 to 158 kg/cap·a). This increase was not continued for the 2004 to 2010 period however, with a similar absolute plastic consumption of 1300 kt/a and a slight decrease in per capita consumption (158 to 154 kg/cap·a), due to the drastic decrease of demand after the 2008 economic crisis. From a waste management perspective, the overall trend is to move away from landfills (from 84 % to 29 % and 2 % from 1994 over 2004 to 2010), highlighting the effect of the landfill ban of 2004.[31] This waste diverted from landfills has mainly gone to incineration (from 10 % to 59 % and 77 %), although the mechanical recycling rate has increased as well (from 6 % to 11 % and 21 %).

Moving away from Austria, Bogucka and Brunner[64] report a landfill rate of 91 % in 2004 for Poland, whereas only 5 % were recycled and 3 % incinerated. Finally, Mutha et al.[71] estimate a recycling rate for India of 47 % for 2000/2001, achieved by a high contribution from the informal sector, while the rest was disposed of in a controlled or uncontrolled way.

5.2 Plastic packaging flows in the Austrian waste management system

The flows of plastic packaging in the Austrian waste management system for 2013 are presented in Figure 5.3, both on a product type and polymer basis. The total incoming waste stream amounted to 300 kt/a±3% (35 kg/cap·a), composed primarily of large and small films (28 % and 19 % respectively), followed by small hollow bodies (18 %), PET bottles (16 %), others (13 %), large hollow bodies (6 %), and large EPS (1 %). From a polymer perspective, the composition of the incoming waste stream was determined to be as follows: LDPE (46 %±6%), PET (19 %±4%), PP (14 %±6%), HDPE (11 %±6%), LLDPE (5 %±5%), PS (3 %±5%), EPS (2 %±4%), and PVC (<1 %±6%).

The following paragraphs will discuss the results as subdivided by product type, while the results per polymer can be found in the Supplementary data of Paper II. The performance of the waste management system regarding separate collection, sorting and recycling is displayed in Figure 5.4. Of the total waste stream, 58 %±3% was collected separately, whereas 30 %±4% was discarded into the RW and 12 %±4% into the BCW. The highest CR values were achieved for the films, EPS large and PET bottles (64-77 %). The sorting efficiencies (i.e. the output of the storing plant sent for mechanical recycling divided by the input into the sorting plant) across the product types showed two distinct groups. PET bottles and the categories of large products were sorted more efficiently (efficiencies ranging from 73 to 88 %) than was the case for small hollow bodies (47 %), small films (34 %) and others (12 %). Combining these efficiencies with the respective CR values shows that the highest SR values were achieved for the EPS large, PET bottles and films large categories, while the SR for the overall waste stream amounted to 34 %±3%.

Plastics that were not collected separately were disposed of in either the RW or BCW. The former was largely treated directly in grate WtE plants (84 %), whereas for the latter this was only the case for about 51 % of the waste stream. The remaining shares of both waste streams were first pretreated mechanically, after which the outputs of this process were mainly sent to fluidised bed incineration (67 %), next to the cement industry (28 %) and landfill (5 %).

Figure 5.5 displays the shares of each of the product types in terms of the final treatment processes. Overall, an RR of 26 %±7% was reached, whereas the remaining was sent to energy recovery in WtE plants (40 %±3%) and in the cement industry (33 %±6%). Only minor amounts were landfilled (1 %±6%), and although plastics are used for chemical recycling in the steel industry in Austria, in 2013 no waste plastics packaging generated in Austria was used. The output of the mechanical recycling process was used for 12 % in food-grade applications (from PET bottles

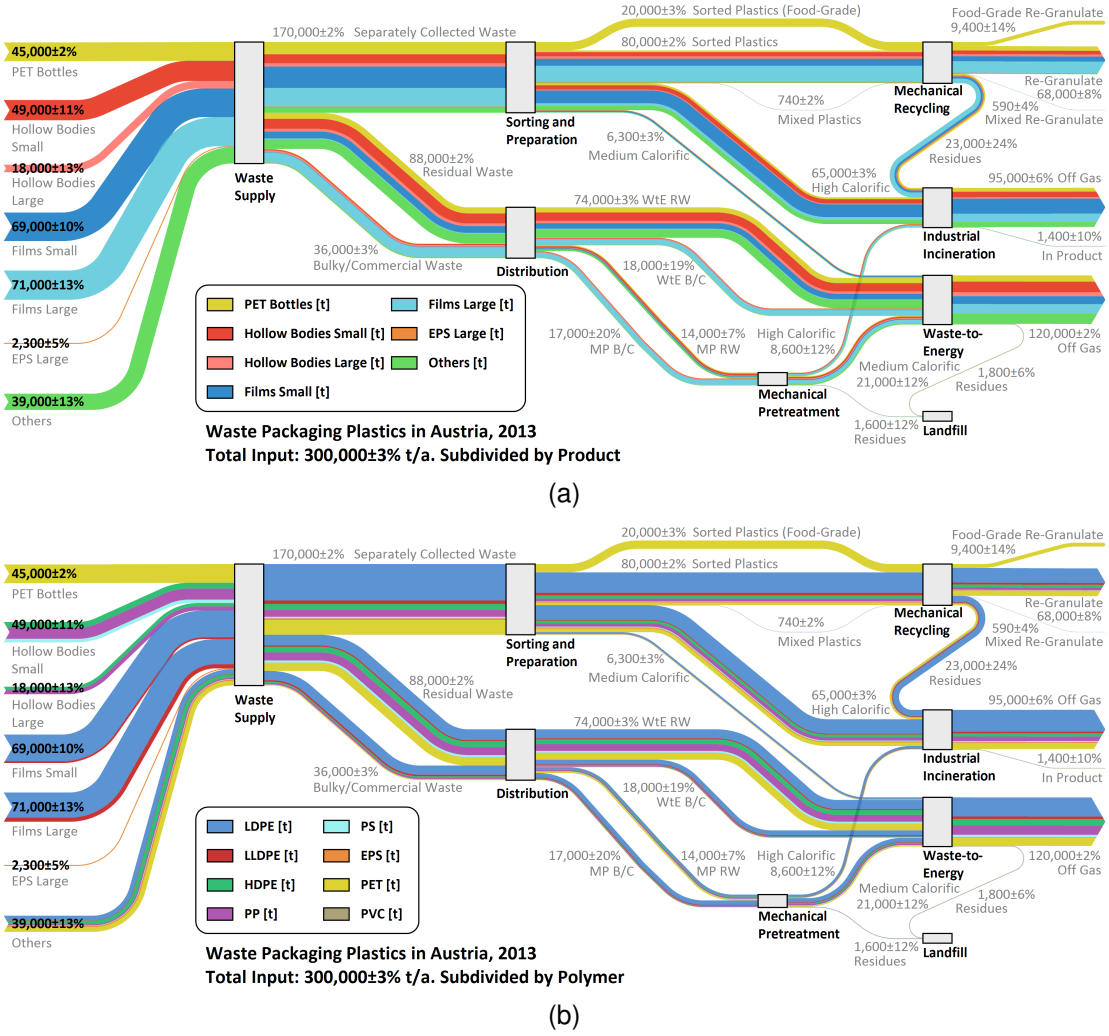

Figure 5.3: Plastic packaging flows in the Austrian waste management system in 2013, subdivided by (a) product type and (b) polymer. The flow values are given by the mean with two significant digits and the relative standard deviation. From Figure 2 of Paper II.

only), 87 % in other single-polymer products, and 1 % for products with mixed-polymer re-granulate. For PET bottles specifically, 46 % of the re-granulate was used for food-grade and 54 % for non-food-grade applications, which represents 21 % and 24 % of the PET bottles waste stream, respectively. However, this distribution depends heavily on the market situation. At the food-grade recycling plant only a part of the waste stream is processed for food-grade applications, as depending on the current demand the rest is sold directly for the production of PET fibres.

Comparing these results with the waste generation from the packaging sector for 2010 (from Paper I), an increase from 270 kt/a (not including drink cartons) to 300 kt/a (35 kg/cap·a) is observed, corresponding to a growth rate of about 8 % over three years. Eurostat[140] reports a waste plastic packaging generation of 34 kg/cap·a for

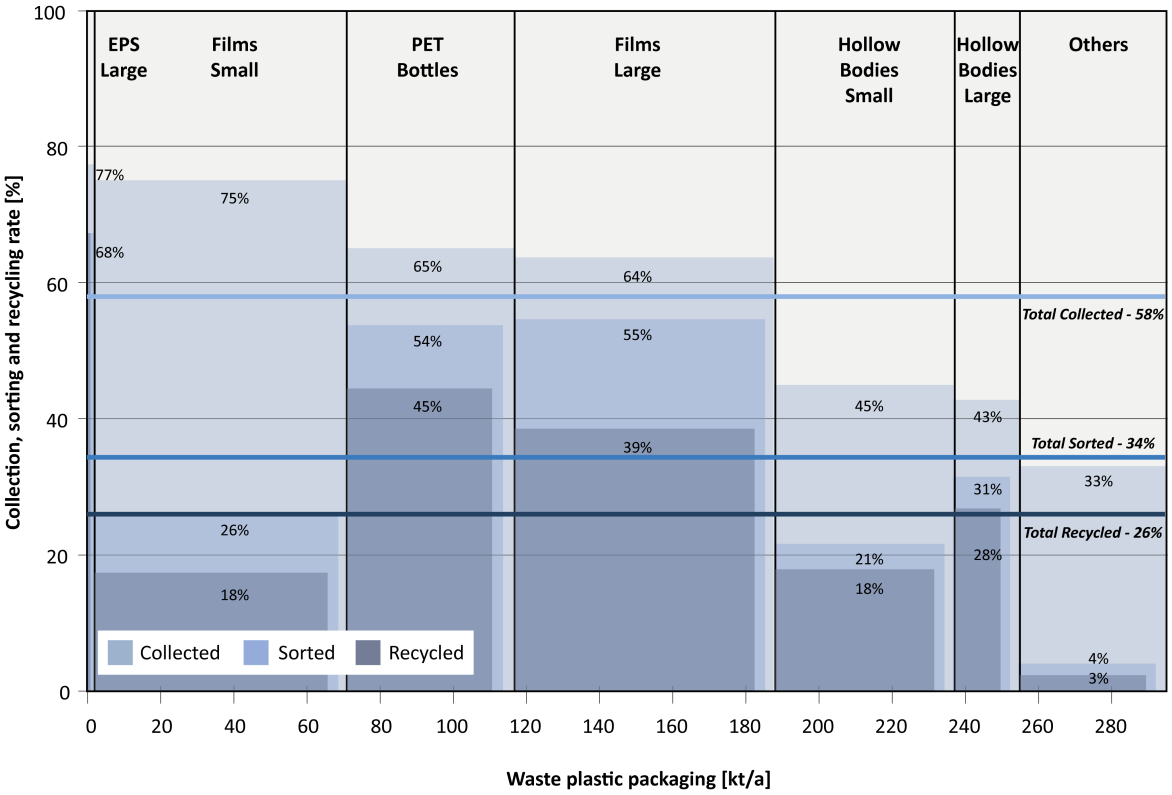

Figure 5.4: Collection, sorting and recycling rates for each of the product types in relation to the respective mass in the input. The rates for the total waste stream are shown by the horizontal lines. From Figure 3 of Paper II.

Austria, which is above the average of 30 kg/cap·a for the EU27. The EU member state with the highest reported waste plastic packaging production in 2013 was Luxembourg (50 kg/cap·a) while the lowest waste production was achieved by Bulgaria (13 kg/cap·a), and overall a clear connection between waste plastic packaging generation and GDP can be observed.[140] For the waste plastic packaging production from households only, so excluding commercial wastes, Brouwer et al.[77] report an amount of 20 kg/cap·a for the Netherlands while Dahlbo et al.[78] report an estimate of 18 kg/cap·a for Finland, both for 2014.

Regarding final waste treatment, the overall RR in 2010 for all wastes treated (i.e. including production and imported wastes) amounted to 21 % (see Section 5.1), which is not that much lower than the RR of 26 % calculated for packaging alone. PlasticsEurope[141] report a recycling rate of 29 % for waste plastic packaging in Austria in 2013, although it is not clear if this refers to the SR (34 %) or the RR (26 %). For most of the EU member states, similar recycling rates between 30 and 40 % for plastic packaging are reported, and the main difference lies in the amount that is incinerated instead of landfilled. On a polymer level, according to Kawecki et al.[65], the packaging polymers with the most recycling and reuse in Eu-

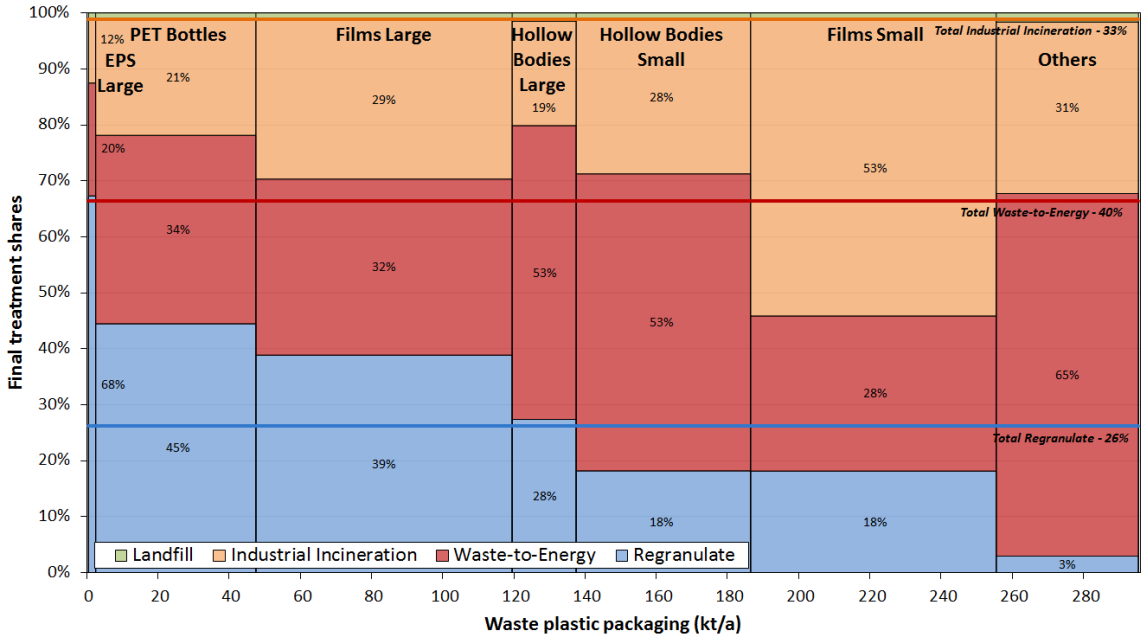

Figure 5.5: Final treatment of each of the product types in relation to the respective mass in the input. The results for the total waste stream are shown by the horizontal lines. From Figure 5 of Paper II.

rope are PET and HDPE, followed by LDPE and EPS. This is different from the results for Austria, where the RRs for the seven polymers are ordered as follows: PET>EPS>LLDPE>LDPE>HDPE>PP>PS>PVC.

Of the considered product types and polymers, PET and PET bottles have received the most attention in the literature. Welle[142] reports a strong increase of the CR for PET drinking bottles in the EU, reaching an average of 48 % in 2009 and peaks to 94 % in Germany (due to the deposit system), whereas in Austria, 65 % of all PET bottles (not only drinking bottles) were collected in 2013 according to this work. In the USA however, the CR has stagnated over the past decades, with a value of 29 % in 2009,[142] and the share of domestically recycled waste, as opposed to the amount that is exported, has even decreased in recent years.[72] Finally, for Switzerland, a CR of 85 % of PET bottles is reported, whereas in the end 26 % is reprocessed into new bottles,[76] compared to the 21 % for Austria.

As mentioned in Section 1.4, member states of the EU need to meet minimum recycling targets for waste packaging. For plastics, this target is set to 22.5 % since the end of 2008,[35] which was adopted by the Austrian implementation as well.[143;144] This target was recently increased however to 50 % by 2025 and 55 % by 2030.[36] Furthermore, the calculation point for reporting the recycled amount is now explicitly defined as the amount of waste entering the final recycling process (i.e. the SR

as defined in this thesis),[36] which is more or less equal to the interpretation which is currently used by most member states.

As shown above, an SR of 34 %±3% was reached in 2013, thereby achieving the current EU and Austrian targets. With respect to the increased EU target of 55 % by 2030, three product types (EPS large, films large, and PET bottles, representing 40 % of the total waste amount) currently have an SR around or above the required value, although major additional efforts will nevertheless be needed to increase the collection and sorting efficiencies to reach this target for the overall waste stream. To illustrate the required process efficiencies to achieve the increased future target, a theoretical scenario was built by changing the collection and sorting efficiencies for the seven product types up to a point where the required 55 % of waste packaging enters a mechanical recycling process (see Table 5.1). This effort shows that quite high values are needed to reach the increased target of 55 %, and thus major improvements in both the collection system and sorting technologies will be necessary. This is especially the case since the reported SR in Austria has been stagnating over the past decade: 31 % was already reached in 2003,[140] compared to 34 %±3% in 2013.

Table 5.1: Change of the collection and sorting efficiencies per product type from the status quo to the future EU target scenario. From Table 1 of Paper II.

Product type	Collection efficiency (%)	Sorting efficiency (%)
PET bottles	*65* → 90	*83* → 90
Hollow bodies small	*45* → 80	*47* → 70
Hollow bodies large	*43* → 80	*73* → 90
Films small	*75*	*34* → 50
Films large	*64* → 80	*86* → 90
EPS	*77* → 80	*88*
Others	*33*	*12*

5.3 Environmental assessment of plastic packaging waste management

An environmental assessment was made of the status quo of plastic packaging waste management, using the waste composition and flows as described in Section 5.2, which was compared to three alternative scenarios regarding the amount of waste sent to mechanical recycling. For the scenarios representing a WMS reaching the increased recycling target, the collection and sorting efficiencies reported in Table 5.1 were used. The LCIA results of these four scenarios for all sixteen considered ICs, subdivided by treatment process, are shown in Figure 5.6.

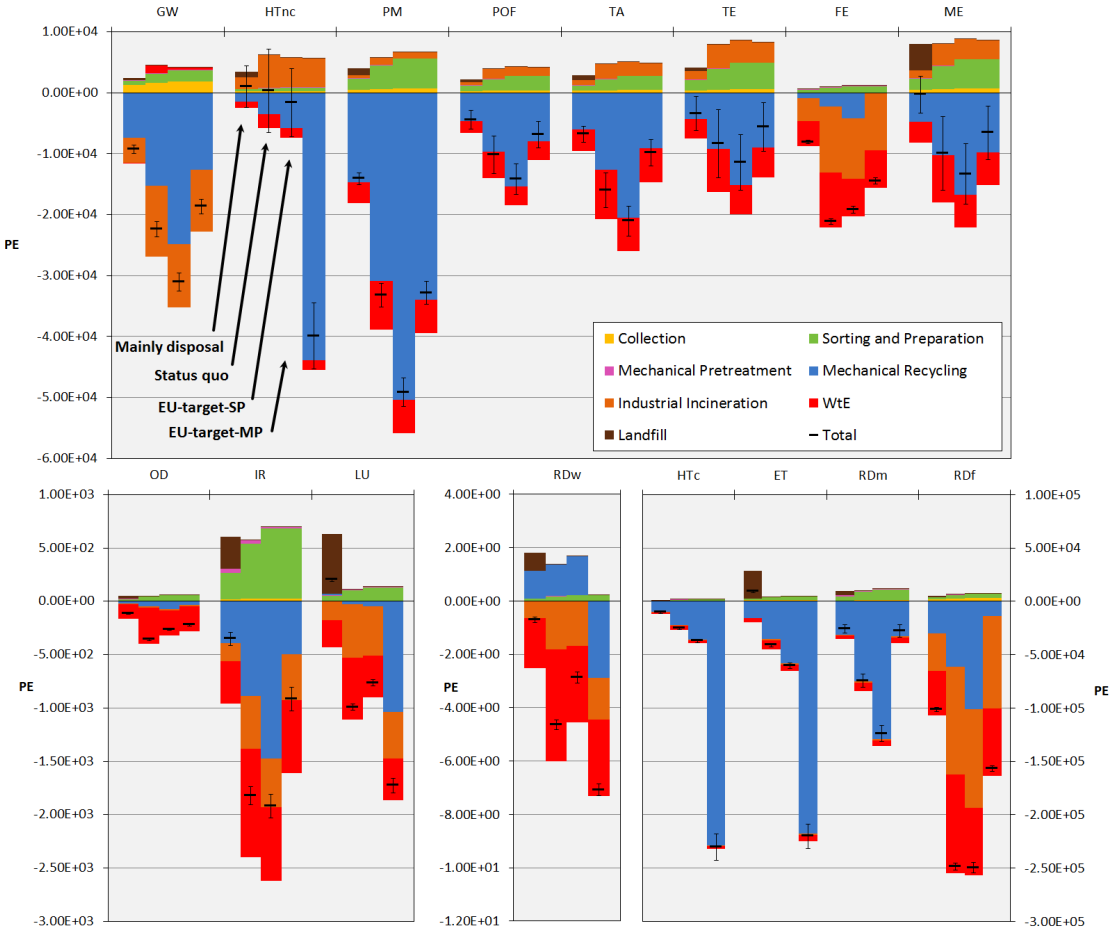

Figure 5.6: LCIA results of the status quo and the three scenarios (Mainly disposal, EU-target-SP, EU-target-MP), subdivided by treatment process, for all 16 considered ICs.

For the status quo, higher benefits than impacts are achieved for all ICs, resulting in net negative impacts, with the exception of HTnc. For OD, LU, RDw and HTnc, however, the break-even point lies within one standard deviation of the result. Relatively low burdens are caused by the collection, sorting, pretreatment and landfill processes. Mechanical recycling causes rather low burdens but achieves large benefits for most ICs, whereas industrial incineration leads to both considerable burdens and benefits of about the same order of magnitude, which balances to net burdens for six of the 16 ICs. For WtE on the other hand, both burdens and benefits are generally lower than for industrial incineration but net burdens are only caused for GW. For most of the ICs the benefits are therefore mainly achieved by the mechanical recycling processes, although, especially for FE, OD, IR, LU and RDf, the incineration processes make important contributions as well. Of the eight distinct mechanical recycling processes which were considered, EPS recycling has the highest specific

net benefit across most ICs, whereas overall PET food-grade and fibre recycling and LDPE recycling achieve most net benefits when taking the processed waste amount into account.

Bringing the net burdens or benefits of the treatment processes in relation to the amounts which are processed shows that for GW the highest net specific benefit is achieved by mechanical recycling ($-0.16\,\mathrm{PE/t}$), followed by industrial incineration ($-0.12\,\mathrm{PE/t}$), landfill (close to $0.00\,\mathrm{PE/t}$) and WtE ($0.01\,\mathrm{PE/t}$). This emphasises the importance of separate collection, as this allows capturing the wastes for mechanical recycling, when a sufficient purity after sorting can be reached, or for industrial incineration, which has a higher energetic efficiency compared to WtE, although the air pollution control system is usually less efficient. This conclusion is confirmed when looking at the LCIA results from a collection perspective, i.e. subdivided by collection route (SCW, RW, or BCW, see Figure 5.7). The treatment of the SCW achieves the vast majority of all benefits of the overall WMS, although only 58 % of the input waste is collected separately.

Looking at the results of the three alternative scenarios (see Figure 5.6), it is clear that for ten out of 16 ICs, the more material is mechanically recycled, the higher the overall benefits are. For FE, RDf, OD, IR, LU and RDw however, the incineration activities contribute considerably to the benefits, causing a similar or even decreased net benefit for the EU-target-SP scenario, due to the decrease of waste material directed to these processes. The high net benefits achieved by the EU-target-MP scenario for the toxicity-related ICs (HTc, HTnc, ET) are explained solely by the substitution of cast iron, which is used in one of the substituted products (street bench). This strong sensitivity regarding the choice of substituted product is further discussed in the scenario analysis. For the other ICs, net benefits similar to or lower than those of the status quo are achieved by the EU-target-MP scenario, indicating the validity of the focus by the EC on high-quality single-polymer recycling.[39]

From the mainly disposal scenario to the status quo an increase of 17 percentage points for the SR is reached, whereas from the status quo to the EU-target scenarios an increase of 22 percentage points is required. When relating the results of the status quo and the mainly disposal and EU-target-SP scenarios to the respective SR values, three types of relationships can be observed (see Figure 5.8 for eight selected ICs and Figure S8 in the Supplementary data of Paper III for the other eight ICs). First, HTnc shows an increasing marginal net benefit with respect to the SR, due to the large net impacts of industrial incineration, the share of which increases strongly for the status quo but decreases again for the EU-target-SP scenario. Second, eight ICs display decreasing marginal benefits or even an absolute decrease in the net benefits when comparing the status quo and the EU-target-SP scenario, due to (a combination of) various reasons: the large net impact of landfilling in the mainly

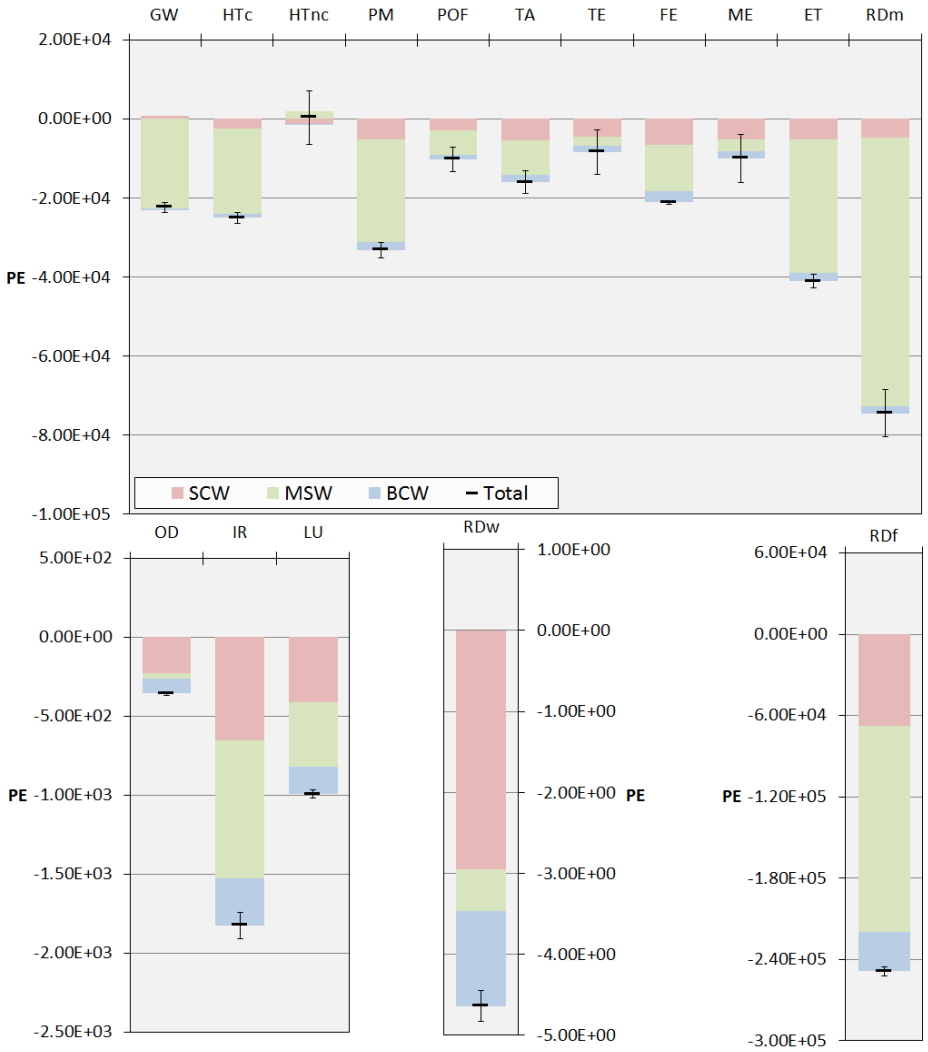

Figure 5.7: LCIA results of the status quo, subdivided by collection route, for all 16 considered ICs.

disposal scenario (ET, ME, LU, RDw), the overall dominance of the incineration processes with respect to the net benefits (OD, IR, FE, RDf, LU, RDw), the large net impact of the sorting process (IR, ME), as well as the fact that mechanical recycling has a net impact itself (RDw). Third, for the remaining ICs an approximately linear (RDm) or slightly decreasing marginal benefit (GW, HTc, PM, POF, TA, TE) is apparent, generally due to the relative dominance of mechanical recycling in achieving the net benefits.

This decreasing marginal benefit and decreasing absolute net benefit with increasing SR, for four ICs each, suggests that an environmentally optimal SR lies below 100 %, depending on the IC. This is further reinforced by the fact that in the scenarios for the EU target, no non-linear effects of e.g. increasing separate collection on transport distances and sorting efficiency are included, which can potentially further decrease

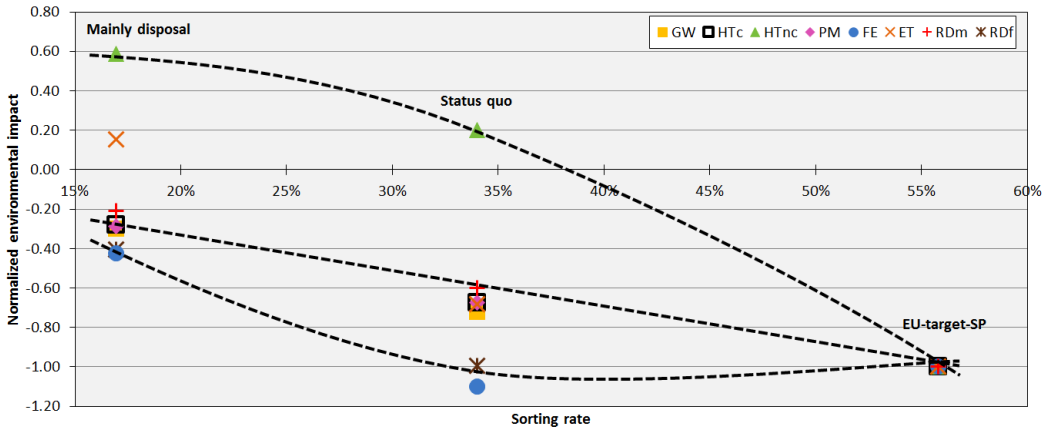

Figure 5.8: Normalised (EU-target-SP=-1) LCIA results for the mainly disposal, status quo and EU-target-SP scenarios in relation to the SR for eight selected ICs, while the results for the other ICs are provided in Figure S8 in the Supplementary data of Paper III. Three types of relationships between SR and impact are observed, as indicated by the stylised trend lines. Adapted from Figure 4 of Paper III.

the benefits for this scenario.[145] Furthermore, the varying response of the different ICs with respect to an increasing SR shows that no optimal SR exists for all ICs simultaneously. To create a sound basis for proposing recycling targets leading to an environmentally optimal outcome, the effects of an increasing SR should be further examined, and it should be made clear which type of environmental intervention is to be optimised.

The investigation of the sensitivity of the model regarding 250 parameters shows that 23 parameters constitute the set of the five most important parameters across all 16 ICs, mostly related to industrial incineration, followed by WtE and mechanical recycling. The energy substitution factor in the cement kiln treating sorting residues is present in the top five of all ICs, and is even the most influential parameter for twelve of them. Combining these sensitivities with the input uncertainties of the parameters allows quantifying the share of each parameter's uncertainty in the overall uncertainty of the result. Overall, 48 parameters out of 250 cause 90 % of the uncertainty of the result across all ICs, although this varies considerably depending on the IC. For some, namely HTnc, POF, TA, TE, and ME, these are entirely governed by 1-3 parameters, while for others (GW, IR, FE, ET, LU, RDw), 15-20 parameters are needed to reach 90 % of the total uncertainty.

To examine the influence of the choices, which were made as part of the consequential modelling (see Section 4.3), regarding the marginal background technology for

the energy substitution in WtE and industrial incineration as well as the material substitution of mixed-polymer mechanical recycling, a scenario analysis was carried out. For WtE, this shows that compared to the substituted electricity produced by hard coal, the change to natural gas or the market mix does not result in a shift from net benefits to net impacts, or vice versa, for any of the ICs. The most important effects include a considerably lower net benefit for GW and RDf when changing to either alternative marginal technologies, and for HTnc a very large increase of the net impact for natural gas, but substantial decrease for the market mix, when changing to these marginal technologies, respectively. For industrial incineration, changing the substituted fuel from hard coal to natural gas results in lower net benefits for four ICs (GW, FE, RDw and RDf), whereas the reverse is true for the remaining 12 ICs, which might be counter-intuitive. However, this is caused not by the incineration of the fuel in the cement kiln, but rather by supply chain of the fuel itself, which causes higher impacts for natural gas (mainly from Russia) than for hard coal (from Western Europe) for most ICs in the case of the Austrian supply mix. Finally, for the EU-target-MP scenario, the choice of substituted product of the mixed-polymer recycling process, specifically the difference between products with and without cast iron, has a sizeable influence especially on the toxicity ICs. As the selection of the substituted product is highly speculative for such a large amount of re-granulate (130 000 t/a or 15 kg/cap·a), the validity of the results for these ICs is to be questioned.

Other studies looking at the full WMS of plastic packaging had quite different results in some instances. Ferrão et al.[101] for example reported net burdens for GW and RDm+RDf for Portugal, whereas this work found net benefits for these ICs, and Ferreira et al.[102] calculated net burdens for FE and net benefits for HTnc for the Setúbal Peninsula in Portugal, whereas the reverse was the case for Austria. For one single product type however, the results of Turner et al.[91] for the United Kingdom (UK) and Haupt et al.[111] for Switzerland are much closer to the ones for Austria. This signifies that results for one product-polymer combination are comparable across different studies with potentially varying geographical and temporal scopes, whereas this is less the case for studies investigating full packaging WMSs, where the importance of the local waste composition plays an important role.

Chapter 6

Conclusions and outlook

In this work, the flows of plastics were analysed on a national level and the WMS for plastic packaging was investigated from a material flows and environmental performance perspective. The results discussed in Chapter 5 are summarised below in order to provide responses to the research questions introduced in Chapter 2.

(i) The plastics flows in the Austrian society are characterised by a primary and secondary production that covers the national consumption from a mass perspective (1100 kt/a primary and 160 kt/a secondary production compared to 1300 kt/a). Mainly polyolefins are produced, however, and thus large trade flows are needed to cover all kinds of plastics applications. The most important of these applications is packaging, followed by building & construction, non-plastics, transport, and others. The building & construction sector is responsible for by far the largest stock increase, whereas about half of the post-consumer waste stream is caused by the packaging sector. Waste management is largely achieved by incineration in WtE (46 %) and the cement industry (21 %), while the rest is mostly recycled mechanically (21 %) or chemically (10 %).

(ii) Due to the continuing strong increase of the societal stocks of plastics and the relatively low recycling rates, even in a country with a comparably advanced WMS, the resource potential of plastic wastes on a national level remains quite low. About 160 kt/a of re-granulates were produced in Austria in 2010 from all waste inputs (including imports), which only amounts to about 12 % of the total national plastics demand. Therefore, to achieve a more circular plastics economy, plastics consumption needs to drop considerably to achieve a steady state, and efforts to increase separate collection and recycling are to be intensified.

(iii) The waste production from plastic packaging in 2013 amounted to 300 kt/a, which corresponds to about 35 kg/cap·a, mainly composed of large and small films and small hollow bodies, including PET bottles. The polymer composition was thus dominated by LDPE (46 %), PET (19 %) and PP (14 %). About 58 % of the

waste stream was collected separately, and after sorting, an RR of 26 % was reached, whereas the rest was incinerated in WtE plants (40 %) and the cement industry (32 %).

(iv) The current recycling target of 22.5 % from the EU and Austrian implementation (calculated as the SR) was reached comfortably. However, to achieve the increased target of 50 % by 2025 and 55 % by 2050, major steps will be needed with respect to both collection and sorting. Three product types, which represent 40 % of the total waste mass, already had an SR around or above the required future target of 55 %: EPS large (68 %), films large (55 %) and PET bottles (54 %). However, the other categories are far away from the target, with required increases of 24 to 51 percentage points. Potentials for improving the current situation can be identified from Figure 5.4 by taking into account the amounts lost during collection, sorting and reprocessing (on the vertical axis) as well as the incoming mass of a certain product group (on the horizontal axis). It is clear that for collection, efforts should be focused on the hollow bodies (except PET bottles) and others product types. Moreover, improving sorting of especially the small product types (films small and hollow bodies small) as well as the others has a large potential to increase the mass of plastic packaging sent to the recycling process. A further potential measure to increase collection could be the introduction of a deposit system, especially for PET drinking bottles, which has shown to be able to reach collection rates of around 90 % in Germany and the Nordic countries.[142;146]

(v) The environmental performance of the status quo of the WMS for plastic packaging shows negative impacts (due to the zero burden assumption), and thus net benefits are achieved for 15 out of the 16 investigated ICs. For most ICs, the benefits are mainly achieved by the mechanical recycling process, although for FE, OD, IR, LU and RDf, the incineration processes make important contributions as well. Compared to the benefits of these final treatment operations, the collection, sorting, pretreatment and landfill processes cause relatively low burdens. The most important parameters are generally those that determine the amount of substitution from the recycling and recovery processes. For mechanical recycling, these parameters thus include collection, sorting and recycling efficiencies as well as the substitution factors of the secondary resources. For the WtE process, the efficiency of the energy recovery plays and important role, whereas the same is the case for industrial incineration, although for the latter this is not determined by plant characteristics but rather by a physical parameter (i.e. the heating value). Finally, for industrial incineration the importance of the emission factors for SO_2, NOx, NH_3 and Hg highlight the importance of the air pollution control system for this process.

(vi) Two alternative scenarios depict situations with a lower SR and mainly landfill disposal, as well as with an SR which reaches the increased EU target. These scenar-

ios indicate for ten out of 16 investigated ICs that the more material is mechanically recycled, the higher the overall benefits are, whereas for the other ICs similar or even decreased net benefits are achieved by the EU-target-SP scenario compared to the status quo. When putting these results in relation to the SR, half of the ICs show a decreasing marginal benefit or even absolute decrease in the net benefit, whereas for seven of the ICs an approximately linear or slightly decreasing marginal benefit is apparent. This thus suggests that depending on the IC the optimal SR from an environmental point of view may lie below 100 %. Furthermore, the response of the different ICs to an increasing SR varies widely, which shows that no one optimal SR exists for all ICs. The effects of increasing the SR on the environmental performance of the WMS should thus be investigated in detail to create a sound basis for proposing recycling targets leading to an environmentally optimal outcome.

These effects of an increasing SR can include increasing separate collection efforts and transport distances which are required for increasing collection amount. Furthermore, separately collecting increasing amounts of waste material can potentially have an effect on the quality of the collected material which could negatively affect the sorting efficiency. These kinds of non-linear effects were already explored for PET bottles in Switzerland,[145] but for a more general investigation of the full plastic packaging waste stream, a considerable amount of further data is needed which thus requires additional research. This should allow further refining the scenarios which reach the increased EU target, which could also be extended to include measures going beyond waste management per se, such as waste prevention.

The environmental assessment of this work has focused on the wastes of only one sector, albeit one that is generally considered to be the most relevant. Plastic wastes from other sectors can pose very different challenges, which therefore may require different assessment approaches. Plastic wastes from the building & construction sector for example can contain legacy substances which may pose health risks and limit the potential for recycling.

This chemical composition regarding chemical substances (such as phthalates and BRFs) was not taken into account in the modelling, although the chemical composition with respect to elements was considered. Although packaging products generally contain less potentially harmful additives compared to e.g. electronic equipment or building materials,[147;148] these substances could nevertheless be of concern, especially for mechanical recycling. This aspect could not be considered due to a lack of data regarding the concentrations in the plastic wastes, and because assessing the impact of these substances in the products of mechanical recycling is challenging due to a lack of data on the exposure from secondary resources. Further research is thus needed to include this type of impacts into LCA. The chemical composition also relates to the quality aspect of mechanical recycling, which is especially relevant

for estimating the substitution potential of the secondary resource. In this work, this was done using the market prices of secondary and primary resources as a first-order approximation, but further work is needed to include the technical properties and market response for secondary plastics to more accurately assess which primary plastics in which products could be substituted by secondary material.

Finally, this work considers a comparably advanced WMS, where plastic litter is generally less of an issue, which is therefore not included in the modelling. However, the main environmental concern regarding plastics is generally regarded to be the problem of marine litter and its impacts both on the marine environment and on human health due to accumulation through the food chain. To asses all impacts caused by plastics and plastic waste, it is therefore crucial to include the impacts of marine litter into LCA, which has been recognised as an important methodological research gap.[149]

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Appendices

Paper I

Comprehensive analysis and quantification of national plastic flows: The case of Austria

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Comprehensive analysis and quantification of national plastic flows: The case of Austria

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ABSTRACT

Plastics have been increasingly used in a wide range of applications, generating important waste streams, but overall information on their flows through society is generally not available. Therefore, the national plastic flows in Austria were analyzed and quantified from the production stage up to the waste management stage, for the reference year of 2010. To achieve this, material flow analysis was used to set up a model quantitatively describing the Austrian plastics budget, and the quality of the data sources was assessed using uncertainty characterization. The results show that about 1.1 million tonnes (132 kg/cap·a ± 2%) of primary plastics were produced in Austria, whereas about 1.3 million tonnes (156 kg/cap·a ± 5%) of plastics products were consumed. Roughly one third of the consumed amount contributed to net stock increase in all consumption sectors, and about half of this increase occurred in building and construction, whereas packaging waste constituted approximately half of total post-consumer wastes (70 kg/cap·a ± 4%). Of the total waste amount (including traded and production waste, 91 kg/cap·a ± 3%), the majority was incinerated in waste-to-energy plants or in the cement industry (46% and 21% respectively), whereas the rest was mainly recycled mechanically or chemically (21% and 10% respectively). The results identify the major national flows and processes of plastics, and evaluate the overall data availability for quantifying these flows. Furthermore, the increasing amounts of plastic wastes, due to large stocks having been built up in sectors with long product lifetimes, necessitate assessing which processing capacities are needed and which treatment priorities are to be set in waste management.

1. Introduction

Plastics have become one of the most used materials globally, showing a stronger growth over the past 50 years than any other engineering material (Allwood et al., 2011), and reaching a global production of more than 300 million tonnes in 2014 (PlasticsEurope, 2015). This is due to the key properties this class of materials possesses: they are inexpensive, lightweight, strong, and very durable. The diversity of the numerous general purpose and specialty high performance polymers means that they are used in a vast range of products and applications, whereas the inclusion of various additives can modify the properties and enhance the performance to make the polymers even more versatile (Murphy, 2001; Thompson et al., 2009).

This combination of the use of many different polymers with various additives makes that in general, plastics form a highly complex resource stream which leads to large and very diverse material flows into society. However, this causes a number of environmental concerns as well. The vast majority of plastics are synthetized from non-renewable fossil resources, which leads to the fact that around 4% of the annual petroleum production is converted into plastics, whereas an additional 3–4% is needed for the energy supply during production (Al-Salem et al., 2009; Hopewell et al., 2009). At the other end of the life-cycle of plastics, waste flows have become increasingly important. The durability of the polymers causes accumulation of improperly discarded items in the natural environment, particularly in marine habitats (Barnes et al., 2009; Browne et al., 2011), where they can cause physical problems for organisms such as entanglement and ingestion (Gregory, 2009), whereas monomers, oligomers, and the included additives of the plastic itself, as well as adsorbed hydrophobic contaminants from the surroundings, can be transferred to organisms and have biolog-

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ical consequences as well (Teuten et al., 2009). Proper management of waste plastics can therefore contribute to reducing both resource consumption and environmental impacts.

This is recognized by the European Commission, which in their circular economy strategy pushes for increased recycling rates of wastes in general, as well as for various materials in packaging wastes, including plastics (EC, 2015). However, to understand what potential for recycling is available to justify these targets, detailed knowledge on the current situation of material flows and stocks is needed. This kind of knowledge is quite well established for some materials, with more than 350 studies published on more than 1000 individual elemental material cycles, especially on metals (Chen and Graedel, 2012), e.g. steel (Hatayama et al., 2010), aluminium (Buchner et al., 2014), and copper (Spatari et al., 2002). However, for plastics, information from government and industry is much sparser, and therefore only limited research on the cycles of individual polymers, or plastics in general, has been performed. The main publications describing flows and/or stocks of plastics in a country or region are subsequently shortly described.

A static material flow analysis (MFA) for Austria was established for 1994 by Fehringer and Brunner (1997), determining the flows and stocks of total plastics as well as specifically for polyethylene (PE) and polyvinylchloride (PVC), and for 2004 by Bogucka and Brunner (2007), who compared these results with data for Poland. Furthermore, Joosten et al. (2000) used supply and use tables to examine the flows of plastics in the Netherlands in 1990, whereas Patel et al. (1998) analyzed past flows of plastic production and consumption in Germany from 1976 to 1996, and used various scenarios to forecast future waste volumes and plastic stocks until 2050. Similarly, Mutha et al. (2006) studied past plastic flows in India from 1960 to 2000, and estimated future developments until 2030 using the national relationships of per capita plastics consumption with the respective gross domestic product (GDP), combined with projections on the development of the GDP in India. On the level of single polymers, Kuczenski and Geyer (2010) conducted an MFA of polyethylene terephthalate (PET) in the United States (US) over the period of 1996–2007, whereas Nakamura et al. (2009) determined the flows of PVC in Japan in 2000 using input-output analysis. Finally, Kleijn et al. (2000) used the case of PVC in Sweden to examine the delaying mechanisms of stocks with respect to waste production in a dynamic MFA, using various scenarios regarding the distribution of input streams and life spans of the products.

These analyses provide valuable understanding on the regional material flows of plastics. However, the mentioned studies present MFAs that are 10 to 25 years old, do not always present the full overview of plastic flows from chemical industry up to and including waste management, and do not distinguish different sectors of plastics consumption. Furthermore, some use a limited amount of different data sources, mainly relying on trade statistics, and none of the studies take data uncertainties into consideration, so no assessment of the quality of the data and subsequently of the robustness of the results can be made.

The aim of this paper is therefore to build on, update and expand the previous work of Fehringer and Brunner (1997) and Bogucka and Brunner (2007), and thus establish the plastics budget of Austria in 2010 to investigate plastics production, use and associated waste flows. Due to constraints in data availability, as data sources are usually updated rather infrequently and not all data are available on an annual basis, 2010 was chosen as the reference year. The data sources needed for the analysis of the Austrian plastics budget are provided, including data quality evaluation and uncertainty assessment, to enable future updates of the system for resource accounting to be done more easily. The model illustrates the structure of the processes and flows of plastics in Austria with a high resolution, which enables a thorough understanding of production, demand and waste generation of plastic materials in Austria. This serves as a basis to identify opportunities for increasing overall resource efficiency through effective waste management, and thus reduce negative impacts from production and consumption of plastics.

2. Materials and methods

To achieve the above stated aim, a model of all relevant processes, and the flows that connect these processes, is set up and quantitative information is provided, using material flow analysis. Furthermore, the uncertainties of the mass flows are characterized, assessing the quality of the data sources, to provide probable ranges with the best estimates of the results, and to highlight potential limitations of these applied data sources. These methods are described more in detail in the following sections.

2.1. Material flow analysis

Material flow analysis comprehensively assesses the flows and stocks of materials through a system that is defined in space and time. It is used to connect and quantify the sources, pathways, and intermediate and final sinks of the material in physical units. The calculations are based on the law of conservation of matter, by using a material balance to compare all inputs, stocks, and outputs of a process. The analysis can be carried out on the level of goods (e.g. wood), or on the level of specific chemicals (e.g. carbon, usually called substance flow analysis, SFA) (Brunner and Rechberger, 2004; van der Voet, 2002). Furthermore, different ways exist to model the system's flows and stocks, from static modelling, representing a snapshot of the material flow system, over the modelling of time-series in the past to keep track of flows and stocks, to dynamic modelling, where time is included as a modelling parameter to predict future behavior of the system (van der Voet, 2002; Zoboli et al., 2015). The procedure of an MFA begins with defining the goals, and selecting the relevant substances, system boundaries, and processes. Subsequently, the mass flows and substance concentrations are determined, considering the uncertainties, and finally the results are presented. This procedure should be carried out iteratively, to continuously check, refine and optimize the results (Brunner and Rechberger, 2004).

