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Waste-to-energy is compatible and complementary with recycling in the circular economy

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Abstract

This paper reviews the role of conventional waste-to-energy, i.e. incineration of (mainly) municipal solid waste with energy recovery, in the circular economy. It shows that, although waste-to-energy figures on a lower level in the European waste hierarchy than recycling, it plays, from an overall sustainability point of view, an essential, complementary and facilitating role within the circular economy. First of all, waste-to-energy combusts (or should combust) only waste that is non-recyclable for economic, technical or environmental reasons. This way waste-to-energy is compatible with recycling and only competes with landfill, which is lower in the waste hierarchy. Furthermore, waste-to-energy keeps material cycles, and ultimately the environment and humans largely free from toxic substances. Finally, waste-to-energy allows recovery of both energy and materials from non-recyclable waste and hence contributes to keeping materials in circulation. These arguments are elaborated and illustrated with many examples. This paper also points out the pitfalls of a circular economy if it merely focuses on material cycles, disregarding economic, environmental, social and health aspects of sustainability.

Keywords

Waste-to-energy, circular economy, waste, recycling

1. Introduction

The term "circular economy" (CE) expresses a new concept that focuses on maintaining the value of products, materials and resources in the economy for as long as possible, and on minimizing waste generation (European Commission, 2015a). The European Commission (2015a) states that the CE will boost the EU's competitiveness by protecting markets against scarcity of resources and volatile prices. Furthermore, the transition towards a CE is believed to create new business opportunities and jobs and will imply innovative, more efficient ways of producing and consuming. It is also presumed that a CE will save energy and will avoid irreversible damages to the environment and to society caused by the consumption of resources at a rate that exceeds the earth's capacity to renew them (European Commission, 2015a).

The CE is a valuable concept with commendable objectives, which are however not always obvious to realize in the present economic and societal context. Of the proposed actions for the transition to a CE, most concrete are the legislative proposals related to waste, amending (1) Directive 2008/98/EC on waste, which is the cornerstone of the European waste policy and legislation, (2) the packaging and packaging waste directive (Directive 94/62/EC), (3) the landfill directive (Directive 1999/31/EC), the directives (4) on end-of-life vehicles (2006/53/EC), (5) on batteries and accumulators and waste batteries and accumulators (2006/66/EC), and (6) on waste electrical and electronic equipment (Directive 2012/19/EU).

Fig. 1 graphically represents the EU waste hierarchy, which is the basis of the EU waste policy and legislation and a key to the transition to a CE. The waste hierarchy lists waste prevention higher than, in decreasing order of priority, reuse, recycling, recovery and disposal, as a way to optimize resource efficiency and minimize environmental effects of waste management. Conventional waste-to-energy (WtE), i.e. incineration of (mainly) municipal solid waste with energy recovery figures on the fourth level of the waste hierarchy (Fig. 1), on a higher level than disposal (i.e. landfill or waste combustion with insufficient energy recovery), but on a lower level than waste prevention, preparing for re-use, and recycling. Whereas it is widely recognized that disposal as a waste treatment option should be avoided, the first three options seem to fit perfectly in the concept of the circular economy. For WtE, the option in between, it is unclear whether it maintains value in the economy for as long as possible or not. Indeed, many stakeholders in the political and societal debate consider WtE as a competitor for re-use and recycling and don't acknowledge its role in a CE, whereas others, on the contrary, consider WtE as indispensable to achieve the CE targets (Malinauskaite et al., 2017).

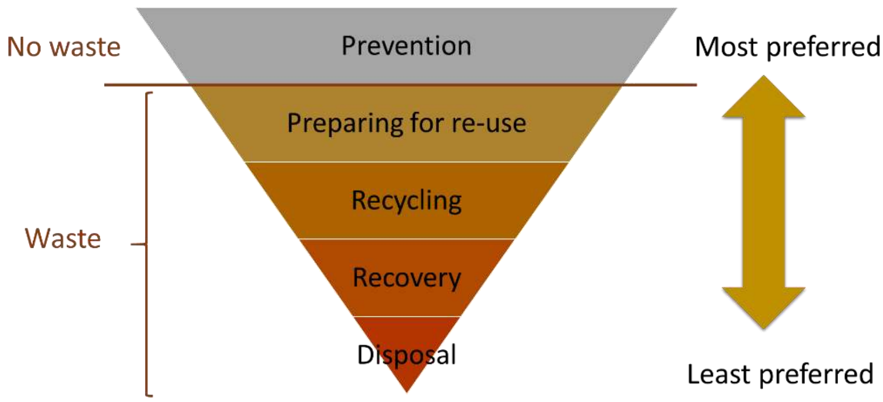

Fig. 1: Schematic representation of the EU waste hierarchy as laid down in Directive 2008/98/EC

The hierarchy can be applied with some flexibility, which means that e.g. for economic, technical or environmental reasons, recovery can be preferred over recycling. This can be the case for e.g. wastes containing toxic substances (European Commission, 2017). Merrild et al. (2012) showed that, although recycling of materials from municipal solid waste (MSW) is commonly considered to be superior to other waste treatment alternatives, for material fractions with a significant energy content this might not be the case if the treatment alternative is WtE with a high energy recovery rate. The authors showed by means of LCA and assuming high-performance technologies for material recycling as well as for waste incineration, that for paper, glass, steel and aluminum, recycling has environmental benefits over WtE. For cardboard and plastic the results were less clear and depended on the level of energy recovery at the WtE plant and on the considered system boundaries and impact categories. After comparing 222 LCA studies, also Laurent et al. (2014) concluded that the environmental impact of MSW treatment systems largely depends on the scope and local conditions such as waste composition, energy recovery rate and avoided emissions and that no general priority in waste management technology can be put forward. More recently, Rigamonte et al. (2018) clearly pointed out that the quality of the materials obtained from recycling is often lower than that of virgin materials so they can only replace virgin materials to a certain, sometimes limited extent. However, in most LCA studies conducted so far, full material substitution is assumed, unrealistically favoring recycling over waste-to-energy. In accordance, Haupt et al. (2018) showed that the credits from material substitution and the assumed energy efficiency of waste-to-energy plants are key variables in LCA of MSW treatment schemes. They pointed out that in countries with extensive separate MSW collection schemes in place, the environmental benefit of further increasing recycling rates is limited compared to the effect of increasing energy efficiency in WtE plants.

From these considerations the obvious overall question is: “Does WtE still have a valuable role to play in a CE?”. The European Commission’s answer to this question is rather prudent, recognizing that WtE can play a role in the transition to a CE (European Commission, 2017), but that, in order to ensure the full environmental and economic potential of a CE, WtE should coincide with higher levels of prevention, re-use or recycling.

This paper shows, by means of five key arguments, that WtE, which to date fulfills an essential role in sustainable waste management, will continue doing so in a CE. Further in this paper "WtE" refers to incineration of MSW and by extension of hazardous waste with energy recovery, which is the most applied type of WtE. Other types of WtE such as gasification or anaerobic digestion are not included in the scope of the arguments below:

1. WtE combusts (or should combust) only waste that is non-recyclable for economic, technical or environmental reasons.
2. WtE can be complementary and compatible with recycling.
3. WtE, and by extension high temperature combustion e.g. in a dedicated rotary kiln, is essential to keep toxic substances out of materials and out of the environment.
4. WtE allows recycling of a large part of the metals and inorganics in the waste.
5. WtE can generate energy with high efficiency, which can partly feed the CE.

