CHEMICAL RECYCLING OF PLASTICS

Technologies, trends and policy implications

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SUMMARY

In recent years, the challenge of plastic waste generation has become a prime concern in the global political arena. At the EU level, a dedicated strategy on plastics was adopted that led to the Single-Use Plastics Directive. In spite of this, plastic waste management data show that achieving a circular economy for plastics in the EU is a long way off. Available studies indicate that plastic waste generation may remain at high levels in the future or even increase in the absence of ambitious circularity policies. The report looks at the challenges associated with plastic waste generation and discusses the potential of using chemical recycling technologies as part of an ecosystem of solutions for increasing the circularity of plastics. It is based on evidence collected through desk-research and inputs provided during a series of stakeholder meetings.

Given the myriad applications of plastics, a mix of recycling solutions, combined with efforts aimed at increasing reuse and waste prevention will be needed. This requires a policy environment that while enabling all recycling options would at the same time provide a level playing field between mechanical and chemical recycling. To achieve such a level playing field, clarification would be needed on how chemical recycling technologies could contribute to achieving recycled content targets. As these technologies scale up, the question about whether there is a need to provide clarity about their position in the waste hierarchy and in the existing recycling definition will also need to be addressed.

There are several data uncertainties about plastic waste feedstocks and composition as well as the emissions and losses in the chemical recycling processes. The publication of methodology guidelines for LCAs comparing different treatment options for waste plastics can support a more informed debate about plastics' circularity. More integrated assessments considering the full spectrum of plastic waste streams and how they can be treated in the most environmentally friendly way can also contribute to this debate.

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1. INTRODUCTION

Plastics have made a wide impact on global economies since their large-scale industrial production began in the 1940s (OECD, 2022a; Al-Salem et al., 2009). Owing to their properties and low-cost, plastics are used in nearly every economic sector and have supported advancements in sectors such as food storage and distribution, transportation, electronics, healthcare and buildings (EEA, 2020; Yates et al., 2021). The rate of growth of plastic production over the decades has been exponential and can only be compared with that of steel and cement (Geyer et al., 2017). It has been estimated that plastic production has seen a 230-fold increase since the 1950s (OECD, 2022b) and more than 8 billion tonnes of plastic have been produced around the globe over the years (Geyer et al., 2017).

The persistent growth in plastic production has been associated with a broad spectrum of environmental, climate and human health impacts occurring throughout the entire life cycle of plastics. An important share of these impacts is attributed to the way plastics are managed at the end-of-life (EOL) stage. According to OECD (2022b), out of the 353 million tonnes of plastic waste produced globally in 2019, only about 9 % was properly recycled. The rest ended up in incinerators (19 %), controlled landfills (50 %) or in unofficial routes often leading to plastic leakage to the environment. For the EU, Plastics Europe estimates that out of the 30 million tonnes of plastic waste produced annually, 35 % is effectively sent to recycling with the rest being sent to incineration with energy recovery (42 %) or to landfilling (23 %)[1]. The practice of incinerating and landfilling such high amounts of plastic each year entails significant environmental externalities including the release of GHG emissions into the atmosphere[2] and leakage of contaminants into terrestrial and marine environments (EEA, 2020; Lange, 2021).

Waste management policies targeting certain plastic waste streams (i.e. packaging) have been in place since the 1990s. In recent years, however, the issue of the plastic waste crisis has received increased attention and has become a priority in the EU political agenda, driven mainly by the increased attention of the media, NGOs and the civil society on the topic (Filho et al., 2019; Nielsen et al., 2019). Plastics emerged as one of the priority sectors addressed by the first EU Circular Economy plan leading to the first EU dedicated strategy on plastics (European Commission, 2018) and a new Directive focused on addressing the impacts of certain single-

[1] As discussed in section 2.2 other studies have estimated that the figure for EU plastic waste generation is higher, while the share of plastic that is effectively recycled lower.

[2] Considering the emissions savings due to the use of plastic for energy recovery, Material Economics (2021) calculates the EU annual CO_2 emissions released into the atmosphere due to the incinerations of plastics at 38 million tonnes.

use plastic items[3]. This action on the EU policy front was matched by pledges by many multinational companies to use more recycled plastics (Kahlert & Bening, 2022). Various national governments in the EU and beyond[4] have put forward commitments on plastics (see OECD, 2022c), while recently there was an agreement by the UN Member States for new legally binding instrument to address plastic pollution (UN, 2022).

Still, the challenges linked to managing the large volumes of plastic waste produced each year persist and could even intensify across the EU in the future if ambitious circularity policies are not adopted[5]. Even though recycling rates for certain plastic streams have seen an increase in recent years, significant improvements in current collection, sorting and recycling systems and technologies are required. As the need for new solutions increases, there has been growing interest and research in leveraging chemical recycling (CR) technologies to improve circularity of plastics. Focusing on producing basic chemicals or other feedstocks from plastic waste, these technologies are considered as complementary options to existing plastic waste management options. However, there have also been concerns about their implementation and potential for delivering substantial environmental benefits (Lee et al., 2021; Manžuch et al., 2021).

This report presents challenges associated with plastic waste generation across the EU and discusses the potential of using chemical recycling technologies as part of an ecosystem of solutions for increasing circularity of plastics. It first summarises current trends linked to plastic use and associated environmental challenges (section 2) and then discusses the key available chemical recycling technologies for managing different plastic waste streams (section 3). The report then presents estimates about future plastic use, waste generation and treatment by different processes including mechanical and chemical recycling (section 4). It then dives into various aspects and policies related to the development of functioning plastics recycling markets including sorting and collection of plastics, application of life cycle assessment (LCA) methodologies, rules for allocation recycled content in plastics and use of ecodesign approaches (section 5). The report concludes with key messages and recommendations (section 6).

The report synthesises 12 months of work by the CEPS research team consisting of desk-based research and the organisation of a series of stakeholder meetings to collect experts' views. The meetings brought together representatives from the full spectrum of stakeholders involved in the discussions on chemical recycling including chemical recycling companies, waste management companies, academics, NGOs and end-users of recycled polymers (see Annex).

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[3] Directive (EU) 2019/904 on the reduction of the impact of certain plastic products on the environment also referred as Single-Use Plastics Directive.

[4] For example, Canada, Chile, Colombia, Korea, see OECD (2022c).

[5] See section 4.2.

2. PLASTICS: CURRENT TRENDS, PATTERNS, AND ENVIRONMENTAL CHALLENGES

2.1 PLASTICS USE, PRODUCTION AND DEMAND

In the public debate, plastic is often used as a generic term for its integral component, a chemical linkage of a series of building blocks, called monomers, to form larger molecules, called polymers (Chamas et al., 2020). These polymers are mostly derived from fossil feedstocks like crude oil, natural gas, or coal. In Europe, the most prevalent fossil-based polymer types are Polyethylene (PE), Polypropylene (PP), Polyvinylchloride (PVC), Polyethylene terephthalate (PET), Polyurethane (PUR), Polystyrene (PS) and Polyamides (PA) (Plastics Europe, 2021). However, it is also possible to produce virgin plastic from renewable, bio-based, feedstocks, like corn, casava, sugar beet, and sugar cane. Such bio-based polymers[6] account only for a marginal share of global plastic production (Hamilton et al., 2019).

Plastics have multiple uses across a variety of sectors. Polyethylene can be divided into low-density Polyethylene (LDPE), medium-density Polyethylene (MDPE) and high-density Polyethylene (HDPE). Reusable bags, agricultural and food packaging film are typical applications of LDPE, while MDPE and HDPE are essential for producing toys, milk bottles, shampoo bottles, houseware, industrial packaging and pipes for water and gas distribution. PP has similar applications in food packaging, for snack wrappers or microwave containers but also automotive parts, such as bumpers, pipes and garden furniture. PET is the common polymer used to produce bottles for drinks or cleaners, and is also used to produce polyester textiles. PVC is mostly found in construction products, such as window frames, floor and wall covering, pipes or electric cables. Building insulation is likewise a prominent application of PUR besides foams, pillows, matrasses, and footwear. PS is frequently used to produce dairy, meat and fish packaging, fridge interiors and is also used in building insulations, or electrical and electronic equipment. Lastly, the automotive and the textile industry are large users of PA for their manufacturing components (Plastics Europe, 2021; Ncube et al., 2021; Andrady & Neal, 2009; Kondo, et al., 2022). Table 1 below summarises the key types of polymer plastics and examples of notable uses.

[6] Polymers like polylactic acid (PLA), plant-derived PET, or Polyhydroxyalkanoate (PHA) are often mixtures of polymers from renewable feedstock and petrochemical derived plastics and fibers (Hamilton et al., 2019).

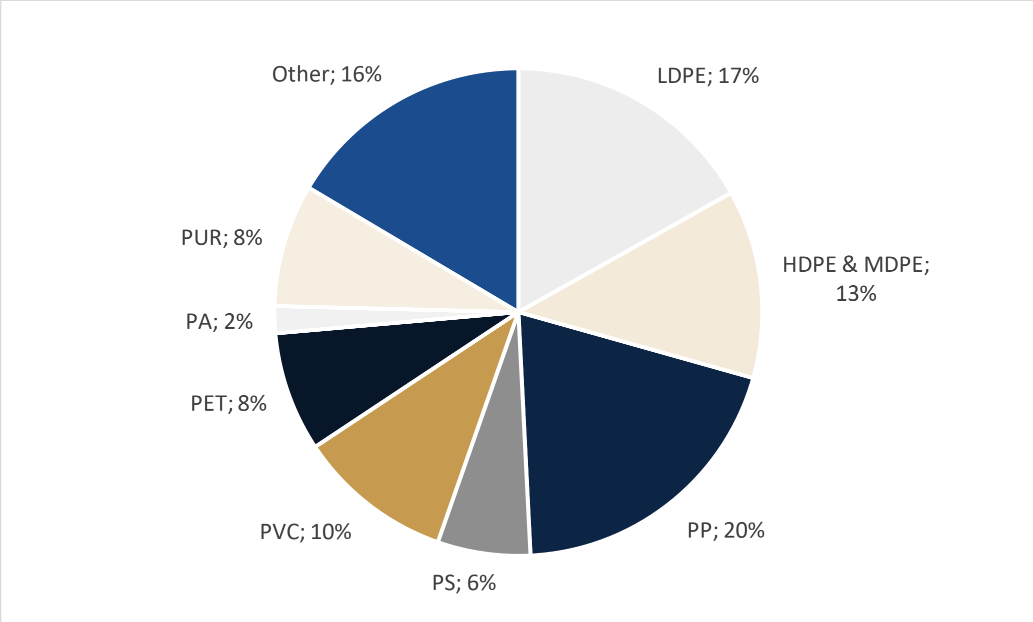
Table 1. Polymer types and usage

Polymer	Name	Use
LDPE MDPE HDPE	Low-Density Polyethylene Medium-Density Polyethylene High-Density Polyethylene (Polyolefins)	LDPE: plastic bags, plastic cutlery, packaging film MDPE, HDPE: food containers, liners, shampoo bottles, pipes, houseware
PP	Polypropylene (Polyolefin)	Food containers, plastic bottles, plastic bags (more heat resistant than PE, very durable)
PA	Polyamides, Nylon	Transportation manufacturing, textiles, automotive industry
PET	Polyethylene Terephthalate (Polyester)	Plastic bottles for drinks and cleaners, fabrics (polyester)
PS	Polystyrene	Dairy, meat and fish packaging, construction/insulation, electrical equipment
PUR	Polyurethane	Different chemical structures - different uses, e.g. foams, coatings, adhesives (e.g. kitchen sponges), mattresses, footwear
PVC	Polyvinylchloride	Building, construction, wires and cable, medical applications (resistant to chemicals and weathering, does not conduct electricity, impermeable to germs)

In terms of plastic production, from the 390.7 Mt of plastics produced globally in 2021, the EU-27+3[7] accounted for a share of around 15 %, equal to 57.2 Mt. The different PE types accounted for 24 % of the total production in the EU-27+3 countries, PP for 16,6 %, PVC for 11.4 %, PET for 5.3 %, PUR for 5.5 %, and PS for 6.1 %. Simultaneously, bio-based plastics represented a share of 2.3 %, and recycled plastics – 10.1 % of the total production (Plastics Europe, 2022).

[7] Including Norway, Switzerland, and the UK.

Figure 1. Plastics demand by polymer type in the EU27+3 (2021)



Source: Plastics Europe (2022). Breakdown of plastics demand in the EU plus the UK, Switzerland, and Norway into the different polymer types in the year 2021.

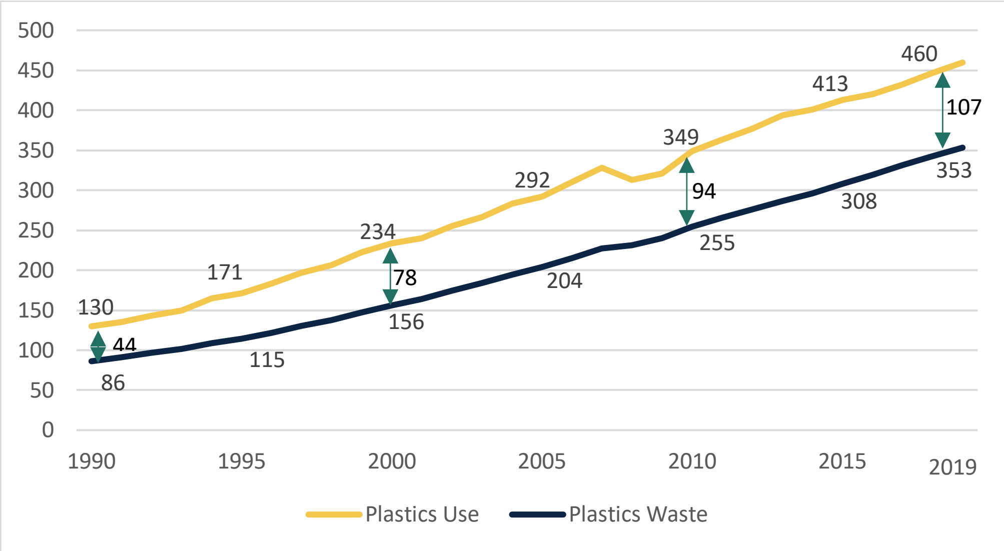
As Figure 1 indicates, the breakdown of the demand for different polymer types in the EU-27+3 countries displays a similar pattern. The major polymers represent a share of around 82 % of the total plastics demand of 50.3 Mt in 2021. Among them, polyethylene, including LDPE, MDPE and HDPE, accounts for around 30 %. Additionally, PP represents a share of 20 %, PVC a share of 10 %, both PET and PUR a share of 8 % each, and PS a share 6 % of the European plastics demand in the given year (Plastics Europe 2022).