The STAN software was used to describe and analyze the system with a standardized method (Cencic and Rechberger, 2008). This software is applied widely for material flow analysis (e.g. Beretta et al. (2013); Chancerel et al. (2009); Yoshida et al. (2013)), is made specifically for conducting an MFA, and is freely available (from stan2web.net). The software has a graphical user interface to build up the model with different layers (goods and substances) and over multiple time periods. It allows for the consideration of data uncertainties, unknown flows can be calculated by balancing the system, and the associated uncertainties of these calculated values are determined using Gaussian error propagation, assuming normally distributed variables. Furthermore, data reconciliation can be performed by the software, provided that more information is available than is required for solving the mass balance equations (i.e. overdetermined system of equations). Data reconciliation resolves conflicts between uncertain input values by forcing the values to comply with given mass balance constraints. The inverses of the variances are used as a weighting factor, resulting in highly uncertain values being changed more strongly relative to less uncertain flows (Laner et al., 2015a). Finally, the material flow results are presented in a Sankey style diagram, to be able to easily recognize the relative size of the individual flows.

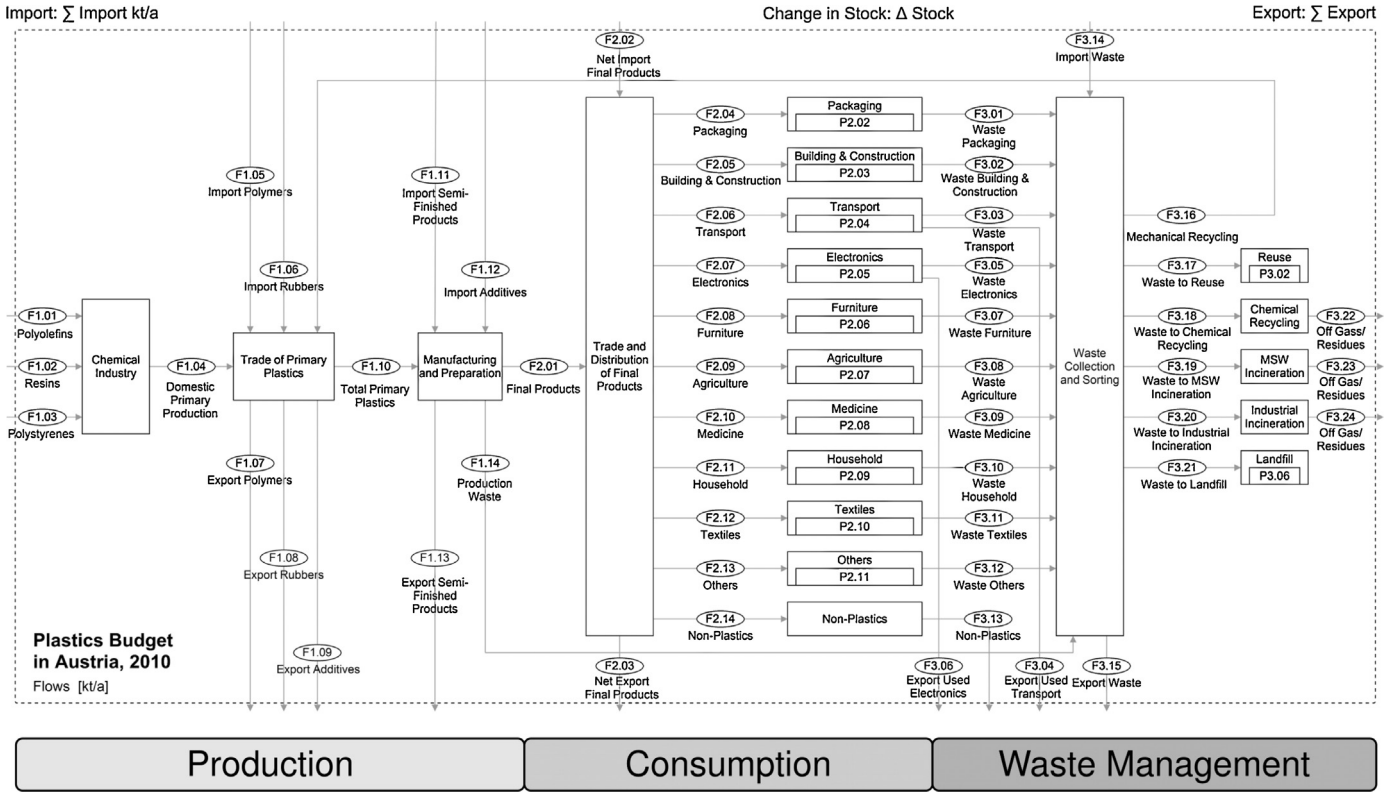

Fig. 1. Model overview of the plastics budget of Austria. The subsystem 'trade and distribution of final products' is presented in Fig. 2.

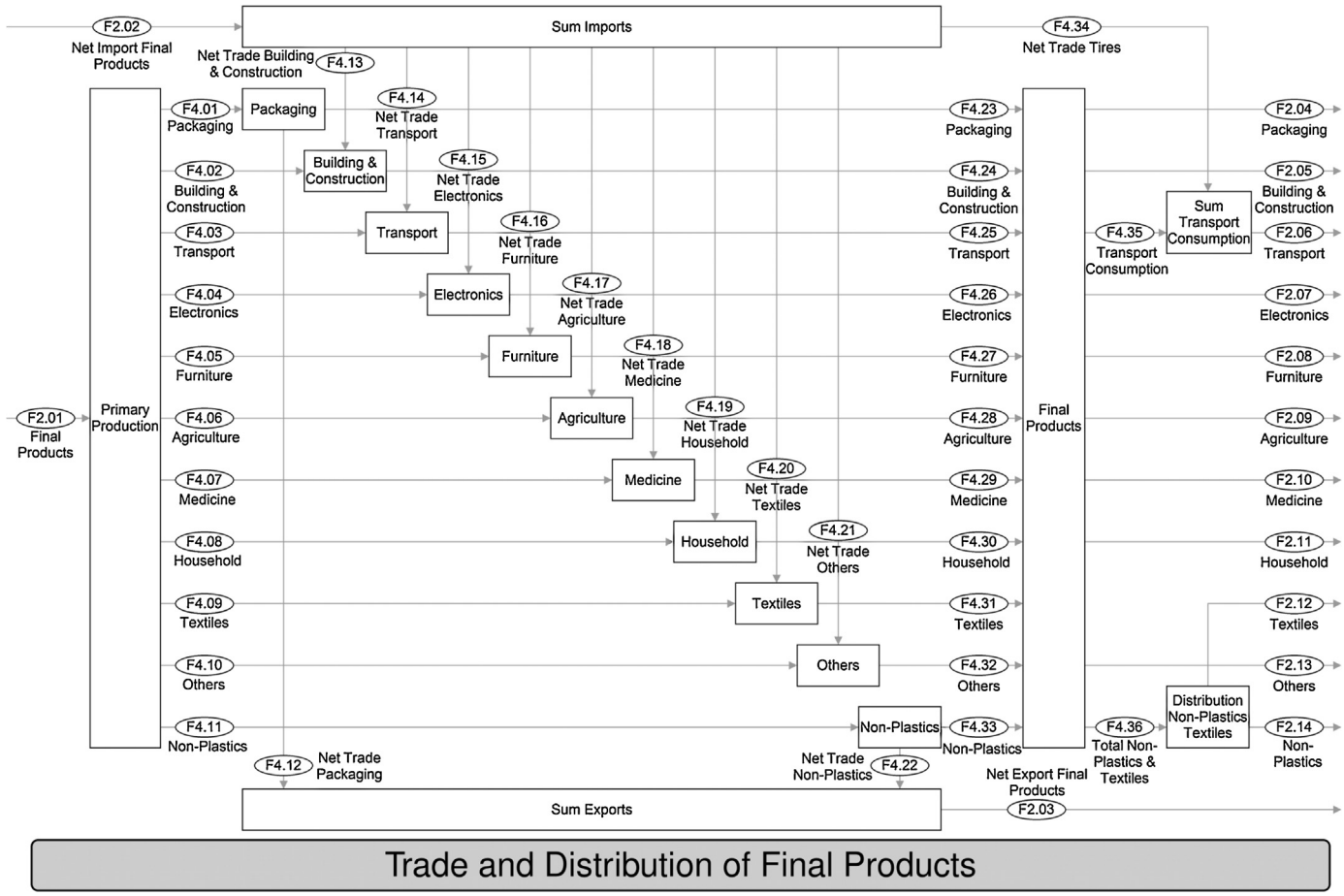

Fig. 2. Subsystem 'trade and distribution of final products' of the overall plastics budget model presented in Fig. 1.

Table 1
Considered consumption sectors and some product examples.

Consumption sector	Product examples
Packaging	Foils, bottles, carrier bags
Building & construction	Pipes, floor coverings, window profiles
Transport	Cars, trucks, bicycles
Electronics	Household appliances, cables, batteries
Furniture	Chairs, matrasses, tables
Agriculture	Silage foils, seed trays, mulch films
Medicine	Syringes, infusion bags, disposable gloves
Household	Kitchen utilities, cutlery, storage containers
Textiles	Clothing, furniture covers, yarns
Others	Toys, sporting equipment, carpets
Non-plastics	Lacquers, adhesives, coloring pigments

2.2. Description of the plastics budget model

In this study, a static MFA is used to investigate the flows of plastics in Austria in 2010. All plastic fluxes are taken into account, including rubbers, as well as the additives added to the plastic material, and the sum of these materials are referred to further on as 'plastics', unless stated otherwise. The full model of the Austrian plastics budget is presented in Fig. 1, whereas the subsystem 'trade and distribution of final products' is introduced in Fig. 2. The overall model is comprised of three main parts, namely the production, consumption, and waste management stages, and consists of 37 processes, 12 of which have stocks, 88 flows, and 10 transfer coefficients (including the sub-system).

First, in the production stage, primary polymers are produced by the chemical industry. In the case of Austria, polyolefins, resins and polystyrenes are synthetized. Further on, primary polymers, rubbers and additives are imported and exported. Finally, these primary plastics are molded into semi-finished (such as sheets and films), as well as final products, in the manufacturing and preparation process. During this process, additional additives are introduced to achieve the required properties of the product. Here, imports and exports of semi-finished products arise as well.

Second, the domestically produced final products are delivered to the consumption stage. Further trade of final products is carried out, which is represented in the diagram as net trade flows, as not all separate imports and exports were available for all consumption sectors. Indeed, the sum of the domestically produced final products and the imported commodities are distributed over various consumption sectors, of which eleven were defined in this study to group the final products. These consumption sectors, with some examples of the included products, are presented in Table 1. The products then enter the stock of the respective consumption sector during the time they are used by. Because of the dissipative nature of the use of non-plastic products, they are not relevant for waste management. Therefore, this sector is not considered in the post-use phase, and is modelled as an export flow.

Third, when the products are discarded by the users, they arise as waste flows coming from the consumption sectors and going to the waste management stage, which encompasses collection and sorting of the waste streams. Furthermore, production waste from the manufacturing and preparation process, as well as imports and exports of plastic wastes are supplied as well. Some waste products, which are in good condition, can be reused after being cleaned and checked for appropriate functionality, which gives these articles a second life as consumer goods. If this option is not available, the waste flow has to be managed through various recycling or recovery operations.

During mechanical recycling, the plastics are reduced in size and extruded to produce re-granulate, which can be used to produce new plastic goods. This flow is thus fed back to the production stage. For this process, the re-granulate output flow is directly displayed in the model overview graph (Fig. 1), and flows of residues going to incineration or landfill are implied within the collection and sorting process for ease of presentation. Chemical recycling on the other hand breaks down the polymers into the constituting monomers, which can then be used as feedstock for other production processes. The use of waste plastics in a blast furnace in the steel industry combines thermal recovery and feedstock recycling, as the plastics act as a reducing agent and thus replace the use of cokes (Trinkel et al., 2015). Therefore, this operation can be considered as chemical recycling (Al-Salem et al., 2009). A further treatment option is the thermal utilization of waste plastics via incineration. These waste plastics can be included in municipal solid waste (MSW) or equivalent waste streams, and thus be incinerated in waste-to-energy plants. Furthermore, high-quality fuels (refuse-derived fuel, RDF), consisting mainly of waste plastics, are used as an alternative energy supply in the cement industry. Finally, wastes and residues containing minor contents of plastics which therefore cannot be utilized (for instance due to their low calorific value) are delivered to landfills for final disposal. Thus, six waste treatment operations were considered in this study, as presented in Fig. 1.

Losses at various points along the life cycle stages of the plastics products, such as dissipative losses and littering, are not considered, as they do not amount to mass flows which are meaningful in the context of this study (order of magnitude of kilotonnes), e.g. around 25 t per year are estimated to leave the country via the Danube river (Hohenblum et al., 2015).

2.3. Data collection

A wide range of data sources was needed to establish the plastics budget of Austria: data on mass flows of polymers or products, as well as average plastics contents of these semi-finished and final products. Data on the domestic primary production were provided by the respective producing companies, through personal communication (Leitner, 2013; Zachhuber, 2013), or via the internet webpage (Sunpor, 2013). Further on in the model, trade data are taken from the foreign trade statistics of the Austrian Statistical Agency (Statistik Austria 2011a), where all positions were scanned for those relevant to plastics production. For primary plastics, the plastics content was estimated to be 100%, while for the semi-finished products, average content data reported by the Swedish Chemicals Agency (2015) were used. The flow of additives is more difficult to estimate, as this usually is confidential company information because of the vital influence of these additives on the properties of the plastic. The imports and exports of additives were therefore determined using expert judgement (Archodoulaki, 2013; Katzmayer, 2013; Zachhuber, 2013). This was also the case for the production waste of the manufacturing and preparation stage, which was estimated to equal around 10% of total production (Pilz, 2013).

For the trade of final products of the respective consumption sectors, statistics from Statistik Austria (2011a) were again analyzed, combined with data on the mass and plastics contents of the products (Chancerel and Rotter, 2009; ICCT, 2013; Kren, 2013; Leibetseder, 2013; McGuire, 2009; Motosuisse, 2015; Schoemaker, 2007; Statistik Austria, 2011c; Swedish Chemicals Agency, 2015; Volvo Trucks, 2015). The exception is the packaging sector, where more detailed data were available from Hauer et al. (2012). The total amount of final products is delivered to the consumption sectors, but for Austria, no relevant data were available on the distribution of plastics consumption over the sectors. Therefore, data from Germany were used (Consultic, 2012), and adapted where needed with information from FAO (2011), Statistik Austria (2011a), and Hauer et al. (2012). These data on domestic consumption, together with the trade of the final products, were used to calculate the

domestic production of plastics for the individual consumption sectors. However, for the transport sector, a different approach was taken, as detailed information from WKO (2012) was available on the consumption of domestically produced transport equipment. Therefore, this value was used together with the total domestic consumption to determine the trade flow of transport products.

To determine the size of the waste flows from the consumption sectors, a number of reports from public authorities and other stakeholders were used:

For the packaging sector, Hauer et al. (2012) estimated the total waste generation. The plastics contents of these products were assumed to be 100%, except for food cartons, where this value was taken from Robertson (2012). The amount of waste from building sites (mass: BMLFUW (2011), their plastics content: Pladerer et al. (2004)), the mass of separately collected waste at demolition sites (ÖAKR, 2015; Vinyl 2010, 2011), general demolition waste (mass: BMLFUW (2011), their plastics content: Kleemann et al. (2016)), and building refuse in bulky waste (mass: Pladerer et al. (2002), plastics content: Swedish Chemicals Agency (2015)) were reported for the building and construction sector. Looking at the transport sector, end-of-life (EOL) cars are collected and shredded, or exported. Kletzmayr (2012) reported the amount of shredded cars and the manually dismantled components at the shredder, as well as the number of exported EOL cars. The number of EOL trucks is calculated from the number of newly registered vehicles and the change in stock (Statistik Austria 2011c, 2011b), and these are exported to a very large extent (Kletzmayr 2015). The plastics contents of these vehicles were taken from the Swedish Chemicals Agency (2015). Furthermore, the amount of separately collected, as well as the estimated amount of exported waste tires was documented (BMLFUW, 2011; ETRMA, 2011; WRAP, 2006). EOL electronics, of which the plastics contents were estimated with data from the Swedish Chemicals Agency (2015), are supposed to be collected through the official take-back systems, which report the mass of collected appliances (EAK, 2011). However, some devices also end up in residual waste (Amt der NÖ-Landesregierung, 2011), at metal scrap dealers, or are exported (Baldé et al., 2015). For the furniture sector, the amount present in the bulky waste was reported by Pladerer et al. (2002), whereas the amount of plastic wastes from agricultural applications was estimated with data from Amt der NÖ-Landesregierung (2011) and UKEA (2001), and medicine wastes were investigated by Obersteiner and Scherhaufer (2008) and BMLFUW (2011). Wastes from the household sector are encountered in residual waste, of which the composition was analyzed by Amt der NÖ-Landesregierung (2011). This gives information on textile waste as well, next to separately collected textiles (Stadtschreiber, 2005) and textiles in bulky waste (Pladerer et al., 2002). Finally, wastes from the others sector appear in residual (Amt der NÖ-Landesregierung, 2011) and bulky waste (Heisterberg-Moutsis et al., 2012; Pladerer et al., 2002). Imports and exports of plastic wastes arise as well, in addition to the domestic production, and are reported by the Austrian statistical office (Statistik Austria, 2011a).

For the waste treatment operations, first of all the reuse of plastic products was investigated, showing that this mainly takes place for goods from the electronics (EAK, 2011) and textiles (FTR, 2008) sectors. The output of the mechanical recycling plants was estimated using declared production capacities and annual statements of accounts in conjunction with prices for plastic re-granulate, whereas the amount of waste plastics used for chemical recycling was reported by Trinkel et al. (2015). For the incineration processes, the included amounts of waste plastics were determined via the so called balance method for waste-to-energy plants (Schwarzböck et al., 2016), and reported by Mauschitz (2014) for industrial utilization in cement kilns. Finally, the residual amount deposited on a landfill was taken from Consultic (2011).

2.4. Uncertainty characterization

Assessing the reliability of the results of an MFA is necessary, as these are inherently uncertain due to data limitations and restricted system understanding. The data sources presented in the previous section have various origins and qualities, and some do not describe the appropriate data point exactly, because of differences with respect to e.g. the geographical frame or time period. It was therefore needed to derive the quantity of interest through up- or downscaling, transformation of data reported for similar systems, or to use expert judgement. Consequently, systematically performing quantitative uncertainty characterization is essential, to allow for the description of the quality of the data and therefore acknowledge data gaps and validate the results (Laner et al., 2014, 2015b).

Characterizing this uncertainty is a crucial step, which should be carried out uniformly and transparently. Therefore, in this study, the approach described by Laner et al. (2015b) is used. This method builds on the data quality assessment scheme proposed by Weidema and Wesnæs (1996), and the assessment of material flow data uncertainty using data classification presented by Hedbrant and Sörme (2001). The combination of these two concepts results in five data quality indicators, which each have four possible scores (ranging from 1: good quality, to 4: poor quality, see Table 2). Here, the reliability indicator focuses on the data source itself, assessing how well the data generation was documented. Completeness evaluates if all relevant mass flows are included in the data point or if the data just represent a fraction of the relevant entities. Temporal and geographical correlation refer to the congruence of the data point available with that of the system under investigation with respect to deviations in time and space, respectively. Finally, the other correlation indicator takes deviations due to other factors into account, such as available data referring to a different product or technology. The qualitative evaluation criteria of the five indicators are presented in Table 2. Furthermore, different sensitivity levels with specific uncertainty characterizations are defined, to express the sensitivity of the studied quantity with respect to deviations in a specific indicator. However, information from expert judgement, which is frequently used in MFA input data as well, is a special case. Here, the reliability of the information provided by the expert is assessed using only one indicator, of which the criteria are presented in the bottom row of Table 2, reflecting the transparency and consistency of the estimate and the information basis available to the expert.

After completing the qualitative data assessment, quantitative uncertainties associated with the data quality indicators are assigned. However, this step remains arbitrary to some extent, especially if no empirical data are available to validate estimates. In this study, the data are assumed to be normally distributed, and the uncertainties are quantified by coefficients of variation (CV, standard deviation divided by mean) which follow an exponential function (i.e. uncertainty increases exponentially with worse scores), as presented in Table 3. After assigning the scores for the respective indicators, the quantitative uncertainties can be taken from Table 3, whereas the overall uncertainty of the data point is determined by aggregating the individual CVs, according to Eq. (1). This total CV value is then used to characterize the uncertainty of the model input data. For expert estimates, the CVs for the individual scores are defined as aggregate, so that the overall uncertainty value of the estimate is consistent with the corresponding uncertainty levels derived for data taken from literature.

$$CV_{total} = \sqrt{CV_{reliability}{}^2 + CV_{completeness}{}^2 + CV_{geogr.corr.}{}^2 + CV_{temp.corr.}{}^2 + CV_{other\ corr.}{}^2} \quad (1)$$

Table 2
Data quality indicators and qualitative evaluation criteria, adapted from Laner et al. (2015b).

Indicator	Score: 1	Score: 2	Score: 3	Score: 4
Reliability	Methodology of data generation well documented and consistent, peer-reviewed data.	Methodology of data generation is described, but not fully transparent; no verification.	Methodology not comprehensively described, but principle of data generation is clear; no verification.	Methodology of data generation unknown, no documentation available.
Completeness	Value includes all relevant processes/flows in question.	Value includes quantitatively main processes/flows in question.	Value includes partial important processes/flows, certainty of data gaps.	Only fragmented data available; important processes/mass flows are missing.
Temporal correlation	Value relates to the right time period.	Deviation of value 1–5 years.	Deviation of value 5–10 years.	Deviation more than 10 years.
Geographical correlation	Value relates to the studied region.	Value relates to similar socio-economical region (GDP, consumption pattern).	Socio-economically slightly different region.	Socio-economically very different region.
Other correlation	Value relates to the same product, the same technology, etc.	Values relate to similar technology, product, etc.	Values deviate from technology/product of interest, but rough correlations can be established based on experience or data.	Values deviate strongly from technology/product of interest, with correlations being vague and speculative.
Expert estimate	Formal expert elicitation with (empirical) database—transparent procedure and fully informed experts on the subject.	Structured expert estimate with some empirical data available or using transparent procedure with informed experts.	Expert estimates with limited documentation and without empirical data available.	Educated guess based on speculative or unverifiable assumptions.

Table 3
Quantitative uncertainties expressed as coefficients of variation for the data quality indicators, adapted from Laner et al. (2015b).

Data quality indicator	Sensitivity level	Score: 1	Score: 2	Score: 3	Score: 4
		Coefficient of variation (CV, in %)			
Reliability	–	2.3	6.8	20.6	62.3
Completeness/temporal/geographic/other correlation	High	0.0	4.5	13.7	41.3
	Medium	0.0	2.3	6.8	20.6
	Low	0.0	1.1	3.4	10.3
Expert estimate	–	4.5	13.7	41.3	124.6

Using normally distributed data has the advantage that error propagation can be conducted analytically, to determine the uncertainty of calculated values based on the Gaussian law of error propagation. Moreover, this concept is implemented in the software STAN, where data reconciliation is used to solve inconsistent datasets through minimization of the sum of squared errors. For a more detailed discussion on this uncertainty characterization method and various alternative implementations, the reader is referred to Laner et al. (2015b).

3. Results and discussion

3.1. Plastics budget for Austria in 2010

The results of the Austrian plastics budget for 2010 are presented in Fig. 3 for the overall system, and in the Supplementary data (Appendix A) for the subsystem 'trade and distribution of final products' (see Fig. S-1). All subsequently quoted quantities are given by the mean value (with two significant digits) and the relative standard deviation. A detailed overview of the MFA input data and the uncertainty characterization is included in the Supplementary data as well (see Tables S-1 to S-3). The results show that the total domestic primary production of polymers amounted to 1100 kt/a ± 2% (1 kt is 1 kilotonne, or 1 Gg). This production was composed of polyolefins (79%), polystyrenes (16%) and resins (5%), reflecting the dominant position of polyolefins in world plastics demand (PlasticsEurope, 2015). Next, major flows of primary polymers were imported and exported (both 1400 kt/a ± 2%). The size of these flows shows that Austria is linked strongly with foreign producers and consumers, and an approximately even trade balance was established. Furthermore, 96 kt/a ± 2% and 49 kt/a ± 2% of rubbers were imported and exported, respectively. These exported polymers and rubbers additionally encompassed about 18 kt/a ± 28% of additives. Balancing these flows means that 1300 kt/a ± 4% of primary plastics were available in Austria for the manufacturing and preparation industries. From this process, there was a positive trade balance concerning semi-finished products (420 kt/a ± 4% imported and 490 kt/a ± 4% exported). An additional additives amount of 24 kt/a ± 168% was introduced as well, whereas 130 kt/a ± 25% of production wastes were generated. This finally resulted in 1100 kt/a ± 5% of final products domestically produced in Austria.

In the consumption stage, first trade of final products occurs. Two consumption sectors reported a net export balance, namely the packaging and non-plastics sectors, which totaled 140 kt/a ± 37%. All other sectors showed a net import of products, which added up to 290 kt/a ± 4%, causing an overall negative trade balance of around 150 kt/a ± 37%. Balancing the domestically produced final products with their imports and exports resulted in a total Austrian plastics consumption of 1300 kt/a ± 5%. Looking at the distribution over the consumption sectors, it is clear that the use of packaging material contributed the most to plastics consumption, with a share of 22%. Next, the building and construction (17%), non-plastics (14%), others and transport (both 12%) sectors had major consumption contributions as well. Only minor shares of plastics products were consumed in the other sectors.

The stock changes were calculated as the differences between the input and output of the respective consumption sectors, and are related to the in-use periods of the products (product lifetimes). Overall, the total change in in-use stock amounted to

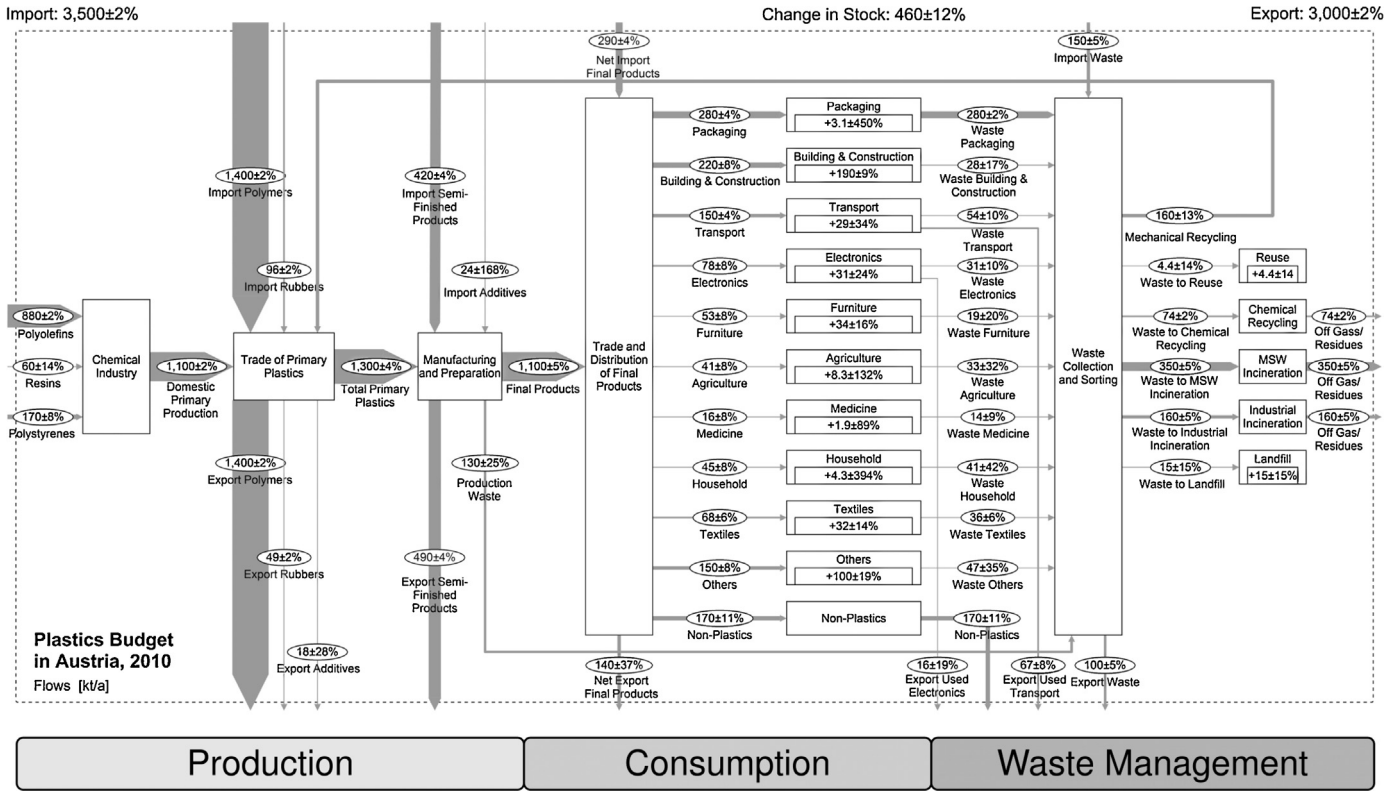

Fig. 3. Results of the plastics budget for Austria in 2010 (exports of plastics include also their chemical or thermal utilization). Flow values are given by their mean value (two significant digits) and relative standard deviation.

+440 kt/a ± 12%, which means that the net increase of the total size of the stocks was about one third of the incoming amount of consumer goods. Although stocks were increasing in all sectors, wide ranges between the sectors are clear. Sectors with products with a short lifetime, such as the packaging and medicine sectors, only had small stock increases (+3 kt/a ± 450% and +2 kt/a ± 89%, respectively), whereas a very large buildup in stock (+190 kt/a ± 9%) took place in the building and construction sector. Especially the latter sector was relevant, as it was responsible for roughly half of the buildup in stock in 2010, and the use of plastics in this sector has been strongly increasing over the past decades. Furthermore, many potentially harmful additives (e.g. brominated flame retardants such as polybrominated diphenyl ethers PBDEs and hexabromocyclododecane HBCDD) were included in building products before being regulated (e.g. by the Stockholm Convention), and are still present in the current building stock (Nie et al., 2015; Vyzinkarova and Brunner, 2013).

Two sectors had important exports of used products as well, namely transport (67 kt/a ± 8%) and electronics (16 kt/a ± 19%), and these then became waste outside of the system boundaries of this study. Furthermore, as mentioned in Section 2.2, the non-plastics sector was not further considered from this point on. The total post-consumer waste production from the consumption stage added up to 580 kt/a ± 4%, or 70 kg/cap·a, of which almost half was caused by the packaging sector (48%). Much lower and similar amounts of waste were produced by all other sectors, with a range from 2% to 9%. Further waste inputs were generated from the manufacturing and preparation stage as production waste (130 kt/a ± 25%), as well as 150 kt/a ± 5% imported and 100 kt/a ± 5% exported amounts of plastic wastes. This brought the total production of waste to be collected and treated by the waste management system to 760 kt/a ± 3%, or 91 kg/cap·a.

Only a minor amount (1%) of this total waste stream was reused, and this was almost exclusively comprised of the reuse of textile products. Furthermore, about 21% of the waste flow was converted by mechanical recycling to re-granulate, whereas residues from this process to be incinerated or landfilled were diverted to the respective treatment operations. Taking into account that it can be assumed that production waste and traded waste will be largely recycled mechanically, it appears that only a limited amount of post-consumer waste from Austria was processed here. Next, 10% was chemically recycled by acting as a reducing agent in the steel industry. Incineration of MSW accounted for 46% of the processed waste stream, whereas the cement industry used 21% for industrial incineration. Finally, the residual 2% was landfilled for final disposal. It thus follows that 98% of plastic wastes were treated by a useful end-of-life operation, be it recycling or thermal utilization. This is much higher than the European average, where around 50% of waste plastics are still landfilled (EC, 2015).

3.2. Uncertainty analysis

Analyzing the uncertainties obtained for the various flows in the system reveals that the data qualities, and therefore the associated uncertainty values, vary over a wide range. As an overall trend, it can be observed that data uncertainties are deemed to be low in the production stage, increase for the consumption flows and further for the generated waste, and again decrease in the case of the waste management treatment operations. This trend is in line with a general tendency of databases for MFAs becoming weaker when moving downstream the material flows, as recognized by Schwab et al. (2016). There are two sources of uncertainty for a certain plastics flow: first, the mass flow itself, and second, the plastics content. The latter is in general the major contributor, due to the huge vari-

ety in products and applications of plastics, and the relatively low availability of information on plastics contents.

In the production stage, generally good data are available, with reliable information on the production of polyolefins from expert information, and on the trade flows of plastics from the Austrian statistical office, with uncertainty ranges of 2–4%, whereas the data quality of the expert judgement on the production of the other polymers and production waste is increasingly lower (8–25%). However, the flows of additives are an exception, as detailed information on additives contents of plastics is generally not accessible, considering these data are usually treated confidentially due to the large influence on the properties of the plastics. Therefore, the data on the exports and imports of additives are (highly) speculative, which is reflected in uncertainty values of 28% and 168% respectively.

The reliability of the data on the trade of final products is largely dependent on the availability of product compositions. Relatively good information is available for the packaging, transport, furniture, household, textiles and electronics sectors, with increasing uncertainties ranging from 5% to 10%. On the other hand, the building and construction, medicine (both 14%), others (22%), and especially non-plastics (77%) sectors have higher uncertainties, where the high value for the latter sector is caused by the high variability and low data availability of plastics contents for these applications. Next, the consumption of plastics is described with relatively low uncertainties, varying from 4% to 11%.

The most challenging flows to quantify were the waste production flows from the consumption sectors, which is reflected by the comparably higher uncertainty values. However, it is clear that sectors with a well-established and strong waste regulation, such as packaging (2%), transport and electronics (both 10%), in general have lower associated uncertainties, due to extensive reporting requirements put in place by the authorities. In contrast, for other sectors less information is available, such as for the furniture (20%), agriculture (32%), others (35%), and household (42%) sectors. In case of the waste treatment operations, the data on chemical recycling and thermal waste utilization were of high quality (2–5%), as relevant and detailed information was available, whereas the results for reuse, mechanical recycling and landfill were more subject to estimations (13–15%).

3.3. Analysis of the data reconciliation

In this study, it was possible to perform data reconciliation due to the presence of an over-determined system of equations. This means that some input values would not have been required to solve the equation system and thus quantify the overall model, but can be used to reduce the uncertainties of the flow estimates. For instance, the waste collection and sorting process was quantified from two sides, i.e. both the amount of waste produced and the amount of waste treated were determined. Moreover, the mass of packaging plastics consumed was derived from the fraction of total plastics consumption in Germany (Consultic, 2012), as well as from data specific for Austria (Hauer et al., 2012). This surplus information together with the uncertainty estimates were subsequently used to reconcile data contradictions and reduce the uncertainty.

The effects of this process are shown in Fig. 4, where the relative changes made by the data reconciliation procedure are presented for the 45 flows for which input data were defined. Furthermore, the error bars around the 0% line represent the input uncertainty of the flows expressed as the standard deviation. Only six of the flows were changed more than 1%, of which only two were altered more than 4%. These flows are F1.12, the import of additives, and F4.22, the net trade of non-plastics, which are also the flows with by far the highest input uncertainty (as mentioned in Section 3.2, and shown in Fig. 4 by the error bars). The error bars also reveal that none of the reconciled flows were altered more than was expected from the uncertainty characterization of the input data, as the mean value changes due to reconciliation all stay well within one standard deviation.

3.4. Comparison with related studies

Quantitatively comparing studies on the flows and processes of plastics in varying regions and time periods can be challenging, as the material stream is very diverse and complex, resulting in differing scopes regarding materials and processes across the studies. Fig. 5 nevertheless compares the results of two previous studies for Austria with similar scope, namely for 1994 (Fehringer and Brunner, 1997) and for 2004 (Bogucka and Brunner, 2007) with those for this study, regarding plastics consumption, waste production and waste treatment.

These results indicate that the growth of plastics consumption was strong between 1994 and 2004, but over the 2004–2010 period, no net growth arose. This is confirmed by statistics from the European plastics industry (see e.g. PlasticsEurope (2010)) and is caused by the drastic decrease of demand after the 2008 economic crisis. Regarding the amount of waste generated however, the three studies show large differences. This may be attributed the use of different methods and sources for data collection. Fehringer and Brunner (1997) used expert judgement to estimate the waste amount, whereas Bogucka and Brunner (2007) used the life spans of the consumed products to calculate this value. Furthermore, in the present study a bottom-up analysis was used (i.e. individual waste flows from the various consumption sectors were added), so these results are to be treated as a lower limit.

In Fig. 5, the delivery of waste plastics to different waste treatment options is compared for the past studies and the present one. To be consistent among studies, the mechanical recycling flows in Fig. 5 represent the volume of actually recycled re-granulate. Furthermore, chemical recycling and thermal utilization were put together in one category, as chemical recycling was practiced only after 2004. The results indicate a trend of moving away from landfilling, towards increased mechanical and chemical recycling and thermal utilization. The increased mechanical recycling rates are in line with the legal targets in this area (see e.g. EC (2015)), whereas a ban on landfilling waste with an organic carbon content higher than 5% from 2004 onwards caused a rapid decline of the amount of waste plastics being deposited (Deponieverordnung, 2004), highlighting the success of this policy change.

Moreover, the European plastics industry reports statistics on the recovery rates of waste plastics in Europe as well, which for Austria in 2010 for post-consumer waste add up to 27% for mechanical and chemical recycling, 70% for thermal utilization, while the remaining 3% is landfilled (PlasticsEurope, 2011). Although these statistics can be compared only to a limited extent (post-consumer versus total waste produced; waste amount delivered to mechanical recycling versus actually recycled re-granulate), they are in the same range as the results of this study (31%, 67%, and 2% respectively). Specifically for plastics packaging waste, the European statistical office reports 32 kg/cap of domestic generation in 2010 (Eurostat, 2016), which is slightly lower than the 34 kg/cap obtained in this study.

Looking at the situation of plastics waste management in other countries, Bogucka and Brunner (2007) report that in Poland in 2004, 91% of the waste flows went straight to landfill, and only 5% and 3% were delivered to mechanical recycling and thermal utilization, respectively. In India, Mutha et al. (2006) estimated that in 2000/2001 around 47% of plastic waste were recycled, while the rest was disposed of in a controlled or uncontrolled fashion. These high recycling rates were achieved by the informal sector, because of the fairly low cost of labor and high demand for second-grade products. Finally, around 12% of waste PET in the US in 2007 was

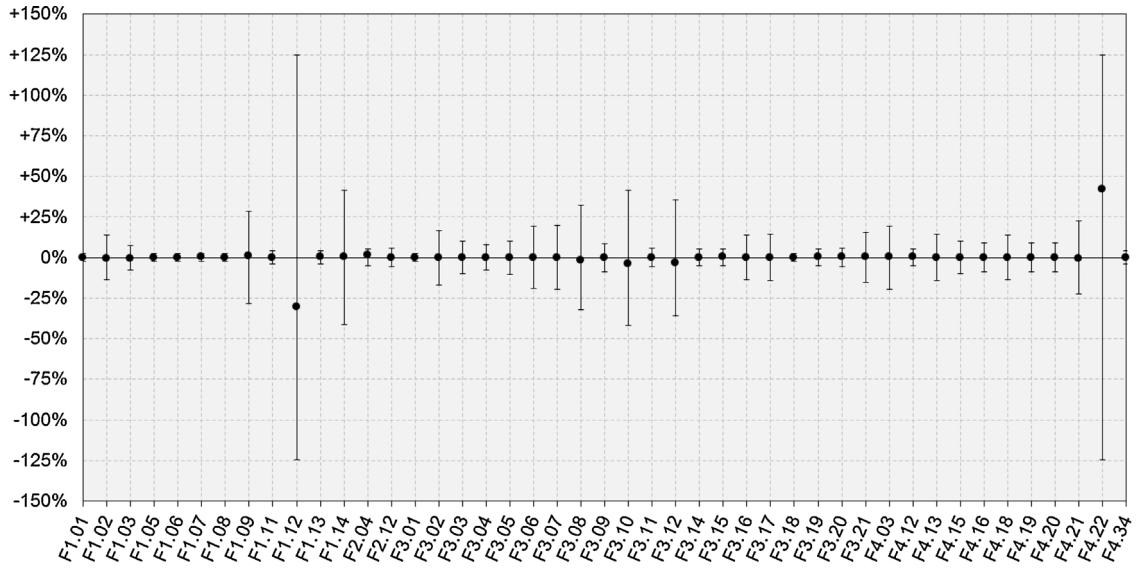

Fig. 4. Relative changes of the original input data by data reconciliation (black dots), and the input uncertainty of these input data expressed as the standard deviation (error bars). The flow numbers correspond to those presented in Fig. 1 (F1–F3) and Fig. 2 (F4).

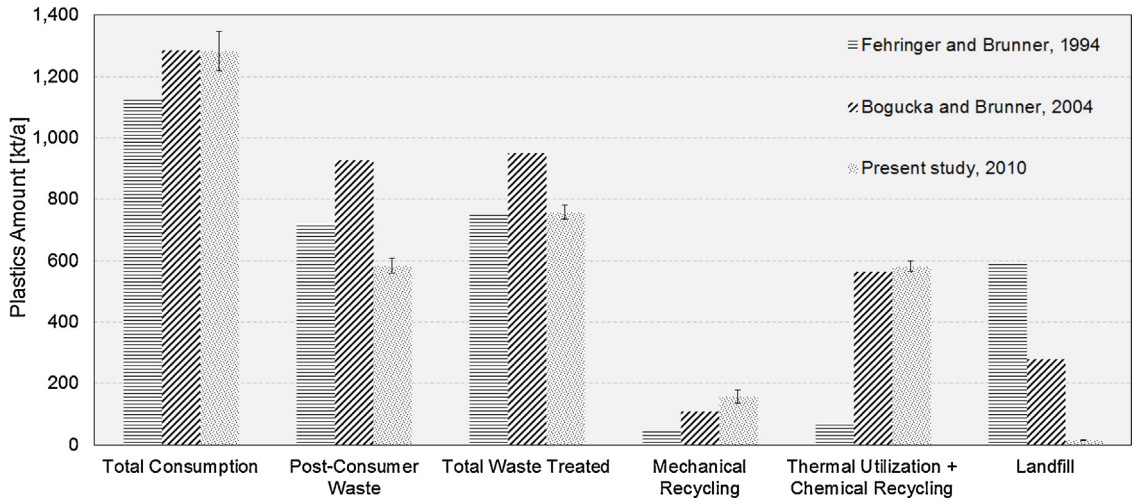

Fig. 5. Comparison of the results for plastic consumption, waste production and waste treatment in Austria from two previous studies (Bogucka and Brunner, 2007; Fehringer and Brunner, 1997) and the present study, for which the standard deviation is given.

recycled according to Kuczenski and Geyer (2010), whereas 81% of the waste flow was landfilled, and the remaining amount was sent to unspecified recovery.

3.5. Recent developments

As this study presents a snapshot of the plastic flows in Austria for 2010, it is relevant to check the validity and representativeness of the model with respect to recent developments of the plastics budget. Other than for the trade statistics, which have yearly editions, most data sources are updated infrequently, leading to substantial lag phases between real phenomenon and reported quantity. However, for some flows across the life-cycle, a comparison of the 2010 plastic flows with values for 2013 is possible.

This comparison shows that in the production and consumption stages, the changes from 2010 to 2013 are relatively small. This is illustrated by the primary production of polyolefins, increasing by 5% (Kravanja, 2016), the imports and exports of semi-finished products, which increased by 11% and 7% respectively, and the imports and exports of products from the packaging, furniture and others sectors, with changes ranging from 1 to 7% (Hauer et al., 2015; United Nations, 2016). Also, the distribution pattern of plastics over the consumption sectors varies little, with relative changes ranging from −6 to 6% (Consultic, 2014), and the consumption of packaging plastics increased by 12% (Hauer et al., 2015). Furthermore, in the waste management stage the amount of packaging waste increased by 9% (Hauer et al., 2015), and the amount of electronic waste collected rose by 4% (EAK, 2014). The treatment of the waste streams shows more variation however, illustrated by an increase of imported and exported waste streams by 33% and 72% respectively (United Nations, 2016), a decrease of 22% for chemical recycling (Trinkel, 2015), and an increase of 21% for industrial incineration (Mauschitz, 2014).

Thus, the plastics budget model is temporally quite robust for primary production, consumption and waste generation, but more variation can be observed with respect to the treatment of waste plastic flows. The waste management stage is therefore more responsive to changes in the policy background (see also Section 3.4), the supply of waste streams and the demand for secondary products. Consequently, if detailed knowledge about plastic waste

generation is available, as through this study, this stage can be receptive to developments to increase its efficiency and sustainability.

4. Conclusions

In this study, the plastics budget of Austria in 2010 is quantitatively described in detail using an MFA model, starting from primary production of polymers, over consumption of the products, up to and including processing the waste plastics. This builds up information on the main flows and processes of the plastics budget, to determine the extent to which resources are utilized effectively and on the potential to optimize this in the future. The quality of the data is assessed using a comprehensive and consistent uncertainty analysis method that characterizes each data source according to various indicators and quantifies the individual uncertainty of the data points.

The results indicate that about 1.1 million tonnes of primary plastics were produced in Austria in 2010, and that, after trade of primary plastics and semi-finished products, about the same amount of domestically produced plastic products was delivered to the consumption market. The latter stage consists of eleven consumption sectors, of which packaging (22%), building and construction (17%), non-plastic applications (14%), transport and others (both 12%) were most important from a mass perspective. During the use phase, about one third of all plastic products contributed to the anthropogenic stock, which increased in every sector, but with large differences between the sectors. Of the post-consumer waste amount, around half was produced by the packaging sector. The total waste amount, i.e. post-consumer waste together with production waste and net imported streams, was mainly incinerated (MSW: 46%; industrial: 21%), followed by mechanical (21%) and chemical (10%) recycling, whereas only minor amounts were reused (1%) or landfilled (2%).

Based on these results for 2010, it can be concluded that Austria has a substantial plastics industry that is strongly connected with international producers and consumers, with an even trade balance for primary plastics, a positive balance for semi-finished products, and a negative balance for final products. Furthermore, the stocks in the use phase increased by about one third of all consumed plastics in 2010, indicating the continuing growth of anthropogenic stocks. It is therefore important to gain knowledge regarding the size and composition (e.g. regarding potentially hazardous additives) of the stocks that are currently being built up, as well as the stocks that were built up in the past, to enable the appropriate handling of the challenges the waste streams arising in future will pose. Finally, the vast majority of plastics are utilized generating a useful product or service, but potential certainly exists for increasing the mechanical recycling rate of post-consumer wastes, especially for those sectors currently lacking appropriate collection, as a measure for increased resource efficiency.

This detailed overview of the flows of plastics in Austria enables a more thorough understanding of the resource potential that is present, as well as highlighting losses of resources in the system. Moreover, considering the quality of the data sources through the uncertainty characterization allows for evaluating the overall data situation. In this study, the main data gaps were related to the amount of additives incorporated, the plastics contents of the products, the distribution of these products over the consumption sectors, waste production from sectors without a well-established regulatory framework (especially the agriculture, household, and others sectors), and the amount of post-consumer plastics that are mechanically recycled (because of the large amounts of traded and production waste). Improved data availability in these areas would therefore further improve the robustness of the results.