2. The role of WtE in a CE

2.1. WtE combusts (or should combust) only waste that is non-recyclable for economic, technical or environmental reasons

The limitations of materials recycling were recently made clear by China's ban on the import of low quality recyclables in the frame of a broader environmental and health protection policy and in order to stimulate domestic recycling. For the last two decades, many European and other industrialized countries have been shipping their low quality recyclables i.e. waste not suitable for local recyclers, such as mixed and soiled plastics, to China or other developing countries where they are further processed or used as cheap fuel, often in rudimentary circumstances (Brooks et al., 2018). Although this practice ensured high "recycling rates" for the exporting countries, it ignored the recyclable's quality problem and the environmental and human health effects of the recycling at its destination. China's ban has made clear that the right target for the recycling industry is not to achieve higher recycling rates, but to produce more high quality recyclables (ISWA, 2018). It also shows that the current technical and economic limitations of recycling should be recognized. Indeed, high quality recycling at reasonable cost is not always feasible e.g. due to insufficient sorting or separation of waste streams at the source (Singh and Ordonez, 2016). Some materials also suffer from degradation upon recycling, ultimately limiting the number of times they can be recycled. A typical example is the shortening of fibres in paper or textile (Delgado-Aguilar, 2015; Ignatyev et al., 2014). Finally, techniques to produce a recycled product or material meeting the quality and/or functional standards or requirements of the consumer at a competitive cost may simply not yet exist (Ignatyev et al., 2014).

The technical and economic limitations of recycling may further be illustrated by the case of household packaging waste, for which well-established separate collection systems are in place in many regions, and efficient sorting processes ensure relatively pure material streams with high recycling potential. Separate collection at the source is indeed essential to obtain *clean* waste streams for which recycling is technically and economically feasible (Janz et al., 2011; Singh and Ordonez, 2016). In Flanders, the Northern region of Belgium, for instance, source separation and collection of household packaging waste is well established. Household packaging waste is collected separately as PMD waste i.e. a mix of Plastic bottles and flasks, Metal packaging such as tins and cans, and Drink cartons. In highly automated sorting centres, the separately collected PMD waste is further separated into four fractions: a plastic fraction (43% of the total mass collected), a metal fraction (28%), drink cartons (11%) and a sorting residue (17%), which is sent to WtE for final treatment (Vervaet et al., 2016). In 2016, 66 707 t of plastic bottles and flasks was recycled through this scheme, i.e. 75.3 mass % of the 88 572 t of plastic bottles and flasks brought on the market. However, this high recycling rate has to be put in the right perspective. Indeed, it can only be obtained because the collected PMD waste is restricted to a limited number of rather easily separable and recyclable material types. For this reason, plastic foil, bags and yoghurt pots, which are composed of polymer types other than polyethylene terephthalate (PET) and high-density polyethylene (HDPE), are for the time being excluded from the PMD waste, because their separation into individually recyclable fractions is economically not yet viable. This incompatible plastic packaging is therefore collected as residual waste and is treated in WtE plants.

This case of Flanders shows that high recycling rates of specific MSW fractions can be established by strict, source-separated collection. However, separation of waste composed of different, difficult to separate materials, such as e.g. food packaging or hygienic waste (diapers, paper tissues, etc.) into recyclable fractions with an acceptable market value seems technically and economically impossible (Singh and Ordonez, 2016). Such non-recyclable waste is typically collected as residual waste, which is, in line with the waste hierarchy, preferably incinerated in WtE plants. This coincides with less waste going to landfill. According to Grosso (2016), there is no generally applicable answer to the question "To which extent should material recycling be encouraged vs. energy recovery?", because the answer is subject to specific local conditions. At the current state of knowledge good quality waste materials (e.g. with minimal contamination), including high grade plastic items, such as PET and HDPE bottles, glass, metals, clean paper, and cardboard, are better recycled. For such materials one should aim at "real" recycling, yielding secondary materials of the same quality level than virgin materials (Grosso, 2016). Lower quality waste materials, such as mixed plastic polymers, are however commonly "downcycled" to lower quality materials (Singh and Ordonez, 2016). In this case, WtE treatment producing thermal and electric energy becomes as a competitive option (Grosso, 2016).

2.2 WtE can be complementary and compatible with recycling

Fig. 2 shows how MSW was treated in different European countries in 2015. A distinction is made between “recycling, composting and digestion”, “waste-to-energy” i.e. incineration with sufficient energy recovery according to the R1 equation given in annex II of Directive 2008/98/EC (R1 value ≥ 60% if in operation before 01/01/2009 or R1 value ≥ 65% if permitted after 31/12/2008) and “disposal” consisting on the one hand of incineration without or with insufficient energy recovery and on the other hand of landfill. A first observation is that even within Europe, large regional differences exist between “Western and Northern” European countries and “Eastern and Southern” European countries. The former have in general a lower disposal rate (in %) than the EU 27 average, which goes along with higher “recycling, composting and digestion” and “waste-to-energy” rates. In fact, in 2015 Germany and Flanders already reached the EU’s circular economy target for “re-use and recycling” of MSW, i.e. 65%, which is only to be reached by 2030 (European commission, 2015b). In several of these Western European countries such as Switzerland, Germany, Sweden, Belgium (including Flanders), Denmark, the Netherlands, Austria, and Norway a MSW landfill ban is effectively in place; they landfill less than 3% of their MSW. In contrast, the Eastern and Southern European countries have generally higher disposal rates than the EU 27 average, which goes along with lower “recycling, composting and digestion” rates and with lower or even zero “waste-to-energy” rates.

These findings based on Fig. 2 are in line with a recent study showing that the dedicated WtE capacity for MSW is unevenly distributed over EU member states (Wilts et al., 2017). Only 6 countries (i.e. France, Germany, Italy, the Netherlands, Sweden, and the UK) account for more than 75% of the total EU’s WtE capacity. The highest per capita WtE capacity is found in Sweden and Denmark, followed by the Netherlands, Austria, Finland and Belgium. In contrast, in most the Southern and Eastern European countries not much dedicated WtE capacity yet exists, hence these countries still predominantly rely on landfill (European commission, 2017).

The major drivers for high recycling and incineration rates in “Northern and Western” Europe compared to “Southern and Eastern” Europe are:

(1) Socioeconomic. France and the countries to the left of France in Fig. 2, except Estonia and Slovenia, are all “wealthy” countries figuring in the top 15 of the GDP/capita list of European Countries (International Monetary Fund, 2018). They have the financial means to support the Capex and Opex of WtE and of installations for recycling, composting and digesting, and have the technology, and the skilled workforce. The good position (low landfill) of Estonia and Slovenia, and the relatively bad position (compared to their ranking according to GDP/capita) of e.g. Malta, Cyprus and Spain (Malinauskaite et al., 2017) shows that this is not the only reason.