The packaging, and the building & construction sectors are the major drivers of European plastics demand, representing 60.4 % (39.1 % and 21.3 % respectively) of the total plastics demand in 2021. Other sectors, such as the automotive sector (8.6 %) or electrical and electronic equipment (6.5 %), play a significantly smaller role (Plastics Europe 2022).

2.2 CURRENT ROUTES FOR END-OF-LIFE PLASTICS

Depending on a variety of factors including product design, consumption patterns, type and sector used, plastic products reach at some point the EoL stage (Geyer, 2020). From this point onwards the different routes for EoL plastics are reuse, recycling, incineration and landfilling. The global in-use stock of plastics and the global waste streams, depicted in Figure 2 indicate a constantly increasing trend.

Figure 2. Global plastics use and waste in Mt (1990-2019)



Source: OECD (2023). Global trends of the in-use stock of plastics and end-of life plastics from 1990 to 2019. Data points are indicated in five-year intervals and in the year 2019. The differences between global plastics use and global plastics waste in million tonnes are highlighted in ten-year intervals and in the year 2019.

According to Plastics Europe (2022), about 30 million tonnes[8] of post-consumer plastic waste were generated across the EU in 2020. Waste disposal in landfills – the least preferable route for end-of-life plastics according to the EU waste hierarchy – remains a common waste management method with about 23 % of plastic waste ending up in landfills. Then approximately 42 % of plastics is incinerated for energy recovery which represents the largest route for plastic waste disposal (Hamilton et al., 2019). The following option in the EU waste hierarchy which as described later has several environmental benefits over incineration and landfilling is recycling. Plastic Europe (2022) estimates that around 35 % of plastic waste was sent to recycling in the EU in 2020, while at the Member States level this figure varies from 21 % to 45 %.

While figures of similar scale about plastic waste generation in the EU have also been reported by EU publications (see European Commission, 2018), other studies have estimated the amount of plastic waste generated across the EU at higher levels. Specifically, based on a material flow analysis, Material Economics (2022) estimates that about 15 million tonnes of plastic waste are not accounted in current statistics[9] and thus the total amount of plastic waste generated could be as high as 45 million tonnes[10]. Out of the 43.6 million tonnes that are treated in the EU, 13 % are effectively recycled[11] with the rest being incinerated (55 %) or landfilled (32 %) (see Figure 3).

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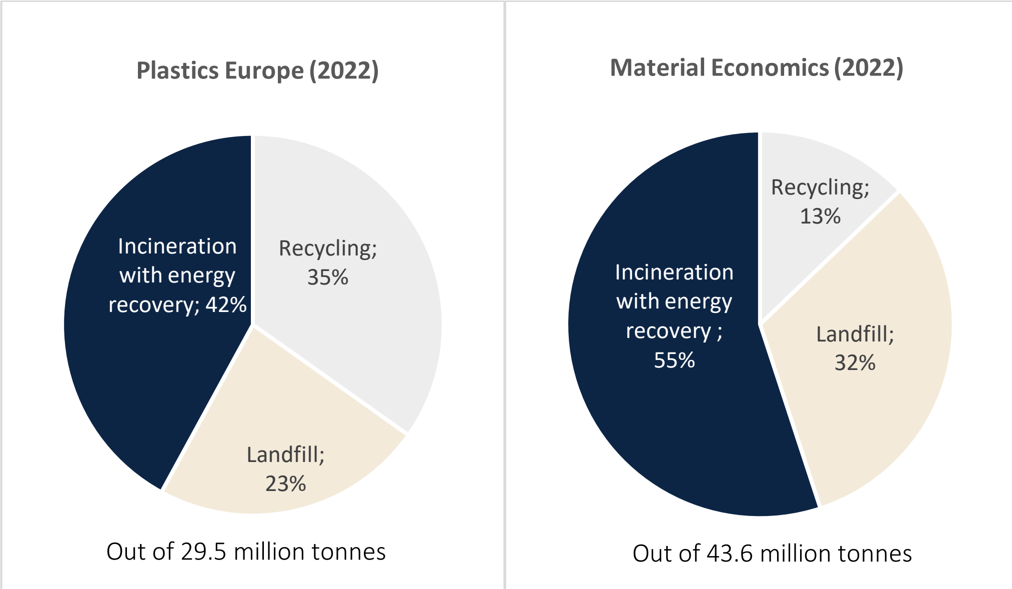
[8] 16% of these 30 million tonnes were exported to countries outside the EU.

[9] SYSTEMIQ (2022) estimates a similar gap in the range of between 8-15 Mt.

[10] Including approximately 1.6 million tons that were exported to countries outside the EU.

[11] This figure includes plastics recycled overseas. Notably, according to the study there are some small quantities of plastic waste that end up in the environment and oceans, though it is difficult to provide precise estimates (Material Economics, 2022).

Figure 3. Share of plastic waste treatment methods in the EU27+3



Source: Plastics Europe (2021), Material Economics (2022). Note: The diagram on the left-hand side illustrates the different routes of treatment of plastic waste for the EU plus the UK, Switzerland, and Norway in 2020 reported by Plastics Europe (2021), while the right-hand side diagram reveals the data published by Material Economics (2022).

Notably, reuse of plastics is currently limited to certain applications, for example plastic containers and crates (Al-Salem et al., 2009). Data on reuse of plastics is generally scarce as it is difficult to develop an account of plastics streams included in products that are traded in second-hand markets or exchanges between consumers[12] (Hsu et al., 2021).

Looking into the plastic demand and waste figures, one can notice some further divergences. One reason for this trend can be attributed to the lifespan of different applications of plastics which varies considerably according to the type of application and the sector used. Given the different product lifetimes of the various applications of plastics in different sectors, there are large differences across plastics demand and plastics waste produced across sectors of different applications that are highlighted in Figure 4. For example, while packaging products[13][14] have a mean product lifetime of only 0.5 years, plastic materials in the building & construction sector can have a lifespan of 35 years[15] (Geyer et al., 2017). The usual explanation for the difference between plastics demand and waste streams in different sectors (see

[12] Hsu et al. (2021) estimate that around 183 kt of plastic waste were reused across the EU in 2016, though this figure is a conservative estimate as it does not include informal and second-hand exchanges.

[13] Considering the plastics waste volumes by sector in the EU27+3 countries in 2018, the packaging sector accounts for the largest share of waste volumes with 17,751 kt, while waste volumes in other sectors lie below 2,000 kt (Plastics Europe 2019; Deloitte Belgium, 2021).

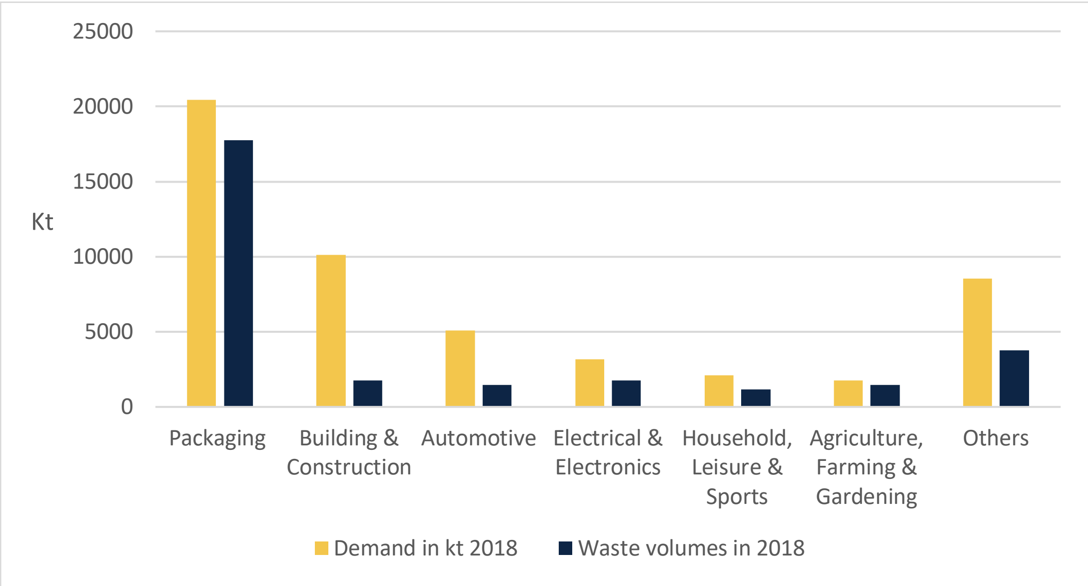
[14] Notably, according to EEA (2022), packaging waste is the only plastic waste stream for which data has been extensively documented since there is a dedicated piece of EU legislation in place targeting this stream (i.e. the Packaging and Packaging Waste Directive).

[15] SYSTEMIQ (2022) reports that plastic materials in the buildings and construction sector could last up to 100 years.

Figure 4) is the long lifespan of some products and the development of in-use stocks of plastics in some sectors (e.g. plastic in cars, buildings, and household goods).

However, according to SYSTEMIQ (2022) there are additional possible explanations to this trend[16]. Firstly, the amount of plastics in mixed waste streams could be significantly larger compared to what is usually accounted for as also discussed above. Secondly, the average product lifetime of plastics products may be underestimated. Thirdly, there are higher levels of exports of secondary goods than reported. Effectively, 8 to 15 Mt of plastics are estimated to remain unaccounted for in the reported waste statistics (SYSTEMIQ, 2022).

Figure 4. Plastics demand and waste in the EU27+3 countries (2018)



Source: Plastics Europe (2019) and Deloitte Belgium (2021). Demand and waste of plastic products across selected sectors in the EU 27 countries plus the UK, Switzerland, and Norway.

2.3 Environmental impacts throughout the plastic lifecycle

With the continuous surge of global plastics use and waste during the last decades, several environmental challenges have emerged along their different lifetime stages of plastics, from the extraction of feedstocks to the production of polymers and the handling of end-of life plastic products.

The extraction of fossil feedstocks like coal, crude oil, and natural gas, represents the first major determinant of environmental impacts along the plastic products value chain. The impact generated by the extraction process varies according to the feedstock type and technology used. Further, in gasification and refining processes, these feedstocks are converted to obtain petrochemicals. The two most common petrochemicals are naphtha, derived from crude oil,

[16] According to SYSTEMIQ's (2022) study, calculating the in use-stock growth based on the gap between plastics demand and waste streams can yield inconsistent results.

and ethane, derived from natural gas. In a second stage, these petrochemicals are processed into monomers in a steam cracker (Hamilton et al., 2019).

Fossil fuel combustion for the supply of power and heat in these industrial processes, in particular the processing stage, is the first and largest source of emissions[17]. Secondly, produced monomers (e.g., olefins) are then bound to produce the different plastic polymer types. The emissions occurring during the production process vary depending on feedstock and the polymer type produced. Due to the high energy intensity required, the production of virgin PS and PET polymers generates higher emissions compared to low, medium, and high-density polyethylene types (Hamilton et al., 2019; Ren et al., 2006; Charles & Kimman., 2023).

Further impacts emerge at the end of life of plastics, which largely depend on the way these products are collected and managed. When end-of life plastics are landfilled, GHG emissions occur during the transporting stage and the handling of plastics prior to reaching the landfill point. Various environmental impacts then emerge during the landfilling of plastics. These include leaching of toxic substances into the soil and waterways[18], as well as emissions from the degradation of plastics (Hamilton et al., 2019). Notably, the rate of surface degradation of end-of life plastics is contingent on the type of polymer[19] (Chamas et al., 2020).

Incineration is an alternative treatment option to landfilling that also entails significant environmental and human health impacts. Specifically, it is estimated that it generates approximately 38 million tons of net CO2 emissions in the EU annually after subtracting the emissions that are offset by replacing other fossil fuels[20] (Material Economics, 2022). Even though the incineration of plastics can replace power generation from other fossil feedstock fuels, the carbon benefits of this process decrease as the share of electricity from renewable energy sources increases (Hamilton et al., 2019).

Besides the impacts occurring at different stages of the plastics' lifetime, the leakage of plastics waste into the environment has been a key cause of concern in the EU and globally. Plastic waste that is not managed properly could reach inland waterways, wastewater systems or be transported by wind and tides to end up in the ocean (Jambeck et al., 2015). The polymers that represent the largest share of mismanaged plastics are those with the highest demand, namely PP, LDPE and PE-MD-HD, representing a share of 45 % of all lost macroplastics (Ryberg et al.,

[17] The largest part of these emissions stems from the steam cracking process, in which feedstocks, for instance ethane and naphtha, are heated in cracker furnaces to produce olefins (Hamilton et al., 2019; Charles & Kimman., 2023; Cabernard et al., 2022).

[18] Landfilling PVC and other polymer types that are contained in waste of electric and electronic equipment (WEEE), for example, entails a high risk of contaminating soil and waterways through the release of leachable components through rainwater when being landfilled (Lange, 2021).

[19] During landfilling and composting, the surface of LDPE degrades up to 11 times higher than PE-HD-MD, while for PET, PVC, and PS no significant degradation occurs (Chamas et al., 2020).

[20] Plastics are typically incinerated in a co-process, burning mixed waste together with coal or biomass in municipal waste-to-energy plants. Therefore, the emissions generated from the incineration process depend on the energy feedstock used as well as the composition of the mixed waste (Hamilton et al., 2019).

2019). Microplastics[21] are another important contributor to the leakage of plastics to the environment. According to Ryberg et al. (2019), they represent around one third of the 9.2 Mt of plastics that are annually lost in the environment[22]. Those are submicron pieces of plastic can leak into the environment for example during the usage or abrasion of plastic products (Lange, 2021).