Furthermore, the results allow for future developments to be (roughly) projected as well. From the results of the consumption stage, it is clear that the stocks of all consumption sectors are growing. This means that, even if the consumption of plastics would reach a steady state level, the waste amounts would nevertheless continue to rise for decades in sectors with long product lifetimes. It therefore follows that further capacities for mechanical and chemical recycling, as well as for thermal utilization, are needed to cope with the treatment of increasing amounts of waste plastics. Which priorities with regards to the different waste treatment operations are to be set in Austria depends on ecological and economic considerations, and is subject to further investigation.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.resconrec.2016.10.017.

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Supplementary data

Comprehensive analysis and quantification of national plastic flows: the case of Austria

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Supporting Information

Contents

1 Results for the subsystem 'trade and distribution of final products'

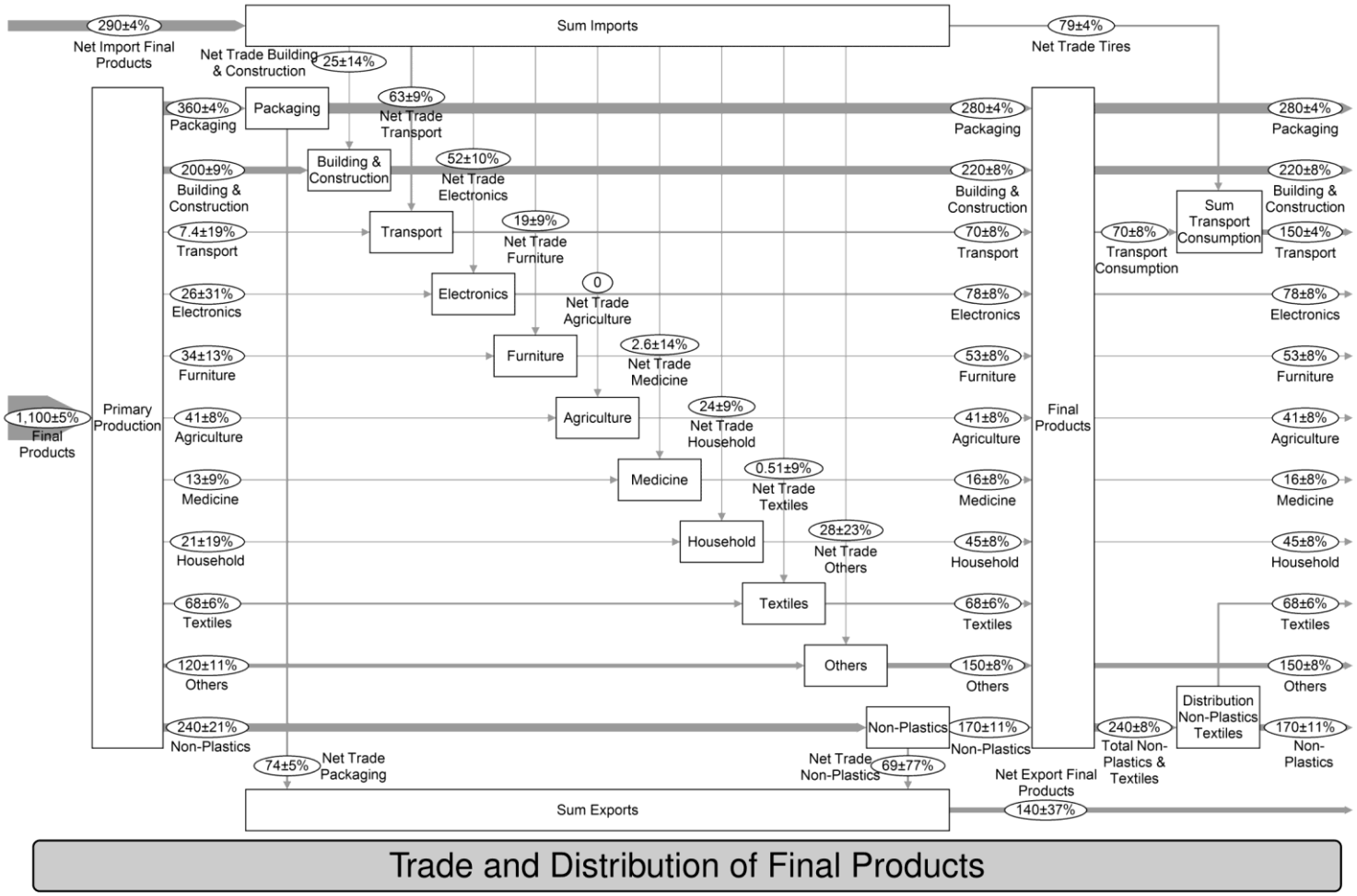

Figure S-1: Results of the plastics budget for Austria in 2010 for the subsystem 'trade and distribution of final products', as shown in Figure 3 of the main article (data given in kt/a).

2 Detailed overview and uncertainty characterization of the input data

Table S-1: Detailed overview of the input data of the **production stage** of the model and uncertainty characterization of each data point. The values of the scores correspond to the coefficients of variation shown in Table 3 of the main article. Urel: relative uncertainty, which is calculated according to the method outlined in Section 2.4 of the main article.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F1.01	**Chemical Industry**	Polyolefins	**877,252**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[1]
F1.02		Resins	**60,000**						13.67%	**13.67%**	[2]
F1.03		Polystyrenes	**175,000**	6.84%	2.26%	2.26%	0.00%	0.00%		**7.55%**	[3]
F1.05	**Trade of Primary**	Import Polymers	**1,387,510**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[4]
F1.06	**Plastics**	Import Rubbers	**96,040**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[4]
F1.07		Export Polymers	**1,363,860**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[4]
F1.08		Export Rubbers	**48,734**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[4]
F1.09		Export Additives	**17,418**	20.64%	13.67%	0.00%	0.00%	13.67%		**28.28%**	[1]
F3.16		Mechanical Recycling	**157,000**						13.67%	**13.67%**	
F1.11	**Manufacturing &**	Import Semi-Finished Products	**416,760**							**4.08%**	
	Preparation	*Mass*		*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*2.26%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F1.12		Import Additives	**35,000**						124.64%	**124.64%**	[6; 7]
F1.13		Export Semi-Finished Products	**483,205**							**4.08%**	
		Mass		*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*2.26%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F1.14		Production Waste	**129,135**						41.28%	**41.28%**	[8]

Table S-2: Detailed overview of the input data of the **consumption stage** of the model and uncertainty characterization of each data point. The values of the scores correspond to the coefficients of variation shown in Table 3 of the main article. Urel: relative uncertainty, which is calculated according to the method outlined in Section 2.4 of the main article.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F4.03	**Trade and Distribution of Final Products**	Transport	**7,412**							**19.43%**	
		Car	***152***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass		*2.26%*	*2.26%*	*2.26%*	*1.13%*	*0.00%*			[10; 11]
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Truck	***3,773***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass		*20.64%*	*13.67%*	*2.26%*	*1.13%*	*0.00%*			[12; 13]
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Bus	***20***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass		*2.26%*	*4.53%*	*2.26%*	*0.00%*	*0.00%*			[14]
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Tractor	***2,304***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass							*41.28%*		
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Trailer	***993***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass							*41.28%*		
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Motorcycle	***150***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass		*2.26%*	*2.26%*	*2.26%*	*1.13%*	*0.00%*			[15]
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Scooter	***0***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass		*2.26%*	*2.26%*	*2.26%*	*1.13%*	*0.00%*			[15]
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Bicycle	***20***	*2.26%*	*0.00%*	*4.53%*	*0.00%*	*0.00%*			[9]
		Unit Mass							*41.28%*		
		Plastics Content		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
F4.12		Net Trade Packaging	**74,006**	2.26%	4.53%	0.00%	0.00%	0.00%		**5.06%**	[16]
F4.13		Net Trade Building & Construction	**24,755**							**14.27%**	
		Mass		*2.26%*	*13.67%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*2.26%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F4.15		Net Trade Electronics	**52,282**							**9.96%**	
		Mass		*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*6.84%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5; 17]

Table S-2: Continued.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F4.16		Net Trade Furniture	**18,948**							**8.88%**	
		Mass		*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*6.84%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F4.18		Net Trade Medicine	**2,565**	2.26%	13.67%	0.00%	0.00%	0.00%		**13.86%**	[4]
F4.19		Net Trade Household	**23,799**							**8.88%**	
		Mass		*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*6.84%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F4.20		Net Trade Textiles	**506**							**8.88%**	
		Mass		*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*6.84%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F4.21		Net Trade Others	**27,893**							**22.57%**	
		Mass Commodity Guide	***32,382***	*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Content Commodity Guide		*6.84%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5; 18]
		Mass Other	***-4,489***	*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content Other							*124.64%*		
F4.22		Net Trade Non-Plastics	**48,639**							**124.66%**	
		Mass		*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content							*124.64%*		[19]
F4.34		Net Trade Tires	**79,142**							**4.08%**	
		Mass		*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[4]
		Plastics Content		*2.26%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
P4.15		Share Packaging	**23.88%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Building & Construction	**18.40%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Transport	**5.84%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Electronics	**6.47%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Furniture	**4.39%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Agriculture	**3.40%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Medicine	**1.33%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Household	**3.73%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Others	**12.56%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
P4.15		Share Non-Plastic Products + Textiles	**20.00%**	2.26%	0.00%	2.26%	4.53%	0.00%		**5.55%**	[20]
F2.04		Packaging	**279,228**	2.26%	4.53%	0.00%	0.00%	0.00%		**5.06%**	[16]
F2.12		Textiles	**68,126**	2.26%	0.00%	4.53%	2.26%	0.00%		**5.55%**	[21]

Table S-3: Detailed overview of the input data of the **waste management stage** of the model and uncertainty characterization of each data point. The values of the scores correspond to the coefficients of variation shown in Table 3 of the main article. Urel: relative uncertainty, which is calculated according to the method outlined in Section 2.4 of the main article.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F3.01	**Waste Production**	Waste Packaging	**280,777**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[16]
F3.02		Waste Building & Construction	**28,194**							**16.84%**	
		Building Site Waste	*9,506*	*6.84%*	*0.00%*	*2.26%*	*0.00%*	*0.00%*			[22]
		Plastics Content Building Site Waste		*6.84%*	*13.67%*	*41.28%*	*1.13%*	*13.67%*			[23]
		Demolition Waste	*6,968*	*6.84%*	*4.53%*	*2.26%*	*0.00%*	*0.00%*			[22]
		Plastics Content Demolition Waste		*2.26%*	*4.53%*	*2.26%*	*4.53%*	*0.00%*			[24]
		Separately Collected	*5,583*	*2.26%*	*13.67%*	*0.00%*	*0.00%*	*0.00%*			[25; 26]
		Bulky Waste	*6,138*	*2.26%*	*13.67%*	*13.67%*	*2.26%*	*0.00%*			[27]
		Plastics Content Bulky Waste		*6.84%*	*0.00%*	*2.26%*	*1.13%*	*0.00%*			[5]
F3.03		Waste Transport	**54,072**							**9.95%**	
		Dismantling Shredder	***2,138***	*2.26%*	*4.53%*	*2.26%*	*0.00%*	*0.00%*			[28]
		Dismantling Garage	***3,500***						*124.64%*		
		EoL Vehicles Shredder	***12,641***	*2.26%*	*4.53%*	*2.26%*	*0.00%*	*0.00%*			[28]
		Waste Tires	***35,793***	*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[29]
		Plastics Content Waste Tires		*6.84%*	*0.00%*	*1.13%*	*0.00%*	*0.00%*			[22]
F3.04		Export Used Transport	**66,852**							**7.69%**	
		Cars	***27,162***	*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[28]
		Plastics Content Cars		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Trucks	***29,835***	*2.26%*	*4.53%*	*0.00%*	*0.00%*	*0.00%*			[11; 30; 31]
		Plastics Content Trucks		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		Tires	***9,855***						*41.28%*		[29; 32]
		Plastics Content Tires		*6.84%*	*0.00%*	*1.13%*	*0.00%*	*0.00%*			[22]
F3.05		Waste Electronics	**31,345**							**10.32%**	
		Collected	***19,045***	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[33]
		Plastics Content Collected							*13.67%*		[5; 17]
		Residual Waste	***2,876***	*2.26%*	*0.00%*	*2.26%*	*2.26%*	*0.00%*			[34]
		Plastics Content Residual Waste							*13.67%*		[5; 17]
		Scrap Metal	***9,425***						*13.67%*		[35]
		Plastics Content Scrap Metal							*13.67%*		[5; 17]
F3.06		Export Used Electronics	**15,806**							**19.34%**	
		Mass							*13.67%*		[35]
		Plastics Content							*13.67%*		[5; 17]
F3.07		Waste Furniture	**18,955**	2.26%	13.67%	13.67%	2.26%	0.00%		**19.60%**	[27]

Table S-3: Continued.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F3.08		Waste Agriculture	**33,281**							**32.17%**	
		Residual Waste	***14,301***	*2.26%*	*0.00%*	*2.26%*	*4.53%*	*41.28%*			[34]
		Separately Collected	***18,980***	*6.84%*	*0.00%*	*20.64%*	*4.53%*	*41.28%*			[36]
F3.09		Waste Medicine	**14,076**							**8.67%**	
		Inside	***13,481***	*2.26%*	*0.00%*	*2.26%*	*0.00%*	*0.00%*			[22]
		Composition Inside		*2.26%*	*0.00%*	*2.26%*	*4.53%*	*0.00%*			[37]
		Plastics Content Inside		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
		In- and Outside	***322***	*2.26%*	*0.00%*	*2.26%*	*0.00%*	*0.00%*			[22]
		Plastics Content In- and Outside							*124.64%*		
		Syringes	***273***	*2.26%*	*0.00%*	*2.26%*	*0.00%*	*0.00%*			[22]
		Plastics Content Syringe		*2.26%*	*4.53%*	*2.26%*	*1.13%*	*0.00%*			[5]
F3.10		Waste Household	**42,273**	2.26%	0.00%	2.26%	4.53%	41.28%		**41.65%**	[34]
F3.11		Waste Textiles	**36,414**							**5.79%**	
		Clothes Collected	***9,490***	*2.26%*	*0.00%*	*6.84%*	*4.53%*	*0.00%*			[38]
		Plastics Content Clothes Collected		*2.26%*	*0.00%*	*2.26%*	*3.42%*	*0.00%*			[21]
		Clothes Residual Waste	***26,029***	*2.26%*	*0.00%*	*2.26%*	*4.53%*	*0.00%*			[34]
		Plastics Content Clothes Residual Waste		*2.26%*	*0.00%*	*2.26%*	*3.42%*	*0.00%*			[21]
		Bulky Waste	***895***	*2.26%*	*4.53%*	*13.67%*	*2.26%*	*0.00%*			[27]
		Plastics Content Bulky Waste							*13.67%*		[21]
F3.12		Waste Others	**48,830**							**35.66%**	
		Bulky Carpets	***7,762***	*2.26%*	*4.53%*	*13.67%*	*2.26%*	*0.00%*			[27]
		Plastics Content Bulky Carpets		*2.26%*	*0.00%*	*13.67%*	*6.84%*	*0.00%*			[39]
		Bulky Leather, Rubber	***1,880***	*2.26%*	*4.53%*	*13.67%*	*2.26%*	*0.00%*			[27]
		Plastics Content Bulky Leather, Rubber							*13.67%*		
		Bulky Sport, Leisure	***489***	*2.26%*	*4.53%*	*13.67%*	*2.26%*	*0.00%*			[27]
		Plastics Content Waste Sport, Leisure		*2.26%*	*0.00%*	*2.26%*	*2.26%*	*13.67%*			[5]
		Residual Plastics	***28,603***	*2.26%*	*0.00%*	*2.26%*	*4.53%*	*41.28%*			[34]
		Residual Others	***10,095***						*124.64%*		[34]
F1.14		Production Waste	**129,135**						41.28%	**41.28%**	[8]
F3.14		Import Waste	**146,132**	2.26%	4.53%	0.00%	0.00%	0.00%		**5.06%**	[4]
F3.15		Export Waste	**100,259**	2.26%	4.53%	0.00%	0.00%	0.00%		**5.06%**	[4]

Table S-3: Continued.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F3.16	**Waste Collection**	Mechanical Recycling	**157,000**						13.67%	**13.67%**	
F3.17	**and Sorting**	Waste to Reuse	**4,399**							**14.43%**	
		Electronics	***319***	*2.26%*	*13.67%*	*0.00%*	*0.00%*	*0.00%*			[33]
		Plastics Content Electronics							*13.67%*		[5; 17]
		Textiles	***4,081***	*2.26%*	*13.67%*	*2.26%*	*4.53%*	*0.00%*			[40]
		Plastics Content Textiles		*2.26%*	*0.00%*	*2.26%*	*3.42%*	*0.00%*			[21]
F3.18		Waste to Chemical Recycling	**73,832**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[41]
F3.19		Waste to MSW Incineration	**346,628**	2.26%	0.00%	4.53%	0.00%	0.00%		**5.06%**	[42]
F3.20		Waste to Industrial Incineration	**159,542**	2.26%	0.00%	2.26%	0.00%	4.53%		**5.55%**	[43]
F3.21		Waste to Landfill	**15,000**	6.84%	13.67%	0.00%	0.00%	0.00%		**15.29%**	[44]

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Paper II

Circular economy of plastic packaging: Current practice and perspectives in Austria

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Circular economy of plastic packaging: Current practice and perspectives in Austria

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ABSTRACT

Plastics, especially from packaging, have gained increasing attention in waste management, driving many policy initiatives to improve the circularity of these materials in the economy to increase resource efficiency. In this context, the EU has proposed increasing targets to encourage the recycling of (plastic) packaging. To accurately calculate the recycling rates, detailed information on the flows of plastic packaging is needed. Therefore, the aim of this paper is to quantitatively and qualitatively investigate the waste management system for plastic packaging in Austria in 2013 using material flow analysis, taking into account the used product types and the polymer composition. The results show that 300,000 ± 3% t/a (35 kg/cap·a) of waste plastic packaging were produced, mainly composed of large and small films and small hollow bodies, including PET bottles. Correspondingly, the polymer composition of the waste stream was dominated by LDPE (46% ± 6%), PET (19% ± 4%) and PP (14% ± 6%). 58% ± 3% was collected separately, and regarding the final treatment, 26% ± 7% of the total waste stream was recovered as re-granulates, whereas the rest was thermally recovered in waste-to-energy plants (40% ± 3%) and the cement industry (33% ± 6%). The targets set by the EU and Austria were reached comfortably, although to reach the proposed future target major technological steps regarding collection and sorting will be needed. However, the current calculation point of the targets, i.e. on the input side of the recycling plant, is not deemed to be fully in line with the overall objective of the circular economy, namely to keep materials in the economy and prevent losses. It is therefore recommended that the targets be calculated with respect to the actual output of the recycling process, provided that the quality of the output products is maintained, to accurately assess the performance of the waste management system.

1. Introduction

Plastics are widely recognized to have an ever increasing importance in waste management. They have become one of the most used materials worldwide, are often used in products with short lifespans, and pose substantial environmental problems due to the accumulation in ecosystems when disposed of improperly (Barnes et al., 2009; Gregory, 2009; Jambeck et al., 2015; Teuten et al., 2009). Ever increasing attention for these negative aspects have stimulated policy initiatives to tackle these problems, especially for plastic packaging, as this is the main application of plastics and makes up the largest share in the post-consumer plastic waste stream (PlasticsEurope, 2015; Van Eygen et al., 2017; World Economic Forum et al., 2016). These initiatives focus on the consumption side, e.g. reductions or bans on lightweight plastic carrier bags (EPC, 2015; Ritch et al., 2009), as well as on the waste management side (Sakai et al., 2011). In case of the latter, the European Union (EU) has imposed a recycling target which currently requires 22.5% of waste plastic packaging to be recycled (EPC, 2004). This target is proposed to increase by 2025 towards 55% (EC, 2015a), further underlining the ambition to increase recycling and reduce landfilling of packaging wastes. This is part of the broader initiative to increase resource efficiency and reduce resource dependency (EC, 2011), and plastics are one of the five priority areas in the EU action plan for the circular economy (EC, 2015b).

This circular economy concept, which foresees a production and consumption system where materials are circulated as wastes are re-used, recycled and recovered, has been increasingly promoted by many governments and international organizations (EEA, 2014; Ellen MacArthur Foundation, 2013; Geissdoerfer et al., 2016; Ghisellini et al., 2016; Haas et al., 2015; Lieder and Rashid, 2016; Winans et al., 2017). To measure the progress towards a circular economy, many indicators can be calculated to quantify this performance (BIO Intelligence Service et al., 2012; Hashimoto and

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Moriguchi, 2004; Haupt et al., 2016; Moriguchi, 2007). One of these indicators is the recycling rate, which is frequently used in policy documents (e.g. from the EU, see above) to quantify the amount of waste materials that is fed back into the economy. However, at which point in the waste management chain these rates are to be measured is part of ongoing discussions (EUWID, 2014). The general consensus for the EU targets seems to be to calculate the recycling rate at the gate of the recycling plant, i.e. the input to the recycling process, although this has not been clearly defined yet. This causes confusion, especially with regard to comparing the performance of different regions or countries, as it is not always clear how reported indicator values were calculated (Haupt et al., 2016).

For the calculation of these recycling rates and to draw the right conclusions on the overall environmental performance and potentially improve the system, detailed mapping of how materials move within the economy is needed (Hashimoto and Moriguchi, 2004; Preston, 2012). In the case of plastic packaging, it is of primary importance to gather information on the different polymers that constitute the waste stream, as these need to be separated in order to be recycled effectively. Furthermore, the environmental benefit achieved by recycling is different for each polymer: polyethylene terephthalate (PET) for example causes relatively high environmental impacts at primary production (Tabone et al., 2010) and has about half the heating value (Phyllis2, 2016) compared to the other major packaging polymers, making it all the more pertinent to increase high-quality mechanical recycling and avoid incineration for energy recovery. Furthermore, it is relevant to have information on the product types in the waste stream, as many collection systems and sorting processes are specific to certain product types.

Therefore, the aim of this paper is to quantitatively and qualitatively investigate the waste management system for plastic packaging in Austria with respect to polymer content and product types and 2013 as the reference year. Based on the results indicators on the performance of the system are calculated and compared with current and future policy targets. Furthermore, the potential for improvements throughout the system are identified, and the implications thereof for reaching future targets are analyzed.

2. Materials and methods

2.1. Material flow analysis

MFA is used to comprehensively assess the flows and stocks of materials through a certain system defined in space and time, thus connecting and quantifying sources, pathways and sinks of the material in question (Brunner and Rechberger, 2004). The software STAN 2.5 was used to perform the MFA calculations using a standardized method (Cencic and Rechberger, 2008). The material flows are calculated on different levels: total waste plastic packaging (i.e. goods) and the various constituting polymers (i.e. substances).

To assess the quality of the input data in describing the desired quantitative information, the uncertainties of these input data were quantified using the approach described by Laner et al. (2016). In this method, the data quality of each input data point is characterized qualitatively using five data quality indicators, which are presented in Table S-1 in the Supplementary data. The quantitative uncertainty value is subsequently derived based on coefficients of variation for each of the data quality indicator scores (as shown in Table S-2 in the Supplementary data), which are described by continuous characterizing functions (see Laner et al., 2016 for more details). This approach introduces two major aspects of subjectivity in the data uncertainty characterization. First, the indicator scores are assigned on more or less stringent criteria, and second, the quantitative uncertainty values for the various scores are estimated. Regarding the first aspect, although most evaluation criteria do not leave much room for interpretation, others are not that unambiguous, relying on the experience and tacit knowledge of the modeler. Concerning the second aspect, although the underlying mathematical functions allow the transparent and consistent characterization of the coefficients of variations within the method, the actual definition of these functions remains up to the modeler's choice (in the present study an exponential-type function is used, see Laner et al., 2016). As empirical data are usually not available as a basis for this choice, the estimates may differ from one MFA study to another. Therefore, although the approach builds on reproducible and internally consistent uncertainty estimates, comparisons of these estimates generated in different MFA studies should be done cautiously (Klinglmair et al., 2016; Laner et al., 2016). The estimated input uncertainties are subsequently propagated through the model using Gaussian error propagation (assuming normally distributed variables), whereas data reconciliation is used to resolve conflicts between input values. The material flow results are given as mean values and relative standard deviations of a normal distribution.

The system boundaries of the MFA are presented in Fig. 1, and are drawn to include all plastic packaging products from becoming waste in Austria until they are processed to provide secondary raw materials or energy, or are deposited on a landfill. The waste stream was subdivided into seven product categories, including PET bottles, small (<5 L) and large (≥5 L) hollow bodies, small (<1.5 m^2) and large (≥1.5 m^2) films, large EPS (≥0.1 kg), and other products. Only products fully composed of plastics are taken into account, so products made from material composites, such as food or drink cartons, are not considered. The quantification of the waste flows through the waste management system was performed separately for each of these seven product categories. On the polymer level, eight polymers were taken into account: low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), high-density polyethylene (HDPE), polypropylene (PP), polystyrene (PS), expanded polystyrene (EPS), polyethylene terephthalate (PET), and polyvinyl chloride (PVC). These polymers account for 99% of all plastics used in packaging in Europe, according to PlasticsEurope (2015).

2.2. Description of the plastic packaging waste flows

Fig. 1 shows the MFA model that quantifies the flows of plastic packaging waste in Austria, and in the further description, the flow numbers from this model are indicated. The plastic packaging products are used in the seven aforementioned categories (F1.01 - F1.07). After becoming waste, the products are either collected separately (SCW; F2.01), or are disposed of in the municipal solid waste (MSW; F2.02) or in bulky and commercial wastes (BCW; F2.03).

The separately collected stream is sorted into 18 sorting fractions, based on polymer, product type and color, which are then sent for single-polymer mechanical recycling (F3.02). Part of the PET waste stream is used for the production of higher value food-grade re-granulate (F3.01), and is therefore included as a separate flow in the model. Furthermore, a mixed-plastics stream is sent for mechanical recycling into mixed-polymer re-granulate (F3.03), used for the production of items such as recycled plastic lumber (RPL), which is then used to substitute wood in e.g. outdoor furniture. Consequently, three types of mechanical recycling processes are taken into account in the model: single-polymer recycling to produce food-grade re-granulate (F4.01) as well as non-food-grade re-granulate (F4.02), and mixed-polymer recycling (F4.03).

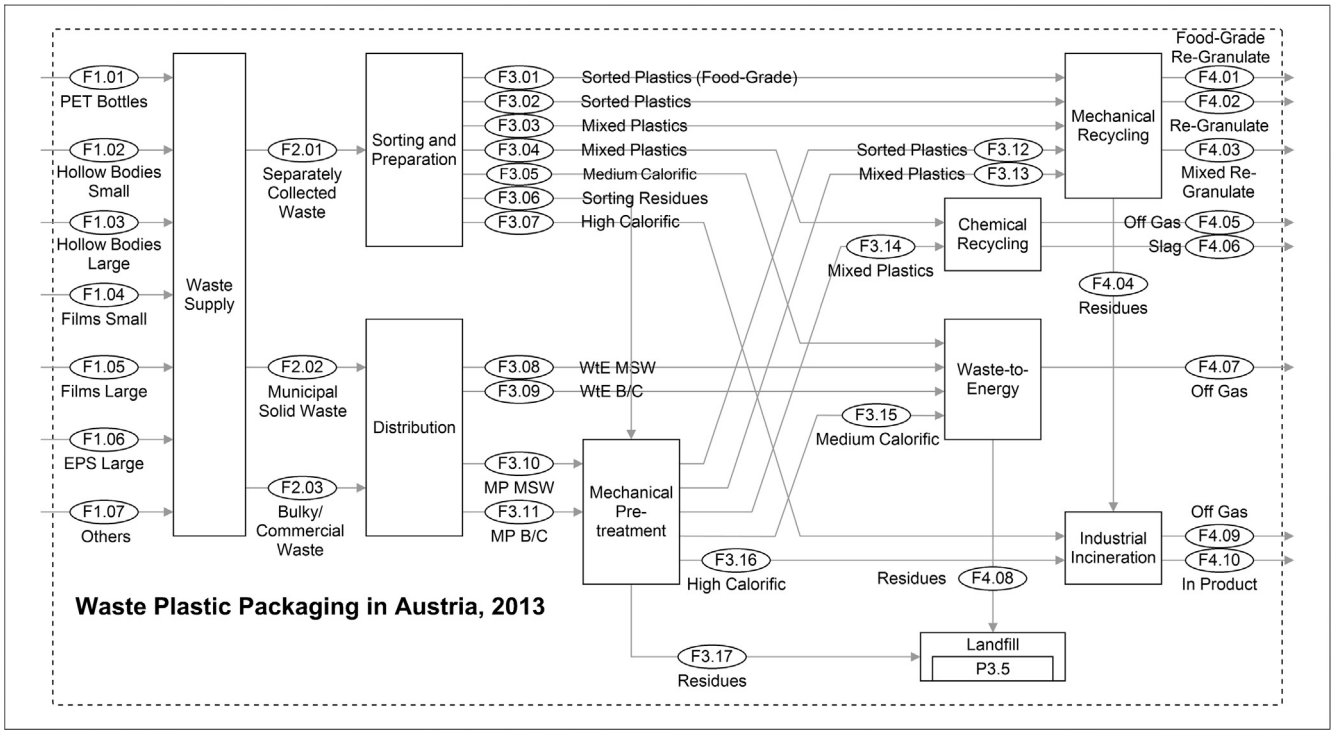

Fig. 1. Scope and model overview of the management of waste plastic packaging in Austria.

Additional mixed-plastics streams can be used for chemical recycling as an alternative reducing agent in the steel industry (F3.04). Furthermore, mixed-plastics streams with a medium calorific value are utilized for the production of energy in grate and fluidized bed Waste-to-Energy (WtE) plants (F3.05). Finally, other sorting residue streams can be sent for further mechanical treatment (F3.06) or are used as a high calorific alternative fuel in the cement industry (F3.07).

In Austria, direct landfilling of waste with an organic carbon content higher than 5% is banned (BMLFUW, 2004). MSW as well as BCW are therefore treated either in a grate WtE plant for energy production (F3.08 and F3.09) or through mechanical pretreatment in a Mechanical Biological Treatment (MBT) or splitting plant (F3.10 and F3.11). The latter process is used to separate materials which can be recycled (F3.12 to F3.14), produce medium calorific fractions (F3.15) for fluidized bed WtE plants and high calorific fractions (F3.16) for the cement industry, as well as a stabilized residual waste stream which can be landfilled (F3.17).

The residues of mechanical recycling are treated thermally in the cement industry (F4.04). Furthermore, the remains of chemical recycling and incineration, which is the ash content of the plastics, is present in the slag for chemical recycling (F4.06), is landfilled in the case of WtE plants (F4.08), or is present in the product in the cement industry (F4.10).

Finally, it should be noted that this is a comprehensive overview of the possible flows of waste plastic packaging, and that not all waste flows and treatment options were in fact present in 2013, as can be seen in the results (see Section 3).

2.3. Data sources and calculations

Market research on packaging consumption and packaging waste for 2013 was carried out by Hauer et al. (2015), providing information on the generation of plastic packaging waste sent to the different disposal routes, subdivided with regard to various product types. The methods used for this analysis are discussed in more depth in the Supplementary data (Section 2). More detailed information was provided for the separately collected waste by ARA (Altstoff Recycling Austria AG), which was the only producer responsibility compliance scheme for household packaging in Austria in 2013. In this capacity, ARA collects information on the separately collected waste it handles. They were thus able to supply data on the mass, polymer composition, product type and waste treatment destination for each of the sorting fractions coming from the sorting and preparation process, as well as on the mass, polymer composition, product type and destination of the mixed-plastics residues from sorting (ARA, 2016). For PET, further data was taken from WKO (2014).

Some additional literature data and calculations were necessary to harmonize the received data into the MFA model. As the product types used in Hauer et al. (2015) and ARA (2016) were specified differently, a combination of these types was needed to define the seven product categories used in this study. In general, for each part of the waste stream information both on the polymer composition and on the product categories was needed. However, this was only available for the separately collected waste. Moreover, the compositions of a few of these sorting fractions were specified to contain two polymers, so the market shares of the respective polymers were used to differentiate between the two (Borealis, 2016; Eurostat, 2016; PlasticsEurope, 2015). For the MSW and BCW, no data on the polymer composition were available, so the polymer distribution in each of the product categories in these waste streams was derived from the data for the SCW.

The share of MSW which was incinerated directly, as opposed to treated in an MBT plant, was available from ARA (2016). For the BCW, this distribution was estimated from the capacities of the MBT (BMLFUW, 2015; Neubauer and Öhlinger, 2008) and incineration plants (BMLFUW, 2015), subtracting the amounts of MSW treated there (BMLFUW, 2011). The transfer coefficients of plastics to the different process outputs of the MBT plant itself were determined by Laner and Brunner (2008). The recycling efficiencies in the mechanical recycling processes were provided by primary data

from recycling facilities for food-grade PET, LDPE and EPS, whereas for the other polymers these efficiencies were taken from Intini and Kühtz (2011) for PET to fiber, Franklin Associates (2011) for HDPE, EASETECH (2012) for PP and PS, and Huysman et al. (2015) for mixed-plastics recycling. Finally, the amount of residues after incineration (i.e. the ash content) for PVC was taken from Eggels et al. (2001), whereas for each of the seven other considered polymers this was calculated from the elemental compositions of these polymers, combined with the elemental transfer coefficients in incineration plants, as provided by Koehler et al. (2011).

A detailed description of all input data values, their uncertainty characterization and data source is provided in the Supplementary data (Tables S-3 and S-4).

3. Results

The results of the MFA are presented in Fig. 2 both on a per product category and per polymer basis. The results for each of the product categories separately are shown in Figs. S-3 to S-9 in the Supplementary data, where tables with the raw results behind the figures are presented in detail as well (see Section S-3). The mass flows in the figures and text are represented by the mean (with two significant digits) and relative standard deviation.

The total mass of waste plastic packaging in Austria amounted to 300,000 ± 3% t/a (tonnes per year, 10^3 kg) in 2013. The largest product categories were large and small films (both 24%), followed by small hollow bodies (17%), PET bottles (15%), others (13%), large hollow bodies (6%), and large EPS (1%). Regarding the polymer composition of the product categories, PET bottles and EPS large were by definition only composed of PET and EPS respectively. The HDPE bottle caps of the PET bottles were counted towards the others category, to be able to compare the results with other sources which report on a pure PET basis (e.g. Kuczenski and Geyer, 2010; WKO, 2014). The small hollow bodies consisted of mainly PP, followed by HDPE, PS, and minor amounts of PVC, whereas the composition of the large hollow bodies was

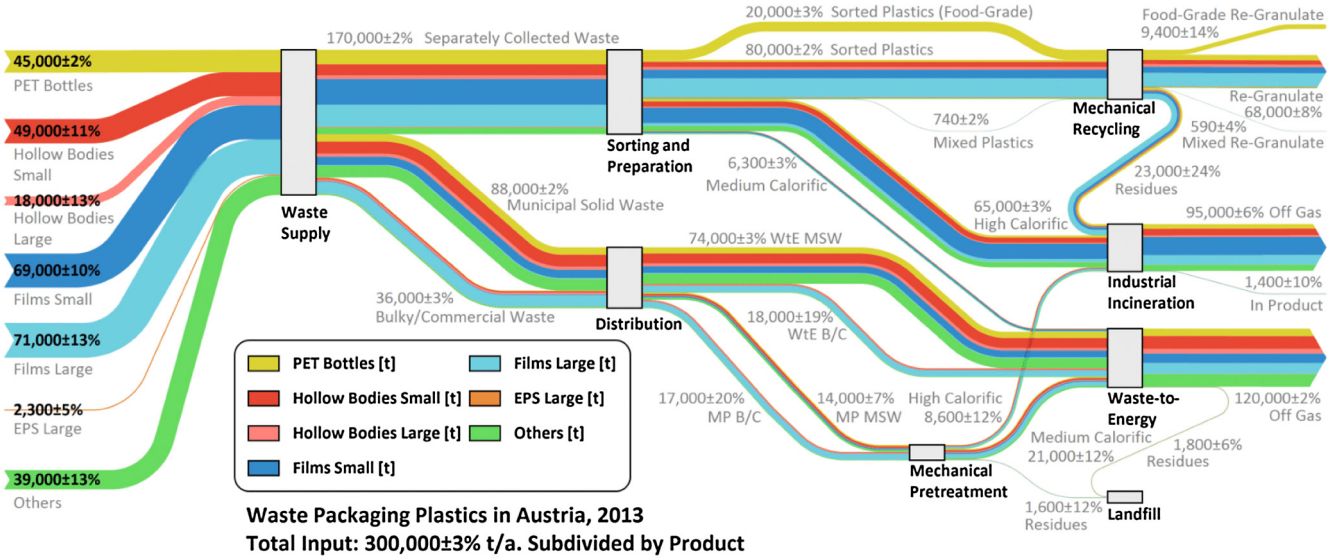

Fig. 2. Results of the material flow analysis for the total waste stream subdivided by (a) product category and (b) polymer. The values are given by the mean (2 significant digits) and the relative standard deviation.

distributed about evenly between HDPE and PP. The large and small films contained mostly LDPE with smaller amounts for LLDPE, as well as minor amounts of PVC in the small films. Finally, the others category contained the whole range of polymers, with mainly PET, LDPE and HDPE. All in all, this amounted to a total waste stream composition of mainly LDPE (46% ± 6%), followed by PET (19% ± 4%), PP (14% ± 6%), HDPE (11% ± 6%), LLDPE (5% ± 5%), PS (3% ± 5%), EPS (2% ± 4%), and PVC (<1% ± 6%) (see also Fig. 6 further on).

Three indicators are subsequently used for describing the extent to which materials are recirculated within the economy: the Collection Rate (the amount collected divided by the total waste amount), the Sorting Rate (the amount sorted and sent to the mechanical recycling processes divided by the total waste amount, as defined by the EU for the targets, see Section 4.3 further on), and the Recycling Rate (the amount of re-granulate produced at the mechanical recycling plant divided by the total waste amount). These three rates are presented in Fig. 3 for each of the product categories. It has to be noted that this definition of the "recycling rate" differs from the way this term is customarily defined, as it is generally used to refer to the amount in the input of the recycling plant (i.e. the Sorting Rate in this study) rather than in the output as defined here. However, we feel that the latter approach is more accurate in describing what these three rates actually represent.

Of the total waste stream, 58% ± 3% was collected separately, whereas 30% ± 4% was discarded into the MSW and 12% ± 4% into BCW. Especially films, EPS large and PET bottles were largely collected separately (64–77%). In the sorting process, two groups of product categories can be distinguished regarding the sorting efficiencies (output of the storing plant sent for mechanical recycling divided by the input into the sorting plant). On the one hand, PET bottles and the categories of large products were sorted efficiently (73–88%). On the other hand, the sorting processes in place were not able to sort out small hollow bodies (47%), small films (34%) and others (12%) to the same extent (see Fig. 4). The Sorting Rate of the total waste stream thus amounted to 34% ± 3%.

MSW was largely treated directly in a grate WtE plant (84%), whereas for BCW this was distributed more evenly (51% to grate WtE). The part passing through mechanical pretreatment first was predominantly treated thermally in fluidized bed WtE plants (67%) or the cement industry (28%), whereas the final 5% was landfilled.

Fig. 5 displays the breakdown of the waste stream in terms of the final treatment processes. The overall Recycling Rate was calculated to be 26% ± 7%, whereas 40% ± 3% was treated in WtE plants and the remaining 33% ± 6% in the cement industry. Minor amounts coming from residues from mechanical pre-treatment (as plastics) and WtE (as ashes) were landfilled (1% ± 6%). Although plastics are used for chemical recycling in the steel industry in Austria, in 2013 no waste plastic packaging generated in Austria was used. The Recycling Rates for the individual product categories ranged from 68% for EPS large to 3% for others. The produced re-granulate was used for 12% in food-grade applications (from PET bottles only), 87% in other single-polymer products, and 1% for products with mixed-polymer re-granulate. For PET bottles specifically, 46% of the re-granulate was used for food-grade and 54% for non-food-grade applications, which represents 21% and 24% of the PET bottles waste stream, respectively. However, this amount of PET bottles going to food-grade applications is to be seen as a lower limit value, as this distribution depends heavily on the market situation. At the food-grade recycling plant not the whole waste stream is prepared for food-grade applications, as a part is sold directly for the production of PET fibers, depending on the current demand.

The share incinerated in a WtE plant, compared to the cement industry, is related to the amount that is separately collected. Categories such as films small had a large Collection Rate but were rejected to a large extent at the sorting plant and thus ended up more in the cement industry, compared to the hollow bodies small

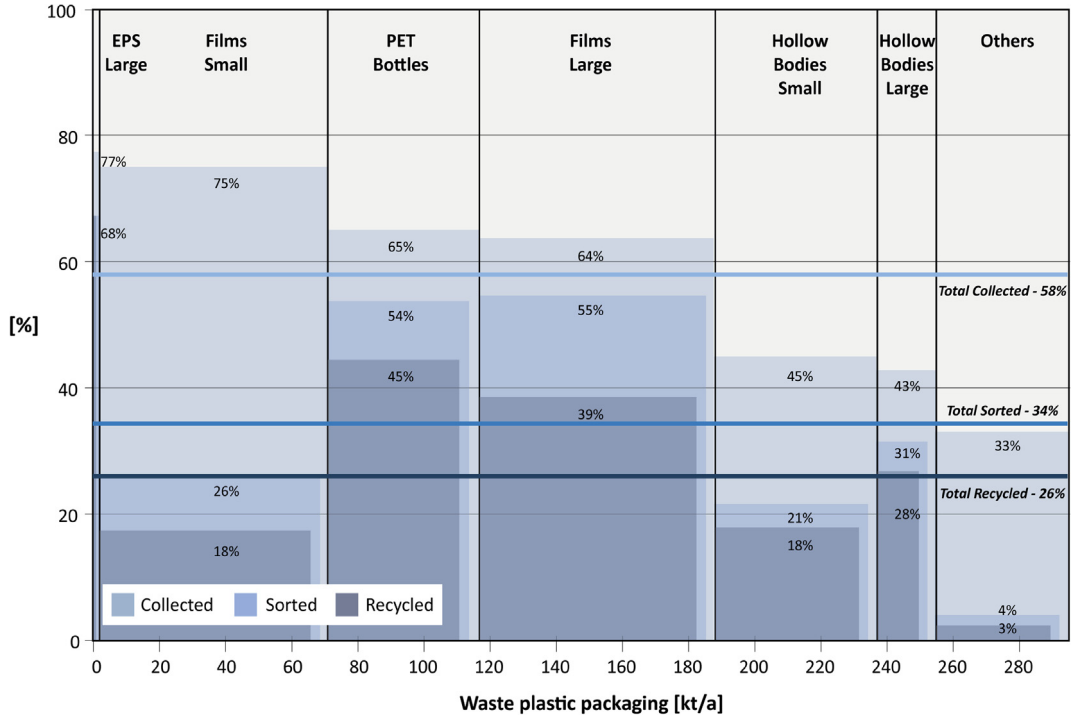

Fig. 3. Comparison of the Collection, Sorting and Recycling Rates, per product category in relation to the respective mass in the input. The respective rates for the total waste stream are shown by the horizontal lines.

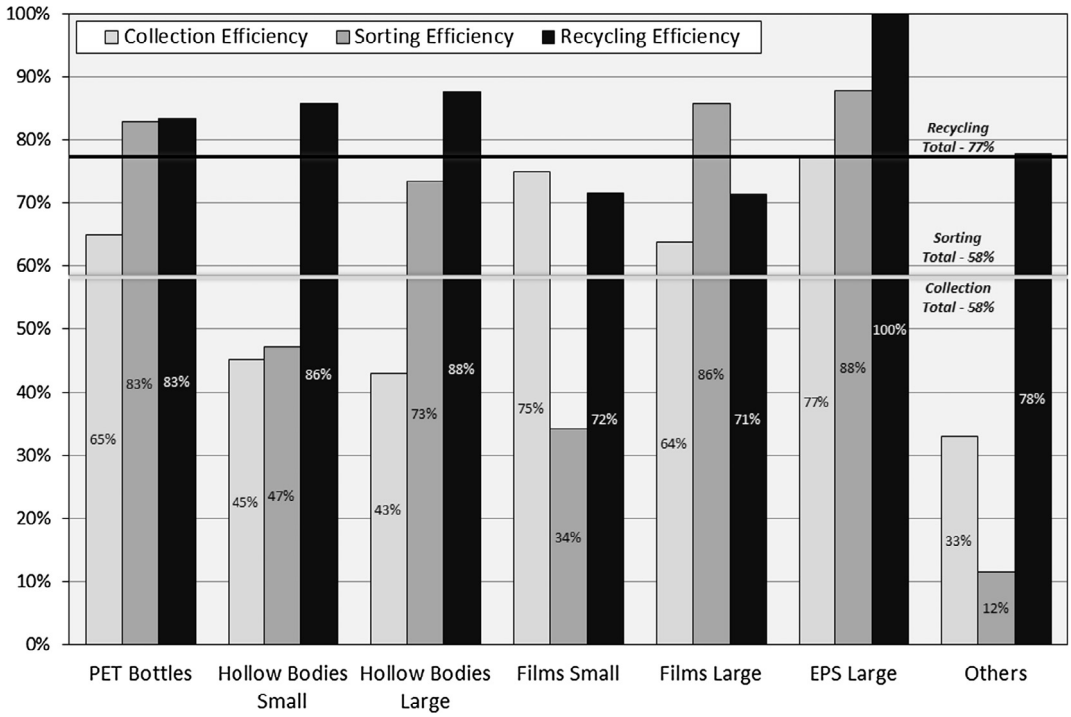

Fig. 4. Efficiencies of the collection, sorting and mechanical recycling processes for each of the product groups. The respective results for the total waste stream are shown by the horizontal lines.

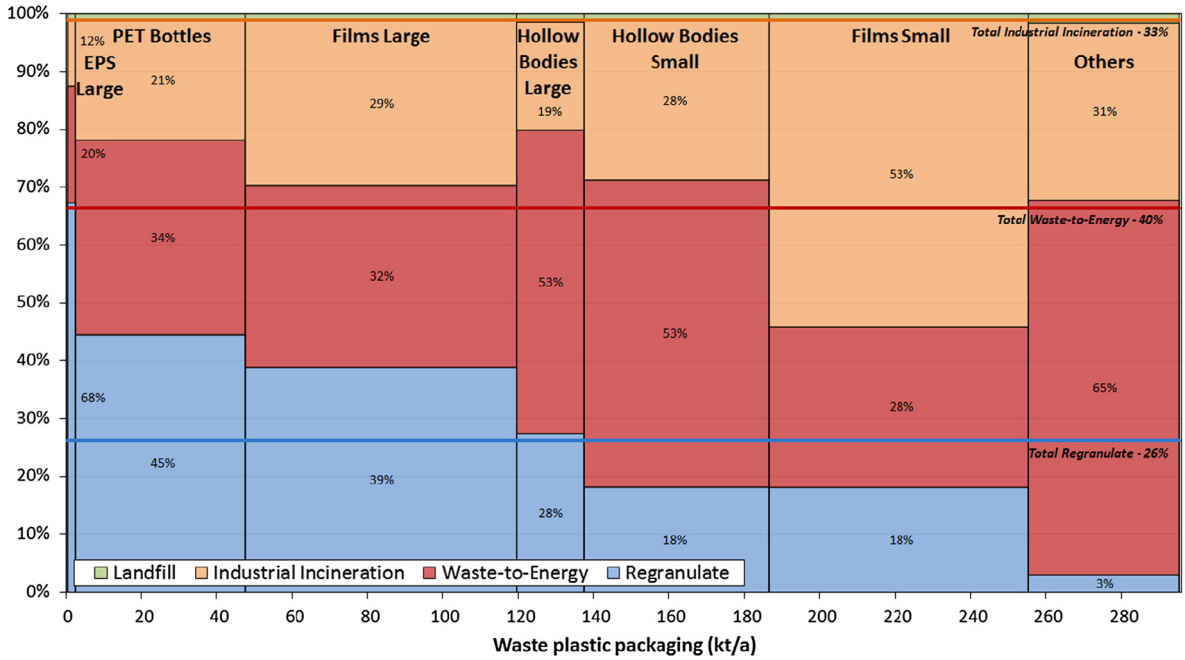

Fig. 5. Final treatment of the product categories in relation to the respective mass in the input. The respective results for the total waste stream are shown by the horizontal lines.

and large and others categories with a low Collection Rate and high share in WtE plants.