(2) Environmental awareness. In e.g. Sweden and Germany, the environmental awareness movement has already for several decades been very active, which is not the case in e.g.

Romania or Poland. High environmental activity motivates governments towards sustainable waste management.

(3) Availability and cost of land suitable or landfill, which is influenced by population density, wealth and production and consumption patterns of the population.

(4) Membership of the EU and for how long is also a driver, as member countries should comply with the European waste legislation.

Fig. 2 also clearly shows that countries with high WtE rates also have high recycling rates. WtE and recycling are thus complementary and compatible, and advanced waste management systems (as applied by the countries at the left in Fig. 2) rely on both. This does however not preclude variation of the relative importance from country to country, depending on the local waste policy and priorities and the occurrence of incinerators and installations for separate collection, sorting, and recycling of waste (see below on overcapacities). As it turns out, countries with the highest rates of WtE, e.g. Denmark, Norway, and Sweden, all incinerating at least 50 % of their waste (Eurostat, 2017), also tend to have high rates of recycling and composting of organic materials and food waste. But one can argue that, if WtE would not be applied at such large scale, these countries with an environmentally conscious image would have even higher recycling rates. Germany, for example, incinerates less than 35% of its MSW and recycles more than 65%, a considerably better recycling rate than the 40-plus % of Scandinavian countries (Seltenrich, 2013).
It can also be argued that, in principle, if WtE complies with the requirement of only combusting waste that is non-recyclable for economic, technical or environmental reasons (see 2.1), no competition exists between WtE and recycling. However, as discussed in the EC's communication on the role of WtE in the circular economy (European commission, 2017), it is essential to avoid future WtE overcapacities, as this is critical to avoid potential economic losses due to 'stranded assets', and might hamper an increase in recycling rates. Again one can distinguish between Member States with low or non-existent WtE capacity and high reliance on landfill (typically Eastern and Southern European countries) and Member States with high WtE capacity (typically Western and Northern European countries). The former should prioritize the development of separate collection systems and effective recycling infrastructure, and thus divert waste from landfill (European commission, 2017). This is, from a climate perspective, particularly urgent for biodegradable waste, as diverting biodegradable waste from landfill reduces methane emissions (Jeswani and Azapagic, 2016; Themelis and Ulloa, 2007). It was shown that diverting 1 ton of biodegradable waste from a landfill towards anaerobic digestion to produce biogas and fertilizer can prevent up to 2 ton of CO_2 equivalent emissions (Bernstad and Jansen, 2012). Indeed, for the separately collected organic fraction of MSW such as kitchen and garden waste, anaerobic digestion with recycling of the digestate as fertilizer could represent an attractive management option (Malinauskaite et al., 2017). Furthermore, the expansion of WtE capacity for treatment of non-recyclable waste should always be considered from a long term perspective, taking into account aspects of waste availability (which can change as

a result of future development of separate collection systems), capacity for co-combustion in industrial processes such as cement production and planned capacity expansion in neighboring countries or regions (European commission, 2017). If expansion of WtE capacity is justified, obviously the new to build plants have to use state-of-the-art technology with the highest possible energy and material efficiency and have to comply with the legislation in place e.g. concerning air emissions (European commission, 2017).

When both WtE and recycling techniques are in place, in the present technological and economic context, a large part of recycling can only stand thanks to legal obligations, taxes, as well as sometimes significant subsidies (in various forms, typically supported by the Extended Producer Responsibility concept). This can be countered by increasing incineration taxes or decreasing financial support for WtE, whether or not combined with increasing landfill taxes. Futhermore, older, less efficient WtE plants can be phased out (European commission, 2017).

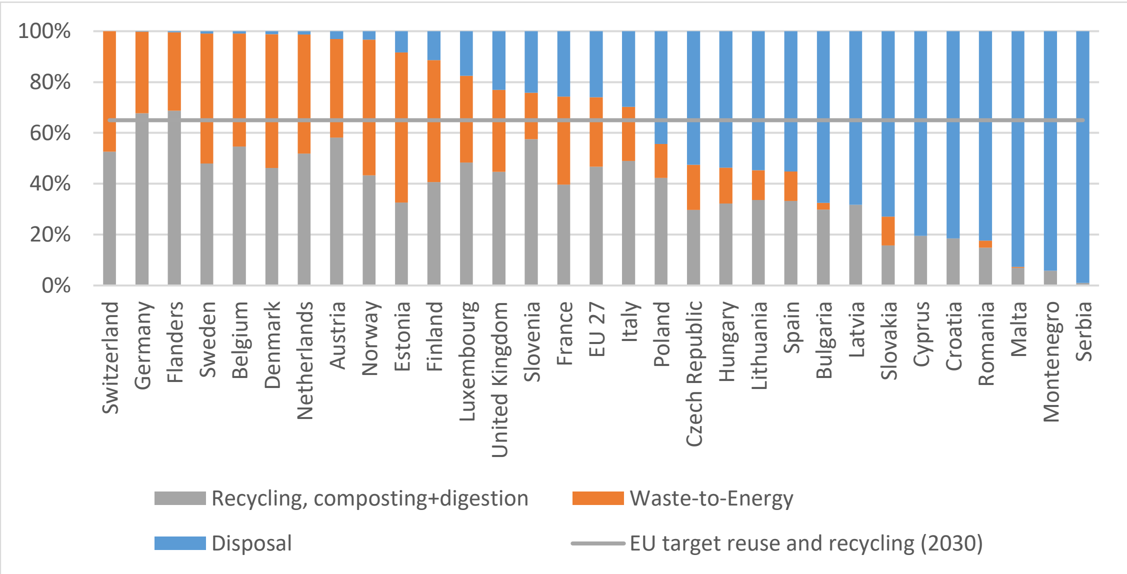

Fig. 2: MSW management in the different EU member states in 2015 (Eurostat, 2017). Data for Flanders were obtained from Vervaet et al., 2016

The data in Fig. 2 do not take into account the amount of household waste that is littered, i.e. waste that unintentionally ends up in the environment or is sometimes even intentionally dumped. Examples are waste transported by wind or animals from landfills into the environment, which is avoided in WtE plants where waste is contained in closed bunkers, or waste abandoned e.g. in road shoulders, woods or rivers. In Flanders for instance, in 2013 about 15 000 t of litter was collected by the local communities, corresponding to 2.3 kg per person per year i.e. about 1.6% of the amount of collected residual waste (148 kg per person) (KplusV, 2014; De Groof and Vandecruys, 2014). However, the amount of litter that is not collected and accumulates in the different environmental compartments is probably much higher.