2.4 Mechanical recycling of plastics

To alleviate the environmental impact and resource inefficiency of plastics there is a consensus among scholars that augmenting recycling should be targeted above energy recovery methods and landfilling, besides reducing plastics production, promoting reuse, and raising consumer awareness on waste handling. The standard route for plastic waste recycling, that was already commercialised globally in the 1970s, is mechanical recycling (Al-Salem et al. 2009). Within the mechanical recycling process, five major steps need to be followed to convert plastic waste into secondary raw materials. First, the waste is separated and sorted according to its shape, density, size, colour or chemical composition. Second, if the plastic is not processed where it is sorted, it runs through a bailing process to facilitate its transport. Third, all contaminants of the plastic waste must be removed within a decontamination process (such as washing). Fourth, grinding reduces the size of the plastic products to flakes. Lastly, the flakes are reprocessed into a granulate within a compounding and pelletising process to facilitate its use for converters to produce new products (Ragaert et al. 2017).

The literature concludes that mechanical recycling techniques are usually the preferred recycling routes for end-of-life plastics from an environmental point of view (see section 4.2). Nevertheless, these processes have their own limitations. A key constraint is that mechanical recycling can only cover certain (homogenous) plastic streams and requires extensive pre-sorting and cleaning. Thus, it is still unsuitable for a considerable share of collected end-of-life plastics (Padhan & Sreeram, 2019; Al-Salem et al., 2009). Furthermore, mechanically recycled plastics tend to continuously degrade in the process and are thus unable to retain their quality after one or more recycling loops. This is especially problematic with regards to food-grade and contact-sensitive plastic applications. There may be examples of technologies where high-purity mechanical recycling is able to generate food-grade output, such as for PET and, potentially, polystyrene (Welle, 2023a and 2023b). However, in many other cases, mechanical recycling faces difficulties in complying with requirements for food-grade and contact-sensitive applications (Padhan & Sreeram, 2019). Despite the potential for mechanical recycling to be further improved through innovation and design for recycling (see section 5.1), and increased collection and sorting (see section 5.2), it is likely that some waste streams will continue to be

[21] Microplastics are '[…] often defined as plastic particles up to 5 mm in dimensions with no defined lower size limit' (Leslie et al., 2022, p. 1).

[22] Tires represent the largest source of microplastics leaked into to the environment, with a share of 47 % percent, followed by road markings with 19 % and city dust with 15 % (Ryberg et al., 2019).

untreatable via mechanical recycling, especially when a certain level of output quality is required. This brings forward the need for complementary recycling approaches for plastics.

3. CHEMICAL RECYCLING TECHNOLOGIES AND APPLICATIONS

As previously discussed, while mechanical recycling plays a vital role in dealing with plastic products at EoL, there are limits to what mechanical recycling can achieve. A group of recycling processes that focus on changing the chemical structure of plastics to produce basic chemicals or other feedstocks and are referred to as chemical recycling technologies are considered as complementary solution to mechanical recycling. Although these processes have been in the development stage for decades, they have been recently increasingly discussed in both academic and policy circles (Quicker et al., 2022; Lee et al., 2021). Nevertheless, there is still a lack of clarity and common understanding among stakeholders about what chemical recycling is and where its potential lies (Manžuch et al., 2021).

Chemical recycling is an umbrella term describing a range of different technologies to recycle EoL plastics. Different conceptualisations of CR have emerged in existing reports and studies. Most of these approaches agree that CR refers to a group of technologies that induce changes in the chemical structure of plastics, via chemical agents, catalysts, or thermochemical processes (Crippa et al., 2019). The aim of these diverse chemical engineering processes is to produce basic chemicals or feedstock for the chemical industry that can be turned into plastics (Rollinson & Oladejo, 2020). As this definition is rather broad, it is applicable to a range of different conversion technologies and recycling pathways.

There is debate on which technologies can fit under the umbrella of CR, with varying categorisations existing in technology reports and research papers. Here, we distinguish between two different categories of technologies, depending on the level of decomposition of the plastic waste input: thermal depolymerisation (e.g. pyrolysis and gasification) and chemical depolymerisation. These different technologies can treat different input streams, vary in their specific processes and result in different outputs. Some other sources also include solvent-based purification[23] in the list of technologies that can fit under the chemical recycling umbrella. In this report, however, we have not included it in the list of core chemical recycling technologies, since it does not break down plastic waste further than the polymer stage, thus essentially not changing the chemical composition.

[23] In the process of solvent-based purification, plastic waste is shredded and dissolved in a solvent that selectively targets the polymer, but not the contaminants. Thus, the contaminants retain their solid state and can be separated from the dissolved polymer. After several purification steps, the polymer is recovered from the solution through precipitation, i.e. re-solidifying the polymer with an anti-solvent (Li et al., 2022). The pure streams of polymer created in this process can then be reformulated into plastic.

3.1 THERMAL DEPOLYMERISATION: GASIFICATION AND PYROLYSIS

Thermal depolymerisation processes utilise heat treatment to convert polymers into multiple simpler molecules. Thermal depolymerisation either decomposes plastics to the corresponding monomers, as is the case with, e.g., polystyrene, or further down into hydrocarbon molecules, depending on the polymer input (see Figure 1). High temperatures[24] are used to decontaminate and degrade plastic waste inputs. Still, further purification steps are often needed before converting the resulting hydrocarbons into monomers (Hann & Connock, 2020). Thermal depolymerisation technologies can produce recycled polymers of virgin-like quality and address the issue of food-grade and contact-sensitive recycling, but they entail yield losses which depend on the specific technology and the quality of waste input. Whilst pyrolysis converts certain polymers into monomers, such as polystyrene into styrene, thermal depolymerisation converts many polymers into chemical intermediates, such as light hydrocarbons and co-products that can also be used to produce fuels – a point of much contestation among stakeholders. Moreover, additional by-products (such as char and gas) lead to production losses and varying yields (Hann & Connock, 2020).

While there are different thermal depolymerisation techniques, two of the most discussed approaches in the literature and among stakeholders are pyrolysis and gasification. These are the main techniques for chemically recycling polyolefins and polystyrene, with the diversity of possible output products described above.

3.1.1 Pyrolysis

The process of pyrolysis (or *cracking*) uses heat (typically in the range of 300–650°C) and an oxygen-deprived environment to break down polymers and obtain simpler hydrocarbon molecules in the form of pyrolysis oil and non-condensable gases. The pyrolysis oil can in exceptional cases already contain monomers (specifically derived from polystyrene waste) or can be converted to monomers for plastics manufacturing or for other petrochemical products in a steam cracker or refinery (Crippa et al., 2019). Pyrolysis can treat mixed polyolefin plastic waste (including multilayers and multi-materials) with higher contamination levels than is possible for mechanical recycling (Li et al., 2022), even though it has its own limitations in terms of the quality of feedstock it can use (Kusenberg et al., 2022a). To meet current steam cracker specifications, pyrolysis oils generated from post-consumer plastic waste needs to undergo different pre-treatment and upgrading steps[25]. Only then can pyrolysis oil be integrated into existing petrochemical sites, to partially substitute naphtha. In contrast to (controlled) chemical depolymerisation (see section 3.2 below), the scission of bonds usually occurs at random points in the polymer chain (Hann & Connock, 2020). Therefore, the resulting pyrolysis oil is composed

[24] Different temperature ranges are recorded in the literature. Generally, pyrolysis techniques utilise lower temperatures than gasification techniques. Pyrolysis temperatures range from 300–650°C, while gasification temperatures are at around 500–1300°C (Saeba et al., 2020).

[25] According to the literature, hydrogen-based technologies are currently the most effective upgrading technique (Kusenberg et al., 2022a).

of a range of hydrocarbon products and requires further purification before it can be used as a feedstock for polymer production (Hann & Connock, 2020).

As of now, pyrolysis is mainly used for mixed polyolefins (PE and PP), i.e. packaging, bags, films and mixed plastic waste, as well as polystyrene (PS), e.g. from insulation and food packaging, or for rubber tires (see Figure 5) (Hann & Connock, 2020). Compared to mechanical recycling, pyrolysis has a higher potential[26] to utilise waste from mixed streams for food contact applications (Kusenberg et al., 2022b). A challenge linked to the application of pyrolysis processes refers to the steady flow of consistent quality feedstock in large amounts needed to make the complex purification and separation processes economically viable (Qureshi et al., 2020; Ragaert et al., 2017). The treatment of by-products generated throughout the process can also be challenging and costly. Despite these challenges, there have recently been increasing examples of pyrolysis plants operating on pilot and commercial scale, aiming to produce chemical feedstocks for plastics manufacturing or other chemicals. In some cases, these plants combine their processes with fuel production to increase the economic viability of their operations (Hann & Connock, 2020).

3.1.2 Gasification

Gasification heats plastic waste (around 500–1300°C) under limited oxygen supply in order to generate so-called syngas, a mix of carbon monoxide and hydrogen. Syngas can be used as a feedstock to a range of chemical processes (SYSTEMIQ, 2022), including conversion into polymers via intermediates such as methanol or light hydrocarbons. As a result of the high operating temperature, gasification technologies have a higher feedstock flexibility. In theory, any plastic can be used as a feedstock for gasification (see Figure 5), and the process mostly targets mixed plastic streams. Gasification technologies have, however, mainly been commercially applied to fuel production instead of plastic-to-plastic recycling (Hann & Connock, 2020). Moreover, there are significant carbon losses that occur from the process of partial oxidation of plastic waste.

As discussed during the CEPS stakeholder meetings, several gasification projects are currently in a planning phase, and research is focused on increasing feedstock flexibility, efficiency and scale (Lee et al., 2021). For example, the Canadian company Enerkem has started several projects in Spain and the Netherlands to produce ethanol and methanol from sorted municipal solid waste (Zero Waste Europe, 2019).

3.2 Chemical depolymerisation

Chemical depolymerisation turns plastics back into their monomer or oligomer (short polymer) building blocks. During the process, polymer chains are broken down under controlled conditions through the use of chemicals (Hong & Chen, 2017). Afterwards, oligomers or

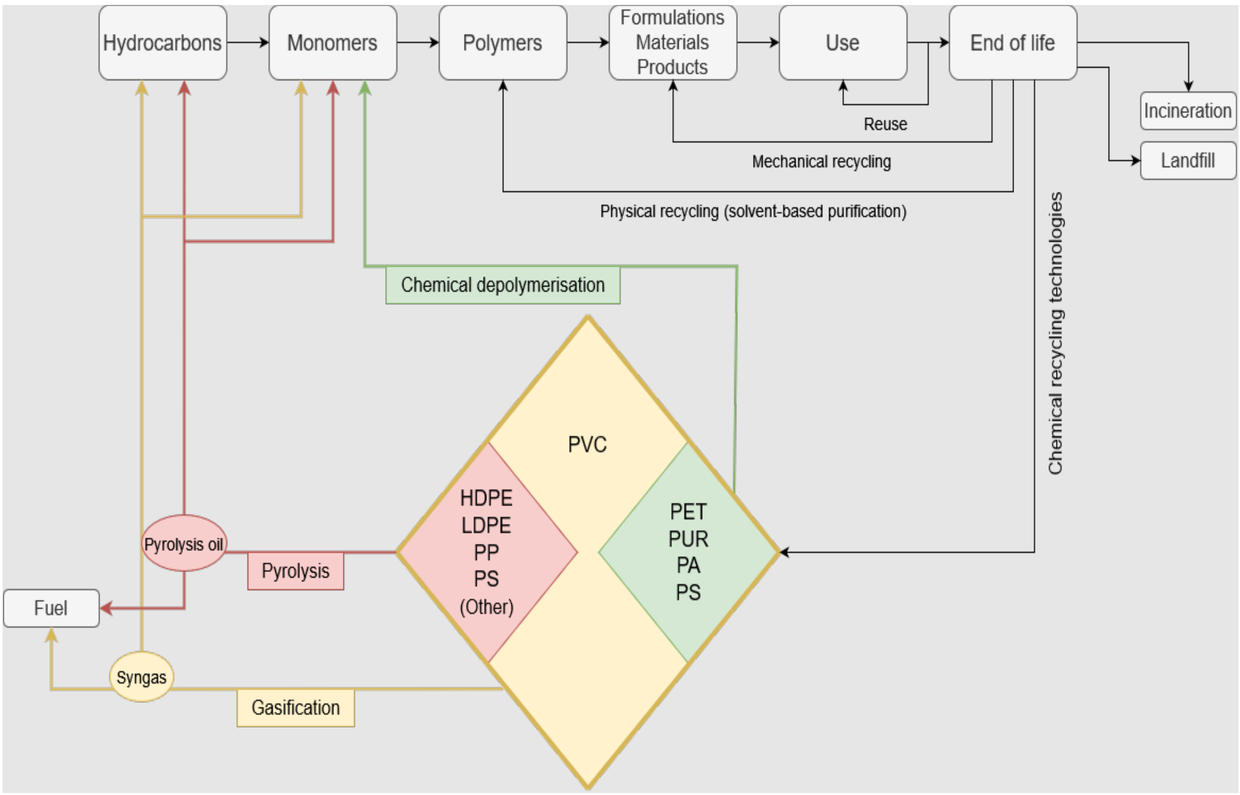
[26] It should be noted, however, that quality of feedstock is also an issue for the pyrolysis process and beyond certain contamination levels there could be risks for the industrial process (Kusenberg et al., 2022a).

monomers are recovered from the reaction mixture and separated from contaminants (Hann & Connock, 2020). The recovered molecules are then used, separately or blended with virgin monomer, to create virgin-grade polymers (Crippa et al., 2019). Thus, chemical depolymerisation can address the issue of food-grade recycling requirements. There are different chemical depolymerisation techniques, depending on the chemical agent used for polymer scission. Examples include glycolysis, hydrolysis and methanolysis (Ragaert et al., 2017).

Potential feedstock for chemical depolymerisation is depicted in Figure 5 and includes PA (polyamide), PET (polyethylene terephthalate), PUR (polyurethane) and PLA (polylactic acid) (Zero Waste Europe, 2019; Hann & Connock, 2020; Crippa et al., 2019). Treatable waste streams thus include bottles, food packaging, textiles, mattresses, fishing lines and nets and beverage cups. Technologies for additional polymers, such as polystyrene (PS), are coming into play as well. A disadvantage of chemical depolymerisation is its limitation in processing pre-sorted and, for economic reasons, relatively clean, mono-streams (i.e., uncontaminated by other polymers or non-polymers).

Despite the existence of a variety of different chemical depolymerisation technologies currently being tested, commercialisation of these technologies at scale has been limited, and many existing plants are at the pilot stage (Hann & Connock, 2020). Most industrial pilots depolymerise PET from packaging and textiles (Hann & Connock, 2020; Zero Waste Europe, 2019). A few companies have developed chemical depolymerisation processes for recycling PUR (polyurethane) foams.