The results can also be presented on a per polymer basis. The Collection, Sorting and Recycling Rates are shown in Fig. S-10 in the Supplementary data, and the breakdown in terms of the final treatment processes is shown in Fig. S-11. Because of the low amounts and therefore bad data quality of PVC in the input, the results of the PVC streams were not deemed to be reliable, so in the results PVC will not be discussed.

The results for the seven polymers are closely correlated to those for the product categories in which they are mainly used. EPS, LDPE, PET and LLDPE, which are primarily used in the categories EPS large, films and PET bottles, thus had the highest Collection Rates (79–56%). Conversely, for HDPE, PP and PS, these rates were progressively lower (45–33%). Within the sorting process, especially PET and LDPE had high sorting efficiencies of around 75%, followed by LDPE and HDPE (57% and 53%). PP, PS and EPS however had lower sorting efficiencies (38–44%). All in all, looking

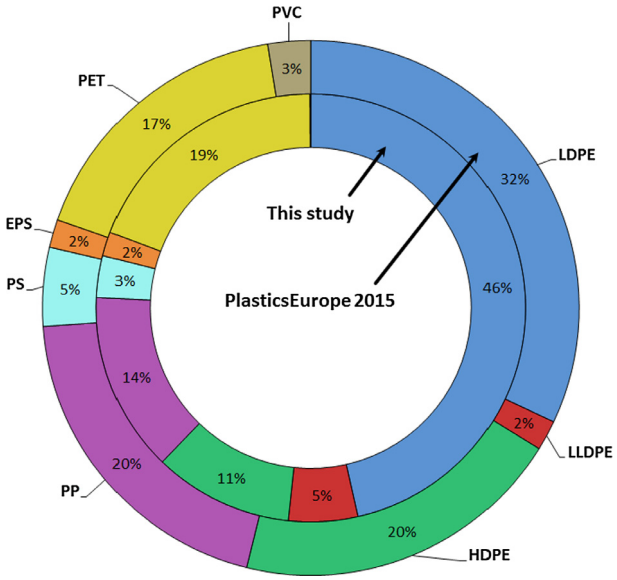

Fig. 6. Comparison of the polymer composition for the year 2013 of plastic packaging demand in Europe from PlasticsEurope (2015), and of waste plastic packaging in Austria.

at the final treatment, PET had the highest Recycling Rate (38% ± 7%), followed by EPS and LLDPE (30% ± 14%), LDPE (26% ± 15%), HDPE (23% ± 15%), PP (15% ± 15%), and PS (11% ± 14%). The rest was incinerated roughly evenly in WtE plants and the cement industry for PET, EPS, LLDPE and LDPE, whereas substantially more was treated thermally in WtE plants for HDPE, PP and PS.

4. Discussion

4.1. Input composition

The calculated composition of the incoming waste material in Austria can be compared with statistics on the demand of plastic packaging by polymer for the EU27, Norway and Switzerland in 2013, obtained from PlasticsEurope (2015). To differentiate between LDPE and LLDPE, data from Eurostat (2016) and Borealis (2016) were used (see also Section 2.3). This comparison, as displayed in Fig. 6, shows that in the waste in Austria, the share of LDPE was substantially higher than in the demand across Europe, offset mainly by a lower share for HDPE and PP.

The values in Fig. 6 can be expressed per capita as well, taking into account the total amount of waste plastic packaging and demand of plastic packaging respectively, which results in similar amounts of 35 kg/cap for Austria and 36 kg/cap for Europe. Therefore, the same trends as was the case for the relative distribution can be seen regarding the input composition. It is therefore clear that in Austria, soft plastics used in films are overrepresented in contrast to hard plastics, compared to Europe.

4.2. Comparison with related results

Van Eygen et al. (2017) discussed the flows of all plastics from all consumption sectors in 2010. Here, the amount of waste plastic packaging was determined to be 281,000 t/a including drink cartons, which were not taken into account in this study. Without these drink cartons, the waste plastic packaging amount for 2010 was 274,000 t/a, which corresponds to a growth rate of 8% over three years compared with the 300,000 ± 3% t/a calculated in this study. This is in the same range as the growth rate over this period of other streams in the plastic budget (Van Eygen et al., 2017). In this study, the overall mechanical Recycling Rate for all eight packaging polymers was calculated to be 26% ± 7%. This result is similar to the value of 21% obtained in Van Eygen et al. (2017) for all plastic wastes. However, the latter also contained production wastes, which are mainly used for mechanical recycling, so the mechanical Recycling Rate of post-consumer wastes is expected to be considerably lower. It can thus be concluded that the packaging sector has a relatively high Recycling Rate, compared to post-consumer wastes from other consumption sectors such as building and construction and electronics, which might be explained by the composition of the plastics in the latter sectors (i.e. higher use of potentially problematic additives) and the long-standing legislation in place for the separate collection of packaging waste.

PlasticsEurope, the association of European plastics manufacturers, yearly reports data on plastics and the plastics industry. For 2013, recycled amounts for waste plastic packaging in Austria of around 29% were reported (PlasticsEurope, 2015), although it is not clear if the Sorting Rate (34% ± 3% in this study), or the Recycling Rate (26% ± 7% in this study) is meant. In a further report prepared for PlasticsEurope however, the 34% ± 3% Sorting Rate found in this study is confirmed (Consultic, 2015). Most of the EU countries reported similar recycled amounts between 30 and 40%, and the main difference lies in the amount that is incinerated instead of landfilled. In this regard, the countries with a landfill ban (i.e. Austria, Luxembourg, Germany, Switzerland, Denmark, Sweden, Belgium, the Netherlands, and Norway) reported recycled and incinerated amounts over 95% (PlasticsEurope, 2015), indicating the success of this legislation in diverting plastics away from landfills. Furthermore, Eurostat (2017) reports the performance of the EU member states on the management of packaging waste. For Austria in 2013, a somewhat lower waste plastic packaging amount of 288,714 t/a is stated, compared to the 300,000 t/a in this study. Thereof 99,258 t/a is reported to be sent for mechanical recycling, amounting to a Sorting Rate of 34%, which is the same result as calculated in this study (see Fig. 3).

Of the polymers and product types used in packaging, the flows of PET and PET bottles have attracted the most attention in the scientific literature. According to Welle (2011), the amount of sold PET bottles in the EU that were collected has seen a spectacular increase over the past decades, with growth rates between 10 and 20% per year. In 2009, an average Collection Rate of 48% was reached, with the highest Collection Rate achieved in Germany (94%, only for drinking bottles) because of the deposit system (cf. 65% in Austria for all kinds of PET bottles from this study). Although the USA started with much higher Collection Rates for PET bottles in the late nineties compared to those in the EU, these have only about doubled since then, resulting in a Collection Rate of 28% in 2009 (Welle, 2011). This is confirmed by Kuczenski and Geyer (2010), who conducted an MFA for PET in the USA for the 1996–2007 time period, and concluded that although the collection of post-consumer PET bottles has indeed about doubled over this period, the amount going to domestic recyclers has not followed this trend, with an ever increasing amount being exported. Domestic Recycling Rates even declined from about 19% to about 11% because of the increase in waste PET exports. Finally, for Brazil and Japan, Collection Rates of 56 and 78% respectively were reached in 2009 (Welle, 2011), whereas this was 85% in Switzerland, with 26% of the PET bottles waste stream reprocessed into new bottles (Haupt et al., 2016), compared to 21% in Austria.

4.3. Circular economy and EU targets

As mentioned in Section 1, member states of the EU need to meet targets for waste packaging, with minimum recycling and recovery rates set for the overall waste stream as well as for five individual materials. For plastics, since the end of 2008 the

minimum target is defined to be 22.5% by weight, counting exclusively material recycled back into plastics (EPC, 2004). In 2015, a proposal was made which sees the target for waste plastic packaging rise to a minimum of 55% by the end of 2025. Furthermore, the calculation point for reporting the amount recycled will then be explicitly defined as the amount of waste entering the final recycling process (i.e. the Sorting Rate as defined in this article), which is more or less equal to the interpretation which is currently used by most member states (EC, 2015a). The future clear definition of the targets is important, as up until now, the implementation of the EU directive into national law was not carried out in a harmonized way. Moreover, member states have shown reluctance in reporting the methods used to obtain the data on the recovery and recycling targets, which makes comparisons between member states even more challenging (Tsiarta et al., 2015).

Since 2006, the Austrian implementation of the EU directive has adopted the same recycling target of 22.5% and uses the Sorting Rate as calculation method (BMLFUW, 2006, 2014). As shown in Section 3 and in Fig. 3, a Sorting Rate of 34% ± 3% was reached in 2013, thereby achieving the EU and Austrian targets. With respect to the proposed EU target of 55% by 2025, three product categories (EPS large, films large, PET bottles) currently have a Sorting Rate around or above the required value, nevertheless major additional efforts will be needed to increase the collection and sorting efficiencies to reach this target for the overall waste stream. The current efficiencies for the collection, sorting and recycling processes are shown in Fig. 4.

To illustrate the required process efficiencies to achieve the proposed future target, a theoretical scenario was built by changing the collection and sorting efficiencies for the seven product categories up to a point where the required 55% of waste packaging enters a mechanical recycling process. The results of this effort are shown in Table 1. It was assumed that for PET bottles, collection (e.g. through a deposit system) and sorting (e.g. through chemical markers) efficiencies towards food-grade recycling of up to 90% could be reached. For the large and small hollow bodies, films large and EPS large, the same collection efficiency of 80% was set. The sorting efficiency of the large hollow bodies and films was set to 90%, whereas this was set to 70% for hollow bodies small and 50% for films small. The collection efficiency for films small and the sorting efficiency of EPS large (already quite high), as well as both values for the others category (most difficult category to collect and sort due to diversity), were not changed. The amount going to mixed-plastics recycling was not changed as well, but this plays a minor role (1% of the mass going to mechanical recycling). All in all, a Sorting Rate of 55.8% is reached using these assumed efficiencies and the waste stream characteristics of the status quo.

This effort shows that quite high values are needed to reach the proposed target of 55%, and thus major improvements in both the collection system and sorting technologies will be necessary. This is especially the case since the reported Sorting Rates in Austria have been stagnating over the past decade: 31% was already reached in 2003 (Eurostat, 2017), compared to 34% ± 3% in 2013. The required efforts could be somewhat moderated however by allowing a certain maximum amount of impurities in the sorted waste stream, which are not subject to a final recycling process, to be counted towards the Sorting Rate. This has been suggested in the new proposal of the EU directive, where an impurity content of up to 10% would be allowed (EC, 2015a). Including this maximum amount of impurities would lower the target to a minimum of 49.5% (55%–10% * 55%) with respect to the actual amount of plastics in the input of the recycling process. This proposed limit on the impurity content should also prevent cases, as reported by EUWID (2014), where some sorting plants are listed as a recycling process, and therefore the total input of the sorting plant is counted as "recycled", although often around 50% of this input goes to incineration as sorting residues. The question remains though how to accurately (and routinely) measure the impurity content for reporting purposes.

The calculation point of the targets remains of major importance, as the closer this is placed towards the end of the recycling chain (i.e. after the recycling plant), the more the actual amount of produced re-granulate is accounted for. As currently the calculation point is placed before the recycling plant, increasing targets could lead to a decreasing purity of the material in the input, as the system is incentivized to include more material in the outputs of the sorting plants. Therefore fewer impurities (non-plastics as well as unwanted polymers) would be sorted out, causing higher losses at the recycling plant as well as jeopardize the quality of the final re-granulate. Conversely, moving the calculation point towards the output of the recycling plant could nonetheless have adverse effects on the quality of the re-granulate as well, by encouraging the shift of high-quality recycling with relatively high losses to processes with lower-grade applications which can handle higher amounts of impurities. In general, it is thus crucial to consider the quality of the recycling products, besides the merely mass-based perspective of the recycling rates where including impurities into the product is encouraged.

This tendency of including impurities manifests itself at the sorting process, where mixed-plastics streams can be generated at the sorting plant by design, which are subsequently recycled into mixed-polymer re-granulate (used e.g. as a substitute for wood). However, the environmental benefits of these re-granulates have been called into question (see e.g. Astrup et al., 2009; Corsten et al., 2010). The latter trend can be seen in the Netherlands for example, where much higher targets are set than in Austria (45% in 2015, rising each year to 51% in 2021, see MIM, 2014) and therefore higher Sorting Rates of up to 50% in 2014 are reported (Eurostat, 2017). This has led the share of mixed-polymer re-granulate to reach more than half of the produced re-granulate from household packaging waste (Nusselder and Odegard, 2016), compared to about 1% of all packaging waste in Austria.

As a final note, moving the calculation point towards the output of the recycling plant could increase the administrative burden for reporting, as it would require collecting data from the many individual recycling companies, which can be located internationally.

Table 1
Change of the collection and sorting efficiencies per product group from the status quo to the future EU target scenario. The status quo of these values can be seen in Fig. 4 as well.

Product categories	Collection efficiency (%)	Sorting efficiency (%)
PET bottles	*65* → 90	*83* → 90
Hollow bodies small	*45* → 80	*47* → 70
Hollow bodies large	*43* → 80	*73* → 90
Films small	*75*	*34* → 50
Films large	*64* → 80	*86* → 90
EPS large	*77* → 80	*88*
Others	*33*	*12*

5. Conclusion and outlook

In this study, the management of waste plastic packaging generated in Austria in 2013 was investigated using MFA on a polymer and product category basis. The results show that around 300,000 ± 3% tonnes of waste plastic packaging were produced in Austria in 2013, which corresponded to about 35 kg per capita and year. These were mainly composed of large and small films, and small hollow bodies, including PET bottles. Correspondingly, the polymer composition of the waste stream was dominated by LDPE (46% ± 6%), PET (19% ± 4%) and PP (14% ± 6%). Overall,

58% ± 3% of the waste stream was collected separately, whereas the rest was present in MSW and BCW. After sorting, 26% ± 7% of the total waste plastic packaging stream was transformed into secondary raw materials as re-granulates, whereas the rest was treated in WtE plants (40% ± 3%) and the cement industry (32% ± 6%).

The current target from the EU, as well as the Austrian implementation, were reached comfortably. However, to achieve the proposed increased target, major steps will be needed with respect to both collection and sorting of waste plastic packaging. Moreover, these targets, calculated with respect to the amount of waste in the input of the mechanical recycling process, are not completely in line with the overall objective of the circular economy, namely to keep materials in the economy and prevent losses. To accurately assess the performance of the waste management system, it is thus recommended that the targets be calculated with respect to the actual output of the recycling process, provided that the quality of the output products is maintained, and separately for mixed-polymer re-granulates. This ensures that the reported rates are not elevated by including more mixed-plastics streams or impurities after sorting, causing increased production of mixed-polymer re-granulate, or increased losses in the recycling process, which would be counterproductive to the goals of the circular economy. Moving the calculation point would then in turn necessitate reevaluating the proposed future recycling targets as well.

Finally, moving beyond mass-based indicators is needed by assessing the waste management system with respect to environmental performance as well. After all, a major objective of the circular economy is to reduce the environmental impacts of the production and consumption system. Therefore, it is necessary to assess the environmental performance of current waste plastic packaging management as well as with respect to increasing targets. These considerations are subject to further research.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.wasman.2017.11.040.

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Supplementary data

Circular economy of plastic packaging: current practice and perspectives in Austria

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Supplementary Data

Abstract

Plastics, especially from packaging, have gained increasing attention in waste management, driving many policy initiatives to improve the circularity of these materials in the economy to increase resource efficiency. In this context, the EU has proposed increasing targets to encourage the recycling of (plastic) packaging. To accurately calculate the recycling rates, detailed information on the flows of plastic packaging is needed. Therefore, the aim of this paper is to quantitatively and qualitatively investigate the waste management system for plastic packaging in Austria in 2013 using material flow analysis, taking into account the used product types and the polymer composition. The results show that 300,000±3% t/a (35 kg/cap·a) of waste plastic packaging were produced, mainly composed of large and small films and small hollow bodies, including PET bottles. Correspondingly, the polymer composition of the waste stream was dominated by LDPE (46%±6%), PET (19%±4%) and PP (14%±6%). 58%±3% was collected separately, and regarding the final treatment, 26%±7% of the total waste stream was recovered as re-granulates, whereas the rest was thermally recovered in waste-to-energy plants (40%±3%) and the cement industry (33%±6%). The targets set by the EU and Austria were reached comfortably, although to reach the proposed future targets major technological steps regarding collection and sorting will be needed. However, the current calculation point of the targets, i.e. on the input side of the recycling plant, is not deemed to be fully in line with the overall objective of the circular economy, namely to keep materials in the economy and prevent losses. It is therefore recommended that the targets be calculated with respect to the actual output of the recycling process, provided that the quality of the output products is maintained, to accurately assess the performance of the waste management system.

Contents

1 Subsystems "Mechanical Recycling and "Waste-to-Energy"

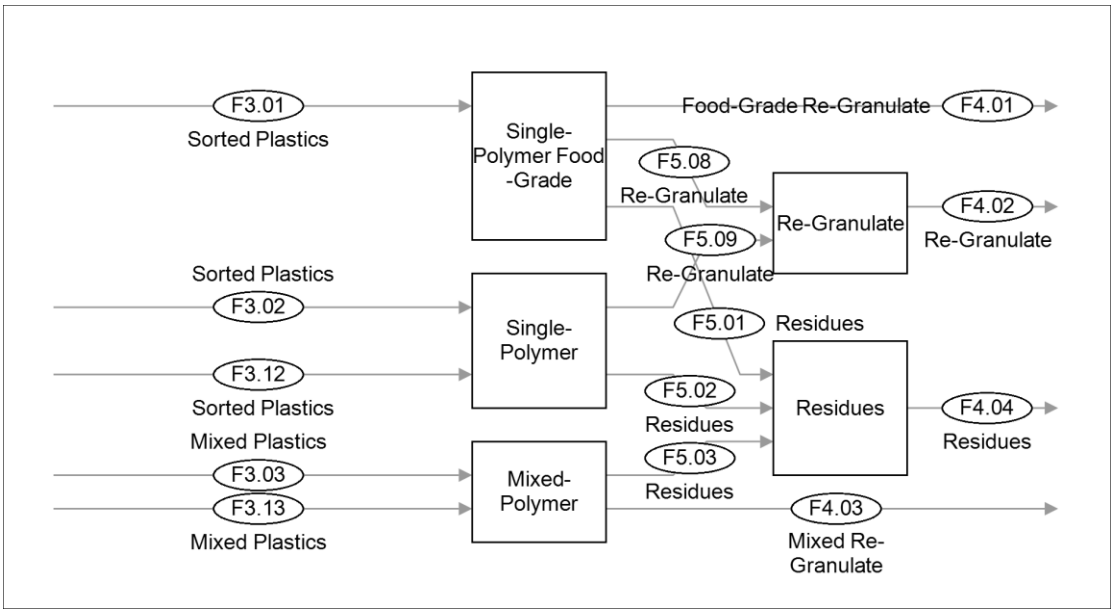

Figure S-1: Subsystem "Mechanical Recycling" of the MFA model as displayed in Figure 1 of the main article.

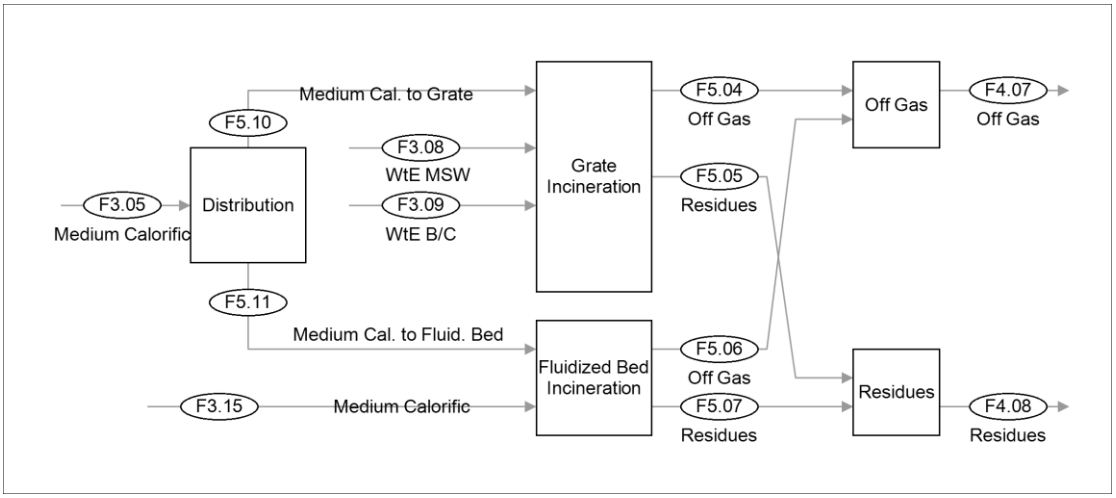

Figure S-2: Subsystem "Waste-to-Energy" of the MFA model as displayed in Figure 1 of the main article.

2 Input data and uncertainty characterization

The confidential report by Hauer et al. [1] is a large source of input data for this study. In this report, the amounts of packaging that were put on the market as well as the waste generated from packaging were surveyed. This was done for various material groups: plastics, composite materials, paper, metals, wood, glass, textile fibers, ceramics, and packaging from biological materials. For the estimation of the waste generation, a combination of various methods was used. First, Austrian-wide waste composition analyses were performed on the municipal solid waste, as well as on bulky, commercial and industrial waste. This was complemented by analyses on the amounts of packaging waste that was collected separately by the various collection systems, as well as by so-called selfdisposers, which are companies that do not choose to delegate their extended producer responsibility obligations to third party collection systems. Finally, other minor waste streams, such as wastes from markets, the building sector and medicine, were analyzed for packaging wastes as well.

These results were subsequently validated by two other approaches. First, the amount of packaging sold was analyzed through market research on the sales of relevant product types, such as beverages. This was done using retail scanner datasets, which were then extrapolated to the full market. Second, the amount of packaging put on the Austrian market was quantified using data on domestic production, imports and exports of empty packaging. This was complemented by estimations on the production, imports and exports of packed goods. Both these validation approaches demonstrated the sufficient accuracy of the estimation of the amount of waste generated.

Table S-1: Data quality indicators and qualitative evaluation criteria, adapted from Laner et al. [2].

Indicator	Score: 1	Score: 2	Score: 3	Score: 4
Reliability	Methodology of data generation well documented and consistent, peer-reviewed data.	Methodology of data generation is described, but not fully transparent; no verification.	Methodology not comprehensively described, but principle of data generation is clear; no verification.	Methodology of data generation unknown, no documentation available.
Completeness	Value includes all relevant processes/flows in question.	Value includes quantitatively main processes/flows in question.	Value includes partial important processes/flows, certainty of data gaps.	Only fragmented data available; important processes/mass flows are missing.
Temporal correlation	Value relates to the right time period.	Deviation of value 1 to 5 years.	Deviation of value 5 to 10 years.	Deviation more than 10 years.
Geographical correlation	Value relates to the studied region.	Value relates to similar socio-economical region (GDP, consumption pattern).	Socio-economically slightly different region.	Socio-economically very different region.
Other correlation	Value relates to the same product, the same technology, etc.	Values relate to similar technology, product, etc.	Values deviate from technology/product of interest, but rough correlations can be established based on experience or data.	Values deviate strongly from technology/ product of interest, with correlations being vague and speculative.
Expert estimate	Formal expert elicitation with (empirical) database – transparent procedure and fully informed experts on the subject.	Structured expert estimate with some empirical data available or using transparent procedure with informed experts.	Expert estimates with limited documentation and without empirical data available.	Educated guess based on speculative or unverifiable assumptions.

Table S-2: Quantitative uncertainties expressed as coefficients of variation for the data quality indicators described in Table S-1, adapted from Laner et al. [2].

Data quality indicator	Sensitivity level	Score: 1	Score: 2	Score: 3	Score: 4
		Coefficient of variation (CV, in %)			
Reliability	-	2.26	6.84	20.64	62.32
Completeness/ temporal/ geographic/other correlation	High	0.00	4.53	13.67	41.28
	Medium	0.00	2.26	6.84	20.64
	Low	0.00	1.13	3.42	10.32
Expert estimate	-	4.53	13.67	41.28	124.64

Table S-3: Detailed overview of the **mass input data** of the model and uncertainty characterization of each data point. The codes of the flows correspond to those in Figure 1 in the main article and Figure S-1 and Figure S-2 in this document. The values of the scores correspond to the coefficients of variation shown in Table S-2. Urel: relative uncertainty, which is calculated according to the method outlined in Laner et al. [2]; SF: Single-polymer Fraction after sorting; MF: Mixed-polymer Fraction after sorting; MSW: Municipal Solid Waste; BCW: Bulky/Commercial Waste.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F1.01	**Waste Production**	PET Bottles	**45,487**							**2.26%**	
		ARA SF	*22,723*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[3]
		Other Systems SF and MF	*1,718*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[4]
		ARA MF	*5,116*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[3]
		ARA MF Other Polymers	*0*								
		MSW	*14,884*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[1]
		BCW	*1,046*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[1]
F1.02		Hollow Bodies Small	**49,176**							**11.28%**	
		ARA SF	*9,811*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		Other Systems SF and MF	*544*	*6.84%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[1]
		ARA MF	*10,920*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		ARA MF Other Polymers	*964*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[3]
		MSW	*24,390*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
		BCW	*2,547*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
F1.03		Hollow Bodies Large	**18,309**							**13.21%**	
		ARA SF	*3,113*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		Other Systems SF and MF	*2,626*	*6.84%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[1]
		ARA MF	*1,948*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		ARA MF Other Polymers	*167*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[3]
		MSW	*6,505*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
		BCW	*3,950*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
F1.04		Films Small	**69,428**							**9.58%**	
		ARA SF	*16,708*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		Other Systems SF and MF	*684*	*6.84%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[1]
		ARA MF	*31,726*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		ARA MF Other Polymers	*2,846*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[3]
		MSW	*16,413*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
		BCW	*1,051*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]

Table S-3: Continued.

Code	Process	Flow	Mass	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[t]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F1.05		Films Large	**71,185**							**12.99%**	
		ARA SF	*22,814*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		Other Systems SF and MF	*16,000*	*6.84%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[1]
		ARA MF	*6,024*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		ARA MF Other Polymers	*541*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[3]
		MSW	*1,736*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
		BCW	*24,070*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
F1.06		EPS Large	**2,328**							**5.17%**	
		ARA SF	*1,226*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[3]
		Other Systems SF and MF	*344*	*6.84%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[1]
		ARA MF	*210*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[3]
		ARA MF Other Polymers	*14*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*6.84%*			[3]
		MSW	*16*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[1]
		BCW	*518*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[1]
F1.07		Others	**39,475**							**12.84%**	
		ARA SF	*1,393*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[3]
		Other Systems SF and MF	*0*								
		ARA MF	*5,645*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*0.00%*			[3]
		ARA MF Other Polymers	*5,995*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*20.64%*			[3]
		MSW	*24,075*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
		BCW	*2,367*	*2.26%*	*0.00%*	*0.00%*	*0.00%*	*13.67%*			[1]
F2.01	**Collection**	Separately Collected Waste	**171,820**	2.26%	0.00%	0.00%	0.00%	0.00%		**2.26%**	[3; 1]
F2.02		Municipal Solid Waste	**88,019**	2.26%	4.53%	0.00%	0.00%	0.00%		**5.06%**	[1]
F2.03		Bulky/Commercial Waste	**35,549**	2.26%	4.53%	0.00%	0.00%	0.00%		**5.06%**	[1]
F3.01	**Sorting and**	Sorted Plastics (Food-Grade)	**20,151**	2.26%	2.26%	0.00%	0.00%	0.00%		**3.20%**	[3; 4]
F3.02	**Preparation**	Sorted Plastics	**79,554**	2.26%	2.26%	0.00%	0.00%	0.00%		**3.20%**	[3; 1]
F3.03		Mixed Plastics	**737**	2.26%	2.26%	0.00%	0.00%	0.00%		**3.20%**	[3; 1]
F3.05		Medium Calorific	**6,269**	2.26%	2.26%	0.00%	0.00%	4.53%		**5.55%**	[3; 1]
F3.07		High Calorific	**65,109**	2.26%	2.26%	0.00%	0.00%	4.53%		**5.55%**	[3; 1]

Table S-4: Detailed overview of the **transfer coefficient input data** of the model and uncertainty characterization of each data point. The codes of the flows correspond to those in Figure 1 in the main article and Figure S-1 and Figure S-2 in this document. The values of the scores correspond to the coefficients of variation shown in Table S-2. Urel: relative uncertainty, which is calculated according to the method outlined in Laner et al. [2]; MR: Mechanical Recycling.

Code	Process		Coefficient	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
F2.02 -> F3.09	**Distribution**		**84%**						13.67%	**13.67%**	[3]
F2.02 -> F3.11			**16%**						13.67%	**13.67%**	[3]
F2.03 -> F3.10			**5%**						41.28%	**41.28%**	[5; 6]
F2.03 -> F3.12			**49%**						41.28%	**41.28%**	[5; 6]
∑ -> F3.17	**Mechanical**		**67%**	6.84%	2.26%	3.42%	0.00%	4.53%		**9.17%**	[7]
∑ -> F3.18	**Pretreatment**		**28%**	6.84%	2.26%	3.42%	0.00%	4.53%		**9.17%**	[7]
∑ -> F3.19			**5%**	6.84%	2.26%	3.42%	0.00%	4.53%		**9.17%**	[7]
∑ -> F4.10	**Industrial Incineration:**	LDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		LLDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		HDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PP	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PS	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		EPS	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PET	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PVC	**5%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[9]
∑ -> F4.01	**Single-Polymer Food-Grade MR**		**46%**						13.67%	**13.67%**	[10]
∑ -> F5.08			**38%**						13.67%	**13.67%**	[10]
∑ -> F5.09	**Single-Polymer MR:**	LDPE	**71%**						13.67%	**13.67%**	[10]
		LLDPE	**71%**						13.67%	**13.67%**	[10]
		HDPE	**93%**						13.67%	**13.67%**	[11]
		PP	**81%**						13.67%	**13.67%**	[12]
		PS	**81%**						13.67%	**13.67%**	[12]
		EPS	**100%**						13.67%	**13.67%**	[10]
		PET	**78%**						13.67%	**13.67%**	[13]
		PVC	**0%**								
∑ -> F4.03	**Mixed-Polymer MR**		**80%**	2.26%	4.53%	1.13%	2.26%%	0.00%		**5.66%**	[14]

Table S-4: Continued.

Code	Process		Coefficient	Scores						Urel	Data
				Reliability	Completeness	Temporal	Geographical	Other	Expert Estimate	total	Source
			[%]	[%]	[%]	[%]	[%]	[%]	[%]	[%]	
∑ -> F5.05	**Grate Incineration:**	LDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		LLDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		HDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PP	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PS	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		EPS	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PET	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PVC	**5%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[9]
∑ -> F5.07	**Fluidized Bed Incineration:**	LDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		LLDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		HDPE	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PP	**1%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PS	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		EPS	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PET	**2%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[8]
		PVC	**5%**	2.26%	13.67%	1.13%	1.13%	2.26%		**14.13%**	[9]

3 Results

The results of the material flow analysis, based on the input data as presented in Section 2 of this document, are supplied in a separate spreadsheet file.

4 Material Flow Analysis diagrams for the product categories

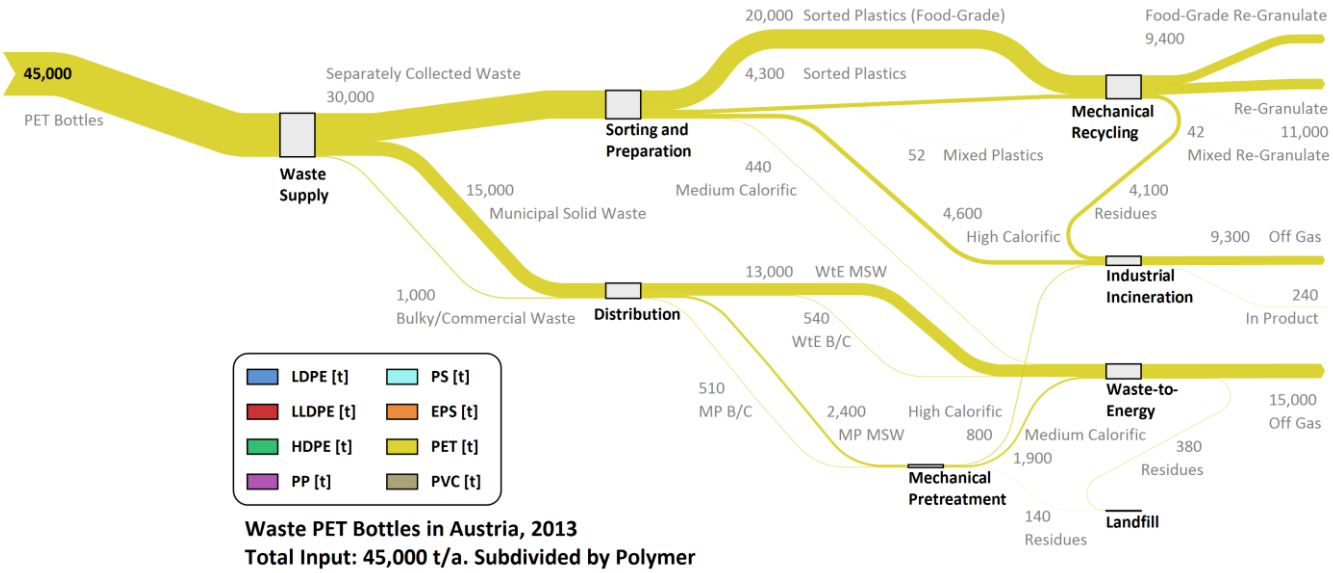

Figure S-3: Results of the material flow analysis for PET bottles

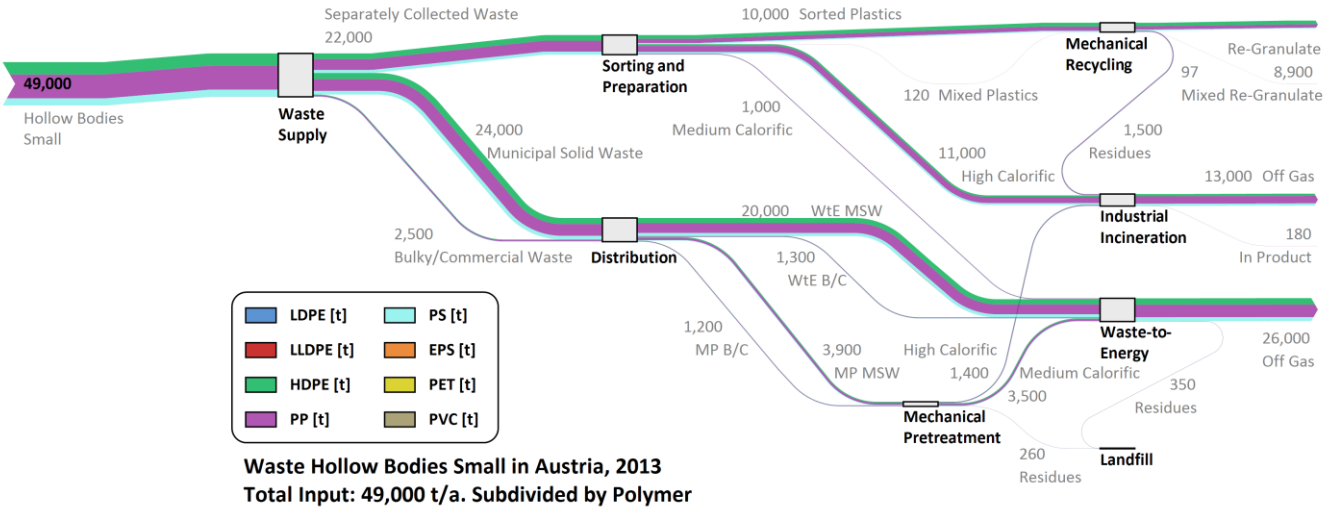

Figure S-4: Results of the material flow analysis for hollow bodies small

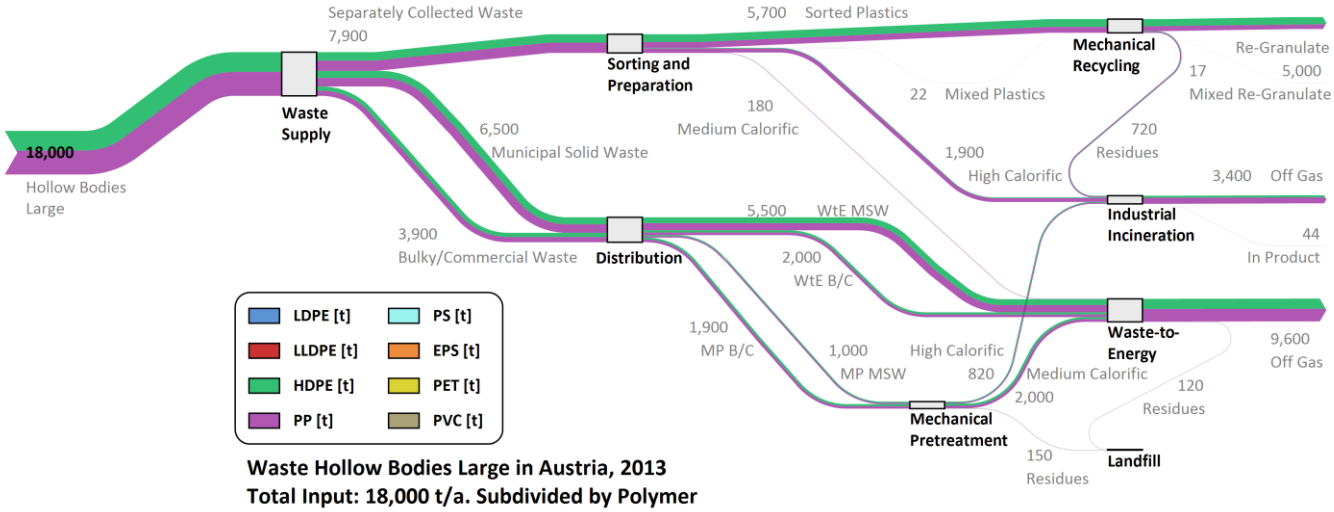

Figure S-5: Results of the material flow analysis for hollow bodies large

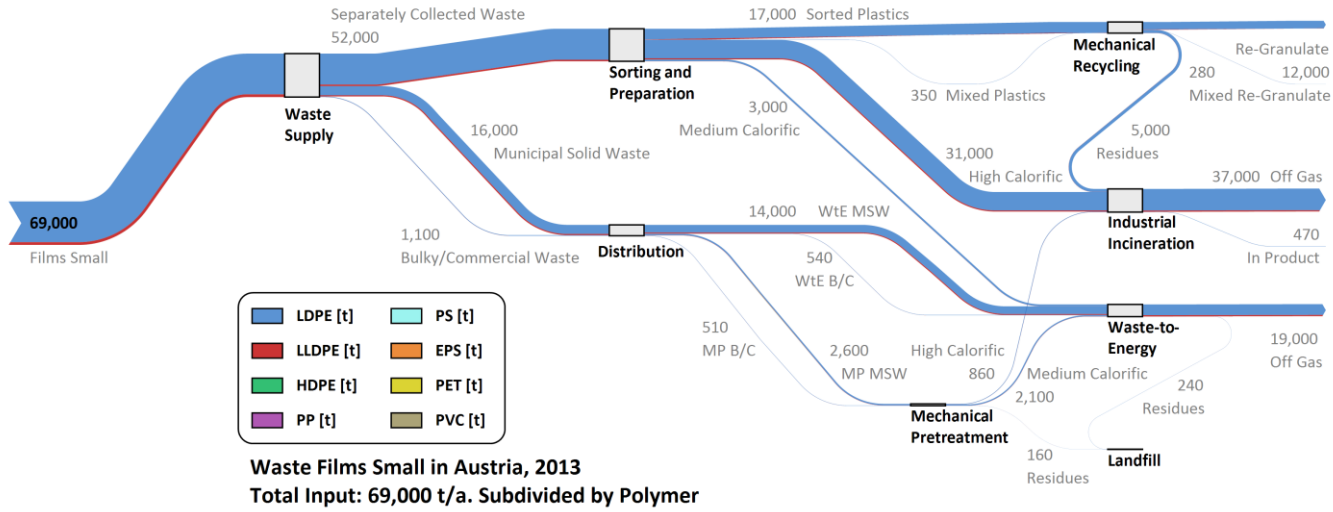

Figure S-6: Results of the material flow analysis for films small

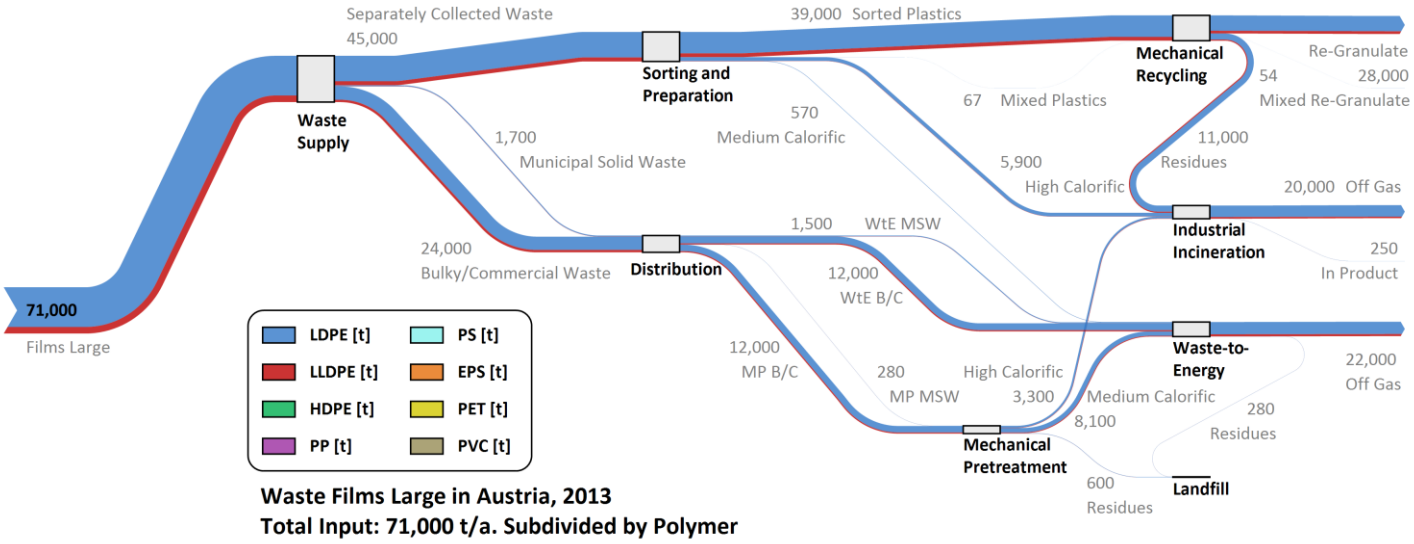

Figure S-7: Results of the material flow analysis for films large

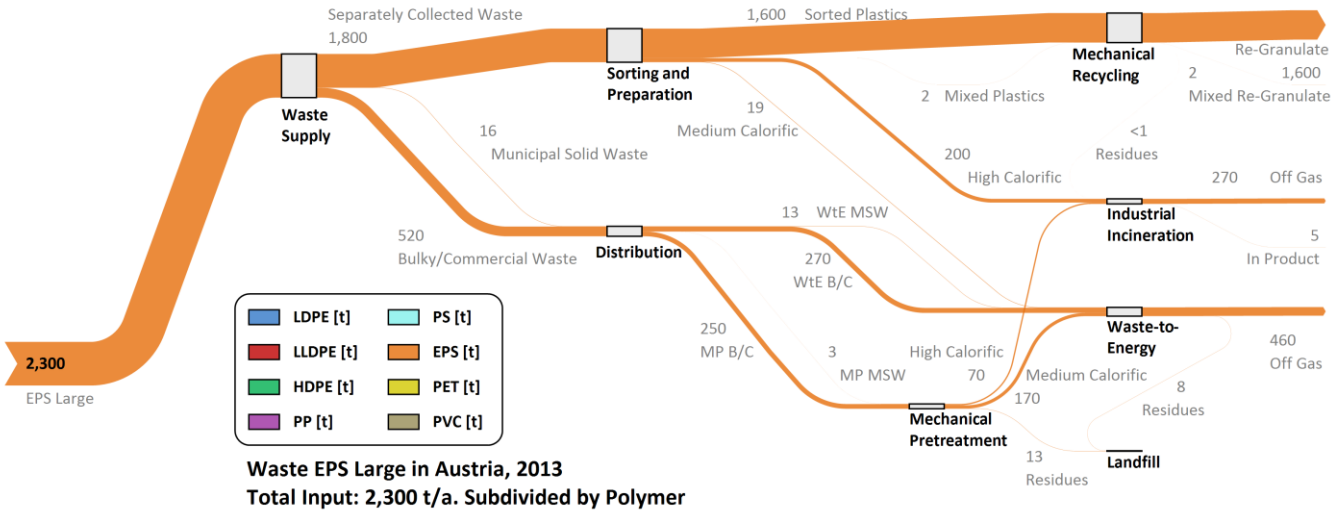

Figure S-8: Results of the material flow analysis for EPS large

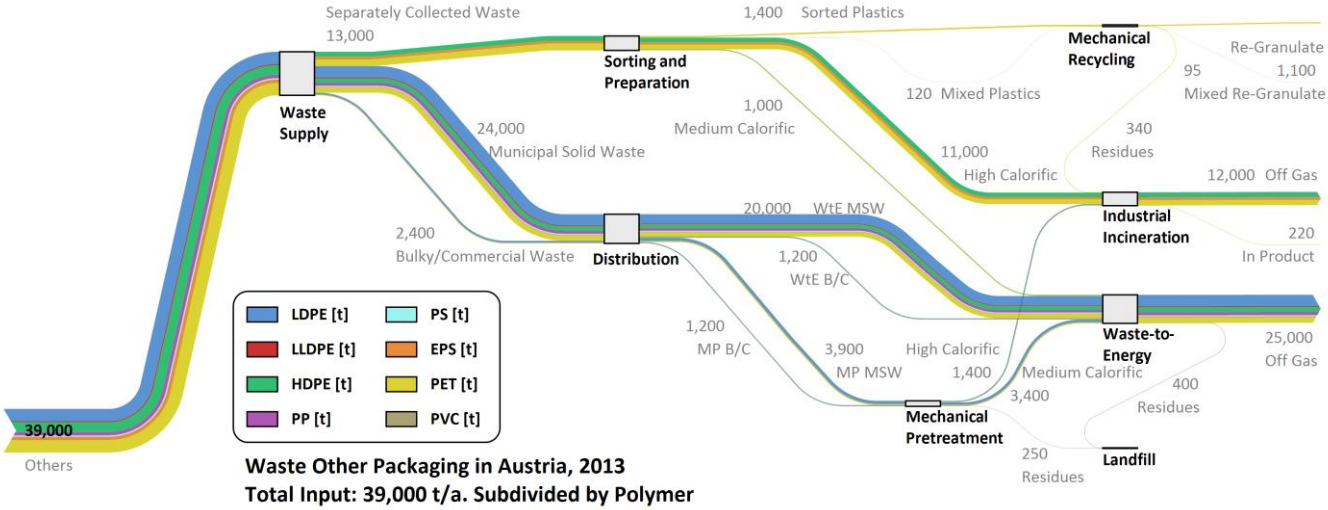

Figure S-9: Results of the material flow analysis for other packaging

5 Results on a polymer basis

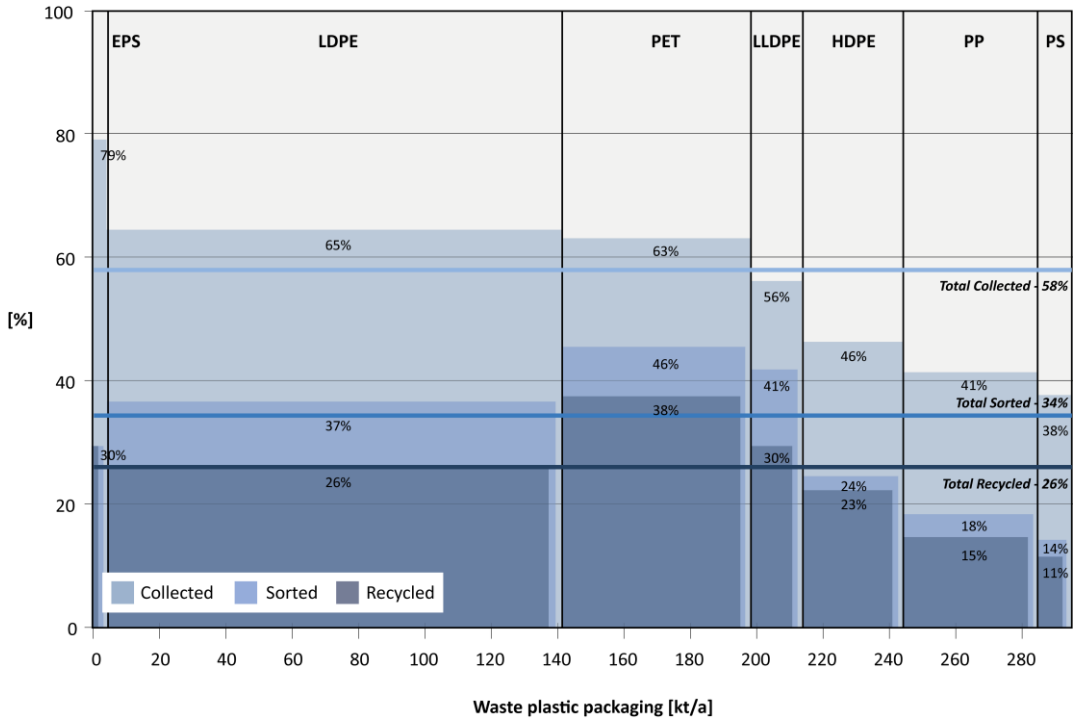

Figure S-10: Comparison of the Collection, Sorting and Recycling Rates, per polymer in relation to the respective mass in the input. The respective rates for the total waste stream are shown by the horizontal lines.