The discussion above only considers Europe. On a global level, 40% of MSW is disposed of in landfills, 19% is recycled or composted, and 11% is recovered in WtE plants. The remaining 33 % of MSW is littered or openly dumped (Kaza et al., 2018). Waste management varies considerably by income and region. Open dumping predominantly prevails in low income countries, where about 93% of the waste is openly burned or dumped on roads and open land or in waterways. As the income of a country increases, a first step towards more sustainable waste management is construction and use of landfills, whereas only middle- and high income countries apply recycling, composting and WtE (Kaza et al., 2018). This evolution in waste management can be illustrated by the case of China: in 1980 the MSW collection rate in this country was only about 30% and waste management consisted mainly of open dumping. In 2003 the MSW collection rate had increased to 70% and about 85% of the collected waste was landfilled, 10% composted and 5% incinerated. By 2013, the MSW collection rate had increased to 90%, with 68% of the collected MSW landfilled, 2% composted or recycled and 30% incinerated. The number of WtE plants increased from 47 in 2003 to 166 in 2013 with a corresponding increase in incineration capacity from 3.7 10^6 to 46.34 10^6 t per year (Zhang et al., 2015). A next step in China's waste management policy is the introduction of separate collection and recycling (Zhang et al., 2015). Nevertheless, the statement that high income countries apply recycling, composting and WtE may not hold true for all developed countries. In the US for instance, 34.7% of the generated MSW is recycled or composted, 12.8% is treated in WtE plants, and the remaining 52.5% is still landfilled (US Environmental Protecion Agency, 2018). So, although the US is higher in the GDP per capita list than all but a few western European countries (International Monetary Fund, 2018), landfill remains the most applied MSW treatment option. Yet, also in the US, there is a trend towards more recycling and WtE, although less pronounced than in the western European countries: the percentage of MSW that is recycled increased from 9.6% in 1980 to 25.8% in 2015, and the share of WtE increased from less than 2% to 12.8% over the same period of time.

2.3. WtE, and by extension high temperature combustion e.g. in a dedicated rotary kiln, is essential to keep toxic substances out of materials and out of the environment.

2.3.1 Toxic substances in recycled materials

Special attention has to be paid to the dissipation of toxic substances in recycled materials, instead of focusing only on maintaining materials in circulation as long as possible. This can be illustrated by the example of bisphenol A (BPA) in recycled paper. BPA is added as a color developer in thermal paper, commonly used for e.g. sales receipts. Due to its estrogenic properties, BPA is considered an endocrine disruptor and is classified as a category 3 reproductive toxic substance, meaning it is alarming for human fertility (Geens et al., 2012a). Dermal contact to paper is recognized as one of the human exposure routes to BPA (Geens et al., 2012b). In Europe, paper is recycled on average 3.5 times, and about 50% of the raw material for the pulp and paper industry is used paper (CEPI, 2013). As thermal paper is typically collected together with other waste paper, it is introduced in the paper recycling loop and this way BPA is dissipated over a variety of paper products. This was confirmed by Pivnenko et al. (2015), who showed that in Danish paper and board samples, the BPA concentration ranged from 0.3 to 480 µg/g. They pointed out that recycling of waste paper was the main source of BPA contamination in paper products, which is sustained by the chemical and physical properties of BPA. An interesting general conclusion of this study was that mass based recycling targets do not ensure high quality recycled materials, as was already discussed in Section 2.1.

A recent report issued by the European Commission acknowledges that the collection and recycling targets in the European waste legislation merely focus on increasing the amounts of recycled materials, not on their quality regarding the absence of toxic substances (Goldenman et al., 2017). The report also states that no quality standards exist for toxic substances in recycled materials (as well as virgin materials), except for the end-of-waste criteria, which exist only for a few material types. Hence, the European legislation in place does not encourage activities to decontaminate waste or recycled materials.

Another type of pollutants possibly present in materials are the persistent organic pollutants (POPs). Because of their particular combination of physical and chemical properties they remain intact for many years or even decades once released into the environment. As a consequence they become widely distributed over the different environmental compartments, where they constitute a chronic source of exposure for plants and animals and also for humans, directly or indirectly. The toxic effects of POPs, typically accumulating in fatty tissue, include cancer, allergies, damage to the nervous system, endocrine disruption and developmental effects (Li et al., 2006). The Stockholm convention, which entered into force in 2004, currently lists 26 POPs whose production and use is eliminated or reduced or for which measures have to be taken to reduce unintentional release. Listed POPs include insecticides such as DDT, industrial chemicals such as polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs), and unintentionally released POPs such as polychlorinated dibenzodioxins and furans (PCDD/Fs). Because of their historical release and persistence, POPs have become ubiquitous in the environment. This was demonstrated amongst others by Jamieson et al. (2017), who analysed POP levels in amphipods caught in the Kermadec and Mariana Trench in the North Pacific at 7 – 10 km below sea level, which is believed to be one of the most remote and inaccessible habitats on earth. Surprisingly, the POP contaminant levels in the deep sea amphipods were considerably higher than documented for sediments or other marine fauna in nearby heavy industrialized regions.

A danger of recycling without eliminating POPs or materials contaminated with POPs is that they are not removed from the material cycle as illustrated by Pivnenko et al. (2017) for brominated flame retardants in plastics. POPs can have much farther reaching effects when they enter the food chain. As one of many food contamination incidents, the Belgian PCB/dioxin incident of 1999 clearly illustrates the dangers and potential far reaching consequences of recycling products contaminated with POPs. 50 to 100 litres of discarded transformer oil “Askarel”, containing 40–50 kg of PCB oil and 1 g of dioxins were accidently added to the household waste fat collection point in a waste recycling centre. The contaminated fat was processed together with other (waste) fats and used for the production of 500 ton of animal feed that was delivered to more than 1500 animal and dairy farms. The true cause and scope of the contamination became only clear four months after the original contamination in the recycling centre and after fat of affected animals had already been re-recycled into the animal food chain (Van Houte and Paque, 2000). Eventually, all contaminated food was removed from the market and thermally destructed, partially in WtE plants. Van Larebeke et al. (2001) estimated the total number of cancers resulting from human exposure during this incident to range between 40 and 8000.

Another aspect to consider is that for some waste types, recycling is more harmful than the production from virgin material. An example is the recycling of WEEE plastics containing toxic polybrominated diphenylethers (PBDEs) as a flame retardant. Evidence from studies suggests that the main effect of long term exposure to low doses of lower brominated diphenyl ethers is damage to the reproductive system (ATSDR, 2017). The main route of exposure for the general population is ingestion of house dust and inhalation of indoor air containing PBDEs emitted from consumer products, particularly electronic equipment (ATSDR, 2017). Because of their persistence, bioaccumulation, and toxicity, tetra, penta, hexa, hepta and decabromodiphenyl ether were added as POP to Annex A of the Stockholm convention, meaning that their production and use is to be eliminated. In view of this, article 8 (2) and annex VII of Directive 2012/19/EU on waste of electrical and electronic equipment (WEEE) state that “proper treatment” and recycling or recovery of WEEE has to include, as a minimum, the removal of plastic parts containing brominated flame retardants (BFRs) such as PBDEs. A first problem in recycling these removed WEEE plastics is that they may contain prohibited PBDEs that are not destroyed in the recycling process and hence are dissipated over the recycled products (Zennegg et al., 2014). This way, the PBDEs remain in the material cycle and the recycled products remain a source of human exposure to these prohibited, toxic substances. Another problem in the recycling of PBDE-containing plastics is that upon recycling, highly toxic polybrominated dibenzodioxins and furans (PBDD/Fs) can be formed that are, either immediately or during later use, emitted from the recycled plastics making them a potential source of human exposure (Schlummer et al., 2007; Zennegg et al., 2014). Schlummer at al. (2007) stated that besides sophisticated waste stream management and input control measures, extractive processes removing bromine from the waste plastics before extrusion can eliminate PBDEs from the plastic material cycle and prevent formation of PBDD/Fs. They acknowledge, however, that these extraction technologies are not available on an industrial scale, probably because the high energy and infrastructural demands of these processes increase the recycling cost, so that the recycled plastics become too expensive compared to plastics made from virgin materials. As a consequence, in the present economic context, WtE remains the only end-of-life technology in place for PBDE containing plastics that ensures destruction of these toxic flame retardants with recovery of the calorific value and with only very limited, well controlled emission of

PBDD/Fs (Vehlow et al., 2000). This case of WEEE plastics shows that, for wastes containing toxic substances, recycling is not always the preferential treatment option from a human health perspective.