Figure 5. Chemical recycling technologies and feedstock



Source: Own design based Crippa et al. (2019) and von Vacano et al. (2022), as well as stakeholder inputs.

3.3 COMPLEMENTARITY WITH MECHANICAL RECYCLING

Chemical recycling technologies have potential to treat more heterogenous plastic waste streams with certain levels of contamination that cannot be managed by mechanical recycling processes (Dogu et al., 2021). Still, these technologies also face limitations. First, the high energy-intensity of the processes renders them economically challenging and dependent on demand for recycled feedstocks. Another issue is that of production losses and resulting varying yields (which are highly impacted by feedstock quality[27]) and, potentially, a range of co-products such as fuels (Hann & Connock, 2020). Finally, CR plants require stable and continuous feedstock supply in sufficiently large quantities in order to be economically viable (Ragaert et al., 2017). Diverting plastic waste from landfills and incinerators to recycling routes can be a challenging task, though policy developments such as the inclusion of incinerators in the EU Emissions Trading System (EU ETS) and mandatory recycled content targets can provide incentives for diverting plastic feedstock to recycling.

While there is a need for diverting plastic waste from landfills and incinerators to recycling routes, one key aspect is how mechanical and chemical recycling technologies can advance together while avoiding feedstock competition. This need was also repeatedly expressed by different stakeholders at the stakeholder events organised by CEPS. Potential levers for complementarity can occur at the levels of input, process as well as output.

At the input level, potential for complementarity lies in differences in input waste stream capacities. Mechanical recycling is not suitable for polluted, highly degraded, complex or heterogeneous plastic. Although most CR technologies also require certain levels of homogeneity and purity of feedstock, they could generally allow for the processing of some inputs that cannot be treated by mechanical recycling, as an alternative to incineration and landfilling.

Concerning the process, CR is more energy-intensive and expensive than mechanical recycling. Therefore, it seems unlikely that there would be a competition with mechanical recycling unless the output quality of CR was significantly higher. Still, stakeholders have argued about the need for regulatory safeguards in order to ensure that the more environmentally friendly recycling process is always preferred (Zero Waste Europe, 2019).

Regarding the output level, via the mechanical route, polymers cannot be recycled infinitely without significant reduction of quality and properties. Thus, mechanical recycling generally leads to downcycling polymers over time and cannot replace the production of virgin polymers. As described above, CR technologies may have the potential to retain better polymer quality and properties over time.

As such, an opportunity for complementarity of both recycling routes may arise from recycling materials through CR either as an alternative to mechanical recycling, where food-grade

[27] See Gendell and Lahme (2022).

recycling or virgin quality due to safety reasons (e.g., automotive plastics) is required, or as an added step to the mechanical recycling process, to enable a circular plastic economy. Furthermore, mechanical recycling generally is not able to deal with the challenge of polymer contamination, whereas some of the chemical recycling technologies have the capacity to effectively handle contaminated input streams (Zero Waste Europe, 2019; Hann & Connock, 2020).

4. FUTURE TRENDS: WHAT WILL PLASTIC VOLUMES LOOK LIKE IN THE COMING YEARS?

As discussed in the previous sections, only a share of plastic waste is currently recycled, mainly through mechanical recycling processes. A key question in this context is how plastic volumes will evolve in the near and long-term future, and what different recycling processes could contribute. Different projections or scenarios assessing the evolution of both plastic production and waste volumes in the EU have been recently developed. Whilst uncertainty remains as to the impacts of policy and technology developments on future plastics trends, forward-looking assessments of this kind provide a useful indication of the scale of plastic recycling capacity required to meet EU targets, the types of plastic waste streams that are likely to increase the most and, crucially, the recycling technologies mix that can more effectively process them. Building above these assessments and other existing evidence, this section highlights the major drivers and examines the relative policy implications.

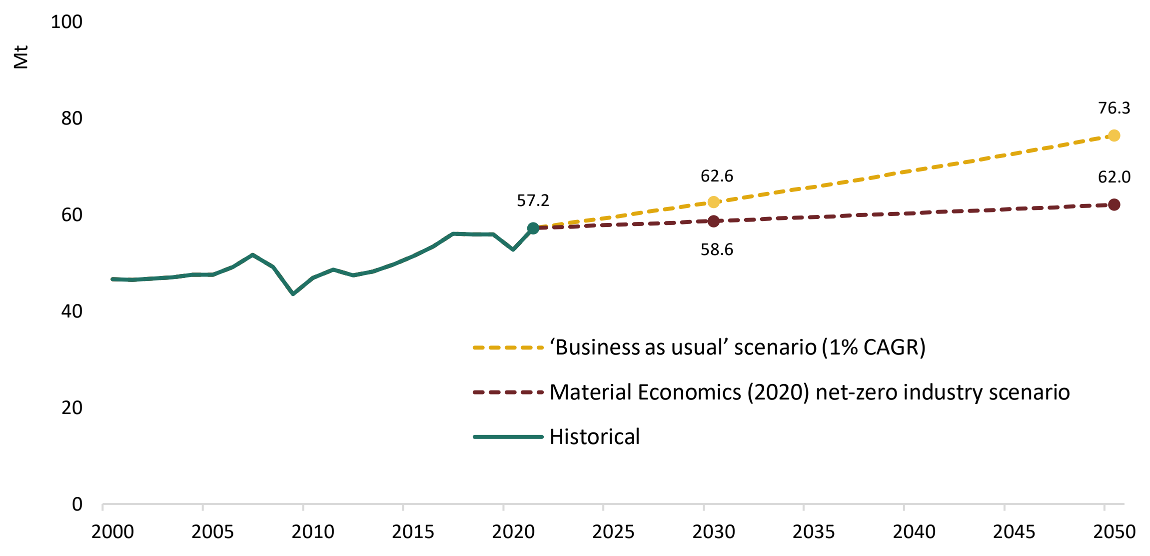
4.1 PLASTIC PRODUCTION AND USE

According to Plastic Europe (2022), plastic production in the EU27+3 was 57.2 Mt in 2021. If solely based on the historical trend of plastics production since the early 2000's[28] (i.e., in the absence of major policy or technology shifts), this volume would increase to about 63 Mt in 2030 and 77 Mt in 2050[29]. However, as the EU builds momentum on policies for plastic circularity, the growth in plastic production will likely be more moderate. For instance, a significantly more conservative figure is reported by Material Economics (2019; 2020), whose net-zero industry scenario expects EU plastic production to reach only 62 Mt by 2050. Figure 6 below presents both the evolution of waste production based solely on historical trends and the estimates by Material Economics.

[28] See Eurostat (2022).

[29] These estimates are based on a 1 % CAGR assumption from now until 2050, in line with the historical evolution trend observed in 'manufacturing of plastics in primary forms' in the EU27 from 2000 to 2021, reported by Eurostat (2022).

Figure 6. Historical and expected evolution of EU plastic production



Notes: Scope is EU27+3. Production includes primary production and mechanical recycling output.
Source: Authors' elaboration based on Plastics Europe (2022), Eurostat (2022) and Material Economics (2020).

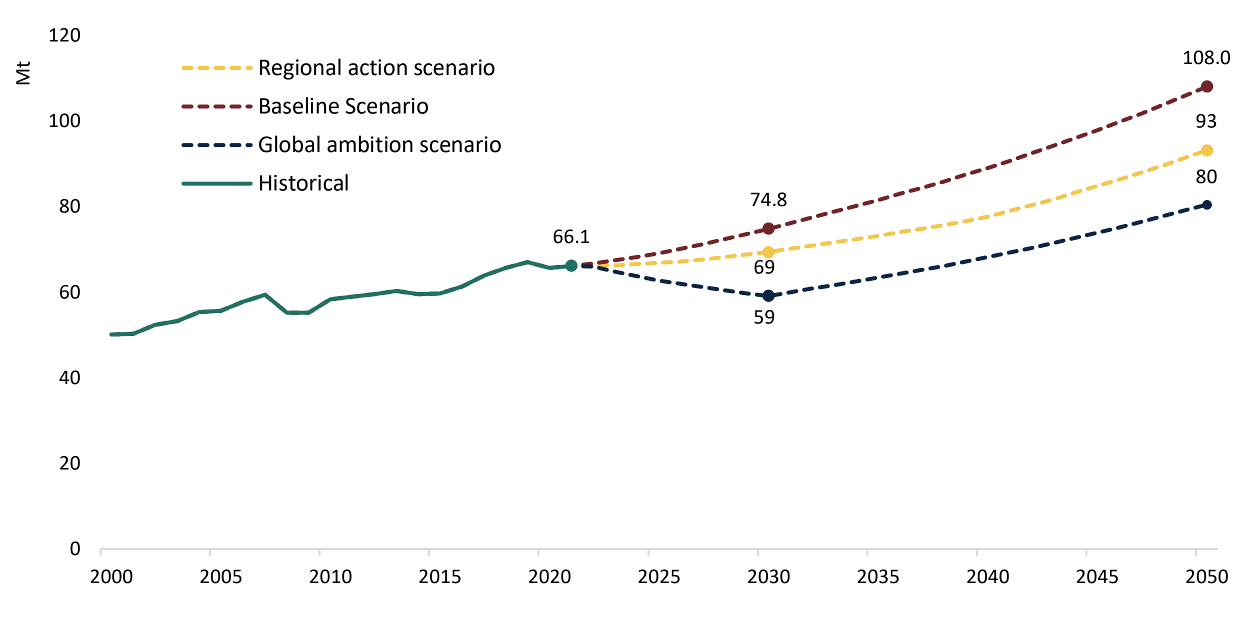
To showcase how circular economy policies can impact future plastic use[30], the OECD (2022) outlines three policy scenarios with varying level of ambition. The *Baseline* scenario assumes no major actions are taken towards plastic circularity, and estimates EU plastic use to increase from the current 66 Mt to about 75 Mt by 2030. This trend aligns with the plastic production 'business as usual' scenario described above in the short-term, though the curve is expected to become increasingly steeper in the long term, exceeding 100 Mt of plastic use in the mid-2040s[31].

In the *Regional Action* scenario – which assumes a mix of fiscal and regulatory policies to be applied at all plastics life-stages, yet with different ambitions among OECD and non-OECD countries – a slightly slower uptake in plastic use in EU countries is foreseen (-7 % in 2030 and -14 % in 2050 compared to the *Baseline* scenario). Finally, the *Global Action* scenario – where policy action is adopted with more ambitious targets, more rapidly and at a global scale – foresees plastic use in the EU to decrease to 59 Mt in 2030 and only increase thereafter up to 80 Mt in 2050 (-21 % in 2030 and -26 % in 2050, compared to the *Baseline* scenario). In all three scenarios, plastic use is expected to increase in 2050 irrespective of the ambition of circular policies for plastics. The scenarios are depicted in Figure 7.

[30] It should be noted that the figures by the OECD (2022) do not refer to plastic *production* but plastic *use*. While these quantities are not directly comparable, it can reasonably assumed the two to have fairly similar evolutions over time.

[31] OECD (2022)'s Baseline Scenario foresees a 1.37 % CAGR of plastic use from 2021 to 2030, and of 1.85 % 2030 to 2050 in the EU27.

Figure 7. Historical and expected evolution of EU plastic use



Notes: Scope is EU-27.
Source: Authors' elaboration based on OECD (2022).

Over the last 20 years, the major driver of EU plastics' production has been the packaging sector, which accounts today for about 40 % of the whole EU plastic demand[32] (Plastics Europe, 2022). While future plastic demand is likely to rise across all its applications, the OECD (2022) expects the global growth of plastics demand to be mostly driven by three predominant segments, namely packaging, construction and transport. In particular, transport is expected to see the largest relative growth across sectors – x3.2 times by 2060 compared to 2019 levels – with packaging and construction growing by 2.7 and 2.5 times over the same period, respectively (OECD, 2022). SYSTEMIQ (2022) reports similar trends in the EU context, yet with a more prominent role of construction over the automotive sector. For the former, a 48 % increase in plastic demand is foreseen by 2050 (from 10 to 15 Mt), as a result of a rapidly increasing plastic use within the sector, e.g. for insulation purposes. As for the automotive, plastic demand is expected to grow by 25 % by 2050, again resulting from an increasing plastic intensity in products (up to a third more plastic per vehicle by 2050) (SYSTEMIQ, 2022).

In terms of polymers, the above trends are likely to reflect an increasing use among those mostly employed within the fastest-growing sectors, such as PP (packaging) PET (packaging and automotive) and PVC (construction) (OECD, 2022).

[32] Meant as demand from plastic converters.

4.2 PLASTIC WASTE GENERATION

As shown in the previous sections, there have been different assessments on plastic waste generation in the EU, with varying methodologies and assumptions employed. This section refers to 'reported' plastic waste volumes, which according to Plastic Europe (2022) currently lay at 29.5 Mt[33]. Taking into account the historical evolution of post-consumer plastic waste reported by Plastics Europe, it can be estimated that in a 'business as usual' scenario this volume could grow to roughly 33-34 Mt in 2030, and further to 43-44 Mt in 2050[34]. Still, as we saw for plastic production and use, the actual future trajectory of plastic waste volume is likely to deviate from such a growth path depending on how ambitious the circular policies adopted for plastics are.

Alternative scenarios looking at future development of plastic waste from the (non-industrial) packaging, construction, household and automotive sectors in the EU have been developed by SYSTEMIQ (2022)[35]. In a *Current Action Scenario* – which reflects current actions taken by governments and industry to enhance plastic circularity – SYSTEMIQ expects plastic waste from these sectors to increase from the 25.2 Mt in 2020 to 26.4 Mt in 2030 and 34.2 Mt in 2050[36]. Applying the same trajectory to the whole plastic waste group (i.e. assuming the same growth path for sectors not in-scope, namely industrial packaging, agriculture and electronics), this would lead to about 31.8 Mt in 2030 and 41.2 Mt in 2050. Notably, this would represent only a marginal improvement compared to a business-as-usual scenario.

To quantify the impact of more ambitious policies, SYSTEMIQ (2022) also outlines a *Circularity Scenario*, which sets the pathway to 'substantially increase plastics system circularity' (SYSTEMIQ 2022, p. 39). Based on this scenario – and again assuming the same trend for all waste streams – overall plastic waste volume can be expected to decrease to 27.6 Mt in 2030 and further beyond towards 2040, and increase thereof up to 29.5 Mt (i.e., 2020 level) by 2050[37]. Notably, over 10 Mt of plastic waste per year could be saved by 2050 under such an ambitious scenario. Fig 8 below presents the estimates about future evolution of EU plastic waste based on the historical trends (i.e. 'business as usual' scenario) and the two scenarios by SYSTEMIQ.