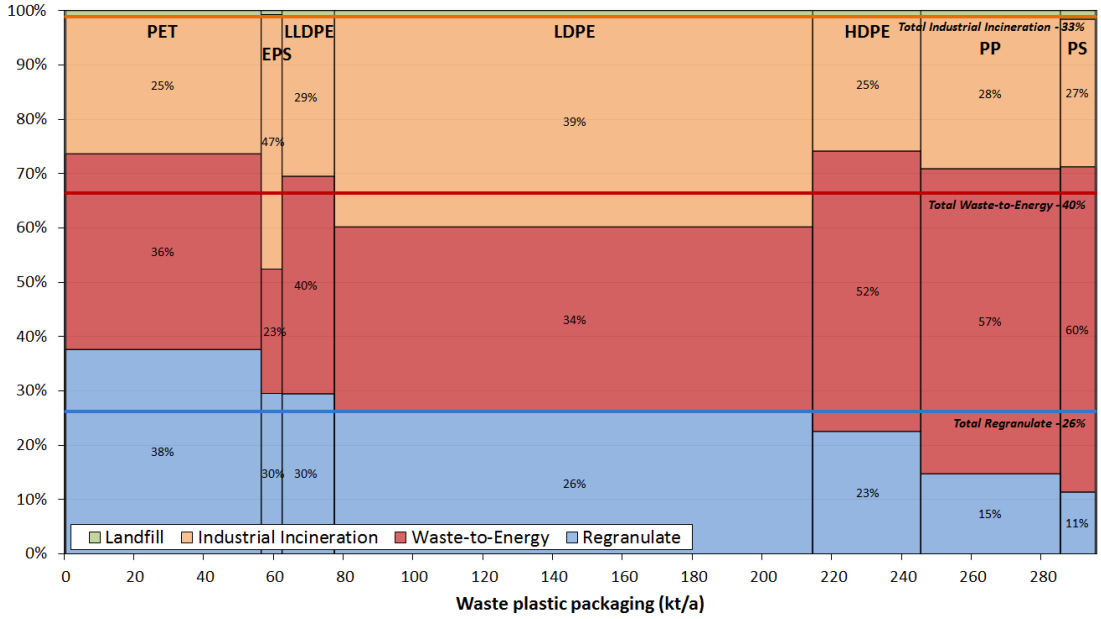

Figure S-11: Final treatment of the polymers in relation to the respective mass in the input. The respective results for the total waste stream are shown by the horizontal lines.

6 References (Supporting Information)

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Circular economy of plastic packaging: current practice and perspectives in Austria

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Supplementary Data

Results of the MFA of plastic packaging in Austria for 2013
The codes and names of the flows correspond to those in Figure 1 of the main article.
The results correspond to those presented in Figure 2 of the main article and Figures SI-3 to SI-9 of the Supporting Information.

Total Waste Stream

		Total		LDPE		LLDPE		HDPE		PP		PS		EPS		PET		PVC	
		Mass (t)	**Urel (%)**	Mass (t)	Urel (%)	Mass (t)	Urel (%)	Mass (t)	Urel (%)	Mass (t)	Urel (%)	Mass (t)	Urel (%)	Mass (t)	Urel (%)	Mass (t)	Urel (%)	Mass (t)	Urel (%)
F1.01	PET Bottles	**45,487**	**2%**	0		0		0		0		0		0		45,487	2%	0	
F1.02	Hollow Bodies Small	**49,176**	**11%**	0		0		14,064	11%	26,948	11%	8,094	11%	0		0		70	11%
F1.03	Hollow Bodies Large	**18,308**	**13%**	0		0		8,262	13%	10,046	13%	0		0		0		0	
F1.04	Films Small	**69,428**	**10%**	65,001	10%	4,384	10%	0		0		0		0		0		43	10%
F1.05	Films Large	**71,185**	**13%**	61,060	13%	10,125	13%	0		0		0		0		0		0	
F1.06	EPS Large	**2,328**	**5%**	0		0		0		0		0		2,328	5%	0		0	
F1.07	Others	**39,476**	**13%**	11,158	13%	924	13%	8,794	13%	3,076	13%	1,219	13%	3,042	13%	11,217	13%	47	13%
F2.01	Separately Collected Waste	**171,820**	**2%**	88,651	2%	8,660	2%	14,375	2%	16,589	2%	3,513	2%	4,251	2%	35,684	2%	97	2%
F2.02	Municipal Solid Waste	**88,019**	**5%**	26,151	5%	2,929	5%	13,931	5%	19,667	5%	5,244	5%	546	5%	19,494	5%	58	5%
F2.03	Bulky/ Commercial Waste	**35,549**	**5%**	22,417	5%	3,844	5%	2,814	5%	3,814	5%	556	5%	573	5%	1,526	5%	5	5%
F3.01	Sorted Plastics (Food-Grade)	**20,151**	**3%**	0		0		0		0		0		0		20,151	3%	0	
F3.02	Sorted Plastics	**79,554**	**2%**	49,833	3%	6,372	3%	7,552	3%	7,244	3%	1,299	3%	1,571	3%	5,684	3%	0	
F3.03	Mixed Plastics	**737**	**2%**	397	3%	23	4%	70	3%	96	3%	23	4%	27	4%	100	3%	1	0%
F3.04	Mixed Plastics	**0**		0		0		0		0		0		0		0		0	
F3.05	Medium Calorific	**6,269**	**3%**	3,374	5%	199	5%	593	5%	812	5%	192	5%	233	5%	856	5%	8	0%
F3.06	Sorting Residues	**0**		0		0		0		0		0		0		0		0	
F3.07	High Calorific	**65,110**	**3%**	35,047	6%	2,065	6%	6,160	6%	8,438	6%	1,999	6%	2,420	6%	8,893	6%	87	6%
F3.08	WtE MSW	**73,936**	**3%**	21,967	6%	2,461	6%	11,702	6%	16,520	6%	4,405	6%	459	6%	16,375	6%	49	6%
F3.09	WtE B/C	**18,253**	**19%**	11,510	29%	1,974	29%	1,445	29%	1,958	29%	285	29%	294	29%	784	29%	3	28%
F3.10	MP MSW	**14,083**	**7%**	4,184	14%	469	14%	2,229	14%	3,147	14%	839	14%	87	14%	3,119	14%	9	14%
F3.11	MP B/C	**17,296**	**20%**	10,907	30%	1,870	30%	1,369	30%	1,856	30%	271	30%	279	30%	742	30%	3	30%
F3.12	Sorted Plastics	**0**		0		0		0		0		0		0		0		0	
F3.13	Mixed Plastics	**0**		0		0		0		0		0		0		0		0	
F3.14	Mixed Plastics	**0**		0		0		0		0		0		0		0		0	
F3.15	Medium Calorific	**21,165**	**12%**	10,179	23%	1,578	25%	2,427	15%	3,374	15%	748	14%	247	24%	2,605	13%	8	13%
F3.16	High Calorific	**8,645**	**12%**	4,158	24%	644	26%	991	17%	1,378	17%	306	16%	101	25%	1,064	16%	3	16%
F3.17	Residues	**1,569**	**12%**	755	24%	117	26%	180	17%	250	17%	55	16%	18	25%	193	16%	1	16%
F4.01	Food-Grade Re-Granulate	**9,354**	**14%**	0		0		0		0		0		0		9,354	14%	0	
F4.02	Re-Granulate	**67,719**	**8%**	35,595	14%	4,552	14%	6,999	14%	5,867	14%	1,052	14%	1,571	14%	12,084	10%	0	0%
F4.03	Mixed Re-Granulate	**589**	**4%**	317	7%	19	7%	56	6%	76	6%	18	7%	22	7%	80	6%	1	6%
F4.04	Residues	**22,779**	**24%**	14,317	34%	1,825	34%	567	169%	1,395	58%	251	57%	5	3976%	4,417	40%	0	23%
F4.05	Off Gas	**0**		0		0		0		0		0		0		0		0	
F4.06	Slag	**0**		0		0		0		0		0		0		0		0	
F4.07	Off Gas	**117,836**	**2%**	46,441	4%	6,133	5%	15,964	4%	22,375	4%	5,539	5%	1,213	4%	20,105	5%	65	15%
F4.08	Residues	**1,787**	**6%**	589	12%	78	12%	202	13%	290	13%	92	13%	20	11%	514	13%	3	12%
F4.09	Off Gas	**95,121**	**6%**	52,851	10%	4,478	14%	7,622	13%	11,068	9%	2,514	7%	2,485	10%	14,016	13%	86	18%
F4.10	In Product	**1,413**	**10%**	670	17%	57	20%	97	19%	143	17%	42	16%	41	17%	358	19%	5	15%

		PET Bottles		**Hollow Bodies Small**					**Hollow Bodies Large**			**Films Small**				**Films Large**			**EPS Large**		**Others**								
		Total	PET	**Total**	HDPE	PP	PS	PVC	**Total**	HDPE	PP	**Total**	LDPE	LLDPE	PVC	**Total**	LDPE	LLDPE	**Total**	EPS	**Total**	LDPE	LLDPE	HDPE	PP	PS	EPS	PET	PVC
		Mass (t)	Mass (t)	**Mass (t)**	Mass (t)	Mass (t)	Mass (t)	Mass (t)	**Mass (t)**	Mass (t)	Mass (t)	**Mass (t)**	Mass (t)	Mass (t)	Mass (t)	**Mass (t)**	Mass (t)	Mass (t)	**Mass (t)**	Mass (t)	**Mass (t)**	Mass (t)	Mass (t)	Mass (t)	Mass (t)	Mass (t)	Mass (t)	Mass (t)	Mass (t)
F1.01	PET Bottles	**45,487**	45,487	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F1.02	Hollow Bodies Small	**0**	0	**49,176**	14,064	26,948	8,094	70	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F1.03	Hollow Bodies Large	**0**	0	**0**	0	0	0	0	**18,308**	8,262	10,046	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F1.04	Films Small	**0**	0	**0**	0	0	0	0	**0**	0	0	**69,428**	65,001	4,384	43	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F1.05	Films Large	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**71,185**	61,060	10,125	**0**	0	**0**	0	0	0	0	0	0	0	0
F1.06	EPS Large	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**2,328**	2,328	**0**	0	0	0	0	0	0	0	0
F1.07	Others	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**39,476**	11,158	924	8,794	3,076	1,219	3,042	11,217	47
F2.01	Separately Collected Waste	**29,557**	29,557	**22,239**	6,414	12,280	3,513	32	**7,853**	3,544	4,309	**51,964**	49,495	2,437	32	**45,379**	39,156	6,223	**1,794**	1,794	**13,033**	0	0	4,417	0	0	2,457	6,127	32
F2.02	Municipal Solid Waste	**14,884**	14,884	**24,390**	6,938	13,302	4,116	34	**6,505**	2,935	3,569	**16,413**	14,573	1,830	10	**1,736**	1,474	262	**16**	16	**24,076**	10,105	837	4,057	2,796	1,129	530	4,610	13
F2.03	Bulky/ Commercial Waste	**1,046**	1,046	**2,547**	711	1,366	466	3	**3,950**	1,782	2,167	**1,051**	934	117	1	**24,070**	20,431	3,639	**518**	518	**2,367**	1,053	87	320	280	90	55	480	1
F3.01	Sorted Plastics (Food-Grade)	**20,151**	20,151	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F3.02	Sorted Plastics	**4,291**	4,291	**10,356**	4,317	4,740	1,299	0	**5,738**	3,235	2,504	**17,391**	16,877	514	0	**38,814**	32,956	5,858	**1,571**	1,571	**1,393**	0	0	0	0	0	0	1,393	0
F3.03	Mixed Plastics	**52**	52	**121**	21	77	23	0	**22**	3	18	**353**	333	20	0	**67**	63	4	**2**	2	**119**	0	0	45	0	0	0	25	48	0
F3.04	Mixed Plastics	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F3.05	Medium Calorific	**445**	445	**1,033**	182	655	192	3	**184**	27	157	**3,005**	2,835	167	3	**571**	539	32	**19**	19	**1,012**	0	0	384	0	0	214	412	3
F3.06	Sorting Residues	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F3.07	High Calorific	**4,619**	4,619	**10,729**	1,893	6,808	1,999	29	**1,909**	279	1,630	**31,214**	29,449	1,736	29	**5,927**	5,597	330	**202**	202	**10,509**	0	0	3,988	0	0	2,218	4,274	29
F3.08	WtE MSW	**12,503**	12,503	**20,488**	5,828	11,173	3,457	29	**5,464**	2,466	2,998	**13,787**	12,241	1,537	9	**1,458**	1,238	220	**13**	13	**20,224**	8,488	703	3,408	2,349	948	445	3,872	11
F3.09	WtE B/C	**537**	537	**1,308**	365	702	239	2	**2,028**	915	1,113	**540**	479	60	0	**12,359**	10,490	1,869	**266**	266	**1,216**	541	45	164	144	46	28	247	1
F3.10	MP MSW	**2,381**	2,381	**3,902**	1,110	2,128	659	6	**1,041**	470	571	**2,626**	2,332	293	2	**278**	236	42	**3**	3	**3,852**	1,617	134	649	447	181	85	738	2
F3.11	MP B/C	**509**	509	**1,239**	346	665	226	2	**1,922**	867	1,055	**512**	454	57	0	**11,711**	9,940	1,771	**252**	252	**1,152**	512	42	156	136	44	27	234	1
F3.12	Sorted Plastics	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F3.13	Mixed Plastics	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F3.14	Mixed Plastics	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F3.15	Medium Calorific	**1,949**	1,949	**3,468**	982	1,884	597	5	**1,998**	902	1,096	**2,116**	1,879	236	1	**8,086**	6,864	1,223	**172**	172	**3,375**	1,436	119	543	394	151	75	655	2
F3.16	High Calorific	**796**	796	**1,416**	401	769	244	2	**816**	368	448	**864**	767	96	1	**3,303**	2,803	499	**70**	70	**1,379**	587	49	222	161	62	31	268	1
F3.17	Residues	**145**	145	**257**	73	140	44	0	**148**	67	81	**157**	139	17	0	**599**	509	91	**13**	13	**250**	106	9	40	29	11	6	49	0
F4.01	Food-Grade Re-Granulate	**9,354**	9,354	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F4.02	Re-Granulate	**11,002**	11,002	**8,892**	4,001	3,839	1,052	0	**5,026**	2,998	2,028	**12,422**	12,055	367	0	**27,725**	23,540	4,184	**1,571**	1,571	**1,081**	0	0	0	0	0	0	1,082	0
F4.03	Mixed Re-Granulate	**42**	42	**97**	17	62	18	0	**17**	3	15	**283**	267	16	0	**54**	51	3	**2**	2	**95**	0	0	36	0	0	20	39	0
F4.04	Residues	**4,096**	4,096	**1,488**	320	916	251	0	**717**	237	479	**5,040**	4,889	151	0	**11,103**	9,429	1,674	**0**	0	**336**	0	0	9	0	0	5	321	0
F4.05	Off Gas	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F4.06	Slag	**0**	0	**0**	0	0	0	0	**0**	0	0	**0**	0	0	0	**0**	0	0	**0**	0	**0**	0	0	0	0	0	0	0	0
F4.07	Off Gas	**15,049**	15,049	**25,944**	7,266	14,230	4,412	36	**9,552**	4,256	5,296	**19,204**	17,216	1,975	12	**22,193**	18,891	3,302	**463**	463	**25,431**	10,334	856	4,443	2,849	1,127	750	5,056	16
F4.08	Residues	**384**	384	**352**	92	184	73	2	**123**	54	69	**244**	218	25	1	**281**	240	42	**8**	8	**396**	131	11	56	37	19	12	129	1
F4.09	Off Gas	**9,274**	9,274	**13,449**	2,582	8,385	2,453	29	**3,399**	874	2,525	**36,652**	34,666	1,958	28	**20,079**	17,606	2,472	**268**	268	**12,000**	579	48	4,166	159	61	2,217	4,742	29
F4.10	In Product	**237**	237	**184**	33	109	41	2	**44**	11	33	**466**	440	25	1	**255**	223	31	**4**	4	**223**	7	1	53	2	1	37	121	2

Paper III

Integrating high-resolution material flow data into the environmental assessment of waste management system scenarios – the case of plastic packaging in Austria

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Main article

Integrating high-resolution material flow data into the environmental assessment of waste management system scenarios – the case of plastic packaging in Austria.

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Abstract

The environmental performance of the waste management system of plastic packaging in Austria was assessed using a combination of high-resolution material flows and input-dependent life cycle inventory data. These data were used to evaluate different configurations of the waste management system, reflecting the system structure as it was in 1994 in Austria and still is in some of the new EU member states, as well as a situation achieving the increased circular economy targets to be met by 2030. For the latter, two options, namely single-polymer recycling and mixed-polymer recycling, were investigated. The results showed that the status quo achieves net benefits for 15 out of 16 impact categories evaluated. Regarding the alternative scenarios, for most impact categories these benefits increased with increasing recycling rates, although for four impact categories the highest net benefit was achieved by the status quo. For many impact categories the marginal environmental benefit decreased at higher recycling rates, indicating that there is an environmentally optimal recycling rate below 100%. The results also highlight the importance of high-quality single-polymer plastics recycling from an environmental perspective because utilizing mixed polymer recycling to achieve circular economy targets would result in lower environmental benefits than the status quo.

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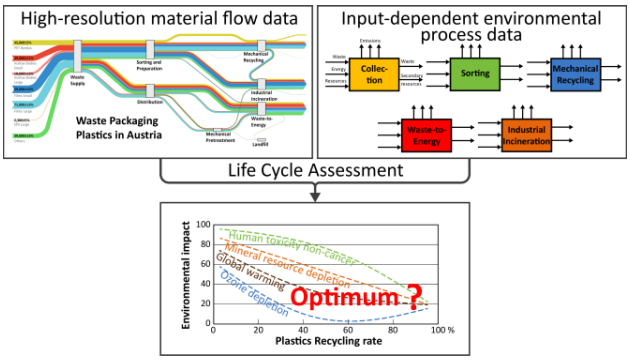

1 Introduction

The European Commission (EC) has described the circular economy in general and a responsible use and proper waste management of plastics in particular as key elements for developing a sustainable and resource efficient economy.[1, 2] To encourage more recycling-oriented waste management systems (WMS), a considerably increased recycling target was adopted for plastic packaging, which constitutes the largest post-consumer plastic waste stream,[3] from currently 22.5%[4] to 55% by 2030,[5] a target calculated in terms of the mass entering the final recycling facility (i.e. the recycling rate). However, formulating quantitative recycling targets without underlying environmental assessments of both the current situation and the possible effects of increasing recycling rates may be counterproductive since trade-offs within the WMS, such as decreased energy production from waste incineration,[6] might not be accounted for. These recycling targets build on the waste hierarchy, prioritizing prevention, re-use, recycling and energy recovery over other treatment options, in that order.[7] Waste prevention (including reuse) has been generally accepted as the preferred option,[8, 9] but for the other alternatives, the a priori use of this generalized hierarchy has been criticized as the performance of a WMS strongly depends on local conditions, e.g. due to specific waste compositions or treatment process efficiencies.[8, 10] It is thus crucial to assess the environmental performance of the entire plastic packaging WMS of a region from a systems perspective in order to identify optimal system configurations.

Although a number of life cycle assessment (LCA) studies on the waste management of plastic (packaging) have been carried out, many studies focus on the plant or technology level (see e.g. refs[6, 11-19]) often taking only one polymer or product into account (see e.g. refs[20-22]). Other studies have taken a more systemic approach, looking at the overall WMS of plastics specifically (see e.g. refs[23-26]) or as a part of packaging waste (see e.g. refs[27, 28]) or

municipal solid waste (MSW) as a whole (see e.g. refs[29-33]), although detailed information on the actual waste flows is usually lacking. Region-specific knowledge on the structure of the WMS and the amount and composition of the plastic waste, in terms of polymers as well as elements, is thus usually not considered or reported in detail, although this has a considerable influence on the result.[34]

Therefore, the LCA method is combined with material flow analysis (MFA) in this study, enabling detailed information regarding the availability, composition and flows of the waste through the WMS to be connected with input-dependent data on the environmental performance of each of the waste treatment processes. This combination allows for evaluating all the treatment options simultaneously to assess the system as a whole and to identify trade-offs (e.g. between material and energetic utilization) within the WMS. Furthermore, different scenarios are evaluated by changing the configuration of the waste flows, reflecting e.g. policy priorities.

Two studies have taken a similar holistic approach for the assessment of a WMS for MWS. Turner et al.[35] assessed the environmental performance regarding climate change of the existing management of the waste collected in the city of Cardiff, Wales, and compared this with three alternative systems reflecting policy options to enhance food waste management, incineration and dry recycling. The plastic waste stream was characterized by 11 fractions comprising six polymers, which were characterized by three data sets regarding the elemental composition, whereas the modelling of the recycling of plastics used three distinct data sets. Haupt et al.[9] investigated the current WMS in Switzerland, which was compared with five scenarios highlighting improvement potentials regarding recycling of various materials and increased energy recovery efficiencies. Three polymers were considered, and the recycling was modelled using three data sets in two versions each (one domestic and one European).

The present study differs from previous work in three main aspects. First, we focus on one (highly relevant) part of MSW specifically, enabling the assessment of the plastics packaging WMS and the waste compositions in more detail. The MFA on which this work builds[36] quantified the flows through the WMS of seven product types and eight polymers, defined by five distinct elemental composition data sets. A product- and polymer-specific approach furthermore enables the identification of the most effective measures in terms of environmental potential when increasing the recycling rate of plastics packaging as demanded by the EU directive[5]. Second, instead of constructing several scenarios to compare a range of different policy options, this study explicitly explores the relationship between the recycling rate and the environmental performance of the WMS. This acknowledges the fact that waste management policies are often based on minimum recycling rates as targets, as is the case for (plastic) packaging. A specific policy target, the recent EU packaging waste directive,[5] is thus assessed regarding its environmental performance, which is done considering a wide range of impact categories to investigate whether burden shifting goes along with increasing the recycling rate. Guidance can thus be provided on the extent to which a higher recycling target makes sense from an environmental perspective and whether an optimal recycling rate exists beyond which increasing rates are counterproductive (see also Haupt et al.[37]). Third, the parameter uncertainties are comprehensively considered using a global sensitivity analysis framework[38] to assess the robustness of the results and identify the most influential parameters in the model.

The aim of this study is to assess the environmental performance of the status quo (with 2013 as the reference year) of the WMS of plastic packaging in Austria. Furthermore, the effect of changes in recycling rates on the environmental performance of the WMS is explored by assessing plastic packaging waste management in a landfill disposal-dominated

system and in a situation where the recently increased recycling target is met. The environmental performance of increasing recycling targets as a policy instrument is thus investigated.

2 Materials and methods

MFA is used to systematically analyze the flows and stocks of materials in a system which is defined in space and time,[39] whereas LCA is a comprehensive method which quantifies all relevant emissions and consumed resources along the full life cycle of a product or service, and subsequently assesses the resulting impacts on human and environmental health as well as resource depletion.[40]

A high-resolution MFA for the status quo of the plastic packaging WMS in Austria was carried out by Van Eygen et al.[36], who described the waste mass flows, compositions and transfer coefficients throughout the system with regard to polymers and product types. This level of detail is crucial as the environmental benefits of secondary material and energy production depend strongly on the composition of the treated waste stream in each of these processes.[41]

For the LCA, a consequential modelling framework (context situation B according to EC[40]) is applied as it is intended to support decision-making and because actions in the WMS can be expected to have effects on the background system.[42] System expansion by substitution is thus used to solve multifunctionality, and marginal background data are taken for processes with large-scale, structural consequences, as recommended by the EC[40]. Furthermore, a generation-based functional unit is selected,[34] meaning that the treatment of the total amount of waste plastic packaging generated in 2013 in Austria is assessed, whereby the burden of the incoming waste is attributed to the producer (zero burden assumption).[42] The

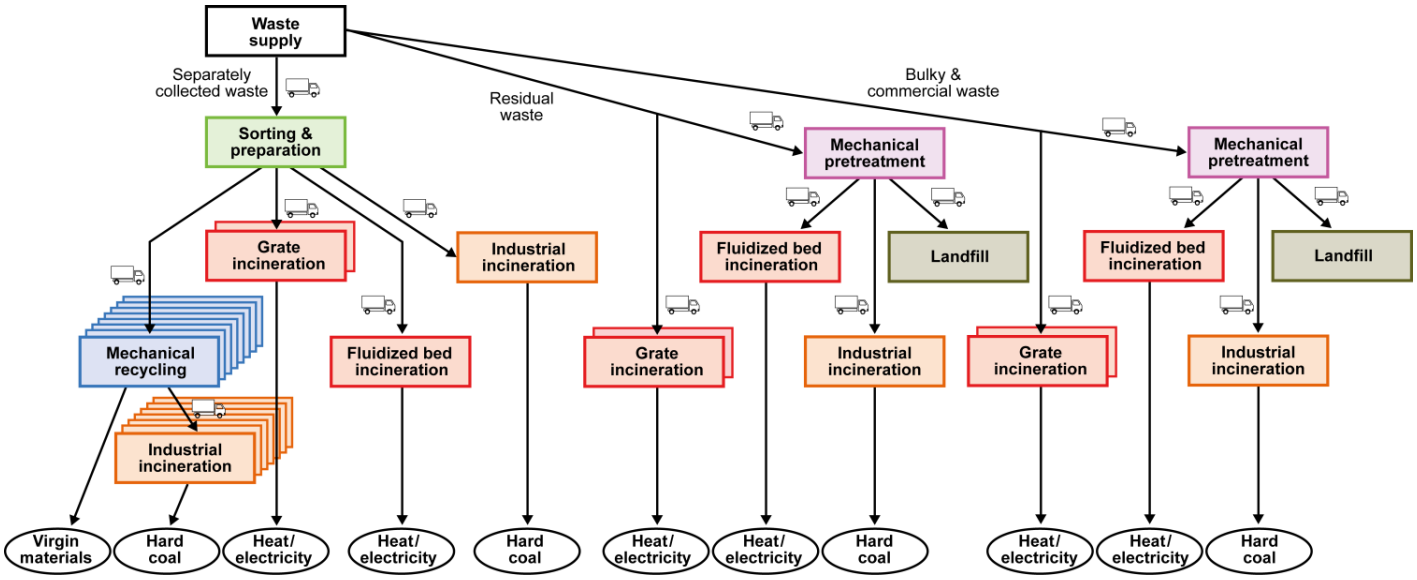

Figure 1: System boundaries and model overview with each of the treatment process modules.

system boundaries are shown in Figure 1, which include capital goods for each of the processes as these may make a significant contribution to the overall results.[43]

The status quo is compared to three scenarios with respect to the management of waste plastics packaging. One represents a WMS where mainly landfill disposal is used (="Mainly disposal"), and is constructed based on how plastic packaging waste was collected, sorted and treated in Austria in 1994, which is comparable to the situation in some of the new EU member states, especially regarding the share of landfilling.[44] Furthermore, two scenarios represent a WMS which achieves the increased recycling target of 55%:[5] one continuing the current focus on (high-quality) single-polymer recycling (="EU-target-SP", single-polymer) and one achieving the required recycling amount through mixed-polymer recycling only (="EU-target-MP", mixed-polymer).

It should be noted that the scenarios only differ with respect to the routing of the waste flows through the system. The mass and composition of the input, and thus the functional unit, as well as the efficiencies of the final treatment processes were kept constant. The alternative scenarios thus represent different technology levels and waste management

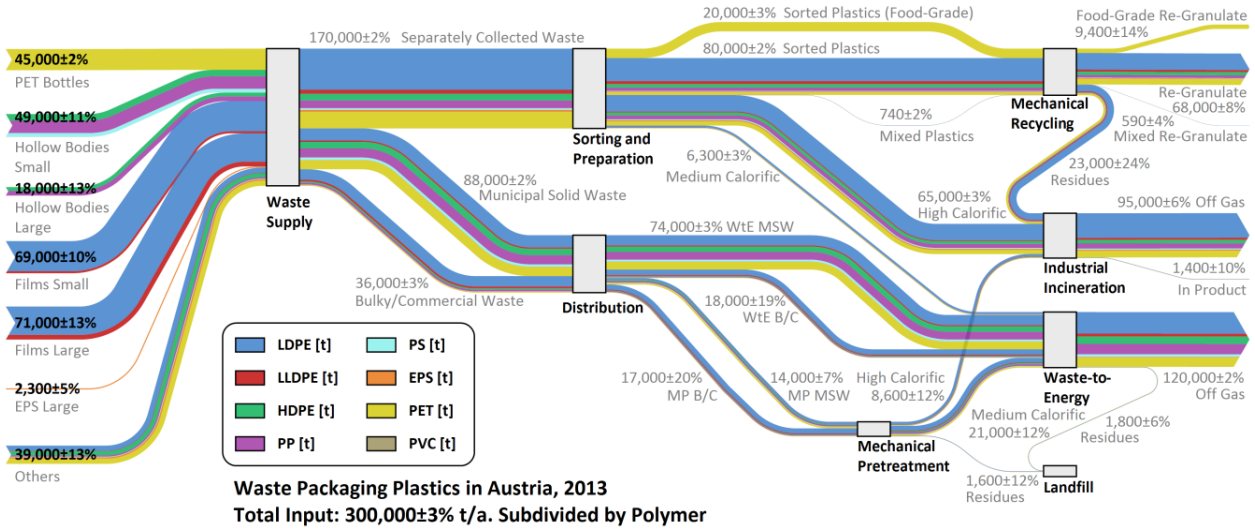

Figure 2: Plastic packaging flows in the status quo of the Austrian waste management system, subdivided by the eight polymers under consideration. Reprinted with permission from Van Eygen et al.[36].

priorities with respect to separate collection, sorting and pretreatment, but do not depict historic and future states of the Austrian system.

2.1 Life cycle inventory

The foreground processes are shown in Figure 1, and for most of these processes primary data were collected, whereas the background system was modelled using the ecoinvent v3.2 database (cut-off system model).[45] The software EASETECH was used, which uses a reference flow which can be described with respect to the material composition, allowing tracking of single materials or elements through the system and input-dependent calculation (i.e. in relation to the waste composition) of the environmental impacts.[46]

In the next sections, the main aspects for each of the treatment process modules will be discussed, whereas a more comprehensive discussion can be found in Section 1 in the Supporting Information (SI), with full tables detailing all input data used in the model.

2.1.1 Waste input and mass flows

The composition of the incoming waste has a large influence on an assessment of a WMS.[34, 41] For the MFA, of which the results regarding the polymer composition are presented in Figure 2, seven product types were taken into account, as well as eight polymers: low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), high-density polyethylene (HDPE), polypropylene (PP), polystyrene (PS), expanded polystyrene (EPS), polyethylene terephthalate (PET), and polyvinyl chloride (PVC). More information on the input composition with respect to product type, polymer and 23 elements is provided in Tables S1 and S2 in the SI.

For the mainly disposal scenario, being based on the situation in 1994, information was available on the separate collection and further treatment of packaging waste flows[47] as well as on the amount present in the residual waste (RW).[48] The separate collection rate amounted to 28% in 1994 (compared to 58% for the status quo), of which 17% was sent to mechanical recycling after sorting (i.e. the recycling rate; 34% for the status quo) while the remaining 11% were stored for later incineration in the cement industry (33% for the status quo, where the cement industry treats residues from mechanical recycling and mechanical pretreatment as well). The RW was mainly landfilled, resulting in an overall landfill rate of 58% (1% for the status quo), whereas 18% of the waste stream was utilized in WtE plants (40% for the status quo). An overview of these waste flows is presented in Figure S1 in the SI.

The scenarios achieving the increased recycling target are based on an effort by Van Eygen et al.[36], where the theoretically required product type-specific separate collection and sorting efficiencies were determined. The fact that only these efficiencies were altered, and no non-linear effects of increasing separate collection on e.g. transport distance or purity of the collected material[37] were taken into account, means that these scenarios can be

184 considered to be optimistic. The waste flows in these scenarios are illustrated in Figure S2
185 and Figure S3 in the SI, and are characterized by a separate collection rate of 74% and a
186 recycling rate of 56%. WtE plants incinerate 27% of the total waste amount, whereas 29%
187 is utilized by industrial incineration and 1% is landfilled.

188

189 **2.1.2 Mechanical recycling**

190 Eight distinct mechanical recycling processes were taken into account, for three of which
191 (PET to food-grade, LDPE and EPS recycling), primary data were available from multiple
192 Austrian plants, whereas literature data were used to model the remaining recycling
193 processes.[15, 49-51]

194 Determining to what extent virgin material is substituted, and hence the avoided burden,
195 has considerable influence on the overall results.[52] To estimate the quality loss and market
196 uptake of recycled materials value-corrected substitution (VCS)[52-55] was used, which is
197 calculated as the ratio of the market prices of secondary to primary material. This method
198 has been used for the assessment of plastics recycling systems,[32, 56, 57] resulting in
199 substitution factors ranging from 0.35 (comparing prices of sorted bales to primary
200 material) to 0.60 and 0.81 (comparing secondary and primary plastics).

201 In this study the VCS method is used for the single-polymer recycling of LDPE/LLDPE,
202 HDPE, PP and PS. Weekly prices for primary[58] and monthly prices for secondary plastics[59]
203 over a time period of two years were analyzed, resulting in the substitution factors ranging
204 from 0.61 to 0.76 (see Tables S11 and S12 in the SI). These factors proved to be relatively
205 stable over time, making them suitable as substitution factors. For the food-grade recycling
206 of PET bottle waste, however, a substitution factor of 1 is taken since this re-granulate is

subject to strict quality and food safety requirements,[20, 60, 61] while the same can be expected for PET to fiber recycling as well.[22]

As the re-granulates of EPS recycling substitute a specific product, and the re-granulates from mixed-polymer recycling substitute other materials rather than plastics, a physical substitution factor was calculated based on the product equivalencies. In the first process, EPS beads are produced for thermal insulation, substituting insulation material made from EPS foam slab, whereas the mixed-polymer re-granulate is assumed to be used for the production of two possible products:[15] a plant tray made from PS concrete (a type of concrete substituting cement by polymers) and a bench made from wood with a cast iron pedestal. For a more detailed discussion on all substitution factors see Section 1.4 in the SI.

2.1.3 Waste-to-Energy

Three plant archetypes were taken into account for the modelling of WtE: two grate incinerators, one with a wet and one with a dry air pollution control (APC) system, as well as one fluidized bed incinerator with wet APC.[62, 63]

The most important parameters for the environmental performance of waste incinerators are the energy efficiency and the substituted energy technology.[32, 64] The marginal electricity technology was determined to be coal-fired combined heat and power (CHP) as the use of coal has seen a decline both in terms of the share of total energy production and the number of operational plants.[65, 66] For the marginal heat technology, the specific situation of each incineration plant needs to be taken into account due to the local nature of district heating networks, resulting in four selected marginal heat technologies across all Austrian plants.[67]

2.1.4 Industrial incineration

Industrial incineration occurs in the cement industry, where waste plastics are increasingly used as a refuse-derived fuel substituting mainly hard and brown coal.[68] No primary data were available from the cement industry, so a modular excel-based technology-specific LCA model from the LCA4Waste tool was used,[69, 70] with which scenarios for clinker production in a cement kiln can be compared with regard to the environmental performance. Two identical scenarios for a typical Austrian cement kiln were constructed, one of which uses hard coal as a fuel while the other uses plastic wastes. For each of the ten industrial incineration modules in the model (see Figure 1), the specific waste composition was taken into account. The substitution of hard coal was done based on the energy of each of the fuels needed to produce one tonne of clinker.

2.1.5 Uncertainty characterization

LCA results are inherently subject to uncertainties, which can be subdivided into three groups: parameter, scenario and model uncertainties.[71, 72] To assess the robustness of the results, the parameter uncertainties of the input data were characterized and propagated through the model, whereas the scenario uncertainties were investigated in a scenario analysis. Model uncertainties, however, were outside the scope of this study.

The characterization of the uncertainty of the input parameters is a common difficulty as these values are usually not available.[34] In this study, where statistical analyses of the input data were not feasible or uncertain input data itself were not present, the pedigree matrix approach, applying the uncertainty factors incorporated into the ecoinvent database,[73] was used. The latter approach is in line with the uncertainty characterization of the underlying

MFA results.[36, 74] The propagation of the uncertainties was performed analytically,[38] and is further discussed in Section 1.1 in the SI.

2.2 Life cycle impact assessment

Including all relevant impact categories (ICs) for the life cycle impact assessment (LCIA) ensures the coverage of a wide range of environmental consequences and prevents potential burden shifting.[34] Therefore, the 14 midpoint ICs and their respective LCIA characterization models, as reported and recommended by the ILCD, were evaluated as these were found to represent best practice.[75, 76] Two ICs were further subdivided though: aquatic eutrophication into freshwater and marine eutrophication, and mineral and fossil resource depletion into one covering minerals and metals and one covering fossil resources. The following 16 ICs were thus taken into account:

- Global warming (GW): IPCC[77]
- Ozone depletion (OD): WMO[78]
- Human toxicity, cancer effects (HTc): USEtox[79]
- Human toxicity, non-cancer effects (HTnc): USEtox[79]
- Particulate matter (PM): Humbert[80]
- Ionizing radiation (IR): Frischknecht et al.[81]
- Photochemical ozone formation (POF): ReCiPe[82]
- Terrestrial acidification (TA): Seppälä et al.[83,] Posch et al.[84]
- Terrestrial eutrophication (TE): Seppälä et al.[83,] Posch et al.[84]
- Freshwater eutrophication (FE): ReCiPe[85]
- Marine eutrophication (ME) : ReCiPe[85]
- Ecotoxicity (ET): USEtox[79]

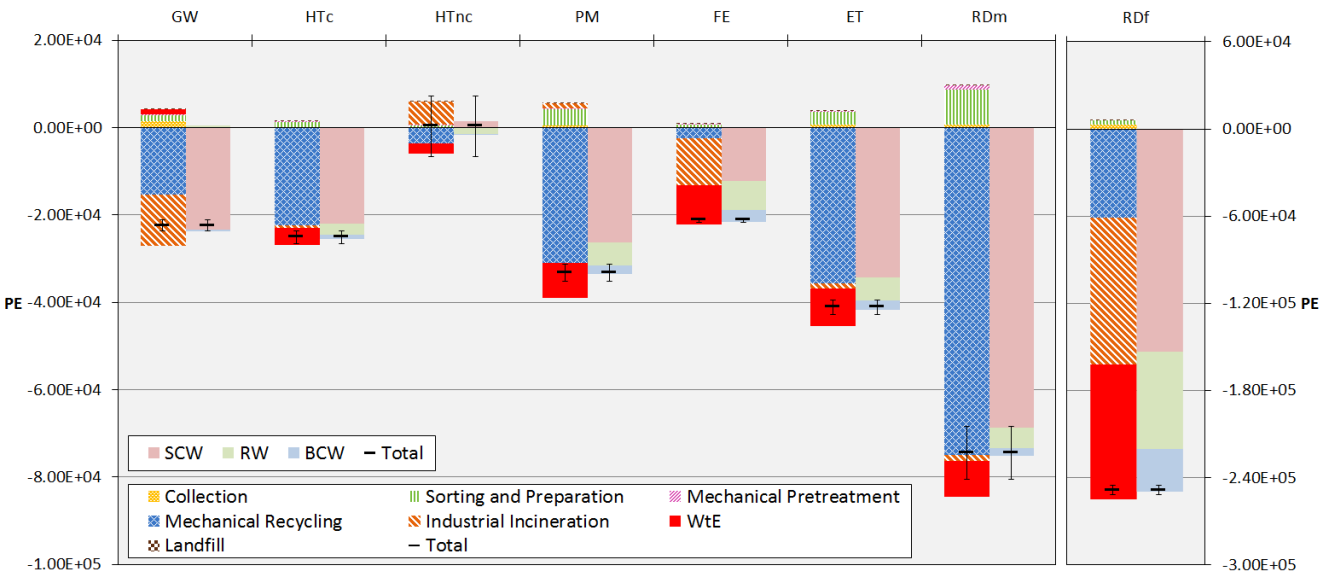

Figure 3: Net LCIA results of the status quo (given in person equivalents PE) for eight selected impact categories, subdivided respectively by treatment process (left bars) and by collection route (right bars). Figure S5 shows these results with separate burdens and benefits. SCW: separately collected waste; RW: residual waste; BCW: bulky & commercial waste.

- Land use (LU): Milà i Canals et al.[86]
- Resource depletion, water (RDw): Frischknecht et al.[87]
- Resource depletion, minerals and metals (RDm): CML[88]
- Resource depletion, fossil (RDf) : CML[88]

The characterized results were normalized to person equivalents (PE) using the global normalization factors for 2010 established by Laurent et al.[89] Finally, a time horizon of 100 years was taken for all ICs.

3 Results

3.1 Status quo

The net LCIA results for each treatment process and collection route for the status quo are presented in Figure 3 for eight selected ICs, whereas the full range of ICs is presented in Figure S4 and Table S34 in the SI and the burdens and benefits for each treatment process

and collection route separately are shown in Figure S5 and Table S35 in the SI. For all ICs, with the exception of HTnc, the benefits achieved by the WMS of the status quo are higher than the impacts, resulting in net negative impacts. For OD, LU, RDw and HTnc, however, the break-even point lies within one standard deviation of the result. The collection, sorting, pretreatment and landfill processes cause relatively low burdens compared to the benefits and burdens related to mechanical recycling, industrial incineration and WtE. Of these latter processes, mechanical recycling causes rather low burdens but large benefits for most ICs, whereas industrial incineration leads to both considerable burdens and benefits of about the same order of magnitude. Balancing these benefits and burdens results in net burdens for six of the 16 ICs. For WtE on the other hand, both burdens and benefits are generally lower than for industrial incineration but net burdens are only caused for GW. So all in all, for most of the ICs the benefits are mainly achieved by the mechanical recycling processes, although, especially for FE, OD, IR, LU and RDf, the incineration processes make important contributions as well.

The PET food-grade and LDPE recycling processes together account for at least 60% of the benefits of mechanical recycling across all ICs, for the latter because of the high mass treated (56% of the total amount) and for the former because of the high specific net benefit. These specific burdens and benefits for each of the mechanical recycling processes are shown in Figure S7 in the SI. EPS recycling has by far the highest specific net benefit for 13 ICs due to the low operational burdens, high-impact substituted product and high substitution factor. Of the more important recycling processes from a mass perspective, PET food-grade has the largest specific net benefit closely followed by PET fiber across most ICs, whereas the LDPE, HDPE and PP processes show similar results. The results of the three WtE process archetypes are governed by the respective energy efficiencies, causing the

benefits for all grate incinerators with wet APC, whereas both grate incinerators with dry APC (GW, HTnc) and fluidized bed incinerators (GW) cause some net burdens as well.

From a mass perspective, the separately collected waste (SCW) amounts to 58% of plastic packaging waste, whereas the RW accounts for 30% and the bulky and commercial waste (BCW) for 12% (see Figure 2). However, it is clear from Figure 3 that the treatment of the SCW achieves the vast majority of all benefits of the overall WMS. This is mainly caused by the high contributions made by the mechanical recycling processes to the overall benefits, but also by the lower benefits of WtE for the GW, FE and RDf ICs compared to industrial incineration, for which about 90% of the input comes from the SCW. The exception is again HTnc, where the net impacts of industrial incineration more than negate the benefits achieved by mechanical recycling.