The examples above show that it is necessary to first isolate toxic substances from the material cycle to prevent them from being dissipated over recycled materials and the environment, ultimately leading to human exposure. The isolated materials then have to be treated in such a way that the toxic substances are either destroyed or definitively removed from the environment. For waste contaminated with low levels of POPs, combustion in WtE plants under conditions as laid down in the Industrial emission directive (IED) i.e. respecting a residence time of the combustion gases of at least 2s at more than 850 °C, is the only treatment option in this regard. For waste containing high POP concentrations , specific high temperature combustion techniques, such as a rotary kiln with a secondary combustion chamber, are applied in practice to guarantee complete destruction (Van Caneghem et al., 2010a). Indeed, whereas in WtE plants incinerating MSW the legal provision of 2s and 850°C is only obtained in the gas phase, in hazardous waste incinerators these conditions are also maintained in the solid phase in the primary combustion chamber, thus guaranteeing "destruction" and "destruction and removal" efficiencies (DE and DRE, see equations 1 and 2) required by UNEP (2015).

$$DE = \frac{(POP\ content\ in\ \ input\ waste - total\ POP\ content\ in\ incinerator\ outputs)}{POP\ content\ in\ input\ waste} \qquad (1)$$

$$DRE = \frac{(POP\ content\ in\ input\ waste - POP\ content\ in\ flue\ gas\ emitted\ at\ stack)}{POP\ content\ in\ input\ waste} \qquad (2)$$

Research in real scale WtE plants showed that POPs, such as PCDD/Fs, PCBs and PBEs, are largely destroyed upon combustion, whereas the amount of PCDD/Fs and PCBs newly formed upon cooling of the flue gases in the steam boiler is limited and to a great extent independent of the POP concentration in the input waste (Vehlow et al., 2000; Van Caneghem et al., 2010a; Van Caneghem et al, 2010b; Vermeulen et al., 2014). For hazardous waste incinerators, DEs above 99.999% and DREs above 99.9999% have been reported for POP wastes (UNEP, 2015). For WtE plants incinerating non-hazardous waste such as MSW or similar commercial waste, almost no DEs or DREs have been reported in literature. Based on the POP input and output data range reported by Van Caneghem et al. (2010b), in the considered fluidised bed combustor, for PCBs DEs between 99.918 and 99.997%, taking into account variations in composition of inputs and outputs, were obtained in case the usual waste feed consisting of 70% RDF and 30% sludge was combusted and between 99.980 and 99.986% in case the waste feed consisted of 25% ASR, 25% RDF and 50% sludge. The given DEs and DRE's show that modern WtE plants are overall POP sinks, since they eliminate more POPs than they produce. It should be noted that DE and DRE depend on the operating conditions (e.g. **t**emperature, **t**ime, **t**urbulence, waste **t**hroughput) and the **t**ype of waste (5**t**s) and are hence different and specific for each installation and waste input (Wang et al., 2007).

2.3.2 Heavy metals

In contrast to toxic organic substances, toxic heavy metals present in waste are not destroyed in a WtE process, but are concentrated in the solid residues. Their partitioning over bottom ash, boiler ash and air pollution control (APC) residue depends on the speciation of the metal in the waste, the presence of other elements with which the metals can bind to give a volatile compound, and on temperature, residence time and mixing conditions in the furnace (Belevi and Moench, 2000). Halogens are instrumental in volatilizing metal elements, since metal halides (chlorides and bromides) typically have melting and boiling points below the temperatures in the furnace. In general, the higher the temperature in the furnace, the higher is the fraction of heavy metals that transfers to the gas phase and the lower the fraction in the bottom ash. With respect to their transfer to the combustion gas, two groups of heavy metals can be distinguished: (1) Cr, Co, Mn and Ni that are mainly transferred to the gas phase by entrainment. Their transfer coefficients are determined by their speciation and distribution in the waste and more than 90% of the amount present in the waste remains in the bottom ash; (2) Cu, Mo, Pb, Sn, Zn, Sb, As, Cd, and Hg for which evaporation is the main transfer process. Their transfer coefficients are not only determined by their speciation and distribution in the waste, but also by chemical and physical conditions and kinetics in the combustion process. The percentage of these elements that remains in the bottom ash ranges from 97% (Cu) to about 1% (Hg) (Belevi and Moench, 2000).

Due to the enrichment of certain heavy metals such as Hg, Pb, Zn and Sb, direct recovery of boiler ash or APC residue as building material is not possible from an environmental and health perspective. Furthermore, boiler ash and APC residue (each about 20 kg/t MSW) represent a much smaller mass than bottom ash (on average about 225 kg /t MSW) (Vandecasteele et al., 2007; ISWA, 2012). Therefore, WtE boiler ash and APC residue are typically stabilized and solidified with cement, and the resulting monolith is discharged on a hazardous waste landfill, which is specially designed with natural and artificial liners and drainage systems to prevent leaching of toxic substances to the environment (Billen et al, 2015a; Block at al., 2015). This way, waste incineration with subsequent solidification and stabilization, and controlled landfill of the generated APC residues prevents heavy metal dissipation in recycled materials and assures their storage in a safe sink.

2.3.3 Emission of air pollutants

Combustion of waste in WtE plants results in the generation of flue gases that contain, besides the products of incomplete combustion mentioned in Section 2.3.1, gaseous pollutants such as SO_2, HCl, NOx, CO and particulate matter (PM). Yet WtE plants have to comply with stringent emission limit values (ELVs), as shown in Table 1 comparing the ELVs for WtE plants and other combustion plants as laid down in the EU Industrial Emission directive (Directive 2010/75/EC). Hence, WtE plants are equipped with extensive flue gas cleaning equipment, generally consisting of selective catalytic or non-catalytic reduction of NOx, neutralization of acid gases by injection of lime or an alkaline solution, adsorption of e.g. dioxins and mercury on activated carbon and dust filtration in a bag filter or electrostatic precipitator (Vandecasteele et al., 2007, Van Caneghem et al., 2012). As a result of this extensive flue gas cleaning, the concentrations of the regulated pollutants in the gases that are emitted at the stack of WtE plants are generally a factor 10 to 100 below the emission limit values (Table 1). The emission of pollutants to air from WtE plants is limited and in each case lower than for conventional energy production, resulting in considerable avoided environmental impacts as shown by e.g. by Billen et al. (2015b) and Jeswani and Azapagic (2016).