[33] As explained earlier, this figure is based on the historical plastic waste volumes and reflects official EU statistics, but other studies have suggested that they do not capture the full amount of plastic waste generated within the EU due to methodological issues. According to Material Economics (2022), up to 15 Mt of so called 'missing plastics' should be added to the present plastic waste reported figure – i.e., about half of the reported volume. SYSTEMIQ (2022) estimates that the plastic waste data gap in official statistics lies between 8-15 Mt.

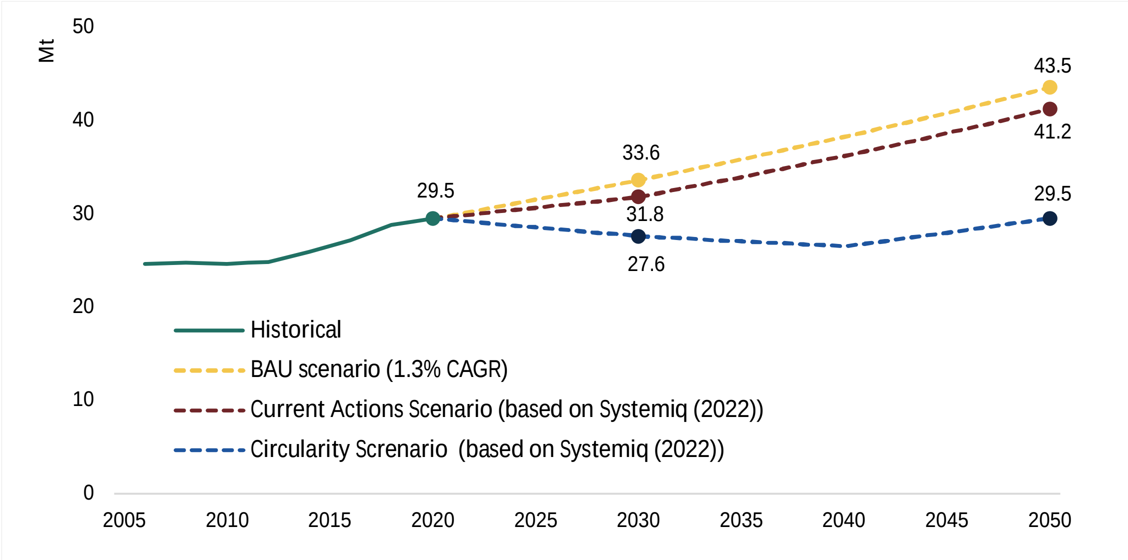
[34] This estimate is based on a 1.3 % CAGR, i.e. slightly higher than the 1 % CAGR assumed for future plastic production.

[35] Sectors out of scope of SYSTEMIQ (2022) assessment include industrial packaging, electronics and agriculture. SYSTEMIQ estimates that altogether the sectors within the scope its assessment cover about 75 % of the EU plastics demand, and 83 % of known post-consumer waste generation.

[36] These figures are net of an expected decrease in plastic waste due to reduced plastic consumption, respectively of a 1.4 Mt in 2030 and 1.6 Mt in 2050, compared to a baseline trajectory.

[37] These figures, adapted to comprise all plastic waste groups, account for an expected decrease in plastic waste resulting from the combined result to reduce plastic consumption and plastic substitution.

Figure 8. Historical and expected evolution of EU plastic waste



Source: Authors' elaboration based on Plastics Europe (2022) and SYSTEMIQ (2022).

In terms of the contribution of different sectors to plastic waste generation, according to Plastics Europe (2022) the current disaggregation sees the packaging sector being by far the most important waste stream (61 %), followed by electrical and electronics tools (6 %) and the building and construction sector (6 %) (See Section 1) [38]. Looking at the future developments, SYSTEMIQ (2022) – which nonetheless excludes agriculture, electronics as well as the industrial packaging sectors from its assessment – foresees packaging plastics waste to still contribute for the largest share of future plastic waste generation, but yet to lose ground to the household, automotive and construction sectors. The construction sector in particular is expected to see the largest growth of plastic waste generation (set to almost triple by 2050), followed by the automotive sector (about +115 % over the same period). This will be the combined result of the above-mentioned current (and expected future) tendency to increase plastic use in these sectors, on one side, and the relatively long lifetimes of these plastic applications (15 to 100 years for the former, about 12 years for the latter) on the other, which leads to a significant time lag before plastic production trends are reflected in waste volumes. A more stable growth path is foreseen for the packaging sector (+23 % by 2050) and households goods sectors (+36 %)[39] .

[38] It should be noted, however, that while Plastics Europe (2022) reports the 'houseware, leisure and sports' products group as having only a marginal impact (4 % of the total, i.e. about 1.4 Mt), according to SYSTEMIQ's (2022) assessment the contribution of the 'household goods' sector is significantly higher (about 5.2 Mt).

[39] Plastic waste generation of household and packaging sectors assume a 1 % CAGR from now until 2050, in line with historical evolution (SYSTEMIQ, 2022).

4.3 PLASTIC WASTE TREATMENT

With regard to plastic waste treatment, over the last 15 years the EU has seen a steady decrease in landfilled volumes, in parallel with the sustained growth of both recycling and incineration. Of the 29.5 Mt of plastic waste collected today, about 12.4 Mt are used for energy recovery (+77 % compared to 2006), 6.9 Mt are landfilled (-47 %) and 10.2 Mt are sent for recycling[40] (+117 %) (Plastics Europe, 2022). EU plastic recycling capacity is today almost entirely based on mechanical recycling, with chemical recycling only playing a negligible role (0.1-0.2 Mt). While these trends are likely to continue in the foreseeable future – with plastic recycling capacity further expanding and landfilled volumes being gradually diverted towards either recycling or incineration – the exact evolution of plastic waste treatment in the EU, for instance in terms of the role of alternative recycling technologies, remains uncertain and dependent on ongoing and future policy developments.

Different studies have investigated how future EU plastic waste treatment could evolve in alternative scenarios. In SYSTEMIQ (2022) *Current Actions Scenario*, the combined output of mechanical and chemical recycling capacity in the EU – i.e. net of material losses occurring during the recycling processes – is expected to double from the current 3.5 Mt to about 7 Mt by 2030, and almost triple by 2050 (10.4 Mt). Mechanical processes are expected to remain the predominant recycling route (with 5.8 and 7.6 Mt of capacity in 2030 and 2050, respectively), but chemical recycling output is also expected to grow (1.1 Mt by 2030 and 2.8 Mt by 2050). While landfilled volumes are expected to halve by 2050 (from the current 6.9 Mt to 3.6 Mt), the amount of plastic waste sent to incineration is expected to increase (12.8 Mt in 2030 and 17.6 Mt in 2050).

In the more ambitious *Circularity Scenario*, SYSTEMIQ (2022) foresees a far larger role for recycling, particularly chemical recycling, which is estimated to reach 3.1 Mt capacity by 2030 and 7.3 Mt by 2050 (i.e., 13% and 21% of the whole plastic waste volume, respectively). Mechanical recycling capacity is also projected to increase faster, yet at a slower pace (5.9 Mt in 2030, 9.8 Mt in 2050). Coupled with a substantial reduction in plastic consumption (up to 9 Mt saved in 2050, compared to the *Current Actions Scenario*) this increase in overall recycling capacity would allow to substantially reduce the role of incineration (7.2 Mt of plastic waste in 2030 and only 2.3 Mt in 2050) and landfilling (3.2 Mt in 2030, and 0.8 Mt in 2050). The evolution of future EU plastic waste recycling capacity up to 2030 has also been assessed by Lease et al. (2022). The authors develop 5 alternative scenarios which are differentiated acccording to the relative importance of mechanical and chemical recycling processes[41].

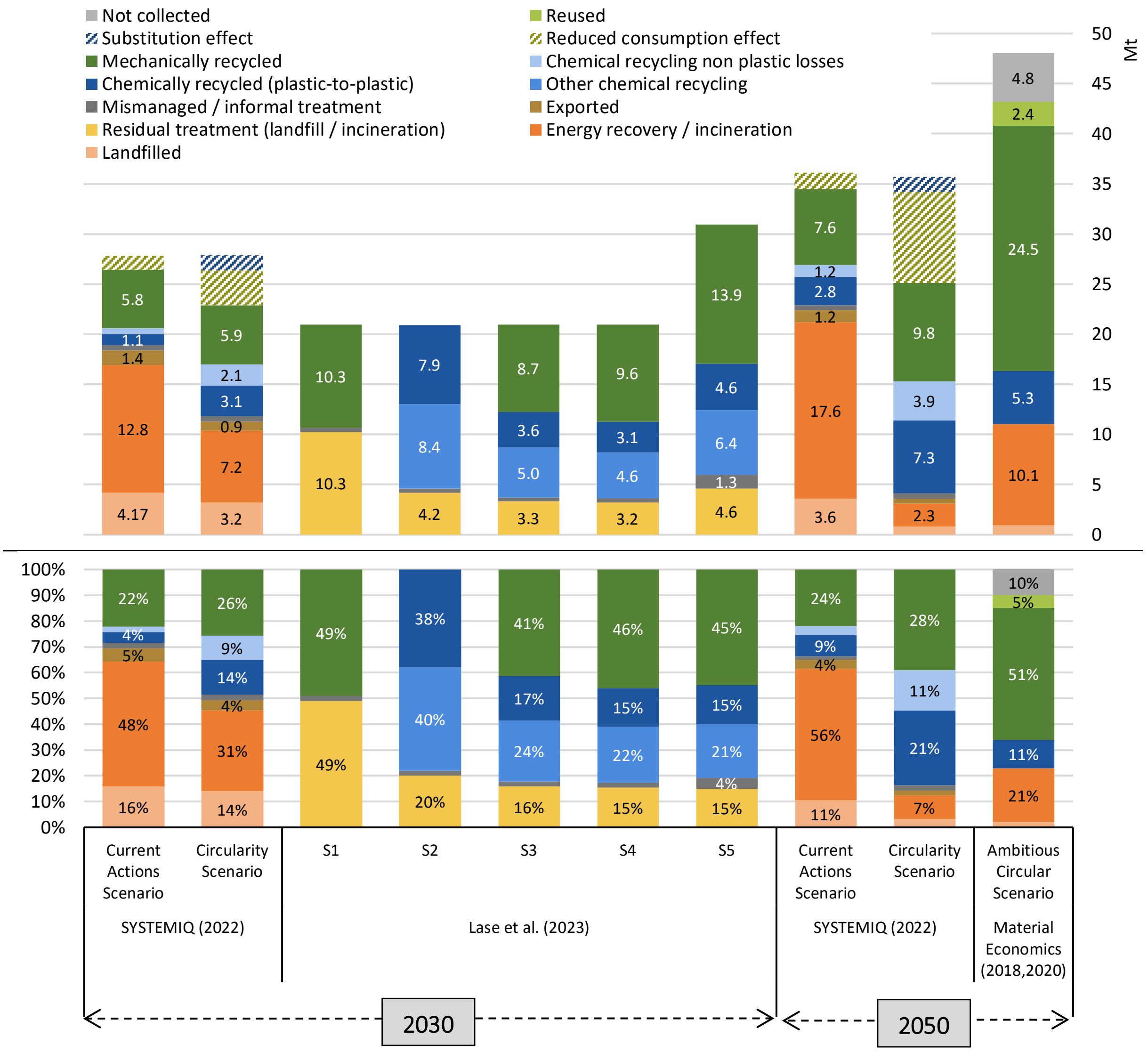
[40] Of these, roughly 1 Mt is exported for recycling and 9.1 Mt enter the recycling process in the EU-27+3 (Plastics Europe, 2021).

[41] The scenarios are defined as follows: in S1, best practices of waste collection, sorting and mechanical recycling are widely applied throughout the EU, though only mechanical recycling is deployed. In S2, chemical recycling outcompetes mechanical recycling, and all sorted plastic waste is assumed to be treated via chemical recycling. In S3, chemical recycling is considered as a 'sub-optimal' alternative to mechanical processes, primarily dealing mixed waste streams. In S4, chemical recycling is considered as a complementary technology to mechanical recycling, used for waste streams that would otherwise be landfilled or incinerated. S5 is identical to S4 though accounts for the 'data gap' in recorded plastic waste (Lase et al., 2023). Scenarios S1-S4 are shown in Figure 9.

Notably, the study shows how the largest outcome in terms of plastic waste volume recycled is achieved when the two technologies coexist in the system – i.e. when neither of the two becomes dominant (scenarios S3 and S4). In these two scenarios, the study estimates that between 8.6 and 9.6 Mt of mechanical recycling capacity and between 3.1 and 3.6 Mt of (plastic-to-plastic) chemical recycling capacity could be reached in the EU by 2030 (that is, enough to process 15-17% of the overall plastic waste volume).

Finally, in Material Economics (2018) projections to 2050 chemical recycling is expected to remain small compared to mechanical routes, being primarily used for waste streams that cannot be easily treated via mechanical recycling. As such, 11 % of overall plastic waste is projected to be chemically recycled, while 51 % is expected to be mechanically recycled. Figure 9 below shows all scenarios to 2030 and 2050 described above.

Figure 9. Breakdown of plastic waste treatment in 2030 and 2050, in Mt (above) and % (below)



Sources: Plastics Europe (2022); Material Economics (2018,2020); SYSTEMIQ (2022); Lase et al. (2023). *Notes*: Plastics Europe (2022) and Material Economics (2022) refer to EU+3. SYSTEMIQ (2022) figures refer to EU27+1 and only consider packaging, household goods, automotive and construction sectors, representing 83 % of the total plastic waste. Lase et al. (2023) refer to EU27+3 in 2018 and only consider the 10 most used polymers within the packaging, agriculture, building and construction, electronics and automotive sectors, which represent about 60% of the total. Lase et al. (2023) S5 scenario and Material Economics (2018,2022) account for the missing volume of plastic waste discussed in previous sections. These assessments may include in their scope solvent-based purification for which as discussed in section 3 there is debate as to whether they should fit under the umbrella of chemical recycling technologies.