3.2 Alternative waste management system scenarios

In general, the more material is mechanically recycled, the higher the overall benefits are as mechanical recycling makes the highest contribution for most of the ICs (see Figure 4a and Figure S6 in the SI). The exceptions are FE, RDf, OD, IR, LU and RDw, where considerable benefits are achieved by incineration activities, and directing waste material away from these processes results in a similar or even decreased net benefit for the EU-target-SP scenario. The high net benefits achieved by the EU-target-MP scenario for the toxicity-related ICs (HTc, HTnc, ET) are explained solely by the substitution of cast iron (commonly used for street benches). This strong sensitivity regarding the choice of substituted product is further discussed in the scenario analysis. For the other ICs, net benefits similar to or lower than those of the status quo are achieved, indicating the validity of the focus by the EC on high-quality single-polymer recycling.[1]

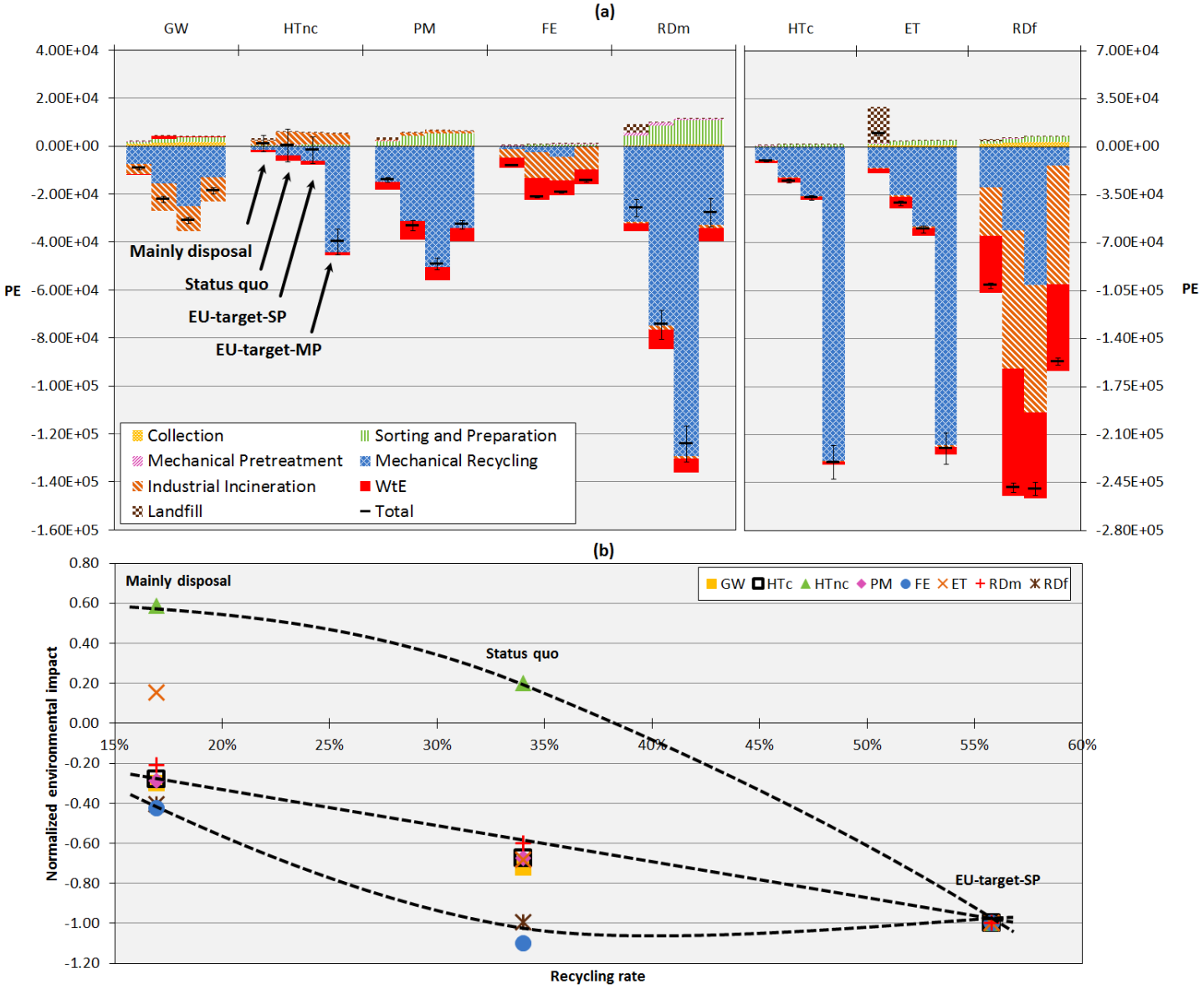

Figure 4: (a) Net LCIA results of the status quo and 3 scenarios subdivided by treatment process for eight selected impact categories, and (b) normalized (EU-target-SP=-1) LCIA results for the mainly disposal, status quo and EU-target-SP scenarios in relation to the recycling rate. Three types of relationships between recycling rate and impact are observed, as indicated by the stylized trend lines.

339 Between the mainly disposal scenario and the status quo an increase of 17 percentage
340 points for the recycling rate was reached, whereas from the status quo to the EU-target
341 scenarios an increase of 22 percentage points will be needed. Three types of relationships
342 between the normalized results and the recycling rate can be observed (see Figure 4b and

343 Figure S8 in the SI), as indicated by the stylized trend lines. An increasing marginal net
344 benefit with respect to the recycling rate is recognized for HTnc due to the large net impacts
345 of industrial incineration, the share of which increases strongly for the status quo but
346 decreases again for the EU-target-SP scenario. On the other hand, eight ICs display
347 decreasing marginal benefits or even an absolute decrease in the net benefits when
348 comparing the status quo and the EU-target-SP scenario, due to (a combination of) various
349 reasons: the large net impact of landfilling in the mainly disposal scenario (ET, ME, LU,
350 RDw), the overall dominance of the incineration processes with respect to the net benefits
351 (OD, IR, FE, RDf, LU, RDw), the large net impact of the sorting process (IR, ME), as well as
352 the fact that mechanical recycling has a net impact itself (RDw). For the remaining ICs an
353 approximately linear (RDm) or slightly decreasing marginal benefit (GW, HTc, PM, POF, TA,
354 TE) is apparent, generally due to the relative dominance of mechanical recycling in
355 achieving the net benefits.

356

357 # 4 Discussion

358 ## 4.1 Sensitivity and uncertainty analysis

359 ### 4.1.1 Sensitivity analysis

360 The sensitivity ratio (SR) represents the relative change in the result divided by the relative
361 variation of a parameter, as calculated in Equation 2 in the SI. The five highest SRs, and thus
362 the five most influential parameters for each of the 16 ICs, are displayed in Table S36 in the
363 SI, in which only 23 parameters are presented as many are shared among the ICs. Ten of
364 these parameters are related to industrial incineration, whereas four are connected to both
365 WtE and mechanical recycling, two to both collection and sorting, and one to mechanical
366 pretreatment. The energy substitution factor in the cement kiln treating sorting residues is

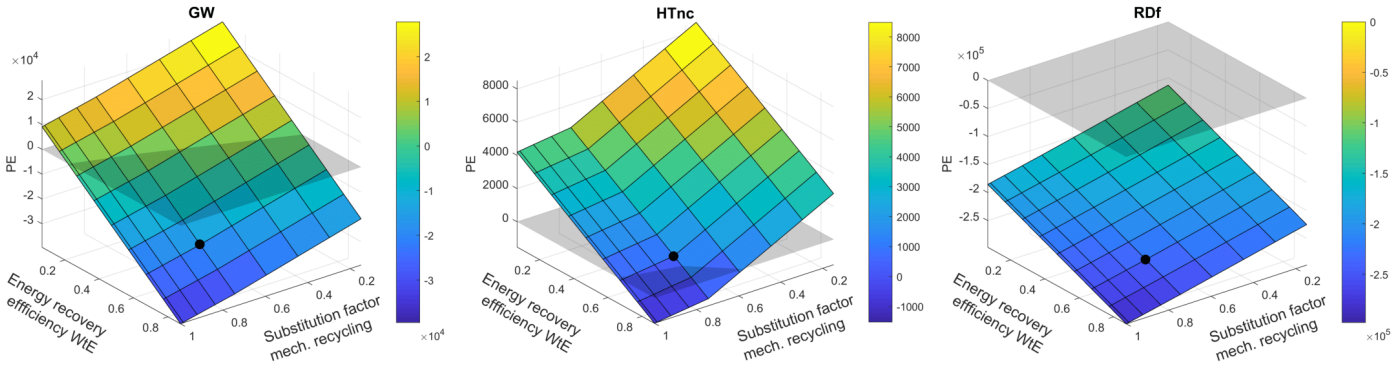

Figure 5: Sensitivity of the results of the status quo with respect to two sets of parameters for one non-toxic, one toxic, and one resource-related impact category. The black dots represent the result with the original parameter sets of the status quo. The axes display the weighted average substitution factor over all polymers in mechanical recycling, and the weighted average energy recovery efficiency over all WtE processes. The grey plane indicates the break-even of burdens and benefits.

present in the top five of all ICs, and is even the most influential parameter for twelve of them.

The substitution factor in mechanical recycling is of great importance, yet the determination (method) of these values is contested. Furthermore, in contrast to the cement industry, where the substitution is based on a physical parameter (heating value), the substitution of energy from WtE facilities is subject to considerable variations due to technological, temporal and geographical differences.[64, 90] To determine the influence of these two parameter sets on the overall results, which are presented in Tables S12 and S13 in the SI respectively, their respective values were varied, i.e. from a 15 to 100% weighted average re-granulate substitution factor and from 7 to 87% weighted average energy recovery efficiency (see Figure 5). The re-granulate substitution factors for some polymers are already equal to 100%, whereas for the others the distance to 100% varies. When simultaneously increasing the substitution factors for every single polymer, these were not allowed to surpass 100%, which causes the bend in the plots of Figure 5 (most pronounced for HTnc).

From Figure 5, it is clear that for GW at current WtE energy efficiencies, the overall system still achieves net benefits, even for re-granulate substitution ratios close to zero. Conversely, at current re-granulate substitution ratios, the average energy recovery efficiency needs to drop to 27% to cause net impacts. The plane is steeper with respect to the energy recovery compared to the re-granulate substitution, indicating a higher sensitivity of the results to the former parameter. A different situation is the case for HTnc however, where net benefits are only achieved for extremely positive parameter values as the large net impact of the cement industry needs to be compensated for, and the results are more sensitive to the re-granulate substitution factor. Finally, for RDf net benefits are reached for all conceivable parameter values due to the substantial net benefit of the cement industry.

4.1.2 Scenario analysis

To assess the influence of choices when building the model, a scenario analysis is carried out. First, the substituted electricity technology was changed from a coal-fired to a natural gas-fired CHP plant and to the Austrian supply mix (about 50% hydropower, 25% imports, mainly from Germany and the Czech Republic, and 20% fossils)[91]. The results (see Figure S9 and Table S38 in the SI) show that no shift occurred from net benefits to net impacts or vice versa for any of the ICs. For most ICs, the differences between the scenarios are quite moderate. However, for GW, the gas CHP and mix scenarios achieve 69% and 86% fewer benefits, respectively. Moreover, in the case of HTnc, the net impact increases by about 1400% for gas CHP substitution but decreases by 30% for the market mix. These large variations are due to the fact that overall, benefits and burdens are almost balanced for this IC, leading to large net effects of small variations in either burden or benefit. Finally, the net benefit for RDf decreases by about 14% and 18% for the two alternatives, respectively. For

406 these ICs, the background electricity system thus has a considerable influence on the overall
407 results.

408 Second, the substituted fuel in the industrial incineration processes was changed from hard
409 coal to natural gas. A comparison of both options (see Figure S9 in the SI) shows that for the
410 four ICs (GW, FE, RDw and RDf), lower net benefits are achieved when substituting natural
411 gas, whereas for all 12 others, the reverse is true. The difference between substituting hard
412 coal and natural gas is not caused by the use of the fuel in the cement kiln, but mainly by the
413 supply chain of the fuel itself, which causes higher impacts for natural gas (mainly from
414 Russia) than for hard coal (from Western Europe) for most ICs in the case of the Austrian
415 mix.

416 Third, alternatives for the substituted product of mixed-polymer mechanical recycling were
417 evaluated as well. The inventory data for this process were taken from Huysman et al.[15],
418 where two substituted products were proposed, namely a plant tray, made from PET or PS
419 concrete, and a street bench, made from cast iron or wood with a cast iron pedestal. For this
420 study both a PS concrete plant tray and a wooden street bench were considered. This
421 choice, however, does not have a meaningful impact on the results of the status quo due to
422 the small mass treated. However, for the EU-target-MP scenario, where all re-granulate is
423 produced through mixed-polymer mechanical recycling, significant impacts on three
424 toxicity ICs (HTc, HTnc, ET, see also Figure 4a) are observed. Here, the difference between
425 the options with and without cast iron substitution is substantial, and as the selection of the
426 substituted product is highly speculative for such a large amount of re-granulate (130,000
427 tonnes or 15 kg/cap/a), the validity of the results for these ICs is to be questioned.

428

429

4.1.3 Uncertainty contribution analysis

The uncertainty contribution analysis quantifies the share of each parameter's uncertainty in the overall uncertainty of the result, and thus takes both the sensitivity and the input variance of each parameter into account.[38, 92] Overall, 48 parameters out of 250 cause 90% of the uncertainty of the result across all ICs (see Table S37 in the SI). The most important parameters related to mechanical recycling are the recycling efficiencies and substitution factors, which have meaningful contributions for most ICs. For WtE, the share of grate incinerators with a wet APC and the energy recovery efficiencies are the most relevant parameters. They contribute mainly to GW, OD and RDf, and to a lesser extent FE and RDw. The parameters related to industrial incineration provide the most important parameter for almost all ICs, except for RDf, OD and ET. Comparing the uncertainty contributions to the 16 ICs, it is noteworthy that for some, namely HTnc, POF, TA, TE, and ME, these are entirely governed by 1-3 parameters, while for others (GW, IR, FE, ET, LU, RDw), 15-20 parameters are needed to reach 90% of the total uncertainty.

4.2 Comparison with related studies

Quantitatively comparing waste management LCA studies can be challenging due to differences in scope, waste composition, and modelling assumptions. Many studies also do not supply sufficient inventory data and provide only graphical and no numerical results. The results of this study were nonetheless compared with two previous studies with a similar scope, namely the assessment of the status quo of a regional plastic packaging WMS: one for Portugal as a whole[27] and one for the Setúbal Peninsula in Portugal[28]. The most noteworthy distinctions between this study and Ferrão et al.[27] are the net burdens reported by the latter for GW and RDm+RDf (2.36×10^{2} kg CO2-eq./t and 1.25×10^{-8} kg Sb-eq./t), whereas this study found net benefits for these ICs (-6.15×10^{2} kg CO2-eq./t and -3.51×10^{4}

455 kg Sb-eq./t). The net benefits for the three other investigated ICs are one order of
456 magnitude lower (POF), about the same order of magnitude (TA) and three orders of
457 magnitude higher (RDw) than the results of this study. Ferreira et al.[28] do not provide the
458 amount of plastic packaging treated, so a quantitative comparison is not possible, but it is
459 noteworthy that net burdens are reported for FE, compared to the net benefits in this study,
460 while the reverse is the case for HTnc. These results thus differ strongly, although the
461 studies deal with the management of a similar waste stream.

462 Next to comparing the results of the full WMS, the model was rerun to just account for the
463 treatment of PET bottles to enable further comparison. Haupt et al.[9] report a GW result
464 of -2.80×10^3 kg CO2-eq./t for the closed-loop (i.e. food-grade) recycling chain of PET bottles
465 in Switzerland per tonne of waste collected separately, whereas Turner et al.[93] report a
466 result of -2.19×10^3 kg CO2-eq./t for PET bottle recycling in the UK. These benefits are higher
467 than the -1.54×10^3 kg CO2-eq./t in this study, which is possibly caused by the fact that in
468 this study, at the food-grade recycling plant not all re-granulate is reprocessed to food-
469 grade quality, depending on market conditions. Nevertheless, these results are in the same
470 order of magnitude, which signifies that LCA studies for one product-polymer combination
471 are comparable, in contrast to both full packaging WMS studies discussed above. This
472 further highlights the importance of the local waste composition for the result of WMS LCA
473 studies. [8, 10]

474

475 **4.3 Limitations**

476 Due to aspects such as data availability and modelling choices, the applicability of the
477 present work may be limited in certain cases. Although the chemical composition with
478 respect to elements was included, the presence of chemical substances (such as brominated

479 flame retardants or phthalates)[94, 95] could not be taken into account because of lacking data.
480 This is less relevant for the incineration processes, but can be of concern for mechanical
481 recycling, although packaging products generally contain less potentially harmful additives
482 compared to e.g. electronic equipment or building materials.[94, 96] However, as information
483 on the exposure from recycled plastics is lacking, further research is needed for the
484 inclusion of this type of impacts into LCA.

485 Waste prevention, including reuse, was not considered due to the focus on an existing
486 amount of waste to be treated and the fact that this is generally accepted as the most
487 preferred option.[8] However, this first priority in the waste hierarchy could and should be
488 able to provide considerable contributions to reaching the EU recycling target, especially
489 when focused on products which are hard to recycle. Including waste prevention into LCA
490 studies brings about methodological challenges though, such as alleviating the zero burden
491 assumption and including rebound effects,[8] which change the scope of the study and were
492 therefore not included.

493 Finally, as discussed in Section 4.2, these results are specific for Austria, and may not be
494 representative for other (European) countries. However, the general conclusion (as detailed
495 below in Section 4.4) that increasing recycling rates do not necessarily lead to an improved
496 environmental performance across impact categories is expected to hold regardless.

497

498 **4.4 Implications for Austrian and European waste plastic packaging management**

499 In Austria in 2013, about 100,000 tonnes of plastic packaging waste were processed in
500 mechanical recycling plants, while about 97,000 tonnes were incinerated in industrial
501 incineration facilities and 120,000 tonnes in WtE plants, and 3,400 tonnes were landfilled.[36]
502 Bringing these waste masses in relation to the GW, mechanical recycling has the highest

503 specific benefit with -0.16 PE/t, followed by industrial incineration with -0.12 PE/t and WtE
504 with 0.01 PE/t, while the landfill process causes impacts close to 0.00 PE/t due to the inert
505 nature of waste plastics regarding methane production.[97] Therefore, to decrease the GW
506 impact of the plastic packaging WMS, it is equally important to direct waste flows away
507 from WtE incineration as from landfills. This is also apparent from Figure 3 (subdivided by
508 collection route), and emphasizes the importance of separate collection, even if a potentially
509 considerable amount of the separately collected material is sorted out and incinerated in
510 the cement industry. Furthermore, Figure 3 suggests that holds for the other ICs as well
511 (with the exception of HTnc) as the net impacts of industrial incineration for some ICs are
512 more than compensated for by the net benefits of mechanical recycling. This trend is further
513 reinforced when considering a relatively clean background electricity system since this
514 strongly increases the net GW impacts of the WtE process as discussed in the scenario
515 analysis.

516 The alternative waste management scenarios indicate that for most ICs increasing recycling
517 rates lead to increased benefits (Figure 4a and Figure S5). However, the marginal benefit
518 decreases with increasing recycling rates for many ICs (Figure 4b and Figure S6), and for
519 four ICs the EU-target-SP scenario achieves lower net benefits than the status quo. This
520 suggests that the environmentally optimal recycling rate is below 100% depending on the
521 IC. This is further reinforced by the fact that in the scenarios for the EU target, no non-linear
522 effects of e.g. increasing separate collection on transport distances and sorting efficiency are
523 included, which can potentially further decrease the benefits for this scenario. Therefore,
524 future research should address these effects to create a sound basis for proposing recycling
525 targets leading to an environmentally optimal outcome.

526

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6 Supporting Information

More detailed information including inventory data, additional results and sensitivity and uncertainty analysis.

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Supplementary data

Supporting Information

Integrating high-resolution material flow data into the environmental assessment of waste management system scenarios – the case of plastic packaging in Austria.

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Contents

List of tables

List of figures

1 Life cycle inventory data

1.1 Uncertainty characterization

The uncertainties of the input parameters were characterized. If uncertain input data were available (from a statistical analysis of primary data or literature), these uncertainties were taken. If this was not available, however, the pedigree matrix approach, as implemented into the ecoinvent database,[1] was used. The mass and composition of the waste input were not treated as uncertain though to ensure the functional unit remains constant.

Due to computational limitations, the propagation of the uncertainties using sampling-based methods such as Monte Carlo simulation was not feasible, so the uncertainty propagation was performed analytically. The approach by Bisinella et al.[2] was used, which combines the sensitivity and uncertainty of each of the parameters in a global sensitivity analysis. Based on the contribution analysis, the most important 250 parameters from a total of 824 in the model were selected for the perturbation analysis, where the results for all impact categories are calculated while varying these parameters by 10% one at a time. The resulting impact scores are used to calculate the sensitivity coefficient (SC) and sensitivity ratio (SR):[2, 3]

$$SC_i^j = \frac{(\Delta result)^j}{(\Delta parameter)_i} \qquad (1)$$

$$SR_i^j = \frac{\left(\frac{\Delta result}{initial\ result}\right)^j}{\left(\frac{\Delta parameter}{initial\ parameter}\right)_i} \qquad (2)$$

with *i=1,…,n* tested parameters and *j=1,…,m* calculated impact categories. The total variance of the result (*Y*) in impact category *j* can subsequently be approximated as the sum over the analytical uncertainties of all parameters X_i:[2]

$$V(Y)^j \approx \sum_{i=1}^{n} \left[\left(SC_i^j\right)^2 \cdot V_{input}(X_i)\right] \qquad (3)$$

Analytical propagation approaches have shown to produce results which are very similar to those obtained through sampling-based methods,[2, 4] which was confirmed for this study using 1000 Monte Carlo runs on an earlier version of the model.

In the tables below, detailing the input data used in the model, for each parameter the mean, variance and distribution (normal or lognormal) is provided, as well as the source of the uncertainty value. This was either from the same source as the parameter value itself ("from source"), from a statistical analysis from multiple values ("statistical analysis"), or from the pedigree matrix. In the latter case, the basic uncertainty, as the variance of the logtransformed data, as well as the scores of the five data quality indicators for the additional uncertainty are provided. The basic uncertainties and uncertainty factor values are taken from Weidema et al.[1].

1.2 Waste input composition

The MFA results of Van Eygen et al.[5] provided the composition of the incoming waste with respect to each of the polymers as well as product types (see Table S1). The elemental compositions of the different polymers with respect to 23 elements were taken from Koehler et al.[6] for all polymers except PVC, for which information from Eggels et al.[7] was used (see Table S2). The water content of plastics in Viennese residual waste was measured to be 15% by Hauer et al.[8], and this value was used throughout the WMS due to the lack of more specific information. Finally, the heating values of the different polymers were taken from Phyllis2[9].

The waste flows of the alternative waste management scenarios subdivided by polymer are presented in Figure S1 for the 1994-Archetype scenario, in Figure S2 for the 2030-Target scenario with a focus on single-polymer recycling, and in Figure S3 for the 2030-Target scenario with mixed-polymer recycling only.

Table S1: Input composition of the plastic packaging waste stream (source: [5]).

Polymer	Input (%)	Product type	Input (%)
LDPE	4.65E+01	PET Bottles	1.54E+01
LLDPE	5.22E+00	Hollow Bodies Small	1.66E+01
HDPE	1.05E+01	Hollow Bodies Large	6.20E+00
PP	1.36E+01	Films Small	2.35E+01
PS	3.15E+00	Films Large	2.41E+01
EPS	1.82E+00	EPS Large	7.88E-01
PET	1.92E+01	Others	1.34E+01
PVC	5.42E-02		

Table S2: Elemental composition (%) of the 5 main polymer groups.

Element	PET [6]	PE [6]	PP [6]	PS [6]	PVC [7]
O	3.47E+01	2.06E+00	2.38E+00	1.97E+00	9.46E-01
H	4.22E+00	1.84E+01	1.62E+01	7.69E+00	4.73E+00
C	6.09E+01	7.94E+01	8.12E+01	8.98E+01	3.93E+01
S	3.99E-02	5.03E-02	4.00E-02	6.99E-02	1.05E-01
N		7.00E-02	6.02E-02	2.00E-01	1.05E-01
Cl	7.01E-02	9.71E-04	9.71E-04	9.01E-02	5.47E+01
Br	9.79E-04	9.85E-06	9.85E-06	1.97E-03	
F	1.03E-03	1.03E-05	1.03E-05		0.00E+00
As				5.06E-04	5.26E-05
Cd	5.63E-05	1.14E-05	1.14E-05	1.26E-05	1.05E-02
Co				4.15E-03	1.05E-04
Cr	6.86E-04	1.25E-03	1.25E-03	6.86E-04	2.63E-03
Cu	9.52E-03	9.06E-03	9.06E-03	1.81E-02	2.63E-02
Hg	7.04E-06	5.21E-05	5.21E-05	7.04E-06	1.05E-05
Mn					2.63E-03
Mo				2.06E-03	

Ni	7.59E-04	7.59E-04	7.59E-04	7.59E-04	1.05E-03
Pb	6.71E-03	9.53E-03	9.53E-03	1.90E-02	5.26E-02
Sb	1.56E-02				1.05E-03
Se					5.26E-05
Sn		1.03E-04	1.03E-04	1.01E-01	5.26E-04
V					5.26E-03
Zn	1.11E-02	9.62E-03	9.62E-03	1.90E-02	4.20E-02

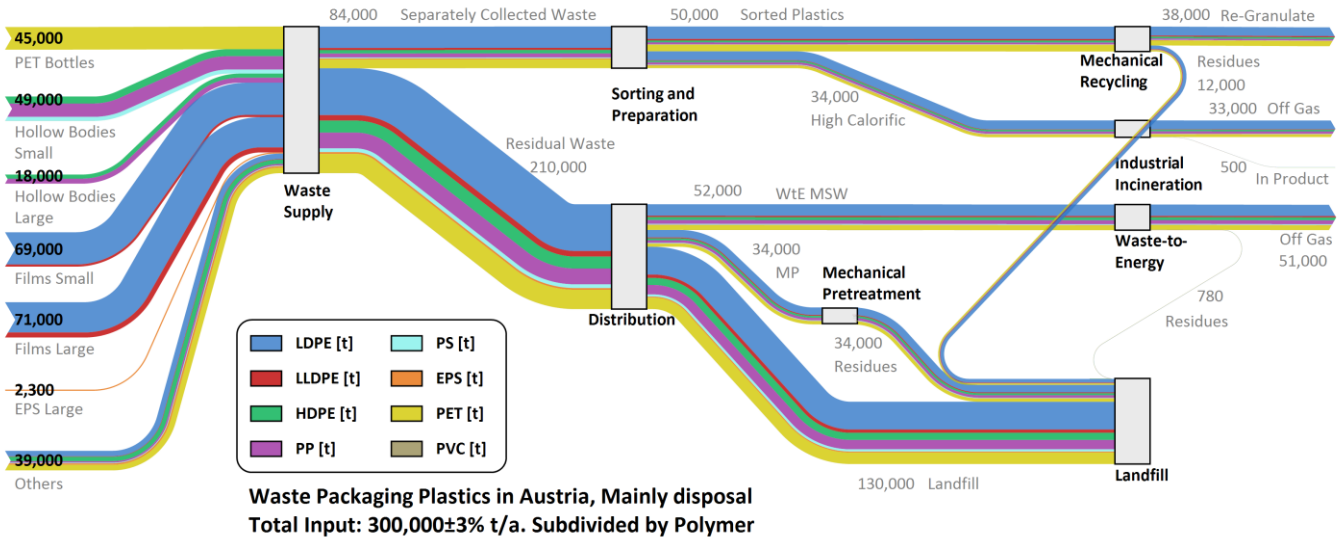

Figure S1: MFA results, subdivided by polymer, of the scenario describing the structure of the waste management system for plastic packaging in the mainly disposal scenario.

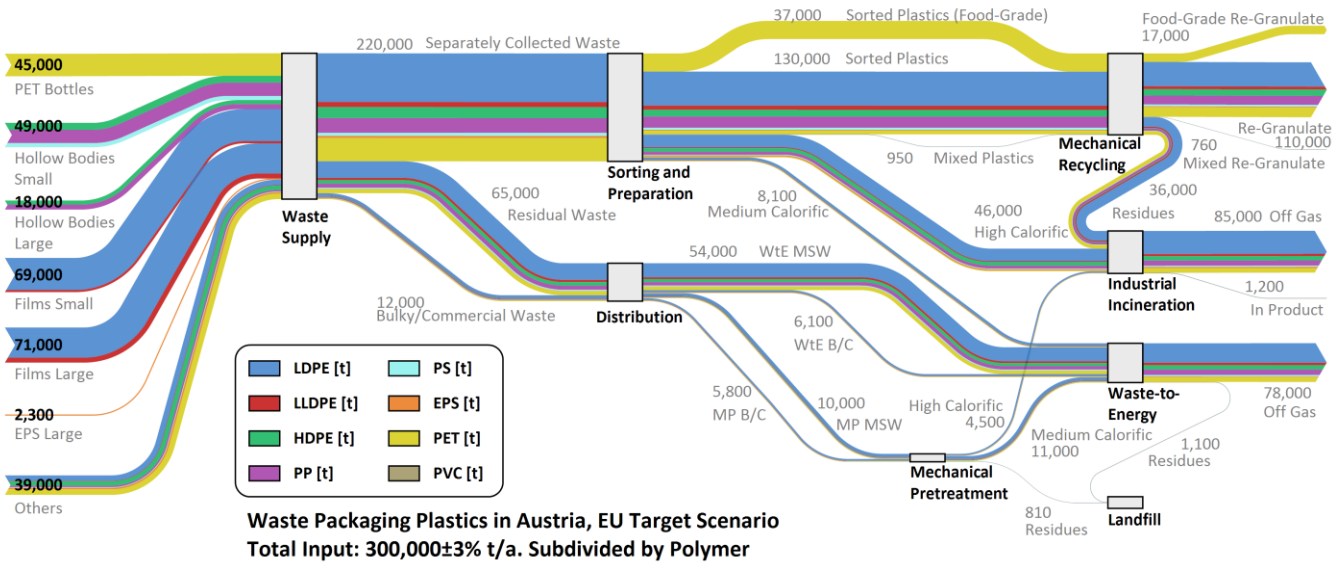

Figure S2: MFA results, subdivided by polymer, of the scenario describing the structure of the waste management system achieving the increased EU recycling target with continuing focus on (high-quality) single-polymer recycling.

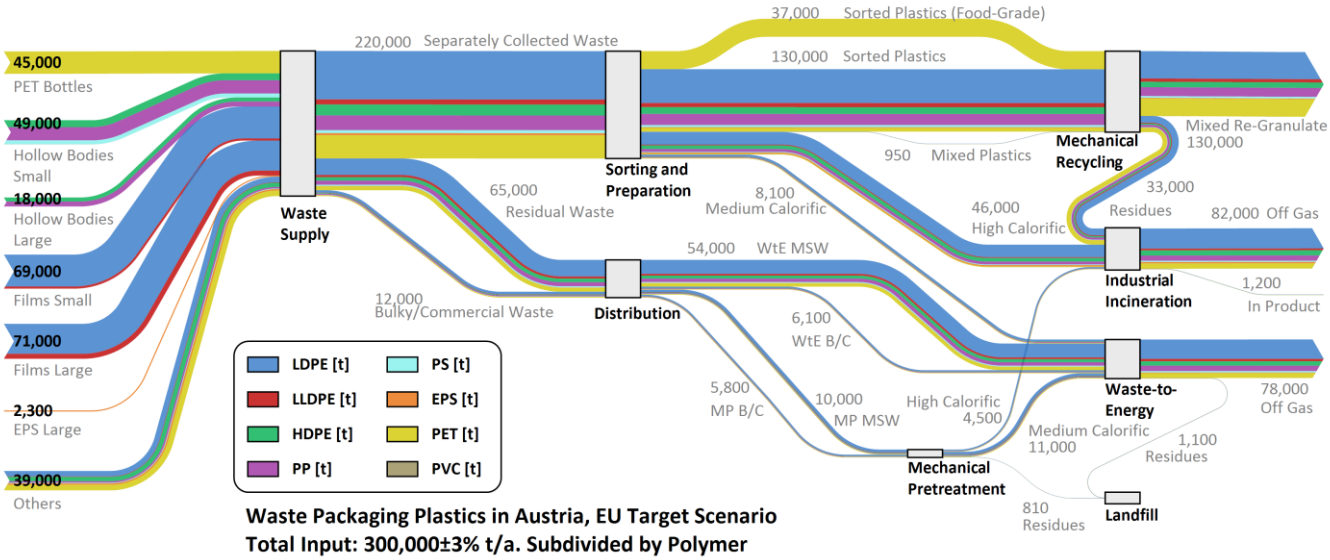

Figure S3: MFA results, subdivided by polymer, of the scenario describing the structure of the waste management system achieving the increased EU recycling target through mixed-polymer mechanical recycling only.

1.3 Collection, sorting, mechanical pretreatment and landfill

The fuel consumption during separate collection of plastic wastes (SCW) was gathered from primary data, which was complemented with the average additional transport distance to the sorting facility.[10] The collection of residual waste (RW) was modelled using average fuel consumptions for urban and rural collection,[11] as well as the share between these two collection types,[10] whereas the collection of bulky and commercial waste (BCW) was assumed to be 30% more efficient. The additional transport for these two waste streams, as well as to further treatment processes, was considered as well (primary data;[10]).

The consumption of electricity and diesel in the sorting process as well as in the mechanical pretreatment process was determined using the weighted average of primary data from two plants for the former and five plants and one literature source[12] for the latter. Finally, the non-hazardous waste landfill was modelled using the process from the EASETECH database.[13] All above-mentioned data are presented in Table S3.

Table S3: Inventory data per kg waste for the collection, transport, sorting, mechanical pretreatment and landfill processes.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Fuel consumption SCW	1.63E-02	3.42E-07	l/kg	Lognormal	0.0006-(2,2,1,1,1)	Primary data
Transport SCW	2.70E+01	9.36E+01	km	Lognormal	0.12-(2,2,1,1,1)	[10]
Fuel consumption RW	4.78E-03	1.77E-06	l/kg	Normal	Statistical analysis	[10, 11]
Transport RW	8.96E+01	1.03E+03	km	Lognormal	0.12-(2,1,1,1,1)	[10]
Transport mechanical recycling	1.50E+02	2.89E+03	km	Lognormal	0.12-(2,2,1,1,1)	Primary data
Transport incineration	5.30E+01	3.60E+02	km	Lognormal	0.12-(2,1,1,1,1)	[10]
Transport landfill	2.20E+01	6.21E+01	km	Lognormal	0.12-(2,1,1,1,1)	[10]
Transport underground landfill wet APC	7.18E+02	6.59E+04	km	Lognormal	0.12-(1,1,1,1,1)	Primary data
Electricity consumption sorting	6.47E-02	3.82E-06	kWh/kg	Lognormal	0.0006-(1,2,2,1,1)	Primary data
Diesel consumption sorting	3.88E-01	1.04E-04	MJ/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data

Electricity consumption mechanical pretreatment	5.82E-02	2.35E-06	kWh/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data, [12]
Diesel consumption mechanical pretreatment	3.94E-02	1.08E-06	MJ/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data, [12]
Infrastructure sorting and mechanical preparation	2.00E-10	1.70E-20	unit/kg	Lognormal	0.3-(3,3,4,3,4)	[14]

1.4 Mechanical recycling

Eight distinct mechanical recycling processes were taken into account:

- Single-polymer PET to food-grade re-granulate
- Single-polymer PET to fiber
- Single-polymer LDPE/LLDPE
- Single-polymer HDPE
- Single-polymer PP
- Single-polymer PS
- Single-polymer EPS
- Mixed-polymer

In the mechanical recycling processes, the incoming sorted waste material is generally subject to several shredding and washing steps, producing secondary plastic flakes, which can be further extruded and pelletized as well. Rejects are utilized by the cement industry as an alternative fuel. The inventory data for the operation of the eight distinct mechanical recycling processes is presented in Table S4 for PET to food-grade re-granulate, Table S5 for PET to fiber, Table S6 for LDPE/LLDPE, Table S7 for HDPE, Table S8 for PP/PS, Table S9 for EPS, and Table S10 for mixed-polymer recycling.

It is assumed that the re-granulates of the single-polymer recycling processes (except EPS recycling) are applied in products substituting virgin plastic resources. Determining to what extent virgin material is substituted, and hence the avoided burden, has considerable influence on the overall results.[15] Many studies on the recycling of plastic wastes use a substitution factor of one, often without justification, suggesting that one kg of produced re-granulate displaces one kg of virgin polymer.[7, 16-21] Schmidt and Strömberg[22] however, suggest a quality loss of 10-20% should be assumed, a supposition adopted by other authors.[23-27] Another approach to estimate the quality loss and market uptake of recycled materials is value-corrected substitution (VCS),[15, 28-30] using the ratio of the market prices of secondary to primary material. This method has been used for the assessment of plastics recycling systems as well,[31-33] resulting in substitution factors ranging from 0.35 (comparing prices of sorted bales to primary material) to 0.60 and 0.81 (comparing secondary and primary plastics).

In this study the VCS method is used for the single-polymer recycling of LDPE/LLDPE, HDPE, PP and PS. Weekly prices for primary[34] and monthly prices for secondary plastics[35] over a time period of two years were analyzed (see Table S11), resulting in the substitution factors ranging presented in Table S12. These factors proved to be relatively stable over time, making them suitable as substitution factors. For the food-grade recycling of PET bottle waste, however, a substitution factor of 1 is taken since this re-granulate is subject to strict quality and food safety requirements,[22, 36, 37] while the same can be expected for PET to fiber recycling as well.[38]

During EPS recycling, EPS beads are produced for thermal insulation with a thermal conductivity of 0.045 $W \cdot m^{-1} \cdot K^{-1}$ and bulk density of 12 $kg \cdot m^{-3}$,[39] which substitute EPS foam slab with a thermal conductivity of 0.0375 $W \cdot m^{-1} \cdot K^{-1}$ and bulk density of 35 $kg \cdot m^{-3}$.[40] The substitution factor was thus calculated as the amount of both materials

needed to provide a thermal insulence of $1 m^2K/W$, resulting in a value of 2.43 $kg_{primary}/kg_{secondary}$. The mixed-polymer re-granulate is used for the production of items such as recycled plastic lumber, which is assumed in this study to substitute two possible products: half of the produced re-granulate replaces a plant tray made from PS concrete (with a substitution factor of 1.39 $kg_{primary}/kg_{secondary}$) while the other half replaces a bench made from wood (0.27 $kg_{primary}/kg_{secondary}$) with a cast iron pedestal (0.34 $kg_{primary}/kg_{secondary}$),[41] which is reflected in the substitution factors in Table S12.

Table S4: Inventory data per kg waste for the PET to food-grade re-granulate mechanical recycling process. The water emissions are reduced in the municipal wastewater treatment plant, which is modelled using the transfer coefficients reported by [42].

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Electricity consumption	5.25E-01	7.92E-04	kWh/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Water consumption	1.35E+00	5.26E-03	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Sodium hydroxide consumption	2.67E-02	2.05E-06	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Detergent consumption	1.23E-03	4.35E-09	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Defoaming agent consumption	1.85E-03	9.78E-09	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Flakes (fiber) production	2.16E-01	1.33E-04	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Flakes (bottle) production	2.23E-01	1.43E-04	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Re-granulate (bottle) production	2.41E-01	1.67E-04	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Sorter flakes (low-quality) production	1.65E-01	7.82E-05	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Reject production	1.55E-01	6.90E-05	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Wastewater production	1.55E+00	6.93E-03	L/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
COD water emissions	9.24E-04	1.12E-07	kg/L	Normal	Statistical analysis	Primary data
Cl water emissions	4.88E-04	2.01E-07	kg/L	Lognormal	Statistical analysis	Primary data
SO_4 water	6.01E-05	2.48E-09	kg/L	Lognormal	Statistical	Primary

emissions					analysis	data
NH_4 water emissions	1.36E-05	4.60E-09	kg/L	Lognormal	Statistical analysis	Primary data
NO_3 water emissions	1.15E-05	9.47E-11	kg/L	Lognormal	Statistical analysis	Primary data
P water emissions	4.09E-05	7.00E-10	kg/L	Lognormal	Statistical analysis	Primary data
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
Virgin polymer substitution factor	1.00E+00	3.46E-05	kg/kg	Lognormal	0.0006-(3,1,1,1,2)	[22, 36, 37]

Table S5: Inventory data per kg waste for the PET to fiber mechanical recycling process.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Sodium hydroxide consumption	7.82E-04	2.32E-09	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Surfactant consumption	1.17E-02	5.21E-07	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Floating agent consumption	7.04E-03	1.88E-07	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Additives consumption	3.76E-02	5.34E-06	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Finishing oil consumption	3.11E-03	3.65E-08	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Additives for wastewater treatment consumption	1.10E-02	4.54E-07	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Diesel consumption	1.68E-01	1.07E-04	MJ/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Heat consumption	2.56E+00	2.47E-02	MJ/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Electricity consumption	9.94E-01	3.74E-03	kWh/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Transports	1.88E-02	4.63E-05	km	Lognormal	0.12-(2,4,2,3,1)	[44]
Natural gas consumption	9.16E-01	3.17E-03	MJ/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]

Water consumption	4.86E+00	8.94E-02	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Reject production	2.24E-01	1.89E-04	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Fiber Production	7.76E-01	2.28E-03	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Wastewater production	4.69E-03	8.34E-08	kg/kg	Lognormal	0.0006-(2,4,2,3,1)	[44]
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
Virgin polymer substitution factor	1.00E+00	3.46E-05	kg/kg	Lognormal	0.0006-(3,1,1,1,2)	[38]
Virgin pellet to filament electricity consumption	6.40E-01	n/a	kWh/kg	n/a	n/a	[45], cited by [38, 46]
Virgin pellet to filament heat consumption	5.00E+00	n/a	MJ/kg	n/a	n/a	[45], cited by [38, 46]

Table S6: Inventory data per kg waste for the LDPE mechanical recycling process. The water emissions are reduced in the municipal wastewater treatment plant, which is modelled using the transfer coefficients reported by [42].

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Electricity consumption	6.50E-01	1.21E-03	kWh/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Heat consumption	3.78E-01	4.10E-04	MJ/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Water consumption	5.25E+00	7.91E-02	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Ferrichloride consumption	9.00E-04	2.32E-09	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Water peroxide consumption	1.40E-03	5.63E-09	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Defoaming agent consumption	8.00E-04	1.84E-09	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Re-granulate production	6.67E-01	1.28E-03	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Byproduct production	4.76E-02	6.51E-06	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data

Reject production	2.86E-01	2.34E-04	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Wastewater production	3.75E+00	4.04E-02	L/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
Waste sludge production	8.50E-02	2.07E-05	kg/kg	Lognormal	0.0006-(1,4,1,1,1)	Primary data
NO_2 water emissions	3.29E-05	5.04E-11	kg/L	Lognormal	0.04-(3,4,1,1,1)	Primary data
SO_3 water emissions	1.00E-05	4.67E-12	kg/L	Lognormal	0.04-(3,4,1,1,1)	Primary data
Ba water emissions	5.00E-06	2.30E-11	kg/L	Lognormal	0.04-(3,4,1,1,1)	Primary data
Co water emissions	1.00E-06	9.20E-13	kg/L	Lognormal	0.04-(3,4,1,1,1)	Primary data
Zn water emissions	2.00E-06	3.68E-12	kg/L	Lognormal	0.04-(3,4,1,1,1)	Primary data
SO_4 water emissions	2.00E-04	1.87E-09	kg/L	Lognormal	0.04-(3,4,1,1,1)	Primary data
BOD water emissions	2.66E-03	3.31E-07	kg/kg	Lognormal	0.04-(3,4,1,1,1)	Primary data
COD water emissions	6.23E-03	1.81E-06	kg/kg	Lognormal	0.04-(3,4,1,1,1)	Primary data
P water emissions	5.66E-05	1.50E-10	kg/kg	Lognormal	0.04-(3,4,1,1,1)	Primary data
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
Virgin polymer substitution factor	6.10E-01	2.05E-03	kg/kg	Normal	Statistical analysis	[34, 35]

Table S7: Inventory data per kg waste for the HDPE mechanical recycling process.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Sodium hydroxide consumption	2.50E-04	9.44E-11	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Defoaming agent consumption	1.41E-03	2.99E-09	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Detergent consumption	1.30E-03	2.54E-09	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Water consumption	4.12E-01	2.56E-09	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Electricity consumption	4.54E-01	3.10E-04	kWh/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Diesel consumption	6.10E-03	5.61E-08	MJ/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Natural gas consumption	2.71E-01	1.11E-04	MJ/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
LPG consumption	1.70E-06	4.38E-15	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Re-granulate production	9.27E-01	1.30E-03	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
Reject production	7.32E-02	8.08E-06	kg/kg	Lognormal	0.0006-(1,3,2,3,1)	[47]
PM10 air emissions	2.13E-05	5.85E-11	kg/kg	Lognormal	0.12-(1,3,2,3,1)	[47]
PM2.5 air emissions	1.39E-05	6.83E-11	kg/kg	Lognormal	0.3-(1,3,2,3,1)	[47]
BOD water emissions	2.78E-04	3.32E-09	kg/kg	Lognormal	0.04-(1,3,2,3,1)	[47]
COD water emissions	1.39E-06	8.29E-14	kg/kg	Lognormal	0.04-(1,3,2,3,1)	[47]
Suspended solids water emissions	2.69E-04	3.10E-09	kg/kg	Lognormal	0.04-(1,3,2,3,1)	[47]
Dissolved solids water emissions	8.43E-06	3.05E-12	kg/kg	Lognormal	0.04-(1,3,2,3,1)	[47]
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
Virgin polymer substitution factor	7.29E-01	2.96E-03	kg/kg	Normal	Statistical analysis	[34, 35]

Table S8: Inventory data per kg waste for the PP mechanical recycling process. Due to a lack of more specific data, these values are used for the PS mechanical recycling process as well.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Electricity consumption	7.60E-01	7.88E-03	kWh/kg	Lognormal	0.0006-(3,4,4,3,1)	[48]
Diesel consumption	2.03E-02	5.60E-06	MJ/kg	Lognormal	0.0006-(3,4,4,3,1)	[48]
Water consumption	2.60E+00	9.22E-02	kg/kg	Lognormal	0.0006-(3,4,4,3,1)	[48]
Natural gas consumption	8.90E-01	1.08E-02	MJ/kg	Lognormal	0.0006-(3,4,4,3,1)	[48]
Re-granulate production	8.10E-01	8.95E-03	kg/kg	Lognormal	0.0006-(3,4,4,3,1)	[48]
Reject production	1.90E-01	4.92E-04	kg/kg	Lognormal	0.0006-(3,4,4,3,1)	[48]
BOD water emissions	6.50E-04	2.35E-08	kg/kg	Lognormal	0.04-(3,4,4,3,1)	[48]
COD water emissions	6.50E-04	2.35E-08	kg/kg	Lognormal	0.04-(3,4,4,3,1)	[48]
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
Virgin polymer (PP) substitution factor	7.57E-01	6.32E-03	kg/kg	Normal	Statistical analysis	[34, 35]
Virgin polymer (PS) substitution factor	6.73E-01	6.94E-03	kg/kg	Normal	Statistical analysis	[34, 35]

Table S9: Inventory data per kg waste for the EPS mechanical recycling process.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Electricity consumption	1.10E-01	2.20E-05	kWh/kg	Lognormal	0.0006-(2,3,1,1,1)	Primary data
EPS beads production	1.00E+00	n/a	kg/kg	n/a	n/a	Primary data
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
Thermal insulation (EPS foam slab) substitution factor	2.43E+00	1.06E-02	kg/kg	Lognormal	0.0006-(2,3,1,1,1)	[39, 40]

Table S10: Inventory data per kg waste for the mixed-polymer mechanical recycling process.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Water consumption	2.70E-01	2.16E-04	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Diesel consumption	4.80E-02	6.84E-06	MJ/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Natural gas consumption	5.04E-01	7.54E-04	MJ/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Polyelectrolyte consumption	2.30E-04	1.57E-10	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Ferrichloride consumption	2.20E-03	1.44E-08	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Sodium hydroxide consumption	3.80E-03	4.29E-08	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Electricity consumption	6.50E-01	1.25E-03	kWh/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Re-granulate production	8.00E-01	1.90E-03	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Reject production	2.00E-01	1.19E-04	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Wastewater production	1.00E-01	2.97E-05	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Infrastructure	2.00E-09	1.47E-18	unit/ kg_{output}	Lognormal	0.3-(3,3,4,2,2)	[43]
PS concrete substitution factor	6.96E-01	1.44E-03	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Wood substitution factor	1.70E-01	8.60E-05	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]
Cast iron substitution factor	1.36E-01	5.50E-05	kg/kg	Lognormal	0.0006-(1,4,1,3,1)	[41]

Table S11: Prices of primary and secondary HDPE, LDPE, PP and PS, and the respective prices ratios.