Table 1: Overview of European emission limit values for WtE plants and conventional solid fuel combustion plants (Directive 2010/75/EC), compared to typical pollutant concentrations in the flue gas emitted at the stack of WtE plants.

	Emission limit values		Typical pollutant concentration range in flue gas emitted at stack
Pollutant	Combustion plants using coal and other solid fuels[a]	WtE plants	WtE plants [b]
Total dust	20	10	0.0-0.8
CO	-	50	6.0-14
TOC	-	10	0.1-1.8
SO_2	400	50	1.6-10.3
NO_x	300	200	65-145
HCl	-	10	0.9-6.1
HF	-	1	-
Heavy metals[c]			
Cd + Tl	-	0.05	0.0001
Hg	-	0.05	<0.0005-0.013
Sum other[d]	-	0.5	0.09
PCDD/Fs[e]	-	0.1	0.001-0.01

[a] Plants with a 50-100 MW thermal input, which is comparable to average size WtE plants, granted a permit after 7 January 2013

[b] Typical values for state-of-the-art MSW grate furnace incinerators as reported by Van Caneghem et al., 2012

[c] Average value over sample period of minimum 30 minutes and maximum 8 hours

[d] Sb, As, Pb, Cr, Co, Cu, Mn, Ni, V

[e] Expressed in ngTEQ.$(Nm^3)^{-1}$

2.4. WtE allows recycling of a large part of the metals and inorganics in the waste

WtE allows material recovery, with possible subsequent recycling, from non-recyclable waste. Bottom ash (BA), which is by far the major residue of WtE, is a heterogeneous material, mainly consisting of ash, stones, and metals, and has a broad particle size distribution, generally from <1 to 25 mm diameter. The inorganic fraction of the MSW (stones, silicates, glass, metals and metal compounds) typically leaves the grate furnace or fluidized bed almost unchanged. In a rotary kiln for treatment of hazardous waste the inorganic fraction generally rather fuses. From MSW BA, the metals, which are in fact the most valuable fraction, and the mineral fraction, which has the largest mass and volume, can be recycled. For recycling as building material, the mineral fraction of BA should comply with local legal concentration and/or leaching limits, which are most often exceeded for Cu, Mo, Sb, Cl^- and SO_4^{2-}. Vandecasteele et al. (2007) described the Indaver wet BA treatment scheme in which, in view of recycling, ferrous metals are removed by overhead magnets, whereas non-ferrous metals are removed by eddy current separation. Moreover, three mineral fractions are obtained, which can be used as building material: two granular fractions (6-50 mm and 2-6 mm, together around 80 kg/t MSW) and a sand fraction (0.67-2 mm, around 70 kg/t MSW). After aging (carbonation) for about 3 months the granulate fractions show low metal leaching and comply with the heavy metal leaching regulations. The major application of these compliant BA fractions is in uncontained bulk applications, such as artificial hills or as subbase in road construction, embankments or noise barriers (Verbinnen et al., 2017). Uncontained application means that no barriers such as liners have to be provided to isolate the ash or the mixture of ash with other building materials from the surrounding soil, and that no intensive monitoring or aftercare of the groundwater has to be provided (Van Caneghem et al., 2016; Verbinnen et al., 2017). The fine sludge fraction (< 0.67 mm, around 20 kg/t MSW) contains the highest concentration of toxic substances and can be used e.g. as cover layer on hazardous waste landfills (Vandecasteele et al., 2007). In general, leaching of heavy metals is the main chemical barrier for uncontained application of WtE BA as bulk building material.

Over the last decade, WtE BA treatment has significantly improved (Verbinnen et al., 2017; ISWA, 2015) to increase the recovery rate and improve the purity of the separated materials. Indeed, in addition to the wet treatment process discussed above, a half dry (moist) treatment, ADR, was developed (Inashco, 2018) in which a ballistic separator is used for separating fines adhering to minerals, improving metal recovery. Dry treatment schemes were also developed, e.g. by Martin (Martin plants and technologies, 2018) and by ZAV Recycling at KEZO in Hinwill (CH) (ZAR, 2018). Moreover, the sorting techniques were improved by using better magnets, Eddy current separators and sensor-based pneumatic sorting (Verbinnen et al., 2017; ISWA, 2015). In addition, some special treatments for reducing leaching of Cu and Sb were developed (Arickx et al., 2007; Van Caneghem et al., 2016). These developments in principle allow almost the whole BA fraction to be recycled.

Besides its application as uncontained building material, WtE BA can also be used as replacement for sand, gravel or cement in construction applications, as raw material in cement production and as feedstock for production of ceramic materials (Verbinnen et al., 2017). In cement and concrete applications, besides heavy metals, also chlorides can limit the use of BA. The chloride concentration in BA depends on the composition of the incinerated waste, but is in general relatively high. Chloride can cause corrosion in the cement kiln and can lower the cement quality making it e.g. more corrosive towards the steel wire-meshing in reinforced concrete. However, to remove chlorides, BA can be pre-treated by washing in dedicated installations, possibly in combination with grinding and sieving (Van Caneghem et al., 2016).

When BA fractions are mixed in concrete as partial sand, gravel or cement replacement (the latter after activation), some problems may occur, such as formation of pop-outs due to reaction between Al^0, Zn^0 and basic cement material leading to hydrogen gas. Moreover swelling and cracking may occur due to formation of ettringite ($Ca_6Al_2(SO_4)_3(OH)_{12}.26H_2O$) around Al^0 grains. These problems can be solved by enhanced removal of metallic Al and Zn from BA (Verbinnen et al., 2017). The potential of recycling WtE BA in cement is sustained by the fact that in the EU28 in 2015, around 170 Mt cement was produced per year (Cembureau, 2017) and that about 1.5 t of raw materials (clay, limestone) is needed per t of cement (Schorcht et al., 2013). Taking into account that up to 10% of raw material can be replaced by properly treated BA (see before for chloride removal) without any negative impact on the end quality of the cement (Verbinnen et al., 2017), around 25 $Mt.year^{-1}$ of BA can be potentially used in cement production. This amount exceeds about 1.5 times the 16 $Mt.year^{-1}$ of MSWI BA actually produced in Europe (Kahle et al., 2015).

Based on data reported in annual or sustainability reports of European WtE plant operators, typically 10-60 kg of ferrous metals and 1-30 kg of non-ferrous metals (mainly Al, Cu and Zn) are recovered per t of MSW incinerated and sold to secondary metal recyclers. This way, WtE enables to re-introduce a part of the metals present in non-recyclable and/or residual waste in the economy.