Overall, the above scenarios suggest that a form of coexistence between mechanical and chemical recycling processes is likely to be required to achieve an optimal systemic level of plastics recycling, particularly in light of the increasing amount of plastics entering the system in the coming years (see Section 4.1 – 4.2). While the projections on the volume of plastic waste treated via chemical recycling processes vary considerably depending on the assumptions and scope of the assessments, even the most conservative scenario foresees a substantial expansion of chemical recycling capacity compared to current levels. If ambitious actions were to be taken to increase plastics' circularity and the role of chemical recycling, in fact, chemical processing capacity could possibly exceed 3 Mt by 2030, and range from 5 to 7 Mt by 2050. In relative terms, this implies that about 15 % of plastic waste could be turned into new plastics via chemical recycling routes by 2030, and over 20 % in 2050.

5. ENABLING FUNCTIONING PLASTICS RECYCLING MARKETS

5.1 ECODESIGN FOR IMPROVED CIRCULARITY OF PLASTICS

The increasing compositional complexity and heterogeneity of many plastic products, especially packaging, considerably affects collection, sorting and recycling rates. Specific plastic design choices can make recycling processes more costly and affect both the quality and value of recycled plastics (European Commission, 2018). Certain additives and colourants make it increasingly difficult to mechanically recycle plastic packaging. The multi-layers of polymers and the presence of certain additives also pose challenges for chemical recycling technologies[42]. Design choices crucially influence how plastic products are managed at EoL and whether they are reparable, reusable or recyclable, and how much virgin material will be necessary for a potential second life. At the same time, the design or products with plastics also influences to what extend these products can incorporate recycled content inputs (Ragaert et al., 2017).

Adopting ecodesign approaches for plastics therefore holds potential to deliver multiple benefits including reduction of waste management costs, increase of overall recycling rates of plastic, improvement of collection efficiency and improvement of recycling output quality (European Commission, 2018; Ellen MacArthur Foundation, 2017). Different ecodesign options for plastics include using mono-materials instead of composites and multilayers, avoiding additives or fillers that could contaminate the material and using as few different types of

[42] For example, PVC layers in films can damage steam crackers and release toxins in pyrolysis processes (Hann & Connock, 2020). In addition, chemical depolymerisation processes currently cannot process certain polymers, and often require relatively clean and sorted feedstock – which is impeded by complex and multi-layered plastic structures.

materials as possible (Van Doorsselaer, 2022). However, the specific design changes that might be useful ultimately depend on the material and the application (Tabrizi, 2021).

While ecodesign approaches for plastics can have benefits for both mechanical and chemical recycling technologies, the various technologies may have different design needs. With regard to chemical recycling, feedstock composition needs vary between the different CR technologies, as thermal depolymerisation can tackle more mixed and multi-layered input materials than chemical depolymerisation (see Section 3.2). Moreover, even though thermal depolymerisation processes can handle a broader range of plastic inputs, they too would benefit from certain design choices. For example, research into depolymerisation behaviour of plastic waste could help identify additives that make pyrolysis and gasification difficult, thereby supporting design for chemical recycling (Dogu et al., 2021). Innovations in polymer design can also support chemical depolymerisation. For instance, design for recycling can facilitate selective depolymerisation under milder conditions, to separate polymers from blends, for example, in textiles and mixed plastic wastes (Sánchez & Collinson, 2011). Simply put, polymers can be designed in a way that depolymerisation is selectively triggered by different external stimuli, such as temperature or light (Hatti-Kaul et al., 2020; Garcia, 2016). However, polymers that can be easily depolymerised may exhibit undesired thermal and mechanical properties and decreased material performance (Hong & Chen, 2017). Therefore, more research in this area is needed.

Design for *mechanical* recycling may aim to anticipate different waste management needs than design for chemical recycling. By adapting plastic design with a view to its EoL management, more valuable plastic waste could be captured by the collection and sorting infrastructure (World Economic Forum et al., 2016), and mechanical recycling processes could be facilitated, thus minimising process losses and increasing recyclate quality. Relevant plastic design considerations include, for example, simplifying material formulations and combinations, designing for facilitated dismantling and disassembly, and avoiding specific chemical additives (Le Blevennec et al., 2019).

The different design needs for chemical and mechanical recyclability raise the question of how to best implement ecodesign in practice. Of course, design choices not only depend on plastic recycling options, but also on product purposes and end markets. A further aspect that needs to be considered is that both mechanical and chemical recycling technologies have not reached the limits of innovation. Added to this, a recent global assessment by OECD of patented innovations in plastics identifies low levels of innovation at the plastic design stage to improve recycling[43] and reuse (Dussaux & Agrawala, 2022). This implies that there is significant room for improving the design of plastics and the overall efficiency of different processes. In line with the priorities of the waste hierarchy, design for improved durability and reparability and increased lifetimes of plastics should be prioritised in order to support waste prevention

[43] Interestingly, according to the assessment, the majority of plastic recycling patents globally have been issued for mechanical recycling and sorting, while only limited patents have been issued for plastic-to-plastic chemical recycling (though there have been some signs of growth recently) (Dussaux & Agrawala, 2022).

(Watkins et al., 2019). Design for recycling would be the subsequent design option, considering both the aspect of designing plastics that can be more effectively recycled at EoL and the question of how to incorporate recycled polymers in new products. Combining these two objectives in the development of design requirements can deliver the higher benefits. Such design requirements should be based on an assessment of the resource efficiency and climate change benefits of the specific design options applied for different product groups in order to achieve the higher overall environmental benefits (Ragaert et al., 2017).

5.2 IMPROVING SORTING AND COLLECTION

As discussed in the previous sections, a large share of EoL plastics today is either incinerated for energy generation or landfilled. On top this, high volumes of plastic are lost in the sorting and recycling process. One reason for low recycling rates across Europe are the insufficient collection and sorting systems in place. As new plastics are developed to allow for improved applications, such as food preservation, the complexity of the materials increases. This leads to challenges for collection, sorting and recycling – both mechanical and chemical recycling, depending on the specific technology. Systems and technologies face difficulties in adapting to continuously developing materials and products, while consumers struggle to understand how to correctly separate household plastic waste (Crippa et al., 2019). To divert recyclable plastic waste from disposal and increase recycling rates, improving current collection, sorting and pre-treatment schemes is crucial (Tallentire & Steubing, 2020; Bapasola et al., 2023).

5.2.1 Current collection and sorting practices for different types of plastics

Plastic packaging makes up the majority of post-consumer plastic waste. On average, the share of packaging collected for recycling (i.e. the collection rate) ranges from 40 to 50 % in the EU (Bapasola et al., 2023). Given that only a fraction of collected waste ends up being recycled due to process losses, this rate would need to increase considerably to meet the 55 % recycling target for plastic packaging waste, set in the European Commission's proposal for a Packaging and Packing Waste Regulation [44]. Even countries with a high efficiency of separate collection, such as Germany and Belgium, are likely to miss the 2030 target included in the Packaging and Packing Waste Directive (Bapasola et al., 2023).

Moreover, current collection and sorting practices display stark variations across and within Member States. Plastic waste is either pre-sorted by consumers at the household level (source separation), or collected commingled with other types of waste (Miranda et al., 2013). In addition, some regions have implemented Deposit Return Systems (DRS), financially incentivising consumers to return EoL beverage containers to a collection point. Source-separated or commingled waste is then usually collected either through kerbside collection or through bring banks, i.e. decentralised collection points with big containers for recyclables (Clausen et al., 2018). In a next step, plastic waste is sorted in a materials recovery

[44] The proposal was released in November 2022 as a revision of the existing Packaging and Packaging Waste Directive (PPWD), with the aim to curb rising plastic waste and to increase circularity (European Commission, 2022).

facility (MRF), using different technologies and pre-treatment steps, including ballistic separation and sensor-based sorting (Clausen et al., 2018). Depending on whether the plastic was collected in single streams or commingled, more pre-treatment steps may be necessary to allow for high-quality inputs for recycling. While commingled collection can reduce collection costs and is easier to understand for end users, thus increasing capture rates, it also makes cross-contamination and decreases in material quality more likely (Miranda et al., 2013).

Differences between collection practices occur across plastic waste streams, and even within the same polymer. For instance, while PET bottles usually serve as a best practice example for effective waste management, overall PET recycling rates are low. Only one quarter of all PET waste in Europe is recycled, despite average recycling rates of 50-55 % for clear PET beverage bottles (SYSTEMIQ, 2023). This is partly due to the fact that beverage bottles only make up around half of the European PET consumption. The rest are textiles and trays, of which only 10 and 20 %, respectively, are sent for recycling.

The comparatively high recycling rates of clear PET bottles are a result of widespread collection and sorting facilities, as well DRS in some Member States, such as Germany and Norway. Still, collection rates of PET bottles vary significantly across Europe, ranging from below 40 % to above 90 % (SYSTEMIQ, 2023). Countries with a functional DRS are estimated to have double the collection rate of countries without DRS (Eunomia and Zero Waste Europe, 2022). Such unharmonised collection systems, along with a lack of dedicated collection schemes and sorting processes for textiles and trays, contribute to varying collection rates of PET waste streams in Europe (SYSTEMIQ, 2023). To achieve the 77 % collection target for 2025 set by the Single-Use-Plastics (SUP) Directive, let alone the 90 % target for 2029, current rates will need to be improved in many Member States.

Waste collection may be even more difficult for other types of plastic packaging. Post-consumer flexible plastics (PCFP), i.e. films mainly serving as consumer packaging, are considered difficult to recycle (Lase et al., 2022). They consist of a mixture of polymers, mainly LDPE, LLDPE and PP, but can also be multi-layered and include other polymers such as PET and PA. Due to this complexity and large amounts of contaminants (Kol et al., 2021), packaging waste is expensive to collect and often ends up incinerated, landfilled or exported (SYSTEMIQ, 2023). The current European waste management system mainly targets rigid plastics. Only in a few Member States, including the Netherlands, Germany and Belgium, is PCFP source separated (i.e., manually pre-sorted at the household level) together with rigid plastics packaging, beverage cartons and metals (Lase et al., 2022). Otherwise, PCFP is directly sent for sorting at an MRF (Bashirgonbadi et al., 2022). Improving the waste management infrastructure of flexible packaging waste could effectively contribute to reaching higher recycling rates (Lase et al., 2022). To improve the current collection and sorting systems and, thereby, ultimately achieve higher recycling rates of EoL plastics, a range of innovations and practices have been proposed.

5.2.2 Innovations and practices for improved collection and sorting

One main hurdle to improving collection, sorting and recycling capacity in Europe are the varying infrastructure and practices between and even within Member States. On the one hand, adapting the collection and sorting systems to the regional context has certain advantages, since local circumstances vary per region. On the other hand, however, such a fragmentation can lead to lower recycling rates, not least because of lack of household compliance for pre-sorting due to confusing rules (Burgess et al., 2021). To improve current waste management, it may be beneficial to take into account both of these aspects; harmonising collection and sorting systems across Europe, while adapting specific solutions to geographic and socioeconomic conditions (Crippa et al., 2019).

One option to harmonise and optimise current collection schemes on a more systemic level, while allowing for regional differences in collection, is to improve mixed waste sorting (MWS). MWS complements existing waste management by recovering recyclable materials from residual waste not captured by separate collection or DRS. While DRS can ensure high collection rates in the countries where they are implemented by providing financial incentives to consumers to contribute to effective sorting, they are currently mostly limited to beverage containers (TOMRA, 2022). Separate collection is relevant to separate different post-consumer EoL products, such as paper, glass packaging, light-weight plastic packaging, textiles and organic waste. Depending on local circumstances, it can be crucial to support recovery of recyclable materials (Bapasola et al., 2023). Still, these systems are insufficient to significantly increase recycling rates on their own (TOMRA, 2022).

Moving from broader systems to technologies, innovation is key for improving collection and sorting rates as well as the quality and yield of recycled plastics. Spectroscopic methods, such as via infrared, can increase accuracy in automated sorting. Tracer-based sorting can recognise different types of polymers, as well as different classes of compounds, which may increase sorting performance. Technologies supported by AI and robotics could improve both source separation and centralised sorting systems, especially when combined with optical or other sorting technologies (Crippa et al., 2019). Sorting technology is continuously evolving and becoming more granular, the limiting factor to innovation often being economic viability rather than technical feasibility.

5.2.3 Best practice examples

While there are countries and collection facilities across Europe with high collection rates, currently no country is set to reach the PPWR targets for packaging recycling (Bapasola et al., 2023). Even high-performing EU Member States, such as Belgium and Germany, could profit greatly from the introduction of mixed waste sorting (Bapasola et al., 2023). Belgium and Germany both have a long-standing separate collection system for packaging recycling.

In Belgium, two producer responsibility organisations for household waste and for commercial and industrial waste, respectively, organise the collection system. The 'blue bag' is used across the country and was expanded from plastic bottles to include other rigid plastic packaging, as well as films between 2019 and 2021 (Bapasola et al., 2023). In Germany, there are two separate waste collection systems for plastic packaging. Through the Duales System Deutschland (DSD), which has introduced the 'yellow bag', household packaging and non-packaging waste are disposed for commingled collection (Picuno et al., 2021). PET bottles are collected through a high-performing DRS for plastic, glass and metal beverage containers (Bapasola et al., 2023). As a result of that system, Germany reports a collection rate for plastic packaging of 75 % (Herrmann et al., 2021). Nevertheless, incorrect separation by consumers remains a challenge, as organic residues contaminate recyclables and plastic packaging disposed with municipal solid waste is lost for the recycling system (Herrmann et al., 2021).

A few mixed waste collection plants have been established across Europe. A case study conducted at the AVR residual sorting plant in the Netherlands suggests that the introduction of mixed waste sorting to the existing system could significantly improve collection rates. Around 12 times the amount of plastic for recycling was captured with the implementation of the new system (TOMRA, 2022). Another study compared different collection systems in Norway, equally suggesting potential benefits from recovering otherwise lost plastics from residual waste. Here, mixed waste sorting increased the collection rates more than twofold when compared with pick-up, drop-off and optibag systems (Deloitte, 2019).

5.2.4 Benefits of improved collection and sorting for mechanical and chemical recycling

Increasing the share of collected and sorted EoL plastics would benefit mechanical recycling. Increased and improved collection would increase the share of plastic waste that is reprocessed, thus securing feedstock for mechanical recycling. Better sorting could improve yields and output quality for mechanical recycling. Plastics packaging, for example, is often highly contaminated with adhesives, organic residues, ink and other materials. These are blended in the recycling process and pose health risks for consumers, especially for food contact materials (Kol et al., 2021).