Month	Primary (€/kg) [34]				Secondary (€/kg) [35]				Ratio (%)			
	HDPE	LDPE	PP	PS	HDPE	LDPE	PP	PS	HDPE	LDPE	PP	PS
Nov.15	1388	1377	1110		910	740	950	1050	66	54	86	
Dec.15	1413	1398	1125		910	710	960	1050	64	51	85	
Jan.16	1369	1357	1091		920	780	860	1060	67	57	79	
Feb.16	1254	1231	954		970	770	890	1010	77	63	93	
Mar.16	1255	1248	954		870	800	880	980	69	64	92	
Apr.16	1318	1300	1027	1365	890	830	840	990	68	64	82	73
May.16	1320	1336	1072	1404	920	800	830	950	70	60	77	68
Jun.16	1275	1314	1050	1396	920	820	830	880	72	62	79	63
Jul.16	1226	1252	1011	1279	910	840	840	880	74	67	83	69
Aug.16	1220	1238	1008	1212	850	820	800	990	70	66	79	82
Sep.16	1229	1264	1053	1221	840	850	800	980	68	67	76	80
Oct.16	1235	1285	1067	1159	860	870	750	900	70	68	70	78
Nov.16	1236	1292	1074	1167	840	800	750	880	68	62	70	75
Dec.16	1218	1250	1031	1243	820	780	760	880	67	62	74	71
Jan.17	1234	1261	1050	1423	920	720	730	880	75	57	69	62
Feb.17	1257	1311	1125	1587	950	730	740	880	76	56	66	55
Mar.17	1298	1360	1218	1735	930	740	770	940	72	54	63	54
Apr.17	1283	1350	1218	1672	950	840	780	860	74	62	64	51
May.17	1249	1325	1205	1420	930	800	800	960	74	60	66	68
Jun.17	1158	1251	1149	1309	880	780	860	980	76	62	75	75
Jul.17	1092	1180	1087	1318	920	800	830	940	84	68	76	71
Aug.17	1077	1203	1079	1286	880	750	800	810	82	62	74	63
Sep.17	1104	1255	1137	1443	930	770	790	840	84	61	70	58
Oct.17	1117	1287	1168	1367	870	740	800	920	78	57	68	67
Nov.17	1077	1217	1099	1300	840	690	830	830	78	57	76	64
								Mean	**73**	**61**	**76**	**67**
								Standard deviation	**5**	**5**	**8**	**8**

Table S12: Substitution factors ($kg_{primary}/kg_{secondary)}$ and substituted products for mechanical recycling of plastics. The point of substitution is taken to be after the recycling plant.

Polymer	Substitution factor	Substituted product
PET to food-grade re-granulate	1	Primary polymer
PET to fiber	1	Primary polymer
LDPE/LLDPE	0.61	Primary polymer
HDPE	0.73	Primary polymer
PP	0.76	Primary polymer
PS	0.67	Primary polymer
EPS	2.43	Thermal insulation
Mix	0.70	PS concrete
	0.14	Cast iron
	0.17	Hardwood

1.5 Waste-to-Energy

Two main classes of WtE technologies are relevant. Grate incinerators handle mainly unprocessed mixed municipal solid waste, whereas fluidized bed incinerators require shredded and homogenized wastes.[49] A further distinction can be made regarding the air pollution control (APC) system between wet and dry or semidry systems.[6] In Austria, three distinct incineration process archetypes are present: grate incineration with wet APC, grate incineration with dry APC, and fluidized bed incineration with wet APC.[6, 50]

Primary data were received from two grate incinerators with a wet APC system, from two with a dry APC system, as well as from one fluidized bed incinerator, which were averaged for each process type using the amount of incinerated non-hazardous waste as a weighting factor (see Tables S13, S16 and S19 in the SI). The allocation of waste plastics to grate incinerators with a wet or a dry APC system, respectively, was determined using the total amounts of waste processed in 2014 in each of the seven facilities.[51, 52]

The weighted average inventory data for grate incineration with a wet air pollution control system (APC) and dry APC, as well as for a fluidized bed incineration plant are given in Table S15, Table S18 and Table S21 respectively.

As mentioned in the main article, the choice of the marginal heat technology for waste incinerators needs to be taken on a plant by plant basis. An overview of all plants in Austria is given in Table S14, stating which types of energy are recovered and which marginal heat technology was assumed. Plants A, B, C, D, H and J are connected to the local urban district heating networks, which otherwise (mainly) rely on natural gas-fired CHP facilities for the base production of heat.[53] The steam produced at facility E is delivered to the adjacent coal-fired CHP plant for electricity and heat production.[50] The district heating network supplied by plant F is only supplied by the WtE plant,[50] therefore the substituted technology is a small-scale natural gas-fired boiler. Finally, plants I and K both deliver their residual heat to nearby fiber and paper manufacturers, respectively. At the site of plant I, energy is also produced from a coal-fired CHP plant,[54] whereas for plant K, auxiliary boilers fired with natural gas are used to ensure the reliability of operation.[50]

Overall, eight combinations of firing technologies, APC systems, energy recovery and marginal heat technologies are present in Austria (see Table S14). For seven of these combinations, primary data on the energy efficiencies were available, which were calculated by dividing the net output of electricity and/or heat by the total energy input. The exception was the plant using a wet APC system producing only heat, for which data was available from Böhmer and Gössl[53]. Furthermore, as plant E delivers the produced steam directly to the nearby coal-fired power plant, data on the efficiency of electricity production from the produced steam in coal-fired CHP plants were included in the calculation as well.[53, 55, 56] These data were then aggregated for each of the process archetypes based on the total amounts of waste treated in each plant in 2014,[51] which resulted in energy recovery efficiencies for each of the three incineration process types, as reported in Table S13. To account for different local conditions and therefore multiple marginal heat technologies, two marginal heat technologies had to be considered within two of the process archetypes. The weighted average energy recovery efficiency across all WtE facilities was 67% (14% electricity, 53% heat), which is quite moderate.[57] Furthermore, the large difference between the recovery efficiency of plants with a wet APC compared to a dry APC is not caused by the APC system itself, but rather by the location of the plants with a dry APC and the resulting low local demand for heat.

The input-based emissions to the flue gas, bottom ash, fly ash, slurry and effluent (the latter two being only relevant for wet APC systems) for the two types of APC systems

were determined using transfer coefficients for each of the considered elements in the input (see Table S2), taken from Koehler et al.[6]. Furthermore, process-based flue gas emissions of 11 substances were taken into account as well using country-specific emissions factors from Koehler et al.[6]. More details on the transfer coefficients and emission factors are given in the Table S16 and Table S17 for wet APC systems and Table S19 and Table S20 for dry APC systems. For the treatment of the fly ashes, based on the current practice it was assumed that these are stabilized with cement and deposited on non-hazardous waste landfills. The inventory data for this process were taken from Huber et al.[58], although this model was rerun using the respective fly ash composition for waste plastics combustion as calculated in this study (see Table S22 and Table S23). The modelling of the bottom ash landfill was taken from the EASETECH database.[59] The slurry is deposited in an underground landfill for hazardous waste in Germany, which is modelled using the respective ecoinvent process[60] and the weighted average transport distance from the relevant incinerators. Finally, the electricity and heat requirements, as well as the elemental transfer coefficients to the effluent of the waste water treatment process were taken from Doka[42].

Table S13: Energy recovery efficiencies of each of the three incineration process archetypes.

Incineration process archetype	Efficiency	Marginal technology
Grate incinerator, wet APC	16%	Electricity from coal CHP
	43%	Heat from natural gas CHP
	16%	Heat from coal CHP
Grate incinerator, dry APC	21%	Electricity from coal CHP
	3%	Heat from small-scale natural gas boiler
Fluidized bed incinerator,	9%	Electricity from coal CHP
wet APC	27%	Heat from natural gas CHP
	10%	Heat from natural gas furnace
	25%	Heat from coal CHP

Table S14: Overview of the Austrian waste incinerators.

	Waste treated in 2014 (tonnes) [51, 52]	APC system	Energy recovery	Marginal heat technology
Grate incinerators				
A	99,000	Wet	Electricity + district heating	Natural gas CHP
B	209,000	Wet	District heating	Natural gas CHP
C	233,000	Wet	Electricity + district heating	Natural gas CHP
D	303,000	Wet	Electricity + district heating	Natural gas CHP
E	461,000	Wet	Steam to nearby coal power plant	Coal CHP
F	94,000	Dry	Electricity + district heating	Small-scale natural gas boiler
G	162,000	Dry	Electricity	n/a
Fluidized bed incinerators				
H	91,000	Wet	Electricity + district heating	Natural gas CHP
I	262,000	Wet	Electricity + heat to fiber industry	Coal CHP
J	186,000	Wet	Electricity + district heating	Natural gas CHP
K	107,000	Wet	Electricity + heat to paper industry	Natural gas industrial furnace

Table S15: Inventory data per kg waste for the operation of the grate incineration process with wet APC.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Share to wet APC	8.36E-01	4.16E-04	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	[51, 52]
Extra light fuel oil consumption	4.14E-02	1.19E-06	MJ/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Natural gas consumption	1.19E-01	9.88E-06	MJ/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Ammonia water consumption	2.44E-03	4.12E-09	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Activated carbon consumption	1.13E-04	8.88E-12	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Brown coal consumption	5.55E-04	2.13E-10	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data

Quicklime consumption	9.85E-04	6.73E-10	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Slaked lime consumption	4.69E-04	1.53E-10	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Sodium hydroxide consumption	5.36E-05	2.00E-12	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Limestone consumption	8.15E-03	4.61E-08	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Hydrochloric acid consumption	6.35E-05	2.80E-12	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Antifouling agent consumption	1.46E-05	1.47E-13	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Flocculant consumption	6.27E-06	2.73E-14	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Ferrichloride consumption	1.13E-04	8.79E-12	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Trimercapto-triazine consumption	2.51E-05	4.37E-13	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Water consumption	6.22E-01	2.68E-04	kg/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Electricity consumption (mass-weighted to plants not producing electricity)	4.17E-05	1.21E-12	kWh/kg	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Energy efficiency to electricity from coal CHP	1.57E-01	1.46E-05	MJ/MJ	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Energy efficiency to heat from gas CHP	4.28E-01	1.09E-04	MJ/MJ	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Energy efficiency to heat from coal CHP	1.56E-01	1.45E-05	MJ/MJ	Lognormal	0.0006-(1,2,1,1,1)	Primary data
Infrastructure	2.50E-10	2.58E-20	unit/kg	Lognormal	0.3-(3,3,5,3,1)	[61]

Table S16: Transfer coefficients to the slag, fly ash, air, slurry and effluent for the incineration processes with wet APC. The water emissions are reduced in the municipal wastewater treatment plant, which is modelled using the transfer coefficients reported by [42].

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O to slag	0.00E+00	0.00E+00	%	Normal	From source	6
O to slag	3.44E+00	0.00E+00	%	Normal	From source	6
H to slag	0.00E+00	0.00E+00	%	Normal	From source	6
C to slag	1.13E+00	6.75E-04	%	Lognormal	From source	6
S to slag	3.15E+01	5.62E+00	%	Normal	From source	6
N to slag	9.98E-01	5.66E-03	%	Lognormal	From source	6
Cl to slag	1.57E+01	1.59E+00	%	Normal	From source	6
Br to slag	1.35E+01	2.59E+01	%	Normal	From source	6
F to slag	4.11E+01	2.82E+01	%	Normal	From source	6
As to slag	4.97E+01	1.14E+01	%	Normal	From source	6
Cd to slag	1.68E+01	6.40E+00	%	Normal	From source	6
Co to slag	8.58E+01	2.34E+00	%	Normal	From source	6
Cr to slag	8.37E+01	6.50E+01	%	Normal	From source	6
Cu to slag	9.30E+01	2.65E+01	%	Normal	From source	6
Hg to slag	4.57E+00	5.11E+00	%	Normal	From source	6
Mn to slag	9.06E+01	1.04E+00	%	Normal	From source	6
Mo to slag	8.40E+01	4.16E+00	%	Normal	From source	6
Ni to slag	8.75E+01	1.61E+02	%	Normal	From source	6
Pb to slag	5.54E+01	4.40E+01	%	Normal	From source	6
Sb to slag	2.39E+01	5.15E+00	%	Normal	From source	6
Se to slag	8.00E+00	2.97E-01	%	Normal	From source	6
Sn to slag	5.61E+01	1.66E+02	%	Normal	From source	6
V to slag	8.88E+01	7.60E+01	%	Normal	From source	6
Zn to slag	4.51E+01	2.62E+01	%	Normal	From source	6
H_2O to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6
O to fly ash	1.14E+00	0.00E+00	%	Normal	From source	6
H to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6

C to fly ash	9.31E-02	5.42E-06	%	Lognormal	From source	6
S to fly ash	5.30E+01	6.66E+00	%	Normal	From source	6
N to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6
Cl to fly ash	3.87E+01	7.90E+00	%	Normal	From source	6
Br to fly ash	3.53E+01	1.21E+01	%	Normal	From source	6
F to fly ash	4.83E+01	8.47E+00	%	Normal	From source	6
As to fly ash	4.97E+01	5.48E+00	%	Normal	From source	6
Cd to fly ash	8.25E+01	1.50E+01	%	Normal	From source	6
Co to fly ash	1.22E+01	1.90E+00	%	Normal	From source	6
Cr to fly ash	1.63E+01	3.80E+00	%	Normal	From source	6
Cu to fly ash	6.97E+00	5.26E-01	%	Normal	From source	6
Hg to fly ash	2.87E+01	7.74E-01	%	Normal	From source	6
Mn to fly ash	9.24E+00	1.06E+00	%	Normal	From source	6
Mo to fly ash	1.59E+01	3.28E+00	%	Normal	From source	6
Ni to fly ash	1.25E+01	3.41E+01	%	Normal	From source	6
Pb to fly ash	4.42E+01	1.43E+01	%	Normal	From source	6
Sb to fly ash	7.60E+01	3.52E+01	%	Normal	From source	6
Se to fly ash	8.37E+01	1.55E+01	%	Normal	From source	6
Sn to fly ash	4.34E+01	2.55E+01	%	Normal	From source	6
V to fly ash	9.79E+00	1.17E+00	%	Normal	From source	6
Zn to fly ash	5.46E+01	1.39E+01	%	Normal	From source	6
H_2O to air	1.00E+02	0.00E+00	%	Normal	From source	6
O to air	9.53E+01	0.00E+00	%	Normal	From source	6
H to air	1.00E+02	0.00E+00	%	Normal	From source	6
C to air	9.88E+01	7.18E-02	%	Normal	From source	6
S to air	2.44E-01	7.15E-03	%	Lognormal	From source	6
N to air	9.89E+01	1.17E+03	%	Normal	From source	6
Cl to air	5.53E-02	2.34E-05	%	Lognormal	From source	6
Br to air	0.00E+00	0.00E+00	%	Normal	From source	6
F to air	6.74E-01	5.50E-02	%	Lognormal	From source	6

As to air	5.95E-01	4.24E-02	%	Lognormal	From source	6
Cd to air	3.72E-01	5.86E-02	%	Lognormal	From source	6
Co to air	9.43E-02	1.14E-03	%	Lognormal	From source	6
Cr to air	1.48E-02	3.16E-05	%	Lognormal	From source	6
Cu to air	2.33E-03	6.80E-07	%	Lognormal	From source	6
Hg to air	2.98E+00	1.14E+00	%	Lognormal	From source	6
Mn to air	3.46E-03	1.51E-06	%	Lognormal	From source	6
Mo to air	0.00E+00	0.00E+00	%	Normal	From source	6
Ni to air	6.08E-02	5.42E-04	%	Lognormal	From source	6
Pb to air	8.50E-02	8.43E-04	%	Lognormal	From source	6
Sb to air	1.24E-01	2.27E-03	%	Lognormal	From source	6
Se to air	1.20E+00	1.72E-01	%	Lognormal	From source	6
Sn to air	2.60E-01	1.23E-02	%	Lognormal	From source	6
V to air	3.91E-01	1.87E-02	%	Lognormal	From source	6
Zn to air	2.35E-02	4.42E-05	%	Lognormal	From source	6
H_2O to slurry	0.00E+00	0.00E+00	%	Normal	From source	6
O to slurry	1.00E-01	0.00E+00	%	Normal	From source	6
H to slurry	0.00E+00	0.00E+00	%	Normal	From source	6
C to slurry	1.12E-03	1.10E-10	%	Lognormal	From source	6
S to slurry	1.14E+01	8.12E-02	%	Normal	From source	6
N to slurry	0.00E+00	0.00E+00	%	Normal	From source	6
Cl to slurry	1.54E-02	2.11E-06	%	Lognormal	From source	6
Br to slurry	4.86E-02	4.27E-04	%	Lognormal	From source	6
F to slurry	5.45E+00	2.63E-01	%	Normal	From source	6
As to slurry	4.97E-02	2.17E-05	%	Lognormal	From source	6
Cd to slurry	3.87E-01	1.34E-03	%	Lognormal	From source	6
Co to slurry	1.89E+00	3.31E-02	%	Normal	From source	6
Cr to slurry	8.71E-03	8.00E-07	%	Lognormal	From source	6
Cu to slurry	1.21E-02	1.34E-06	%	Lognormal	From source	6
Hg to slurry	6.37E+01	1.34E+01	%	Normal	From source	6

Mn to slurry	1.41E-01	1.85E-04	%	Lognormal	From source	6
Mo to slurry	1.76E-01	2.97E-04	%	Lognormal	From source	6
Ni to slurry	5.78E-03	3.60E-07	%	Lognormal	From source	6
Pb to slurry	2.94E-01	7.96E-04	%	Lognormal	From source	6
Sb to slurry	1.74E-02	2.58E-06	%	Lognormal	From source	6
Se to slurry	7.10E+00	4.42E-01	%	Normal	From source	6
Sn to slurry	8.21E-02	2.61E-04	%	Lognormal	From source	6
V to slurry	9.78E-01	8.58E-03	%	Lognormal	From source	6
Zn to slurry	3.03E-01	8.30E-04	%	Lognormal	From source	6
H_2O to effluent	0.00E+00	0.00E+00	%	Normal	From source	6
O to effluent	0.00E+00	0.00E+00	%	Normal	From source	6
H to effluent	0.00E+00	0.00E+00	%	Normal	From source	6
C to effluent	1.12E-03	1.10E-10	%	Lognormal	From source	6
S to effluent	3.89E+00	8.12E-02	%	Normal	From source	6
N to effluent	1.00E-01	2.60E-04	%	Lognormal	From source	6
Cl to effluent	4.55E+01	1.07E+01	%	Normal	From source	6
Br to effluent	5.12E+01	2.60E+01	%	Normal	From source	6
F to effluent	4.51E+00	3.04E-01	%	Normal	From source	6
As to effluent	3.46E-06	2.23E-05	%	Lognormal	From source	6
Cd to effluent	3.34E-03	5.44E-08	%	Lognormal	From source	6
Co to effluent	9.10E-03	3.08E-02	%	Lognormal	From source	6
Cr to effluent	1.43E-03	7.65E-07	%	Lognormal	From source	6
Cu to effluent	2.30E-03	5.43E-06	%	Lognormal	From source	6
Hg to effluent	4.25E-02	6.51E-05	%	Lognormal	From source	6
Mn to effluent	0.00E+00	0.00E+00	%	Normal	From source	6
Mo to effluent	0.00E+00	0.00E+00	%	Normal	From source	6
Ni to effluent	2.10E-03	3.71E-07	%	Lognormal	From source	6
Pb to effluent	5.15E-03	8.88E-04	%	Lognormal	From source	6
Sb to effluent	4.35E-02	4.21E-05	%	Lognormal	From source	6
Se to effluent	1.02E-02	4.63E-01	%	Lognormal	From source	6

Sn to effluent	1.41E-01	1.89E-02	%	Lognormal	From source	6
V to effluent	9.33E-04	7.68E-03	%	Lognormal	From source	6
Zn to effluent	1.53E-03	8.42E-04	%	Lognormal	From source	6

Table S17: Process-specific air emission factors for the grate incineration process with wet APC.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
NOx emissions	7.31E-05	2.22E-09	kg/kg	Lognormal	From source	6
SO_2 emissions	5.93E-06	2.01E-12	kg/kg	Lognormal	From source	6
NH_3 emissions	3.23E-06	1.34E-12	kg/kg	Normal	From source	6
NMVOC emissions	6.13E-06	1.55E-10	kg/kg	Lognormal	From source	6
CO emissions	8.08E-05	1.20E-08	kg/kg	Lognormal	From source	6
HCl emissions	2.51E-06	2.91E-13	kg/kg	Lognormal	From source	6
HF emissions	4.16E-07	5.30E-15	kg/kg	Lognormal	From source	6
CH_4 emissions	6.36E-06	2.11E-12	kg/kg	Lognormal	From source	6
N_2O emissions	6.69E-06	5.75E-12	kg/kg	Normal	From source	6
PCDD/F emissions	5.53E-14	2.39E-26	kg/kg	Lognormal	From source	6
PM10 emissions	6.01E-06	1.26E-10	kg/kg	Lognormal	From source	6

Table S18: Inventory data per kg waste for the operation of the grate incineration process with dry APC.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Share to dry APC	1.64E-01	1.60E-05	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Extra light fuel oil consumption	3.90E-02	9.05E-07	MJ/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Diesel consumption	7.21E-03	3.10E-08	MJ/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Propane consumption	3.68E-07	8.06E-17	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data

Ammonia water consumption	2.58E-03	3.98E-09	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Activated carbon consumption	4.95E-05	1.46E-12	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Lignite coke consumption	4.19E-05	1.04E-12	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Brown coal consumption	6.42E-04	2.46E-10	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Slaked lime consumption	1.03E-02	6.30E-08	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Sodium bicarbonate consumption	2.06E-02	2.52E-07	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Sodium hydroxide consumption	4.14E-05	1.02E-12	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Hydrochloric acid consumption	9.96E-05	5.90E-12	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Water consumption	1.56E-01	1.44E-05	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Energy efficiency to electricity from coal CHP	2.05E-01	2.51E-05	MJ/MJ	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Energy efficiency to heat from gas small-scale boiler	2.92E-02	5.08E-07	MJ/MJ	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Infrastructure	2.50E-10	2.58E-20	unit/kg	Lognormal	0.3-(3,3,5,3,1)	[61]

Table S19: Transfer coefficients to the slag, fly ash, air, slurry and effluent for the incineration processes with wet APC. The water emissions are reduced in the municipal wastewater treatment plant, which is modelled using the transfer coefficients reported by [42].

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O to slag	0.00E+00	0.00E+00	%	Normal	From source	[6]
O to slag	3.43E+00	0.00E+00	%	Normal	From source	[6]
H to slag	0.00E+00	0.00E+00	%	Normal	From source	[6]
C to slag	1.13E+00	6.75E-04	%	Lognormal	From source	[6]
S to slag	3.15E+01	5.62E+00	%	Normal	From source	[6]
N to slag	9.98E-01	5.66E-03	%	Lognormal	From source	[6]

Cl to slag	1.58E+01	1.59E+00	%	Normal	From source	6
Br to slag	1.35E+01	2.59E+01	%	Normal	From source	6
F to slag	4.13E+01	2.82E+01	%	Normal	From source	6
As to slag	4.97E+01	1.14E+01	%	Normal	From source	6
Cd to slag	1.70E+01	6.40E+00	%	Normal	From source	6
Co to slag	8.87E+01	2.34E+00	%	Normal	From source	6
Cr to slag	9.20E+01	6.50E+01	%	Normal	From source	6
Cu to slag	9.52E+01	2.65E+01	%	Normal	From source	6
Hg to slag	4.58E+00	5.11E+00	%	Normal	From source	6
Mn to slag	9.32E+01	1.04E+00	%	Normal	From source	6
Mo to slag	8.79E+01	4.16E+00	%	Normal	From source	6
Ni to slag	9.70E+01	1.61E+02	%	Normal	From source	6
Pb to slag	5.68E+01	4.40E+01	%	Normal	From source	6
Sb to slag	2.34E+01	5.15E+00	%	Normal	From source	6
Se to slag	8.00E+00	2.97E-01	%	Normal	From source	6
Sn to slag	5.58E+01	1.66E+02	%	Normal	From source	6
V to slag	8.99E+01	6.23E+01	%	Normal	From source	6
Zn to slag	4.57E+01	2.62E+01	%	Normal	From source	6
H_2O to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6
O to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6
H to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6
C to fly ash	2.59E-01	7.14E-06	%	Lognormal	From source	6
S to fly ash	6.14E+01	1.27E+01	%	Normal	From source	6
N to fly ash	0.00E+00	0.00E+00	%	Normal	From source	6
Cl to fly ash	8.30E+01	2.17E+01	%	Normal	From source	6
Br to fly ash	8.64E+01	5.76E+01	%	Normal	From source	6
F to fly ash	5.82E+01	3.42E+01	%	Normal	From source	6
As to fly ash	4.99E+01	3.49E+01	%	Normal	From source	6
Cd to fly ash	8.25E+01	9.96E+01	%	Normal	From source	6
Co to fly ash	1.11E+01	2.59E+00	%	Normal	From source	6

Cr to fly ash	7.89E+00	1.28E+00	%	Normal	From source	6
Cu to fly ash	4.76E+00	2.40E+00	%	Normal	From source	6
Hg to fly ash	7.84E+01	1.35E+02	%	Normal	From source	6
Mn to fly ash	6.75E+00	1.14E+00	%	Normal	From source	6
Mo to fly ash	1.19E+01	4.37E+00	%	Normal	From source	6
Ni to fly ash	2.99E+00	5.75E-01	%	Normal	From source	6
Pb to fly ash	4.27E+01	1.66E+01	%	Normal	From source	6
Sb to fly ash	7.57E+01	1.16E+01	%	Normal	From source	6
Se to fly ash	9.20E+01	1.19E+02	%	Normal	From source	6
Sn to fly ash	4.38E+01	1.19E+02	%	Normal	From source	6
V to fly ash	1.00E+01	2.07E+00	%	Normal	From source	6
Zn to fly ash	5.40E+01	7.23E+01	%	Normal	From source	6
H_2O to air	1.00E+02	0.00E+00	%	Normal	From source	6
O to air	9.66E+01	0.00E+00	%	Normal	From source	6
H to air	1.00E+02	0.00E+00	%	Normal	From source	6
C to air	9.86E+01	4.12E-02	%	Normal	From source	6
S to air	7.09E+00	3.24E+00	%	Lognormal	From source	6
N to air	9.90E+01	2.13E+02	%	Normal	From source	6
Cl to air	1.22E+00	4.40E-05	%	Lognormal	From source	6
Br to air	1.40E-01	7.02E-06	%	Lognormal	From source	6
F to air	4.41E-01	2.10E-01	%	Lognormal	From source	6
As to air	4.58E-01	5.93E-03	%	Lognormal	From source	6
Cd to air	5.02E-01	1.61E-02	%	Lognormal	From source	6
Co to air	2.03E-01	9.01E-04	%	Lognormal	From source	6
Cr to air	1.51E-01	4.95E-04	%	Lognormal	From source	6
Cu to air	4.98E-02	2.39E-04	%	Lognormal	From source	6
Hg to air	1.70E+01	7.24E+01	%	Normal	From source	6
Mn to air	6.97E-02	1.06E-04	%	Lognormal	From source	6
Mo to air	2.20E-01	1.06E-03	%	Lognormal	From source	6
Ni to air	5.01E-02	5.48E-05	%	Lognormal	From source	6

Pb to air	5.01E-01	5.45E-03	%	Lognormal	From source	6
Sb to air	9.47E-01	1.96E-02	%	Lognormal	From source	6
Se to air	5.01E-02	5.48E-05	%	Lognormal	From source	6
Sn to air	3.59E-01	4.27E-05	%	Lognormal	From source	6
V to air	5.01E-02	5.48E-05	%	Lognormal	From source	6
Zn to air	3.52E-01	7.01E-05	%	Lognormal	From source	6

Table S20: Process-specific air emission factors for the grate incineration process with dry APC.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
NOx emissions	9.50E-05	1.98E-09	kg/kg	Lognormal	From source	6
SO_2 emissions	1.36E-04	4.32E-09	kg/kg	Normal	From source	6
NH_3 emissions	3.87E-06	7.35E-13	kg/kg	Lognormal	From source	6
NMVOC emissions	8.25E-06	1.83E-10	kg/kg	Lognormal	From source	6
CO emissions	6.42E-05	3.40E-09	kg/kg	Lognormal	From source	6
HCl emissions	5.50E-05	1.44E-10	kg/kg	Lognormal	From source	6
HF emissions	1.94E-07	1.18E-15	kg/kg	Lognormal	From source	6
CH_4 emissions	6.37E-06	2.13E-12	kg/kg	Lognormal	From source	6
N_2O emissions	8.02E-06	3.16E-12	kg/kg	Lognormal	From source	6
PCDD/F emissions	4.15E-14	5.85E-27	kg/kg	Lognormal	From source	6
PM10 emissions	8.70E-06	2.86E-10	kg/kg	Lognormal	From source	6

Table S21: Inventory data per kg waste for the operation of the fluidized bed incineration process with wet APC.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Ammonia water consumption	6.20E-04	4.57E-10	kg/kg	Lognormal	0.0006-(1,3,1,1,1)	Primary data
Activated carbon consumption	2.32E-04	6.40E-11	kg/kg	Lognormal	0.0006-(1,3,1,1,1)	Primary data
Limestone consumption	2.30E-03	6.30E-09	kg/kg	Lognormal	0.0006-(1,3,1,1,1)	Primary data
Quicklime consumption	8.13E-04	7.87E-10	kg/kg	Lognormal	0.0006-(1,3,1,1,1)	Primary data
Energy efficiency to electricity from coal CHP	8.52E-02	4.32E-06	MJ/MJ	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Energy efficiency to heat from gas CHP	2.66E-01	4.22E-05	MJ/MJ	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Energy efficiency to heat from gas furnace	1.02E-01	6.24E-06	MJ/MJ	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Energy efficiency to heat from coal CHP	2.52E-01	3.78E-05	MJ/MJ	Lognormal	0.0006-(1,1,1,1,1)	Primary data
Infrastructure	2.50E-10	2.58E-20	unit/kg	Lognormal	0.3-(3,3,5,3,1)	[61]

Table S22: Inventory data per kg of fly ash for the operation of fly ash treatment.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
Transport	4.30E+01	2.36E+02	km	Lognormal	0.12-(1,1,1,1,1)	Calculated using [58]
Cement consumption	1.00E+00	5.95E-04	kg/kg	Lognormal	0.0006-(1,1,1,1,1)	Calculated using [58]
Electricity consumption	5.00E-02	2.50E-05	kWh/kg	Normal	From source	Calculated using [58]
Slag landfill process specific burdens	2.14E+00	1.34E-03	kg/kg	Normal	From source	Calculated using [58]
Slag landfill construction	3.80E-09	4.22E-21	m3/kg	Normal	From source	Calculated using [58]
Water consumption	1.39E-01	1.34E-03	kg/kg	Normal	From source	Calculated using [58]

Table S23: Transfer coefficients from landfilled fly ash to leachate treatment effluent.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
As to effluent	3.21E-05	9.38E-10	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Cd to effluent	2.17E-06	4.29E-12	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Co to effluent	1.64E-07	2.45E-14	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Cr to effluent	5.46E-09	2.71E-17	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Cu to effluent	4.10E-09	1.53E-17	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Hg to effluent	5.71E-03	2.97E-05	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Mn to effluent	1.80E-08	2.96E-16	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Mo to effluent	3.42E-02	1.07E-03	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Ni to effluent	3.98E-06	1.45E-11	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Pb to effluent	4.32E-10	1.70E-19	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Sb to effluent	3.83E-06	1.34E-11	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Se to effluent	3.19E-03	9.25E-06	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Sn to effluent	1.36E-09	1.68E-18	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
V to effluent	1.59E-08	2.32E-16	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]
Zn to effluent	4.34E-09	1.72E-17	%	Lognormal	0.65-(1,1,1,1,1)	Calculated using [58]

1.6 Industrial incineration

The industrial incineration processes were modeled using a modular excel-based model.[62] As the composition of the input waste has an influence on the results, this was done separately for each of the ten distinct industrial incineration processes. These results are expressed relative to the energy input (i.e. per MJ), as the replaced fuel (hard coal in this case) is substituted on this basis. The substitution factor was calculated using the energy of each of the fuels needed to produce one tonne of clinker: 3200 MJ/t for coal, 3236 MJ/t for PET waste streams and 3234 MJ/t for the other plastic waste streams.[62] These inventory data are presented in Table S24 for the treatment of PET material, Table S25 for PE, Table S26 for PP, Table S27 for PS, Table S28 for the residues of mixed-polymer recycling, Table S29 for the residues of the sorting process, Table S30 for material from the mechanical pretreatment of RW, and Table S31 for material from the mechanical pretreatment of BCW. The inventory data for the utilization of the substituted fuel is presented in Table S32. Finally, the transfer coefficients on which the emission factors in the previously mentioned tables are based are presented in Table S33.

Table S24: Inventory data per MJ of waste material of the mechanical recycling process of PET for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.48E-01	7.79E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.97E-06	1.76E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	7.08E-09	4.59E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	1.56E-09	2.23E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	4.60E-10	1.94E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.11E-11	4.08E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.45E-07	1.93E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.73E-12	5.47E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

Co air emissions	5.07E-11	2.35E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	2.57E-10	6.04E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.51E-11	2.09E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	2.37E-10	5.14E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	1.19E-09	1.30E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.47E-05	2.85E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.32E-04	4.92E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.93E-05	3.87E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.83E-06	1.51E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.32E-15	2.42E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.55E-04	2.93E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.83E-05	1.51E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.39E-02	2.03E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S25: Inventory data per MJ of waste material of the mechanical recycling process of PE for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.05E-01	3.90E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.18E-06	6.32E-14	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	6.59E-09	3.98E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	3.25E-09	9.66E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

Sb air emissions	3.05E-11	8.50E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.11E-11	4.09E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.28E-07	1.50E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.70E-12	5.43E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	5.07E-11	2.36E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	1.47E-10	1.97E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.42E-11	1.84E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	2.46E-10	5.55E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	9.71E-10	8.64E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.48E-05	2.86E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.36E-04	4.96E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.94E-05	3.90E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.84E-06	1.52E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.36E-15	2.44E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.56E-04	2.95E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.84E-05	1.52E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S26: Inventory data per MJ of waste material of the mechanical recycling process of PP for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.07E-01	4.06E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.18E-06	6.32E-14	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]

Cd air emissions	6.59E-09	3.98E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	3.28E-09	9.85E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	3.05E-11	8.50E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.11E-11	4.09E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.28E-07	1.51E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.70E-12	5.43E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	5.07E-11	2.36E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	1.47E-10	1.98E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.42E-11	1.84E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	2.46E-10	5.55E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	9.71E-10	8.64E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.46E-05	2.80E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.20E-04	4.76E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.88E-05	3.75E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.80E-06	1.46E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.20E-15	2.34E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.51E-04	2.83E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.80E-05	1.46E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]

SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S27: Inventory data per MJ of waste material of the mechanical recycling process of PS for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.22E-01	5.33E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.73E-06	1.35E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	6.62E-09	4.01E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	1.27E-09	1.48E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	3.05E-11	8.50E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.86E-11	7.51E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.75E-07	2.80E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.61E-12	5.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	8.10E-11	6.02E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	2.60E-10	6.21E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.42E-11	1.85E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	1.00E-08	9.20E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	1.16E-09	1.24E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

Dust air emissions	1.45E-05	2.74E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.05E-04	4.56E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.82E-05	3.58E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.76E-06	1.40E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.05E-15	2.24E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.45E-04	2.71E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.76E-05	1.40E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S28: Inventory data per MJ of waste material of the mechanical recycling process of mixed-polymers for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.09E-01	4.26E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.67E-06	1.26E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	6.71E-09	4.13E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	3.01E-09	8.30E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	6.22E-11	3.54E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.16E-11	4.29E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.33E-07	1.62E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.70E-12	5.43E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	5.29E-11	2.56E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	1.63E-10	2.43E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.42E-11	1.85E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	9.09E-10	7.57E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	1.00E-09	9.20E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.47E-05	2.84E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.30E-04	4.88E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.92E-05	3.85E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.83E-06	1.50E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.30E-15	2.40E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.54E-04	2.91E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.83E-05	1.50E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S29: Inventory data per MJ of waste material of the sorting and preparation process for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.09E-01	4.26E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.67E-06	1.26E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	6.71E-09	4.13E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	3.01E-09	8.30E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	6.22E-11	3.54E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.16E-11	4.29E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.33E-07	1.62E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.70E-12	5.43E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	5.29E-11	2.56E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

Cu air emissions	1.63E-10	2.42E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.42E-11	1.85E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	9.09E-10	7.57E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	1.00E-09	9.20E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.47E-05	2.84E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.30E-04	4.88E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.92E-05	3.85E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.83E-06	1.50E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.30E-15	2.40E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.54E-04	2.91E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.83E-05	1.50E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S30: Inventory data per MJ of waste material of the mechanical pretreatment process of RW for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.12E-01	4.45E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.56E-06	1.10E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	6.71E-09	4.13E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Hg air emissions	2.91E-09	7.74E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	8.41E-11	6.48E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

As air emissions	2.16E-11	4.29E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.34E-07	1.64E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.70E-12	5.43E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	5.29E-11	2.56E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	1.69E-10	2.60E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.43E-11	1.87E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	9.18E-10	7.73E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	1.01E-09	9.43E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.47E-05	2.84E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.30E-04	4.88E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.92E-05	3.83E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.82E-06	1.49E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.30E-15	2.40E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.54E-04	2.90E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.82E-05	1.49E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S31: Inventory data per MJ of waste material of the mechanical pretreatment process of BCW for the production of clinker.

Parameter	Mean	Variance	Unit	Distribution	Uncertainty source	Source
H_2O air emissions	5.07E-03	9.16E-08	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fossil CO_2 air emissions	1.07E-01	4.04E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
CO_2 (from calcination) air emissions	1.63E-01	9.51E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
HCl air emissions	1.24E-06	6.93E-14	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Cd air emissions	6.62E-09	4.01E-17	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]

Hg air emissions	3.18E-09	9.30E-18	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Tl air emissions	6.67E-12	4.08E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sb air emissions	3.99E-11	1.46E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
As air emissions	2.14E-11	4.18E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Pb air emissions	1.30E-07	1.55E-14	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cr air emissions	7.70E-12	5.43E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Co air emissions	5.16E-11	2.44E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Cu air emissions	1.52E-10	2.13E-20	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Mn air emissions	7.42E-11	5.04E-21	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Ni air emissions	1.42E-11	1.85E-22	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
V air emissions	5.01E-12	2.30E-23	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Sn air emissions	5.41E-10	2.68E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Zn air emissions	9.83E-10	8.86E-19	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
Dust air emissions	1.47E-05	2.85E-11	kg/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
CO air emissions	7.33E-04	4.92E-07	kg/MJ	Lognormal	0.65-(2,1,3,3,1)	Calculated using [62]
VOC air emissions	2.93E-05	3.86E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
Benzene air emissions	1.83E-06	1.51E-13	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
PCDD/F air emissions	7.33E-15	2.42E-30	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NOx air emissions	2.55E-04	2.92E-09	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
NH_3 air emissions	1.83E-05	1.51E-11	kg/MJ	Lognormal	0.04-(2,1,3,3,1)	Calculated using [62]
SO_2 air emissions	2.55E-04	2.31E-10	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Limestone consumption	4.33E-01	6.67E-04	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Clay consumption	6.80E-02	1.65E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Fe-corrective consumption	6.95E-03	1.72E-07	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Calcium hydroxide consumption	6.18E-05	1.36E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Electricity consumption	2.37E-02	2.00E-06	kWh/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Lubricating oil consumption	1.46E-05	7.55E-13	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, basic consumption	5.87E-05	1.23E-11	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, fireclay consumption	2.54E-05	2.30E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Refractory, high alu oxide consumption	4.23E-05	6.39E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Chromium steel consumption	1.81E-05	1.17E-12	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Infrastructure	1.94E-12	1.34E-24	unit/MJ	Lognormal	0.3-(2,1,3,3,1)	Calculated using [62]
Water consumption	7.73E-02	2.13E-05	kg/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]
Road transport	3.49E-01	1.60E-02	kg*km/MJ	Lognormal	0.12-(2,1,3,3,1)	Calculated using [62]
Coal substitution factor	9.89E-01	3.48E-05	MJ/MJ	Lognormal	0.0006-(2,1,3,3,1)	Calculated using [62]

Table S32: Inventory data per MJ of hard coal for the production of clinker.

Parameter	Mean	Unit	Source
H_2O air emissions	5.69E-03	kg/MJ	Calculated using [62]
Fossil CO_2 air emissions	1.22E-01	kg/MJ	Calculated using [62]
CO_2 (from calcination) air emissions	1.66E-01	kg/MJ	Calculated using [62]

HCl air emissions	1.63E-06	kg/MJ	Calculated using [62]
Cd air emissions	5.88E-09	kg/MJ	Calculated using [62]
Hg air emissions	1.73E-09	kg/MJ	Calculated using [62]
Tl air emissions	7.44E-12	kg/MJ	Calculated using [62]
Sb air emissions	3.16E-11	kg/MJ	Calculated using [62]
As air emissions	2.44E-11	kg/MJ	Calculated using [62]
Pb air emissions	8.16E-08	kg/MJ	Calculated using [62]
Cr air emissions	7.88E-12	kg/MJ	Calculated using [62]
Co air emissions	5.44E-11	kg/MJ	Calculated using [62]
Cu air emissions	6.00E-11	kg/MJ	Calculated using [62]
Mn air emissions	7.38E-11	kg/MJ	Calculated using [62]
Ni air emissions	1.57E-11	kg/MJ	Calculated using [62]
V air emissions	5.31E-12	kg/MJ	Calculated using [62]
Sn air emissions	2.69E-10	kg/MJ	Calculated using [62]
Zn air emissions	8.16E-10	kg/MJ	Calculated using [62]
Dust air emissions	1.42E-05	kg/MJ	Calculated using [62]
CO air emissions	6.66E-04	kg/MJ	Calculated using [62]
VOC air emissions	2.66E-05	kg/MJ	Calculated using [62]
Benzene air emissions	1.66E-06	kg/MJ	Calculated using [62]
PCDD/F air emissions	6.66E-15	kg/MJ	Calculated using [62]
NOx air emissions	2.32E-04	kg/MJ	Calculated using [62]
NH_3 air emissions	1.66E-05	kg/MJ	Calculated using [62]
SO_2 air emissions	2.49E-04	kg/MJ	Calculated using [62]
Limestone consumption	4.38E-01	kg/MJ	Calculated using [62]
Clay consumption	6.41E-02	kg/MJ	Calculated using [62]
Fe-corrective consumption	5.94E-03	kg/MJ	Calculated using [62]
Calcium hydroxide consumption	6.25E-05	kg/MJ	Calculated using [62]
Electricity consumption	2.50E-02	kWh/MJ	Calculated using [62]
Lubricating oil consumption	1.47E-05	kg/MJ	Calculated using [62]
Refractory, basic consumption	5.94E-05	kg/MJ	Calculated using [62]

Refractory, fireclay consumption	2.57E-05	kg/MJ	Calculated using [62]
Refractory, high alu oxide consumption	4.28E-05	kg/MJ	Calculated using [62]
Chromium steel consumption	1.83E-05	kg/MJ	Calculated using [62]
Infrastructure	1.96E-12	unit/MJ	Calculated using [62]
Water consumption	7.81E-02	kg/MJ	Calculated using [62]
Road transport	2.16E+00	kg*km/MJ	Calculated using [62]
Hard coal consumption	3.72E-2	kg/MJ	Calculated using [62]

Table S33: Elemental transfer coefficients to the flue gas in the cement kiln process.

Element	TC	Unit	Source
Cd	0.170	%	[62, 63]
Hg	40.000	%	[62, 63]
Tl	1.300	%	[62, 63]
Sb	0.030	%	[62, 63]
As	0.023	%	[62, 63]
Pb	0.050	%	[62, 63]
Cr	0.012	%	[62, 63]
Co	0.019	%	[62, 63]
Cu	0.093	%	[62, 63]
Mn	0.018	%	[62, 63]
Ni	0.030	%	[62, 63]
V	0.052	%	[62, 63]
Sn	0.074	%	[62, 63]
Zn	0.074	%	[62, 63]

2 Results

Table S34: Net LCIA results (in PE) of the four scenarios. For the mechanical recycling, industrial incineration and WtE processes, the benefits and burdens are furthermore displayed separately in Table S35.