Classification of waste as hazardous or non-hazardous is based on Commission Decision 2000/532/EC on the List of Waste (LoW) amended by Commission Decision 2014/955/EU and Annex III of the Waste Framework Directive 2008/98/EC (WFD), amended by regulation 1357/2014 (Klymko et al., 2017). The latter defines 15 hazard properties to consider in the evaluation of the hazardousness of a waste including e.g. flammability (HP3), carcinogenicity (HP7) and corrosiveness (HP8). An assessment of the hazard properties of EU BA showed that it is in general non-hazardous by HP1 to HP13 and by HP15. However, for HP14 (eco-toxicity) the results are debatable: based on the total metal content BA should be considered as hazardous waste (by HP 14), whereas based on the more realistic heavy metal leaching from BA at its natural pH (pH 7 to 12) this residue can be classified as non-hazardous (Klymko et al., 2017). Also in recent legislation regulating the use of WtE BA as building material such as the Dutch decree on soil quality, limit values set to prevent damage to the ecosystem only consist of realistic leached concentrations at natural pH (Dutch decree on soil quality, 2007).

2.5 WtE can generate energy with high efficiency that can partly feed the CE

A WtE plant is generally equipped with a steam boiler for energy recovery from the hot combustion gases. The steam generated in the boiler can be applied in a Rankine power cycle, generating electricity with a turbine, as in a fossil fuel power plant. Part of the electricity is used internally, the remaining is usually supplied to the grid. The net electric efficiency achieved in state-of-the-art WtE plants is generally around 25%, which is modest compared to conventional coal power plants (De Greef et al., 2018). This is because in WtE boilers, the steam temperature is typically limited to 420°C, compared to e.g. 550 °C in power plants, to avoid high temperature corrosion. Indeed, due to the chemical and physical composition of the MSW, combustion gasses in WtE plants contain high concentrations of acids, metal species and dust, leading to phenomena of fouling and corrosion (Lee at al., 2007; De Greef et al., 2013). Through application of advanced cladding and coating of metal heat exchanging surfaces, it is possible to apply higher steam parameters. This is e.g. the case at the AEB WtE plant in Amsterdam, where an advanced steam cycle (440°C, 130bar, re-heating of steam) is applied, which affords a net electrical efficiency of more than 30% (AEB Amsterdam, 2015). Construction of such an installation is however complicated and expensive and also the operation costs are higher than in conventional WtE plants.

In WtE plants, as in conventional power production processes, there are three main possibilities for energy generation and application: (1) electricity is generated through a Rankine cycle and the residual energy (heat) is lost in the cooler after the condenser, which affords a limited overall energy efficiency as shown above, (2) low pressure steam or hot water from the condenser after the turbine is applied as heat source in a combined heat and power (CHP) scheme, which increases the overall energy efficiency of the WtE plant or (3) the high temperature steam from the boiler is (partly) directly provided as a process heat source to nearby industry (Vandecasteele et al., 2007; De Greef et al., 2018). This approach may afford the highest overall energy efficiency of up to 80%. This percentage takes into account, at the input side, besides the energy content of the waste, the energy content of other flows, e.g. combustion air and leakage air flows, and also the typical evaporation losses from the waste and from supplementary liquid flows (e.g. injected ammonia water for SNCR). Moreover, this efficiency also takes into account radiation losses and losses at heat exchangers.

In case (2) the hot water or low pressure steam is used for heating e.g. greenhouses, hospitals, or residences by means of a district heating network. In countries with a high heat demand and district heating infrastructure in place, such as the Scandinavian countries, it has become common practice to use heat from WtE plants for public heating purposes. Cities like Copenhagen, Vienna and Paris have large district heating networks fed by WtE incinerators located in or close to the city (Hofor, 2018; Wien Energie, 2018; CPCU, 2018). This has the additional advantage of requiring only short MSW transportation distances.

In case (3), WtE plants located in industrial areas directly supply high pressure steam to neighbouring companies. A very recent example is the ECLUSE steam network (Ecluse, 2017), which connects, via a steam duct and a condensate return duct, a cluster of WtE plants with neighbouring industrial companies. The WtE cluster consists of three grate furnace incinerators operated by Indaver, mainly combusting household and commercial waste (Vandecasteele et al., 2007) and of three fluidized bed combustors operated by SLECO, combusting sludge and industrial waste (Van Caneghem et al., 2010b), all located on the same site in the Waasland Port, east of Antwerp (Flanders, Belgium). The six incineration lines have an overall waste throughput of around 1,000,000 ton per year and a heat production power of around 250 MW_{th}. The Port of Antwerp is home to the largest chemical cluster in Europe. The chemical industry is very energy intensive and its energy consumption is very stable over time. In the chemical industry heat in the form of steam is typically used for unit operations such as distillation, evaporation or drying, and to drive endothermic chemical reactions. To date, the steam needed by the chemical processes is in general produced on site using natural gas as energy source in a CHP scheme. In 2017, the construction of the new ECLUSE steam network was started, which will deliver high pressure steam (400°C, 40 bar) produced in the WtE cluster to a network of neighbouring industrial (chemical) companies. In a first phase starting beginning 2018, which corresponds to 50% capacity, ECLUSE will provide process steam for 5 chemical companies; in a subsequent second phase the remaining 50% of steam produced will cover the expected increased consumption of the existing consumers, and/or will feed a new chemical cluster. This way, ECLUSE consumers no longer have to produce heat from fossil fuel, avoiding the yearly emission of hundreds of thousands tonnes of CO_2 (Ecluse, 2017). The new steam network fits in the “Green heat action plan” of the Flemish government aiming at increasing the proportion of green heat in the renewable energy supply in the region.

In case no heating network is available, or if it is economically not viable to build such a network, direct supply to a single neighbouring heat consumer is another option to increase the overall energy efficiency of WtE plants (De Greef et al, 2018). An interesting example is the Greater Manchester waste project (4 lines), where annually 850,000 ton of partly pre-treated MSW and solid residues of biomass digestion are treated, and has a power of 51 MW_{th} and 82 MW_{el} in a combined heat and power scheme. The produced high pressure steam and electricity are delivered to INOVYN's Runcorn manufacturing site located next to the WtE plants (Viridor, 2018).

Alternative WtE technologies such as gasification and pyrolysis often claim a higher overall energy efficiency than conventional waste combustion, mainly because the produced calorific gas would allow electricity generation with a higher efficiency. However, starting from MSW, the produced syngas contains high concentrations of gaseous pollutants (such as HCl and SO_2) and tars. Hence, gasification and pyrolysis plants require either extensive waste pre-treatment or gas treatment, which decreases the overall energy efficiency. Furthermore, the higher complexity of gasification plants compared to conventional WtE plants, leads to more difficulties in operation, additional maintenance and reduced reliability resulting in higher investment and operational costs (Lombardi et al., 2015). These reasons explain why in practice almost no cases of long term operation of full scale MSW gasification plants exist and conventional WtE in grate furnaces or fluidized beds remains today the most apt and most applied technology for energy and material recovery from residual, non-recyclable MSW (Quicker et al., 2014; Castaldi et al., 2017). Plasma pyrolysis of certain special waste types containing heavy metals such as galvanic sludges has the advantage that, due to the very high temperatures obtained in the plasma (2000 – 30 000°C) (Quicker et al., 2014), the bottom ash is vitrified, decreasing the metal leaching (Vieira Cubas et al., 2014) allowing the formed "plasma rock" to be recycled as building material instead of landfilled. However, the use of electricity to generate the plasma decreases the overall energy efficiency of the pyrolysis plant (Quicker et al., 2014) and it remains to be proven that the increased costs are offset by the income from the plasma rock. Only a limited amount of data on plasma treatment and waste gasification in general is available in the literature, making it difficult to quantitatively compare their energetic and materials performance with that of conventional WtE (Bosmans et al., 2013).