At the same time, chemical recycling technologies would also benefit from improved collection. Access to sufficient and stable feedstock is often vital to sustain chemical recycling operations, as a certain scale is needed to make them viable (Qureshi et al., 2021; Kusenberg et al., 2022a). While technological innovations for gasification and pyrolysis may reduce dependency on intense sorting (Dogu et al., 2021), waste streams in practice still need to be relatively clean to ensure that operations do not get contaminated and achieve sufficient yield (Hann & Connock, 2020). Some contaminants[45] can cause damage to the CR infrastructure when they are released during the process of polymer decomposition. They also decrease the quality of pyrolysis oil, making further upgrades necessary before it can be used as a substitute for naphtha in a steam

[45] Such contaminants include nitrogen, oxygen, halogens and metals, originating from certain polymers, additives, inks and organic and inorganic residues (Kusenberg et al., 2022b).

cracker (Kusenberg et al., 2022a; 2022b). Therefore, advanced pre-treatment of plastic waste could considerably improve feedstock quality for chemical recycling.

The discussion above highlights that there is still a lot of scope for improvement of both practices and technologies of plastic waste collection and sorting. Improved collection and sorting have the potential to divert waste streams from landfill and incineration. Mixed waste sorting, in addition to separate collection and DRS, is crucial to recovering recyclable plastics otherwise lost to incineration or landfill. Capturing these recyclables has the potential to secure the large levels of feedstock required for cost-effective and functioning chemical recycling. At the same time, higher quality feedstock, as assured through better sorting of polymers, is relevant for both mechanical and chemical recycling. Harmonisation of waste management schemes across regions and Member States may be an important step towards higher collection and recycling rates.

5.3 ADDRESSING TRANSPARENCY CHALLENGES AND COMPARABILITY OF LCA FOR CHEMICAL RECYCLING TECHNOLOGIES

While chemical recycling technologies are still at the development stage with only limited capacities available at the moment across the EU, a debate has been launched about how far they can reduce the environmental impacts associated with plastic waste generation. This is shown by the growing number of academic articles and studies published during the last few years that assess the impacts of chemical recycling processes or compare them with other waste treatment options. This section summarises some of the key messages emerging from these assessments.

In assessing the potential environmental benefits of CR technologies it is important to understand the impact of different CR processes due to the use of chemicals, enzymes or heat treatment to depolymerise EoL plastics. Potential resulting environmental pressures of these technologies can then be weighed against their benefits – such as closing the plastic loop, reducing the reliance on virgin materials and avoiding emissions during potential alternative EoL treatments like incineration. To identify the appropriate plastic waste management strategy, material recovery must be maximised and (environmental) process impacts minimised (Garcia-Gutierrez et al., 2023).

Life Cycle Assessment (LCA) is a prominent tool to investigate environmental impacts of products, services and processes throughout their life cycles. LCA has been applied to different treatment options for plastic waste by modelling impacts such as carbon emissions, resource use and energy consumption. Existing LCAs, however, differ in aspects like underlying assumptions, input data used and technologies and waste treatment options analysed. These differences often make a direct comparison of studies challenging. Nevertheless, a general picture emerges.

5.3.1 Comparison of mechanical and chemical recycling

As raised by many studies and during the CEPS' roundtable meetings, CR should only serve as an EoL treatment option for plastic waste that currently cannot be treated via mechanical recycling[46], or where the necessary output quality cannot be achieved via the mechanical route. In addition, and as pointed out by the Joint Research Centre's recent study, due to data availability limitations it is currently unclear whether there are instances where chemical and mechanical recycling would compete for the same feedstock (Garcia-Gutierrez et al., 2023). These points should be kept in mind when attempting to make comparisons between the environmental benefits of mechanical and chemical recycling.

That said, most studies that directly compare chemical and mechanical recycling routes conclude that mechanical recycling usually performs better environmentally than chemical recycling technologies (Davidson et al., 2021). For example, an LCA by Oeko-Institut finds that mechanical recycling of plastics performs better than pyrolysis since, owing to its relatively low carbon efficiency, more than half of the carbon in plastics is lost in the modelled pyrolysis process and would thus need to be replaced by virgin polymers (Möck et al., 2022). Still, such results should not be simplified and regarded in isolation of other factors. When comparing those two recycling routes, both the output quality – including the need for food-grade or virgin-grade recycling in some applications - and the required feedstock quality and composition should be taken into account.

Moreover, carbon emissions are not the only relevant impact category since there are a range of different environmental impacts to consider when choosing the most appropriate management option for a certain waste stream. According to a recent study by the Joint Research Centre (see Garcia-Gutierrez et al., 2023), establishing a clear ranking of chemical and mechanical recycling technologies in terms of environmental performance is challenging and would require improved knowledge about the complex plastic waste streams. Furthermore, it is currently unclear whether chemical and mechanical recycling would even compete for the same feedstock (Garcia-Gutierrez et al., 2023). As has been highlighted in previous Sections, a combination of both routes, instead of competition between them, can lead to significant savings in GHG emissions in comparison to virgin plastic production (Möck et al., 2022). Given this positioning of CR within the waste hierarchy, a comparison of CR technologies with waste management options other than mechanical recycling, i.e. incineration and landfilling, may be more useful to determine the potential environmental benefits from CR (Ozoemena & Coles, 2023).

[46] See, for example, Davidson et al., 2021, Möck et al., 2022, SYSTEMIQ, 2022, Lase et al., 2023 and Ragaert et al., 2017.

5.3.2 Comparison of chemical recycling with incineration and landfilling

With regards to climate change impacts, there is a general consensus in the literature that the different CR technologies appear to perform better than incineration (Garcia-Gutierrez et al., 2023; Russ et al., 2020; Viveros et al., 2022). While energy recovery from incineration of plastic waste leads to some emission savings, these are not sufficient to outweigh the burdens from emissions generated by combusting fossil carbon. Similarly, whereas landfilling in theory may store carbon contained in plastics instead of emitting it into the atmosphere through incineration (Viveros et al., 2022; Davidson et al., 2021), carbon emissions from landfills occur at a later stage due to plastics degradation over time (see Section 2.3). Landfilling is commonly regarded as the least favourable waste management route from an environmental point of view, as it prevents valuable plastics in the stored waste from being used further in a circular economy (Viveros et al., 2022).

Moreover, with increasing decarbonisation of the electricity mix in the future, the benefits of CR in comparison to incineration are expected to increase (Garcia-Gutierrez et al., 2023; Russ et al., 2020; Voss et al., 2021). Assuming a higher share of renewables in future electricity production, credits for energy recovery from incineration will decrease and relative impacts will rise. At the same time, the environmental advantages of CR are likely to increase (Viveros et al., 2022), underlining the validity of the EU waste hierarchy (Garcia-Gutierrez et al., 2023).

The advantages of CR compared to incineration are also evident for other impact categories, such as ozone depletion (Garcia-Gutierrez et al., 2023). When it comes to fossil resource use, CR performs better than incineration and landfilling, as less virgin material has to be sourced than in a linear system (Garcia-Gutierrez et al., 2023; Russ et al., 2020).

It should be kept in mind, however, that there are individual impact categories where incineration can perform better than CR. This applies to acidification, particulate matter and eutrophication, at least under the current EU energy system (Garcia-Gutierrez et al., 2023; Russ et al., 2020). For some substances emitted to the air, current incinerators achieve lower emissions than the EU average energy production mix, which has a high share of coal, nuclear and heavy fuel power plants. With a cleaner energy mix, however, the gap between recycling and energy recovery will increase in favour of recycling.

5.3.3 Existing gaps

While an increasing number of LCAs has been conducted on different CR technologies, there are still several challenges to overcome to accurately estimate the impacts of these emerging technologies. Common issues refer to a lack in transparency, comprehensiveness and comparability of LCAs (Keller et al., 2022). LCAs often do not transparently disclose underlying assumptions and boundary conditions, limiting their comparability with other studies. Another challenge is a general lack of relevant data. For example, there is little data available on the characteristics and origin of waste inputs to recycling, which is crucial for identifying the appropriate waste management option (Garcia-Gutierrez et al., 2023). Moreover, results of LCAs strongly depend on the technologies and concrete waste streams analysed. Therefore, unique aspects for each investigated CR pathway should be considered when conducting the analysis. In general, while some CR technologies (such as pyrolysis and gasification) offer lower climate change benefits, they are also able to treat a broader range of input feedstock in comparison with other technologies, such as chemical depolymerisation, physical and mechanical recycling (CE Delft, 2020).

Most studies rank mechanical recycling as having the lowest environmental impact, followed by CR, incineration and finally landfilling. Crucially, CR can enable high-quality recycling of low-quality waste streams otherwise not suitable for mechanical recycling, and divert them from incineration and landfill, thereby keeping more plastic in the loop. It has the potential to contribute to reducing carbon emissions from waste treatment and its relevance is expected to increase with a cleaner energy mix in the future (Voss et al., 2021). Nevertheless, given the current lack of data and transparency in LCAs, such results need to be interpreted with caution. While there is scope to improve LCA assessments and address the existing data gaps, further attention should be given to solutions higher up the waste hierarchy, such as reuse where possible and overall waste prevention. The potential of CR lies in its complementarity with other waste treatment options.

5.4 ALLOCATING RECYCLED CONTENT

Mandatory recycled content targets for plastics will be established in EU Regulation for several sectors (such as packaging, automotive, electronics), in order to '[…] increase uptake of recycled plastics and contribute to the more sustainable use of plastics' (European Commission, 2020, p. 9). Meeting these targets requires having a transparent view about the proportion of recycled content in new plastics. However, tracing recycled content for some chemical recycling processes is not always as straightforward as it is for mechanical recycling (Broeren et al., 2022).

Tracking recycled content is particularly challenging for pyrolysis and gasification (i.e. thermal depolymerisation), at any process stage that has waste (-derived) as well as virgin inputs and produces multiple outputs. These technologies break down polymers into their building blocks (i.e. hydrocarbons) to produce certain chemicals that then can be used in chemical production (Broeren et al., 2022). Blending and co-processing recycled with virgin substances and materials in existing petrochemical sites has economic advantages, if the former fulfil similar technical requirements, while at the same time it poses challenges when it comes to tracing recycled content throughout the process (Ellen MacArthur Foundation, 2019). As a result, there is a need for a method to allocate recycled content to final products in a transparent and reliable way, to make valid and credible sustainability claims to consumers and to downstream supply chain players. One potential approach to addressing this particular challenge is using the mass balance chain of custody model.

5.4.1 Chain of custody models

Mass balancing is one among several chain of custody models (CoC). CoC models are tools for tracing desired properties of materials and products throughout value chains. The nature of desired properties depends on the context; examples include origin, embedded raw material consumption, or attributes such as 'recycled', 'organic' and 'renewable'. The aim is to establish a link between inputs and outputs of production processes throughout value chains, in order to create transparency and trust regarding the proportion of the output materials' desired content (Ellen MacArthur Foundation, 2019; Broeren et al., 2022).

Several CoC methods exist to establish these links between input properties and output: identity preservation, segregation, controlled blending, mass balance, and book and claim. The three models commonly used in plastics recycling as defined by ISO 22095 are described in more detail here[47]:

In the **segregation model**, physical mixing of certified and non-certified material of the same commodity is not permitted. However, it does allow for a mixing of different certified inputs of various origins, as long as these are certified according to the same standards. Thereby, materials of different sources are kept physically separate but mixed within one common category (such as organic food). As a result, assurance is guaranteed that products stem from certified sources, even though physical traceability of one product back to its source may not be possible (ISO, 2020; ISEAL Alliance, 2016; Beers et al., 2022).

In the **controlled blending** model, certified and non-certified materials are mixed according to specified standards, resulting in known proportions of those blended properties in all batches of the final output (ISO 22095:2020; Beers et al., 2022).

Mass balance is a CoC model where materials with certain characteristics can be mixed with others in a co-processing procedure. The physical presence of these characteristics varies across the outputs, and on average only match the initial proportions of the input (ISO 22095:2020). There are different calculation and attribution methods within this model, which are described in more detail in the next Section.

[47] The two models at the end of the spectrum have been deemed inappropriate by stakeholders, according to a study by Eunomia (2022). In the **identity preservation** model, certified products from a certified site are kept separate from (certified or non-certified) products from other sources. The physical separation of materials and thus, characteristics, is maintained throughout the supply chain, thus allowing for the highest levels of traceability of desired (certified) content (ISO 22095:2020; ISEAL Alliance, 2016; Beers et al., 2022). In the **book and claim** model, the administrative record flow is not necessarily connected to the physical flow of material or product throughout the supply chain (ISO 22095:2020. This bookkeeping method provides the lowest levels of traceability of desired content among the CoC models. Credits for the desired property are bought from other companies lacking the market demand for these properties, irrespective of who actually sells the product with the physical presence of the property.

5.4.2 Mass balance

In the mass balance CoC model, the physical mixing of certified and non-certified inputs is allowed at any stage in the production process, as long as volumes of certified (i.e., recycled) input are controlled. Thus, volumes of output properties have to be claimed proportionally to the input, in order to establish the link between (mixed) inputs and outputs of the production system, while taking into account losses during the production process (Eunomia, 2021; ISEAL, 2016; Beers et al., 2022). This model is commonly used when the technical process does not allow for a separation of inputs, when it is not economically viable to do so or when the certified (recycled) input is very low compared to the virgin input. The latter is currently the standard case for most value chains in the chemical industry.

While the concept of mass balance is not new, its application in the chemical recycling of polymers is a relatively recent development aiming to support the scale-up of the nascent thermo-chemical recycling infrastructure (see Beers et al., 2022). Use of mass balance approaches in other sectors includes certified cotton, forestry, palm oil and aluminium, where the physical separation of certified and non-certified products is impractical (Beers et al., 2022; Ellen MacArthur Foundation, 2019; Broeren et al., 2022; ACC, 2021).

Regarding chemical recycling of polymers, mass balance can be applied to account for recycled inputs when they are blended with virgin materials, in order to allocate recycled content to specific outputs (Broeren et al., 2022). Not all recycling technologies[48] require an accounting method to keep track of recycled content in their processes. When chemical depolymerisation is used to directly reproduce the same polymers, authors argue that no mass balance is needed to determine recycled content (Broeren et al., 2022). If the recycled monomers are however blended with virgin monomers to produce new plastics or other downstream products, an accounting method is necessary to allocate recycled content to the output (Broeren et al., 2022). Mass balance is of particular relevance to thermal depolymerisation, where plastic waste is in many cases broken down to the hydrocarbon molecules stage and processed in large and highly integrated operations with mixed inputs and outputs.