		Collection	Sorting and Preparation	Mechanical Pretreatment	Mechanical Recycling	Industrial Incineration	WtE	Landfill	Total
GW	Mainly disposal	1.21E+03	7.27E+02	1.23E+02	-7.37E+03	-4.12E+03	-1.59E+02	2.86E+02	-9.30E+03
	Status quo	1.58E+03	1.49E+03	1.13E+02	-1.53E+04	-1.16E+04	1.28E+03	2.66E+00	-2.24E+04
	EU-target-SP	1.80E+03	1.89E+03	5.84E+01	-2.49E+04	-1.03E+04	3.86E+02	1.37E+00	-3.11E+04
	EU-target-MP	1.80E+03	1.89E+03	5.84E+01	-1.26E+04	-1.02E+04	3.86E+02	1.37E+00	-1.87E+04
OD	Mainly disposal	4.89E-01	2.07E+01	2.49E+00	-2.16E+01	-4.94E+00	-1.41E+02	2.71E+01	-1.17E+02
	Status quo	6.36E-01	4.24E+01	2.30E+00	-4.65E+01	-1.42E+01	-3.40E+02	2.50E-01	-3.55E+02
	EU-target-SP	7.25E-01	5.40E+01	1.18E+00	-7.36E+01	-1.30E+01	-2.32E+02	1.29E-01	-2.62E+02
	EU-target-MP	7.25E-01	5.40E+01	1.18E+00	-3.47E+01	-1.22E+01	-2.32E+02	1.29E-01	-2.23E+02
HTc	Mainly disposal	1.61E+01	6.87E+02	1.30E+02	-1.00E+04	-2.18E+02	-1.61E+03	6.00E+02	-1.04E+04
	Status quo	2.21E+01	1.40E+03	1.20E+02	-2.22E+04	-6.27E+02	-3.76E+03	5.54E+00	-2.51E+04
	EU-target-SP	2.57E+01	1.79E+03	6.18E+01	-3.58E+04	-5.88E+02	-2.57E+03	2.86E+00	-3.71E+04
	EU-target-MP	2.57E+01	1.79E+03	6.18E+01	-2.29E+05	-5.40E+02	-2.57E+03	2.86E+00	-2.30E+05
HTnc	Mainly disposal	2.84E+02	2.38E+02	5.77E+01	-1.48E+03	1.89E+03	-9.85E+02	9.62E+02	9.64E+02
	Status quo	2.49E+02	4.86E+02	5.32E+01	-3.54E+03	5.40E+03	-2.32E+03	8.91E+00	3.27E+02
	EU-target-SP	2.31E+02	6.19E+02	2.74E+01	-5.85E+03	4.88E+03	-1.56E+03	4.59E+00	-1.64E+03
	EU-target-MP	2.31E+02	6.19E+02	2.74E+01	-4.39E+04	4.67E+03	-1.56E+03	4.59E+00	-3.99E+04
PM	Mainly disposal	4.10E+02	1.89E+03	1.38E+02	-1.47E+04	3.82E+02	-3.47E+03	1.18E+03	-1.41E+04
	Status quo	5.56E+02	3.86E+03	1.28E+02	-3.09E+04	1.10E+03	-7.97E+03	1.09E+01	-3.32E+04
	EU-target-SP	6.44E+02	4.92E+03	6.58E+01	-5.04E+04	1.01E+03	-5.45E+03	5.64E+00	-4.92E+04
	EU-target-MP	6.44E+02	4.92E+03	6.58E+01	-3.40E+04	9.43E+02	-5.45E+03	5.64E+00	-3.28E+04
IR	Mainly disposal	1.85E+01	2.50E+02	3.60E+01	-3.90E+02	-1.73E+02	-3.96E+02	3.00E+02	-3.53E+02
	Status quo	2.41E+01	5.12E+02	3.32E+01	-8.89E+02	-4.95E+02	-1.01E+03	2.77E+00	-1.82E+03
	EU-target-SP	2.75E+01	6.52E+02	1.71E+01	-1.47E+03	-4.54E+02	-6.90E+02	1.43E+00	-1.92E+03
	EU-target-MP	2.75E+01	6.52E+02	1.71E+01	-4.96E+02	-4.27E+02	-6.90E+02	1.43E+00	-9.16E+02
POF	Mainly disposal	2.31E+02	9.22E+02	5.83E+01	-4.67E+03	5.46E+02	-1.95E+03	4.00E+02	-4.46E+03
	Status quo	3.06E+02	1.89E+03	5.38E+01	-9.68E+03	1.59E+03	-4.39E+03	3.70E+00	-1.02E+04
	EU-target-SP	3.51E+02	2.40E+03	2.77E+01	-1.54E+04	1.46E+03	-3.00E+03	1.91E+00	-1.42E+04
	EU-target-MP	3.51E+02	2.40E+03	2.77E+01	-8.01E+03	1.35E+03	-3.00E+03	1.91E+00	-6.88E+03
TA	Mainly disposal	2.88E+02	8.77E+02	7.50E+01	-6.03E+03	8.33E+02	-3.58E+03	7.25E+02	-6.82E+03
	Status quo	3.77E+02	1.79E+03	6.92E+01	-1.26E+04	2.41E+03	-8.06E+03	6.70E+00	-1.60E+04

	EU-target-SP	4.32E+02	2.28E+03	3.56E+01	-2.05E+04	2.21E+03	-5.51E+03	3.45E+00	-2.11E+04
	EU-target-MP	4.32E+02	2.28E+03	3.56E+01	-9.17E+03	2.06E+03	-5.51E+03	3.45E+00	-9.86E+03
TE	Mainly disposal	3.75E+02	1.65E+03	1.05E+02	-4.33E+03	1.35E+03	-3.18E+03	6.15E+02	-3.42E+03
	Status quo	4.94E+02	3.38E+03	9.67E+01	-9.21E+03	3.90E+03	-7.09E+03	5.68E+00	-8.41E+03
	EU-target-SP	5.66E+02	4.31E+03	4.98E+01	-1.51E+04	3.58E+03	-4.85E+03	2.93E+00	-1.15E+04
	EU-target-MP	5.66E+02	4.31E+03	4.98E+01	-9.03E+03	3.32E+03	-4.85E+03	2.93E+00	-5.63E+03
FE	Mainly disposal	3.81E+00	4.01E+02	1.32E+02	-9.02E+02	-3.76E+03	-4.11E+03	1.13E+02	-8.13E+03
	Status quo	4.95E+00	8.21E+02	1.21E+02	-2.27E+03	-1.08E+04	-9.04E+03	1.04E+00	-2.12E+04
	EU-target-SP	5.65E+00	1.05E+03	6.26E+01	-4.26E+03	-9.91E+03	-6.18E+03	5.37E-01	-1.92E+04
	EU-target-MP	5.65E+00	1.05E+03	6.26E+01	-1.34E+02	-9.30E+03	-6.18E+03	5.37E-01	-1.45E+04
ME	Mainly disposal	4.23E+02	1.83E+03	1.12E+02	-4.74E+03	1.24E+03	-3.49E+03	4.33E+03	-3.00E+02
	Status quo	5.57E+02	3.75E+03	1.03E+02	-1.02E+04	3.59E+03	-7.81E+03	4.00E+01	-9.97E+03
	EU-target-SP	6.39E+02	4.78E+03	5.30E+01	-1.68E+04	3.29E+03	-5.35E+03	2.06E+01	-1.33E+04
	EU-target-MP	6.39E+02	4.78E+03	5.30E+01	-9.80E+03	3.06E+03	-5.35E+03	2.06E+01	-6.59E+03
ET	Mainly disposal	9.78E+02	1.38E+03	2.10E+02	-1.56E+04	-3.76E+02	-3.63E+03	2.62E+04	9.20E+03
	Status quo	9.39E+02	2.82E+03	1.94E+02	-3.56E+04	-1.08E+03	-8.64E+03	2.42E+02	-4.11E+04
	EU-target-SP	9.27E+02	3.59E+03	9.98E+01	-5.82E+04	-1.00E+03	-5.95E+03	1.25E+02	-6.04E+04
	EU-target-MP	9.27E+02	3.59E+03	9.98E+01	-2.18E+05	-9.39E+02	-5.95E+03	1.25E+02	-2.20E+05
LU	Mainly disposal	0.00E+00	5.05E+01	6.35E+00	1.49E+01	-1.76E+02	-2.56E+02	5.62E+02	2.02E+02
	Status quo	0.00E+00	1.03E+02	5.86E+00	-2.45E+01	-5.05E+02	-5.77E+02	5.19E+00	-9.92E+02
	EU-target-SP	0.00E+00	1.32E+02	3.02E+00	-4.39E+01	-4.64E+02	-3.94E+02	2.67E+00	-7.64E+02
	EU-target-MP	0.00E+00	1.32E+02	3.02E+00	-1.04E+03	-4.35E+02	-3.94E+02	2.67E+00	-1.73E+03
RDw	Mainly disposal	-8.62E-03	8.05E-02	2.12E-02	1.02E+00	-6.30E-01	-1.88E+00	6.97E-01	-7.00E-01
	Status quo	-1.12E-02	1.65E-01	1.96E-02	1.17E+00	-1.81E+00	-4.18E+00	6.44E-03	-4.64E+00
	EU-target-SP	-1.28E-02	2.10E-01	1.01E-02	1.45E+00	-1.66E+00	-2.86E+00	3.32E-03	-2.86E+00
	EU-target-MP	-1.28E-02	2.10E-01	1.01E-02	-2.87E+00	-1.56E+00	-2.86E+00	3.32E-03	-7.08E+00
RDm	Mainly disposal	7.20E+02	3.85E+03	1.21E+03	-3.15E+04	-4.76E+02	-3.42E+03	3.68E+03	-2.59E+04
	Status quo	9.37E+02	7.88E+03	1.11E+03	-7.48E+04	-1.37E+03	-8.28E+03	3.40E+01	-7.45E+04
	EU-target-SP	1.07E+03	1.00E+04	5.74E+02	-1.29E+05	-1.25E+03	-5.70E+03	1.75E+01	-1.24E+05
	EU-target-MP	1.07E+03	1.00E+04	5.74E+02	-3.26E+04	-1.18E+03	-5.70E+03	1.75E+01	-2.78E+04
RDf	Mainly disposal	2.09E+03	1.50E+03	2.81E+02	-3.00E+04	-3.52E+04	-4.16E+04	1.51E+03	-1.01E+05
	Status quo	2.72E+03	3.08E+03	2.59E+02	-6.11E+04	-1.01E+05	-9.26E+04	1.40E+01	-2.49E+05
	EU-target-SP	3.10E+03	3.92E+03	1.33E+02	-1.01E+05	-9.27E+04	-6.32E+04	7.21E+00	-2.50E+05
	EU-target-MP	3.10E+03	3.92E+03	1.33E+02	-1.37E+04	-8.70E+04	-6.32E+04	7.21E+00	-1.57E+05

Table S35: Benefits and burdens (in PE) of the mechanical recycling, industrial incineration and WtE processes. For each process, the net benefit or burden is displayed in Table S34.

		Mechanical recycling		Industrial incineration		WtE	
		Benefits	Burdens	Benefits	Burdens	Benefits	Burdens
GW	Mainly disposal	-9.86E+03	2.49E+03	-5.17E+04	4.76E+04	-1.82E+04	1.81E+04
	Status quo	-1.92E+04	3.91E+03	-1.49E+05	1.37E+05	-4.06E+04	4.18E+04
	EU-target-SP	-3.14E+04	6.51E+03	-1.36E+05	1.26E+05	-2.77E+04	2.80E+04
	EU-target-MP	-1.85E+04	5.90E+03	-1.28E+05	1.18E+05	-2.77E+04	2.80E+04
OD	Mainly disposal	-7.38E+01	5.22E+01	-5.00E+01	4.50E+01	-1.47E+02	5.98E+00
	Status quo	-1.23E+02	7.62E+01	-1.43E+02	1.29E+02	-3.52E+02	1.14E+01
	EU-target-SP	-2.00E+02	1.26E+02	-1.32E+02	1.19E+02	-2.39E+02	7.54E+00
	EU-target-MP	-1.46E+02	1.11E+02	-1.24E+02	1.11E+02	-2.39E+02	7.54E+00
HTc	Mainly disposal	-1.25E+04	2.47E+03	-3.63E+03	3.41E+03	-2.20E+03	5.96E+02
	Status quo	-2.64E+04	4.22E+03	-1.04E+04	9.81E+03	-5.08E+03	1.29E+03
	EU-target-SP	-4.26E+04	6.83E+03	-9.57E+03	8.98E+03	-3.44E+03	8.70E+02
	EU-target-MP	-2.35E+05	5.50E+03	-8.98E+03	8.44E+03	-3.44E+03	8.70E+02
HTnc	Mainly disposal	-2.58E+03	1.10E+03	-4.51E+03	6.40E+03	-2.16E+03	1.17E+03
	Status quo	-5.47E+03	1.93E+03	-1.29E+04	1.84E+04	-4.77E+03	2.44E+03
	EU-target-SP	-9.05E+03	3.20E+03	-1.19E+04	1.68E+04	-3.25E+03	1.70E+03
	EU-target-MP	-4.69E+04	2.97E+03	-1.12E+04	1.58E+04	-3.25E+03	1.70E+03
PM	Mainly disposal	-1.68E+04	2.09E+03	-1.50E+04	1.54E+04	-3.92E+03	4.44E+02
	Status quo	-3.38E+04	2.89E+03	-4.31E+04	4.43E+04	-8.88E+03	9.02E+02
	EU-target-SP	-5.51E+04	4.73E+03	-3.96E+04	4.06E+04	-6.05E+03	5.98E+02
	EU-target-MP	-3.81E+04	4.10E+03	-3.71E+04	3.81E+04	-6.05E+03	5.98E+02
IR	Mainly disposal	-9.28E+02	5.38E+02	-8.07E+02	6.35E+02	-4.48E+02	5.27E+01
	Status quo	-1.83E+03	9.39E+02	-2.32E+03	1.82E+03	-1.11E+03	1.02E+02
	EU-target-SP	-3.03E+03	1.56E+03	-2.13E+03	1.67E+03	-7.58E+02	6.74E+01
	EU-target-MP	-1.94E+03	1.44E+03	-2.00E+03	1.57E+03	-7.58E+02	6.74E+01
POF	Mainly disposal	-5.33E+03	6.57E+02	-8.95E+03	9.50E+03	-2.12E+03	1.66E+02
	Status quo	-1.06E+04	9.22E+02	-2.57E+04	2.73E+04	-4.75E+03	3.60E+02
	EU-target-SP	-1.70E+04	1.51E+03	-2.36E+04	2.50E+04	-3.24E+03	2.39E+02
	EU-target-MP	-9.41E+03	1.39E+03	-2.21E+04	2.35E+04	-3.24E+03	2.39E+02
TA	Mainly disposal	-7.20E+03	1.17E+03	-1.69E+04	1.77E+04	-3.85E+03	2.64E+02
	Status quo	-1.44E+04	1.73E+03	-4.84E+04	5.08E+04	-8.61E+03	5.46E+02

	EU-target-SP	-2.34E+04	2.87E+03	-4.44E+04	4.66E+04	-5.87E+03	3.62E+02
	EU-target-MP	-1.18E+04	2.67E+03	-4.17E+04	4.37E+04	-5.87E+03	3.62E+02
TE	Mainly disposal	-5.39E+03	1.05E+03	-1.72E+04	1.85E+04	-3.49E+03	3.06E+02
	Status quo	-1.08E+04	1.57E+03	-4.92E+04	5.32E+04	-7.76E+03	6.69E+02
	EU-target-SP	-1.77E+04	2.56E+03	-4.52E+04	4.88E+04	-5.29E+03	4.44E+02
	EU-target-MP	-1.14E+04	2.33E+03	-4.24E+04	4.57E+04	-5.29E+03	4.44E+02
FE	Mainly disposal	-3.43E+03	2.52E+03	-5.81E+03	2.04E+03	-4.21E+03	1.03E+02
	Status quo	-6.89E+03	4.63E+03	-1.67E+04	5.88E+03	-9.26E+03	2.03E+02
	EU-target-SP	-1.18E+04	7.58E+03	-1.53E+04	5.39E+03	-6.31E+03	1.35E+02
	EU-target-MP	-6.58E+03	6.45E+03	-1.44E+04	5.06E+03	-6.31E+03	1.35E+02
ME	Mainly disposal	-6.02E+03	1.27E+03	-1.61E+04	1.73E+04	-3.87E+03	3.74E+02
	Status quo	-1.20E+04	1.83E+03	-4.61E+04	4.97E+04	-8.61E+03	7.94E+02
	EU-target-SP	-1.97E+04	2.96E+03	-4.23E+04	4.56E+04	-5.87E+03	5.26E+02
	EU-target-MP	-1.22E+04	2.43E+03	-3.97E+04	4.28E+04	-5.87E+03	5.26E+02
ET	Mainly disposal	-2.08E+04	5.26E+03	-1.01E+04	9.67E+03	-5.18E+03	1.55E+03
	Status quo	-4.37E+04	8.15E+03	-2.88E+04	2.78E+04	-1.18E+04	3.12E+03
	EU-target-SP	-7.13E+04	1.31E+04	-2.65E+04	2.55E+04	-8.03E+03	2.08E+03
	EU-target-MP	-2.28E+05	1.04E+04	-2.48E+04	2.39E+04	-8.03E+03	2.08E+03
LU	Mainly disposal	-1.60E+02	1.75E+02	-3.04E+02	1.28E+02	-2.69E+02	1.30E+01
	Status quo	-3.20E+02	2.96E+02	-8.72E+02	3.68E+02	-6.04E+02	2.62E+01
	EU-target-SP	-5.31E+02	4.87E+02	-8.01E+02	3.37E+02	-4.11E+02	1.73E+01
	EU-target-MP	-1.49E+03	4.54E+02	-7.52E+02	3.16E+02	-4.11E+02	1.73E+01
RDw	Mainly disposal	-1.21E+00	2.23E+00	-2.38E+00	1.75E+00	-2.13E+00	2.47E-01
	Status quo	-2.35E+00	3.52E+00	-6.82E+00	5.02E+00	-4.64E+00	4.55E-01
	EU-target-SP	-3.97E+00	5.42E+00	-6.26E+00	4.60E+00	-3.17E+00	3.02E-01
	EU-target-MP	-4.12E+00	1.25E+00	-5.88E+00	4.32E+00	-3.17E+00	3.02E-01
RDm	Mainly disposal	-4.06E+04	9.12E+03	-1.24E+04	1.19E+04	-4.76E+03	1.34E+03
	Status quo	-8.96E+04	1.49E+04	-3.55E+04	3.41E+04	-1.10E+04	2.67E+03
	EU-target-SP	-1.53E+05	2.45E+04	-3.25E+04	3.13E+04	-7.47E+03	1.77E+03
	EU-target-MP	-5.62E+04	2.36E+04	-3.06E+04	2.94E+04	-7.47E+03	1.77E+03
RDf	Mainly disposal	-3.59E+04	5.90E+03	-3.98E+04	4.64E+03	-4.21E+04	5.05E+02
	Status quo	-7.01E+04	8.99E+03	-1.14E+05	1.33E+04	-9.37E+04	9.85E+02
	EU-target-SP	-1.16E+05	1.49E+04	-1.05E+05	1.22E+04	-6.38E+04	6.53E+02
	EU-target-MP	-2.72E+04	1.36E+04	-9.85E+04	1.15E+04	-6.38E+04	6.53E+02

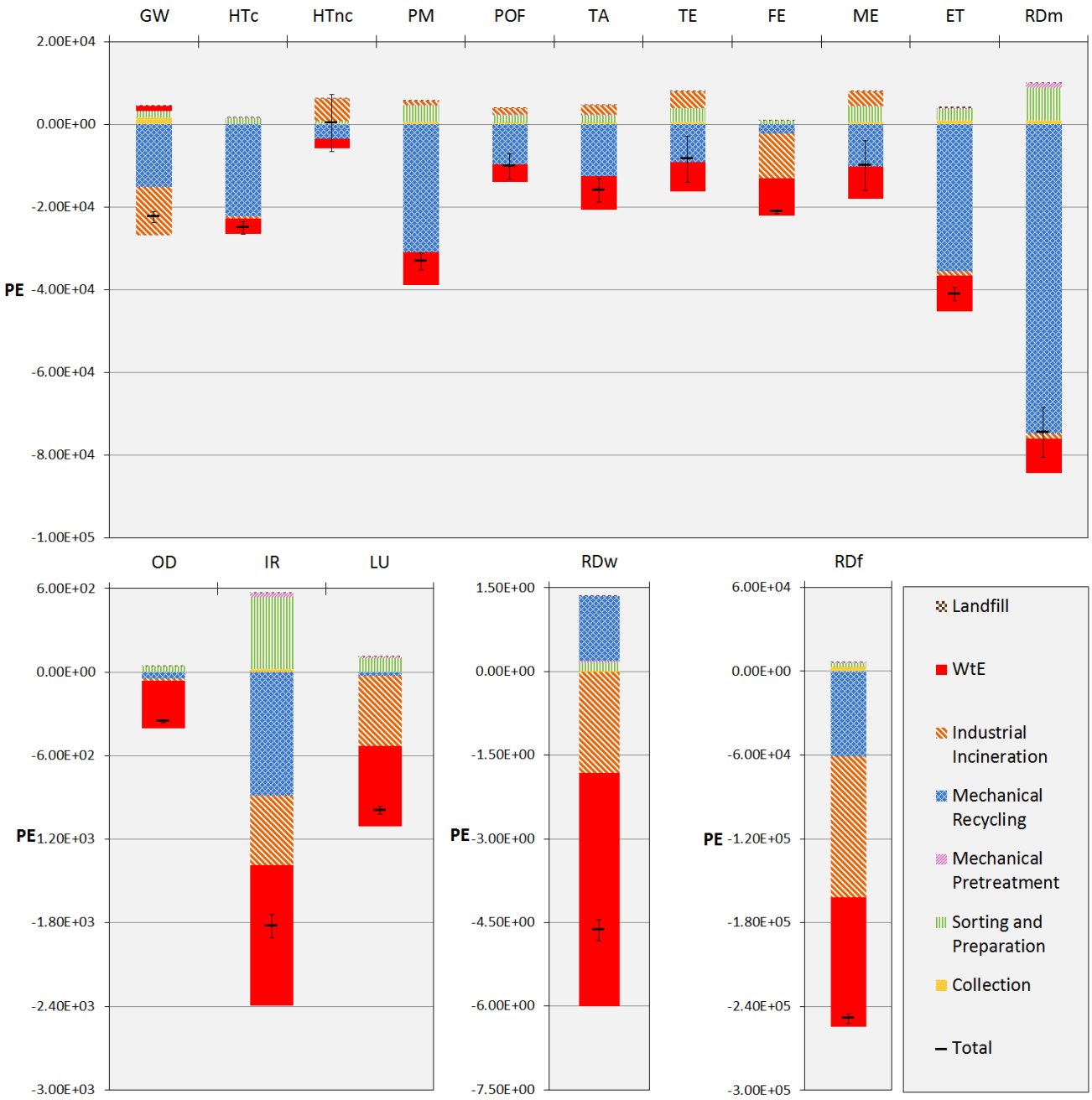

Figure S4: Net LCIA results of the status quo for all 16 impact categories, subdivided by treatment processes. The results with the burdens and benefits shown separately are presented in Figure S5.

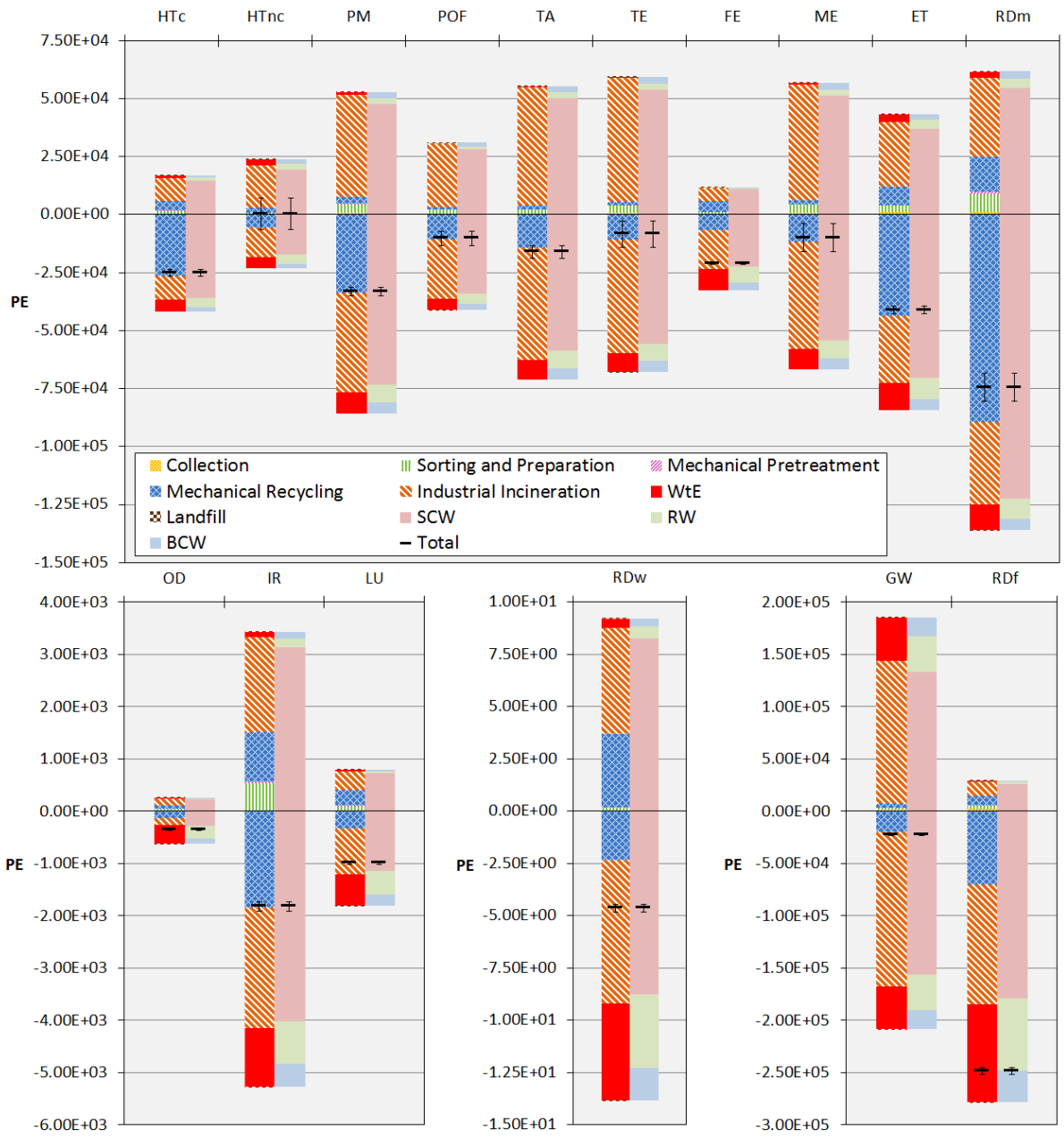

Figure S5: LCIA results for the status quo, subdivided respectively by treatment process (left bars in colors) and by collection route (right bars in greens), displaying both burdens and benefits for each of the treatment processes and collection routes. SCW: separately collected waste; RW: residual waste; BCW: bulky & commercial waste.

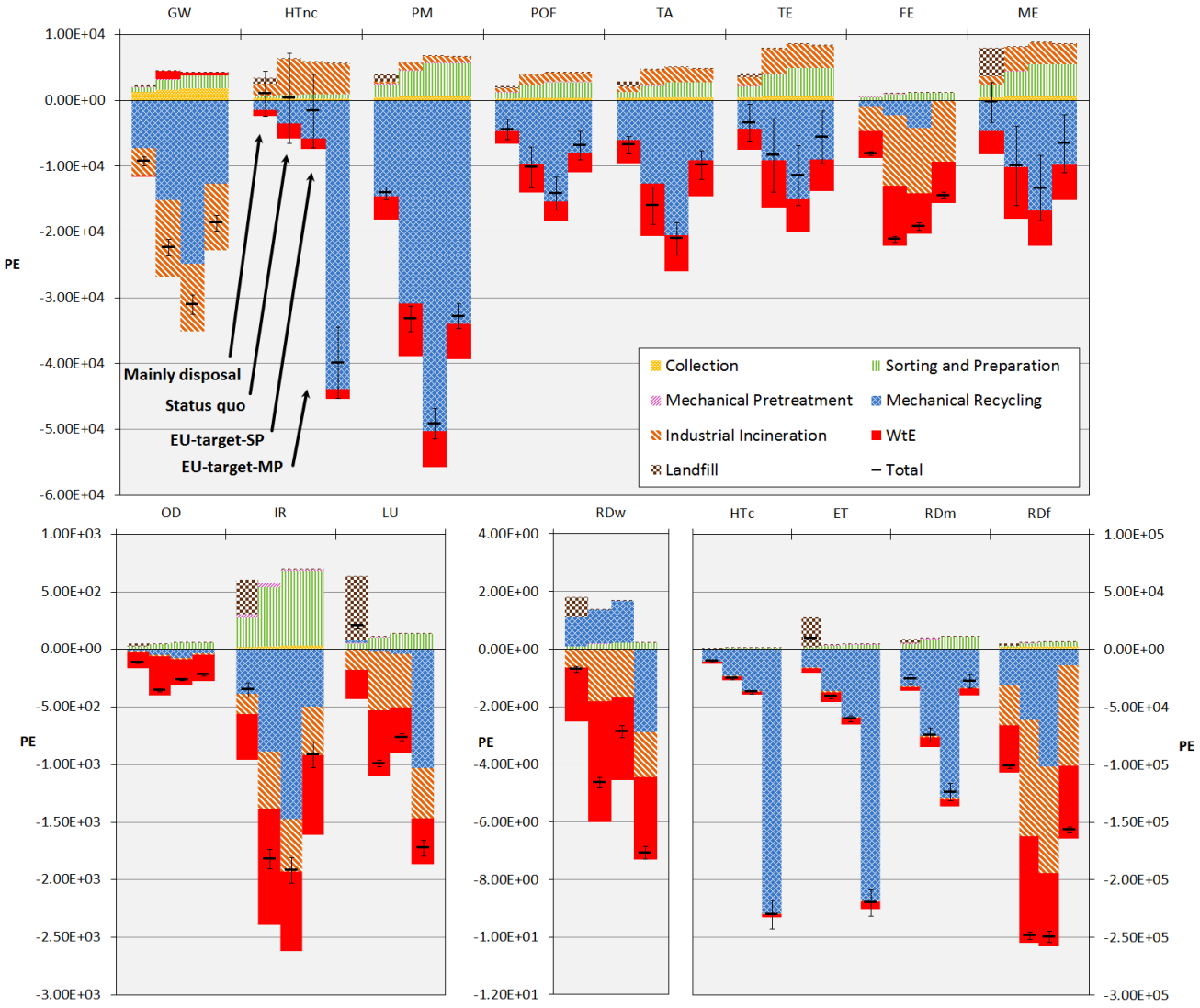

Figure S6: Net LCIA results for the status quo and the 3 alternative scenarios for all 16 impact categories.

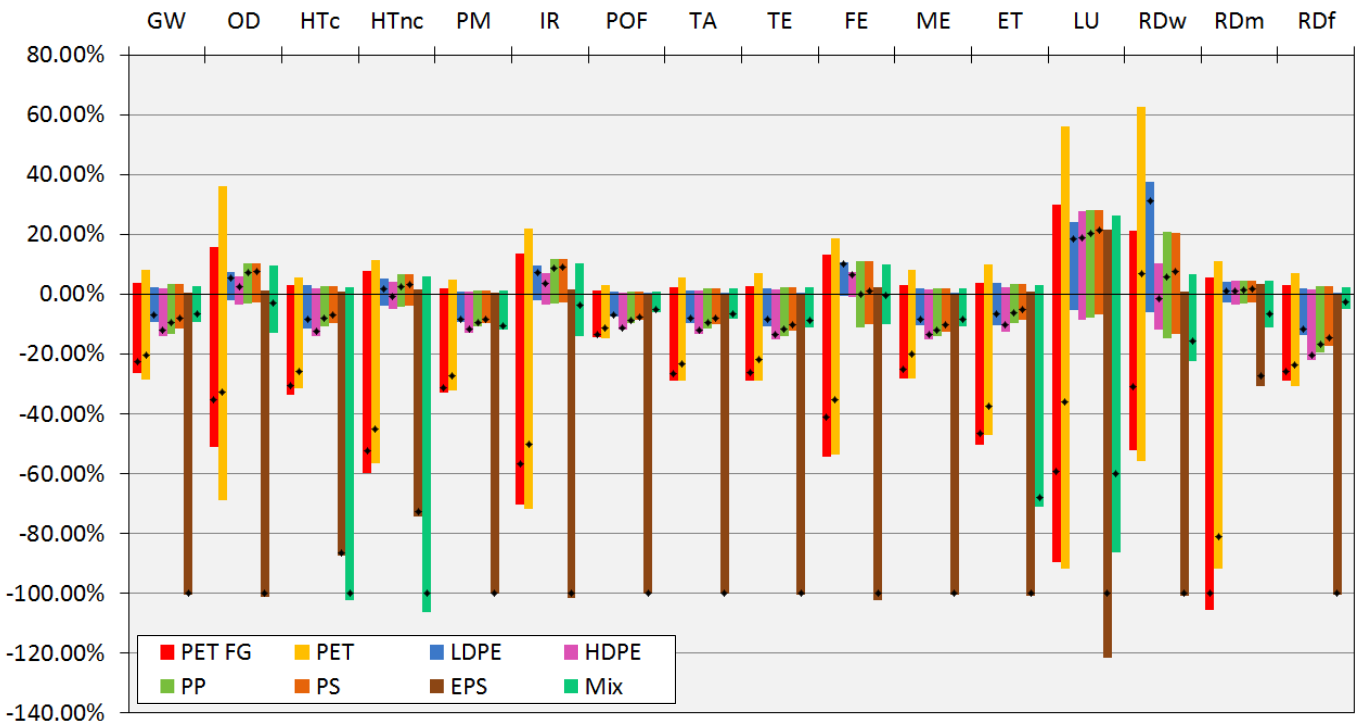

Figure S7: Relative LCIA results for the eight mechanical recycling processes, normalized to the highest net result per impact category. PET FG: PET food-grade.

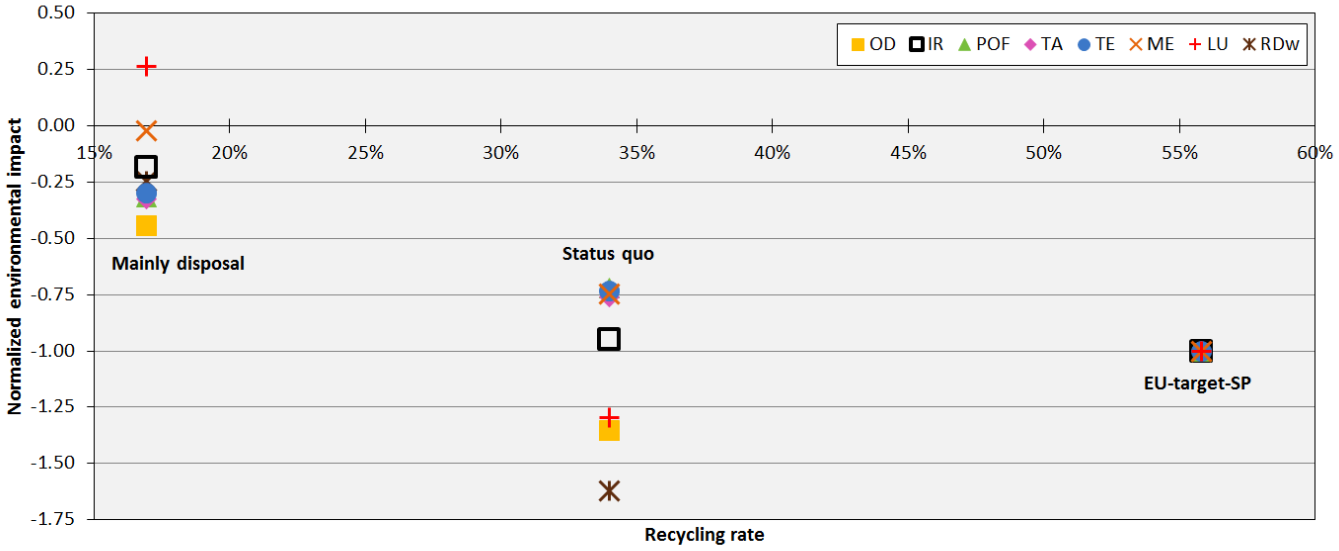

Figure S8: Normalized (EU-target-SP=-1) LCIA results for the eight impact categories not displayed in Figure 4b in the main article for the mainly disposal, status quo and EU-target-SP scenarios, in relation to the recycling rate.

3 Sensitivity and uncertainty analysis

Table S36: Sensitivity ratio (-) of the five most important parameters for each of the impact categories. TC: transfer coefficient.

Parameter	GWP	ODP	HTc	HTnc	PM	IR	POF	TA	TE	FE	ME	ET	LU	RDw	RDm	RDf
PET collection rate			0.41		0.41	0.57				0.17		0.52			1.00	
LDPE collection rate		-0.72		19.86		-0.54				-0.16	-0.55		-0.29	-0.79		
PET to food-grade sorting efficiency			0.33				0.37									0.10
LDPE sorting efficiency										-0.29			-0.30	-0.52		
Mechanical pretreatment TC to fluidized bed incineration		0.20											0.16			
PET food-grade recycling efficiency to granulate production			0.34	-17.04	0.38	0.52		0.31				0.45			0.83	
Grate incineration share with wet APC		0.66		-21.52												
Grate incineration wet energy recovery efficiency to electricity from coal CHP										0.22			0.23	0.46		0.18
Grate incineration wet energy recovery efficiency to heat from gas CHP		0.69														
Grate incineration wet carbon to air TC	-1.58															
Cement kiln CO_2 emission factor from calcination	-3.42															
Cement kiln SO_2 emission factor					-0.65			-1.60								
Cement kiln limestone consumption														-0.43		
Cement kiln PE recycling residues NOx emission factor									-0.79		-0.75					
Cement kiln PE recycling residues energy substitution factor	1.21						0.46	0.55	1.07		0.85					0.08
Cement kiln sorting residues CO_2 emission factor	-1.54															
Cement kiln sorting residues NOx emission factor							-1.13	-0.61	-2.87		-2.71					
Cement kiln sorting residues NH_3 emission factor									-0.65							
Cement kiln sorting residues Hg emission factor				29.69												
Cement kiln sorting residues energy substitution factor	4.38	0.26	0.28	-42.57	0.87	0.82	1.68	2.00	3.88	0.52	3.07	0.47	0.58	0.97	0.32	0.31
Substitution factor PET to food-grade recycling												0.30			0.58	
Substitution factor PET to fiber recycling						0.43						0.30			0.51	
Substitution factor LDPE recycling			0.35		0.30		0.39									0.11

Table S37: Uncertainty contribution analysis: share (%) of the overall uncertainty caused by the parameters needed to represent 90% of the uncertainty of the result for each of the impact categories. TC: transfer coefficient.

Parameter	GWP	ODP	HTc	HTnc	PM	IR	POF	TA	TE	FE	ME	ET	LU	RDw	RDm	RDf
LDPE collection rate	0.76	10.08				2.92							2.20	7.60		
PET collection rate	1.66		1.50		1.44	4.74				2.00		5.06			4.59	
LDPE sorting efficiency					1.17	2.26				11.95		0.72	7.21	10.68		2.20
PET sorting efficiency to food-grade			1.47		1.67	5.52				1.40		5.08			4.41	
Directly incinerated fraction of RW														1.26		
Directly incinerated fraction of BCW		6.13								1.57			2.67	9.52		1.51
Mechanical pretreatment TC to fluidized bed incineration		4.08										0.90	3.50	1.51		1.86
Mechanical pretreatment TC to industrial incineration										2.70			4.14			4.31
Sorting process diesel consumption						1.17										
Sorting process infrastructure consumption												0.91	4.50		43.80	
Transport from sorting to mechanical recycling												0.99				
PET food-grade recycling efficiency to fiber production		1.54				4.49				1.91		2.68		0.96		
PET food-grade recycling efficiency to flakes production			1.08		1.10	4.65				2.23		3.71			3.29	
PET food-grade recycling efficiency to granulate production			1.26		1.30	5.46				2.61		4.36		1.01	3.87	
PET fiber recycling efficiency	0.87	2.10	1.08		1.25	6.11				2.60		3.64	1.00	1.31		1.22
LDPE/LLDPE recycling electricity consumption		1.73				7.89				5.36			0.96			
LDPE/LLDPE recycling efficiency	6.44		9.33		6.82							11.01		1.05		16.32
LDPE/LLDPE recycling fraction of LDPE												0.70				
LDPE/LLDPE recycling waste production	1.11									5.12			2.99			8.13
PP recycling electricity consumption						1.18										
PP recycling efficiency	1.43		0.93									1.08				3.79
Grate incineration share with wet APC		25.77				1.82							0.94			
Grate incineration wet energy recovery efficiency to electricity from coal CHP	13.63									6.47			4.15	7.55		10.30
Grate incineration wet energy recovery efficiency to heat from gas CHP	0.81	27.78				1.76							1.54	0.87		
Grate incineration dry energy recovery efficiency to electricity from coal CHP	0.90															
Fluidized bed incineration energy recovery efficiency to heat from gas CHP		1.30														
Cement kiln CO_2 emission factor from calcination	13.21															

Parameter	GWP	ODP	HTc	HTnc	PM	IR	POF	TA	TE	FE	ME	ET	LU	RDw	RDm	RDf
Cement kiln PE recycling residues NOx emission factor							5.05	4.21	6.63							
Cement kiln SO_2 emission factor					43.12			30.20								
Cement kiln limestone consumption		3.51			6.38	8.65						5.87	7.86	40.57		
Cement kiln Fe-corrective consumption					1.45	1.20						2.32				
Cement kiln basic refractory consumption												7.74				
Cement kiln infrastructure consumption			1.51							22.09		7.07	41.73		31.47	
Cement kiln PE recycling residues Hg emission factor			4.11	6.35												
Cement kiln PE recycling residues electricity consumption						2.10				1.43						
Cement kiln PE recycling residues energy substitution factor	1.64															
Cement kiln sorting residues CO_2 emission factor	2.66															
Cement kiln sorting residues Hg emission factor			46.50	71.83												
Cement kiln sorting residues Pb emission factor			1.30	17.60												
Cement kiln sorting residues dust emission factor					5.70											
Cement kiln sorting residues CO emission factor	7.55						22.91									
Cement kiln sorting residues NOx emission factor					3.42		65.69	54.77	86.27		90.81					
Cement kiln sorting residues NH_3 emission factor					1.51			4.71								
Cement kiln sorting residues electricity consumption	1.57	6.07				27.73				18.84		1.53	3.38	2.54		1.26
Cement kiln sorting residues energy substitution factor	21.61					1.19				2.12			1.56	2.06		1.82
Substitution factor LDPE recycling	13.37		20.51		14.30							24.45		2.24		33.76
Substitution factor HDPE recycling																1.60
Substitution factor PP recycling	0.86											0.72				2.28

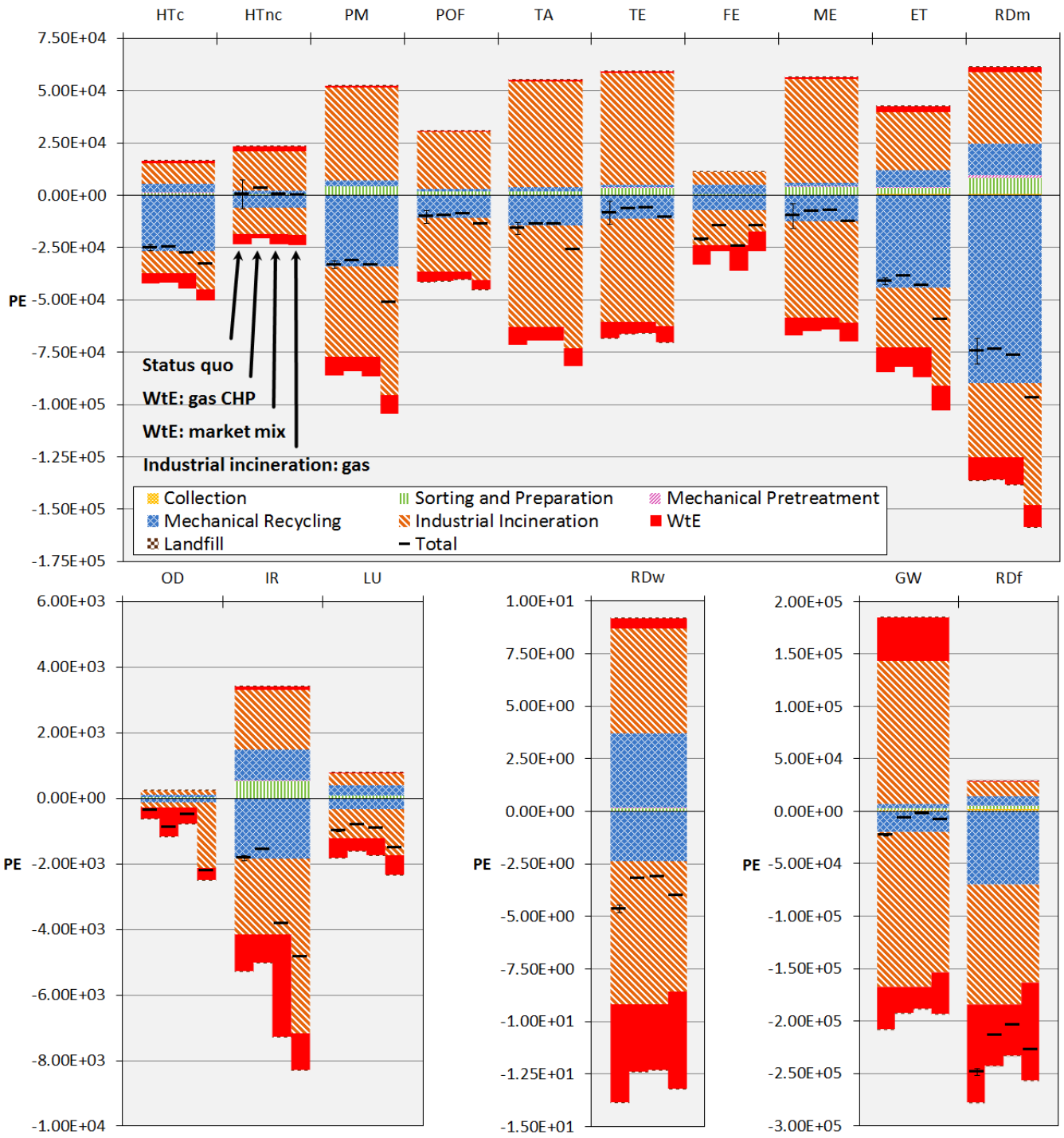

Figure S9: Scenario analysis for the waste-to-energy (change of the substituted electricity technology from coal-fired to gas-fired CHP and the Austrian market mix) and industrial incineration (change of the substituted fuel from hard coal to natural gas) processes. The values shown are presented in Table S38.

Table S38: Results (in PE) for the scenario analysis for the industrial incineration (change of the substituted fuel from hard coal to natural gas) and WtE (change of the substituted electricity technology from coal-fired to gas-fired CHP and the Austrian market mix) processes. The results correspond to those in Figure S9.

		Industrial incineration		WtE	
		Benefits	Burdens	Benefits	Burdens
GW	Status quo	-1.49E+05	1.37E+05	-4.06E+04	4.18E+04
	WtE gas CHP			-2.48E+04	4.18E+04
	WtE market mix			-2.08E+04	4.18E+04
	Industrial incineration natural gas	-1.34E+05	1.37E+05		
OD	Status quo	-1.43E+02	1.29E+02	-3.52E+02	1.14E+01
	WtE gas CHP			-8.72E+02	1.14E+01
	WtE market mix			-4.89E+02	1.14E+01
	Industrial incineration natural gas	-1.99E+03	1.29E+02		
HTc	Status quo	-1.04E+04	9.81E+03	-5.08E+03	1.29E+03
	WtE gas CHP			-4.77E+03	1.29E+03
	WtE market mix			-7.57E+03	1.29E+03
	Industrial incineration natural gas	-1.83E+04	9.81E+03		
HTnc	Status quo	-1.29E+04	1.84E+04	-4.77E+03	2.44E+03
	WtE gas CHP			-1.94E+03	2.44E+03
	WtE market mix			-4.83E+03	2.44E+03
	Industrial incineration natural gas	-1.32E+04	1.84E+04		
PM	Status quo	-4.31E+04	4.43E+04	-8.88E+03	9.02E+02
	WtE gas CHP			-7.07E+03	9.02E+02
	WtE market mix			-9.25E+03	9.02E+02
	Industrial incineration natural gas	-6.13E+04	4.43E+04		
IR	Status quo	-2.32E+03	1.82E+03	-1.11E+03	1.02E+02
	WtE gas CHP			-8.47E+02	1.02E+02
	WtE market mix			-3.12E+03	1.02E+02
	Industrial incineration natural gas	-5.33E+03	1.82E+03		
POF	Status quo	-2.57E+04	2.73E+04	-4.75E+03	3.60E+02
	WtE gas CHP			-4.38E+03	3.60E+02
	WtE market mix			-3.63E+03	3.60E+02
	Industrial incineration natural gas	-2.95E+04	2.73E+04		
TA	Status quo	-4.84E+04	5.08E+04	-8.61E+03	5.46E+02
	WtE gas CHP			-6.39E+03	5.46E+02

	WtE market mix			-6.58E+03	5.46E+02
	Industrial incineration natural gas	-5.83E+04	5.08E+04		
TE	Status quo	-4.92E+04	5.32E+04	-7.76E+03	6.69E+02
	WtE gas CHP			-5.70E+03	6.69E+02
	WtE market mix			-5.50E+03	6.69E+02
	Industrial incineration natural gas	-5.13E+04	5.32E+04		
FE	Status quo	-1.67E+04	5.88E+03	-9.26E+03	2.03E+02
	WtE gas CHP			-2.65E+03	2.03E+02
	WtE market mix			-1.24E+04	2.03E+02
	Industrial incineration natural gas	-1.03E+04	5.88E+03		
ME	Status quo	-4.61E+04	4.97E+04	-8.61E+03	7.94E+02
	WtE gas CHP			-6.36E+03	7.94E+02
	WtE market mix			-5.90E+03	7.94E+02
	Industrial incineration natural gas	-4.88E+04	4.97E+04		
ET	Status quo	-2.88E+04	2.78E+04	-1.18E+04	3.12E+03
	WtE gas CHP			-9.17E+03	3.12E+03
	WtE market mix			-1.39E+04	3.12E+03
	Industrial incineration natural gas	-4.71E+04	2.78E+04		
LU	Status quo	-8.72E+02	3.68E+02	-6.04E+02	2.62E+01
	WtE gas CHP			-4.10E+02	2.62E+01
	WtE market mix			-5.32E+02	2.62E+01
	Industrial incineration natural gas	-1.40E+03	3.68E+02		
RDw	Status quo	-6.82E+00	5.02E+00	-4.64E+00	4.55E-01
	WtE gas CHP			-3.20E+00	4.55E-01
	WtE market mix			-3.11E+00	4.55E-01
	Industrial incineration natural gas	-6.21E+00	5.02E+00		
RDm	Status quo	-3.55E+04	3.41E+04	-1.10E+04	2.67E+03
	WtE gas CHP			-1.03E+04	2.67E+03
	WtE market mix			-1.30E+04	2.67E+03
	Industrial incineration natural gas	-5.79E+04	3.41E+04		
RDf	Status quo	-1.14E+05	1.33E+04	-9.37E+04	9.85E+02
	WtE gas CHP			-5.87E+04	9.85E+02
	WtE market mix			-4.88E+04	9.85E+02
	Industrial incineration natural gas	-9.32E+04	1.33E+04		

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