3. WtE and its role in the circular economy

As discussed in Sections 2.1 and 2.3, for certain types of waste generated in today's society, such as mixed wastes, composites, contaminated materials or wastes containing toxic substances, within the present techno-economical context, recycling might not be the best option from an overall sustainability point of view i.e. taking into account environmental, economic and societal and health aspects. So, a CE should not only focus on maintaining materials as long as possible in the economy and on decreasing resource depletion, but also has to consider economic, environmental, social and health aspects (the three pillars of sustainability), when the best (most sustainable) waste treatment option is selected.

Fig. 3 schematically represents the material flows and the role that WtE and waste recycling could fulfil in a CE. It shows in the circle that human activities in different fields, e.g. agriculture, industry, services and households, generate waste. In a sustainable CE, the most suitable treatment option for the different waste types generated is determined by economic, environmental, as well as societal and health criteria. For probably the largest part of the waste, recycling is the most sustainable treatment option. This waste is, after processing, largely re-introduced in the economy, preferably in the same application. For another, smaller fraction of the waste, such as e.g. hygienic, mixed and contaminated waste or waste containing toxic substances, WtE or, in case the waste is classified as hazardous, dedicated combustion in a hazardous waste incinerator is the preferred treatment option. WtE converts the non-recyclable waste into energy that is delivered back to society, including the recycling industry, and into solid residues that are partly recycled as building material or as secondary metal (indicated by the grey arrow to the recycling industry) and partly stabilized/solidified to be stored in a safe sink. Sorting residues from recycling activities are either sent to WtE for energy recovery or, after appropriate treatment, to a safe sink.

This way, WtE (and more in general incinerators, including dedicated combustors for hazardous waste) fulfils in the CE the role of a gatekeeper enabling material recovery from non-recyclable waste, while keeping recovered materials free from toxic substances.

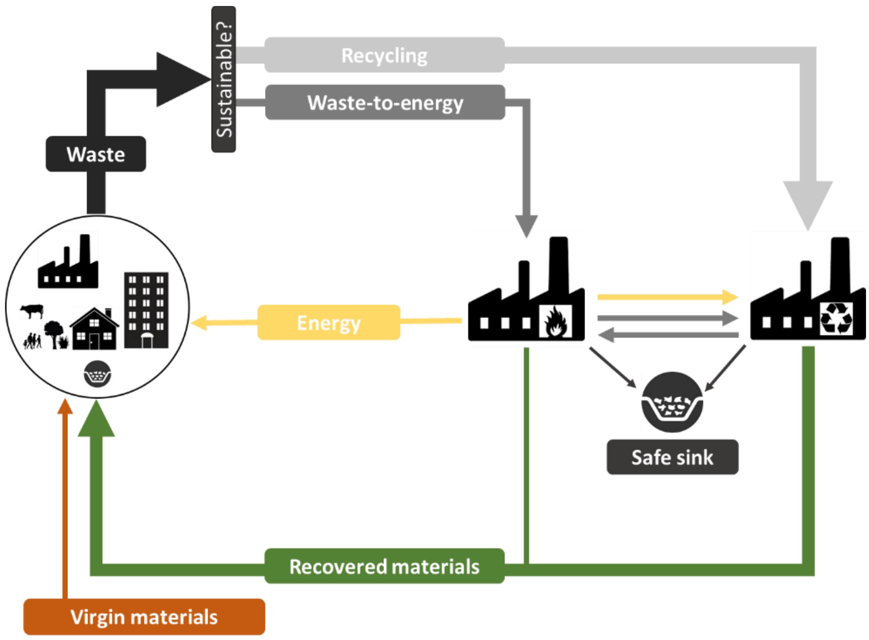

Fig. 3: Schematic representation of the material flows and the role of WtE and waste recycling in a CE.

A key factor to reduce the amounts of non-recyclable waste to be treated in WtE plants is eco-design, which should take into account the sustainability of the end-of-life treatment of a product as one of the design parameters (Singh and Ordonez, 2016). Eco-design can enhance the recyclability of products e.g. by selecting and/or limiting material types, by allowing for quick and easy disassembly and by avoiding the presence of known toxic substances. With regard to the latter, Goldenman et al. (2017) acknowledge that a need exists for strategies and implementation instruments preventing toxic substances from entering consumer goods and material cycles.

4. Conclusions

Enhancing high level material recycling is essential in the transition towards a CE. Although WtE figures on a lower level in the European waste hierarchy than recycling, it plays, from an overall sustainability point of view, an essential, complementary and facilitating role within the CE. Indeed, WtE is the preferred treatment option for waste that in the present context cannot be recycled. Moreover, WtE does not have to compete with recycling, but rather with landfill, which is lower in the waste hierarchy. Furthermore, WtE keeps material cycles, and ultimately the environment and humans free from toxic substances. Finally, WtE not only allows recovering energy from non-recyclable waste, but also allows recycling of some materials from such waste.

Unrestrained allowance in the recycling system of waste containing toxic substances, such as POPs or heavy metals, can increase the environmental and health impact of recycled materials. Legislators should be aware that a one-sided focus on increasing recycling volumes can lead to a quality deterioration and to dissipation of toxic substances in recycled materials. Instead of only increasing recycling volumes or masses, the quality of the recyclables, and the elimination of toxic substances in consumer goods should be the first priority in the transition towards a circular economy. In this regard, WtE fulfils an essential role as a gatekeeper, by destroying and eliminating toxic substances from the material cycles. Furthermore, WtE offers a treatment method for waste that is, e.g. because of its heterogeneous composition, today not recyclable from an economic point of view, whilst recovering valuable energy and materials. As long as contaminated material streams are generated or present in our society, decontamination and hence WtE remain a necessary part of the CE. The ultimate solution towards a fully CE is eco-design that not only takes into account the technical and functional specifications, but also considers the end-of-life aspects of products. Although in an ideal CE scenario WtE is phased out completely, because all material streams are clean and can be fully recycled in an economically viable way, it is more realistic to expect that WtE will remain necessary, at least for many decades, if it can be phased out at all. Indeed, on the one hand, for new materials, although often designed as green alternative to highly contaminating materials, the true environmental and health impact may only be discovered after many years (e.g. asbestos, CFCs, PCBs, etc.) during which the material cycles and/or the environment have already been contaminated. On the other hand, material degradation upon recycling cannot be avoided, entailing non-recyclable material residues. Hence, in the future, WtE is likely to remain a necessary, even essential part of the CE, but will probably treat lower amounts of waste, as further progress is made in waste prevention and reuse, which are both higher in the hierarchy than recycling, as well as in eco-design, in avoidance of toxic substances, and in improvement of recycling technology, which allow all three higher quality recycling.

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