Pyrolysis and gasification both produce basic chemicals (pyrolysis oil that can substitute fossil naphtha or syngas and methanol) that are typically blended with virgin fossil materials before being fed, for example, into a steam cracker, which then produces basic chemicals for a range of plastic and non-plastic products. Due to economic constraints and availability of recycled feedstock, those technologies rely on a mixture of feedstock, instead of recycled inputs only.

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[48] In the literature there is often a debate about whether solvent-based purification can be considered to be among the key chemical recycling technologies. In this report, however, we have not included it in the list of core chemical recycling technologies since it does not break down plastic waste further than the polymer stage, thus essentially not changing the chemical composition.

As such, it is very challenging to tell how much recycled content is part of which of the outputs of the chemical production process. This requires an accounting approach, such as mass balancing, to trace recycled content. However, this needs to be carried out in a transparent way, ensuring that the sustainability claims made are valid, and that recycled content claimed in the outputs does not exceed that of the inputs (Broeren et al., 2022).

Accounting methods

The ISO differentiates between two accounting methods for mass balance, the rolling average and the credit method (ISO 22095:2020). The **rolling average percentage method** is applied to account for fluctuating shares of recycled and non-recycled content in the input over a defined claim period. An average percentage of recycled content is claimed for the output over the claim period. The **credit method** allows producers to transform certified mass balanced output into a credit, in order to claim any amount of recycled content outputs proportional to the inputs, minus process losses and taking into account the allocation method (see next Section) (Eunomia, 2022). Some certification schemes also permit these credits to be transferred within a company under specific conditions. For example, **restricted credit transfer** between sites could apply under certain conditions, such as between identical products only (Eunomia, 2022). ISCC PLUS or REDcert2, for example, allows for credit transfer between different sites, if they are part of the same corporate group or joint venture (ISCC, 2023).

Allocation methods

Other variations in mass balancing stem from the specific allocation method applied. Generally, two different allocation methods can be distinguished: proportional allocation and non-proportional allocation. Non-proportional allocation can further be divided in polymers only, fuel use excluded, auto consumption excluded, and free allocation (Hann et al., 2022).

Proportional allocation attributes recycled content evenly to all outputs of the production process. If n % of recycled input is fed into the process, then each output stream is allocated n % recycled content credits. This approach is generally favoured by environmental NGOs. On the other side of the spectrum, **free allocation** allows for a high degree of flexibility in assigning recycled content to the production outputs. For example, outputs with a high market demand for recycled content could be assigned 100 % recycled content, while the rest of the outputs, where there is no market for recycled content, are claimed to have 0 % recycled content. **Polymers only**, **fuel use excluded**, and **auto consumption excluded** employ non-proportional allocation rules as well. Polymers only requires that only the recycled content in outputs directly related to polymer production can be freely allocated, whereas for fuel use excluded, the recycled content in all outputs except for fuels can be allocated freely (Broeren et al., 2022; Beers et al., 2022). In contrast, only the fuel that is burned for energy generation within the process is excluded in the **auto consumption excluded** approach (Hann et al., 2022). All of these different methods account for production losses. Recycled material that is lost during the production processes cannot be allocated to the output (Broeren et al., 2022).

While the fuel use excluded approach is consistent with existing legislation on recycling targets, according to a Eunomia study for the European Commission, ‘there are strong arguments for a stricter approach that considers the importance of focusing on the credibility of the system and not further reducing the physical links’ (Hann et al, 2022, p. 150). At the same time, some actors argue that less strict approaches can spur investment into CR technologies.

Credit units and calculation rules

Different physical properties can be measured to determine recycled content in inputs and outputs: mass allocation, carbon accounting and lower heating value. Calculations based on **mass** weigh the recycled and virgin volumes in the input and allocate the recycled output content based on weight, which is useful when virgin and recycled material are similar (Broeren et al., 2022). Using **carbon** as the credit unit, the carbon atoms in both feedstock streams are counted to determine the percentage of recycled content in the output, which is suitable when input streams have different compositions and the aim is to trace only the fraction which is to be recycled (Broeren et al., 2022). The **lower heating value** is a representation of the energy content of a material. Using it as a basis for calculations, recycled content is allocated to output based on the lower heating value of both fossil and recycled input material. This method is useful when input streams have highly variating compositions (Broeren et al., 2022). Since waste streams are complex and input compositions often vary in reality, some stakeholders argue for using the lower heating value as a basis for recycled content calculations (Ellen MacArthur Foundation, 2019). In contrast, NGOs have criticised both carbon and lower heating value due to their low levels of physical traceability of recycled content.

6. CONCLUSIONS AND POLICY MESSAGES

Although there have been EU policies in place to address plastic waste for decades, the emergence of the circular economy agenda brought forward a different vision for the management of plastics across their full life cycle. The call for a new approach in terms of how we produce, consume and dispose of plastics has led to the adoption of a dedicated EU strategy on sustainable plastics, a Directive focused on single-use plastic items and various voluntary commitments by multinational companies. While there is a general alignment of objectives by policy, civil society and industry actors, available estimates indicate that the EU will need to make great leaps in the coming years to improve the circularity of plastics.

It is estimated that the EU still produces about 30 Mt of plastic waste annually, of which only about one third is recycled while the rest is directed towards incinerators and landfills. There are also studies that place the amount of plastic waste generated across the EU at much higher levels[49]. Existing assessments about future plastic waste generation do not necessarily provide an improved picture. Specifically, different studies estimate that the amount of plastic waste generation in the EU could range between 28 and 34 Mt in 2030 and then between 30 and 44 Mt in 2050, depending on the level of ambition of adopted circularity policies. While packaging is expected to still account for the largest share of future plastic waste generation, other streams from the automotive and construction sectors are projected to make increasingly important contributions.

New collection, sorting and recycling systems will therefore need to come into play to enable a more sustainable EoL management of the complex plastic waste streams. Most studies assessed for this report conclude that a mix of solutions including both mechanical and chemical recycling technologies will likely be required to achieve optimal levels of recycling and to reduce the share of plastics sent to incineration and landfills to minimal levels. While high heterogeneity exists across the different assessments, even in the most conservative scenario, chemical recycling technologies are projected to expand compared to current levels and to play a complementary role to mechanical ones. The need for a mix of recycling solutions, combined with efforts aimed at increasing reuse and waste prevention was raised during the CEPS' stakeholder meetings.

Expanding plastics' recycling markets would require a policy environment that enables all recycling options while at the same time providing a level playing field between mechanical and chemical recycling technologies. Some areas where action by both policy and industry actors can support further growth of such markets are discussed below.

[49] See section 2.2.

Level playing field

While various chemical recycling plants and joint business collaborations have been recently announced across the EU, a policy environment providing clarity about the role of these technologies is currently under development. One particular area that has been subject to debate in the context of the requirements for single-use plastics and packaging and packaging waste is how to take into account chemical recycling in the calculation of recycled content targets. This is particularly relevant for thermal depolymerisation (pyrolysis and gasification) chemical recycling technologies, for which tracing the recycled content throughout the recycling process is complex. In this context, mass balance is a chain of custody model that is being discussed as a tool to support traceability. Different methods to allocate the recycled feedstocks and to communicate this information to the consumer have been extensively debated. **Defining an optimal system that balances all environmental and economic considerations will be challenging for the thermal depolymerisation options, but there is a need to provide certainty and to clarify how chemical recycling technologies can make recycled content contributions. Such a system would need to be based on clear and transparent rules and claims in order to provide certainty to the market, clarity to consumers and downstream supply chain players, and a level playing field between different recycling technologies.**

Another issue is whether certain provisions in the Waste Framework Directive would need to be re-assessed during future revisions to give room for all technologies, while prioritising the options that deliver the highest environmental benefits. Beyond the current definition of recycling included in the Directive that excludes the use of fuel, there are no further provisions at the moment for chemical recycling technologies. **As these technologies scale-up, there will be an open question on whether there is a need to provide clarity about their position in the waste hierarchy and in the existing recycling definition, or whether to allow for more flexibility.** Either way, increasing data availability will be key for this discussion since, as discussed in the recent study by JRC, establishing an environmental ranking of options for recycling technologies is currently very difficult due to data limitations. Besides environmental considerations, any ranking should take into account the different capabilities of chemical and mechanical recycling. **When choosing the optimal EoL management option, the complexity and level of contamination of input waste streams as well as the desired output quality should be considered.** With continued innovation, the capacity of the different recycling options to treat complex inputs or to produce high-quality (food-grade or contact-sensitive) outputs may improve, which suggests potential benefits of enabling some flexibility in the waste hierarchy.

Data availability

As documented in the literature and discussed during the CEPS meetings, there are still several data gaps related to chemical recycling and plastic waste recycling in general, which creates uncertainties in the public debate and complicates the discussion about the mix of solutions that can deliver the highest environmental benefits. When it comes to the impacts of chemical recycling technologies, there are uncertainties regarding the waste feedstocks to be used as well as the emissions and losses in the process. **There is a general lack of primary data and complete databases for chemical recycling processes which can also be attributed to the fact that commercial deployment of these technologies has only just began. Data limitations, however, extend beyond the effects of chemical recycling technologies to the overall plastic waste composition and generation across the EU.** This is also due to the existence of a very diverse set of plastic products and applications which makes such assessments very complex. While the spotlight is often on the packaging waste stream and on single use plastic items, other streams from buildings and automotive for which available data face further limitations, may increasingly come into play in the future.

In addition, the focus on existing assessments has been on ranking different EoL treatment options, with most studies concluding that from a climate change point of view chemical recycling performs better than incineration but has higher emissions compared to mechanical recycling. **Given the variety of approaches adopted by the growing body of LCA studies assessing different EoL plastic treatment options, the publication of methodology guidelines or handbook for such LCAs would support transparency and comparability in terms of the system boundaries for datasets**. In addition, **to enable a more informed debate about circularity in the plastics sector, a move towards more integrated assessments is needed, taking into consideration what waste streams exist and how they can be treated in the most environmentally friendly way**. This arguably requires higher transparency with regard to the impacts of plastics recycling and improved availability of data across the various segments of the plastics supply chain. This challenge can be higher for plastics compared to other materials given the myriad of plastics applications in nearly every sector of the economy. **The introduction of common frameworks for data sharing across actors along the various lifetime stages of plastics can support improved data availability**. **Policy can be a facilitator through the development of principles and common standards for data sharing that also take into account confidentiality concerns**. This could link to similar efforts currently being undertaken in different sectors (such as batteries, electronics and textiles) with the introduction of Digital Product Passports.

Innovation and ecodesign

A position shared by various experts and existing global assessments for the plastics sector is that there is still significant room for innovation both at the design and EoL management stage of plastics. Innovation on these two fronts would benefit both mechanical and chemical recycling. At the design stage, efforts to support an increased lifetime of plastics should be prioritised, followed by actions to improve recyclability of EoL plastics. Examples of the latter include designing product components that can be easily separated, reducing where possible the multi-layers of polymers and reducing the use of additives and substances. How to incorporate recycled content in new plastic products is a further aspect that needs to be considered at the design stage. **At the EU level, design requirements can be introduced or strengthened through the horizontal Ecodesign for Sustainable Products Regulation or legislations targeting product groups such as the Packaging and Packaging Waste Regulation and the upcoming revised End-of-Life Vehicles Directive. Such requirements should be introduced in a coherent way providing a clear set of harmonised requirements for different product groups**. Given the complexity of different product groups and types of plastics, the resource efficiency and climate change benefits of different design options should be carefully assessed.

Moving to the EoL management stage, the varied plastic waste management systems across the EU coupled with the continuous development of new plastic applications pose challenges for collection, sorting and recycling – both mechanical and chemical. **Harmonising and increasing collection and sorting systems while taking into account the local circumstances can support improved recycling of EoL plastics. Innovations on sorting and collection technologies are also key and would need to be continuously supported through innovation funding by the EU and Member States.** Examples of such innovations include combinations of sorting systems and digital tools for tracking polymers and digital product passports providing information about the plastic products and the substances included.

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Annex

This report has been supported by:

- BASF
- Neste
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- TOMRA

Stakeholder events and speakers

Chemical recycling: What contribution can it make to the EU's unfolding industrial transformation?

- **Kevin M. Van Geem,** Professor, Laboratory for Chemical Technology, Faculty of Engineering, Ghent University

The role of life cycle assessment for chemical recycling deployment and policy decision-making (online)

- **Roh Pin Lee,** Head of Technology Assessment Division, Institute of Energy Process Engineering & Chemical Engineering, TU Bergakademie Freiberg
- **Florian Keller,** Research Associate, TU Bergakademie Freiberg
- **Martijn Broeren,** Senior Researcher, CE Delft
- **Maiju Helin,** Head of Sustainability and Regulatory Transformation, Neste
- **Christian Krüger,** Circular Economy Expert, BASF
- **Lauriane Veillard,** Policy Officer, Zero Waste Europe

How to develop an ecosystem of recycling technologies for different plastic feedstocks?

- **Daniel Schwaab,** SVP, Strategy & Sustainability Recycling, TOMRA
- **Adela Putinelu,** Head of Policy, Plastic Energy

Allocating recycled content in plastics: What approach is needed?

- **Jan Henke,** Director, International Sustainability and Carbon Certification (ISCC)
- **Stefan Gärtner,** Circular Economy Expert, International Sustainability and Carbon Certification (ISCC)
- **Janek Vahk,** Climate, Energy and Air Pollution Coordinator, Zero Waste Europe
- **Lauriane Veillard,** Chemical Recycling and Plastic-to-Fuels Policy Officer, Zero Waste Europe

THE FOLLOWING ORGANISATIONS PARTICIPATED IN THE MEETINGS:

- BASF
- CE Delft
- ChemSec
- Dow
- ECOS
- Ghent University
- International Sustainability and Carbon Certification (ISCC)
- Lux Research
- Neste
- Oeko-Institut
- Plastic Energy
- Plastics Europe
- Plastics Recyclers Europe
- Pryme
- Pyrowave
- REMONDIS
- SABIC
- Shell
- Styrenics Circular Solutions (SCS)
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