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Life cycle assessment of plastic waste and energy recovery

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ABSTRACT

Plastics are essential in our economy and everyday life. However, plastic pollution is a global concern. To address this issue, the European Strategy for Plastics in a Circular Economy was adopted in January 2018. Attention has been raised to the entire life cycle of products, with legislation stating that plastic used throughout the design phase to manufacturing and packaging phases needs to be recyclable by 2030. This study evaluates selected plastic material categories and technologies carrying out a review of Life Cycle Assessment (LCA) analysis from literature. The literature review was carried out, the indicator units for impact categories among the investigated mid-point methodologies as well as the conversion factors for the metrics harmonization were provided and finally a detailed analysis of the environmental impact of several types of plastics was carried out for two options in the waste hierarchy, which are through disposal by sending waste to landfills and incineration with energy recovery. The disposal, treatment and recycling of 2.2 tonnes of general plastic waste including non-recyclable material delivered to a recycling facility was considered for comparison with these methods. An assessment of the comparative advantages of each practice was conducted. The potential for energy recovery was highlighted.

1. Introduction

Plastics are highly versatile materials crucial for the prosperity of modern societies. The array of applications, from packaging and construction to transport and agriculture is enabled by a multitude of functions conferred on plastic materials by design that highlight the usefulness of these materials. Indeed, since the first instance of mass production in the 1950s [1], plastic consumption has catapulted from 1.5 million tonnes in 1950 to 460 million tonnes in 2019, and expected to triple by 2060 [2]. Global plastics production is likewise increasing, from 359 million tonnes in 2018 to 390.7 in 2021 [3], 51% of which was produced in Asia, where China dominates the market with a 32% in 2021 share of global production. In Europe however, effective policies have managed to curb plastic demand from 61.8 Mt in 2018 to 55 Mt in 2020 [4]. Packaging is the primary end use for plastics, accounting for almost 40% of total demand in Europe, with the building and construction sector in second place with just over 20% of total demand. A similar trend is seen in the UK, where approximately 5 million tonnes of plastic is used annually with nearly half of it from packaging [5].

In terms of the EU (European Union), plastic waste management has been in the forefront of both policy and research. Action on plastics was identified as a key priority in the first Circular Economy Action Plan (CEAP), which provided a comprehensive body of both legislative and non-legislative actions aiming to transit the European economy from a linear to a circular model. The EU aims for waste management to be transformed into sustainable material management which embeds the principles of the circular economy, enhances the diffusion of renewable energy, increases energy efficiency, reduces the dependence on imported resources and provides economic opportunities and long-term competitiveness [6]. Specifically, the EU committed to 'prepare a strategy addressing the challenges posed by plastics throughout the value chain considering their entire life-cycle'. The European Strategy for Plastics in a Circular Economy adopted in 2018, the first such strategy in the world, focuses at transforming the way plastic products are designed, used, produced and recycled in the EU. It notes that better design of plastic products, higher plastic waste recycling rates, more and better quality recyclates will help to boost the market for recycled plastics. This strategy should also contribute to reaching the UN Sustainable Development Goals, the global climate commitments and the EU's industrial policy objectives [7]. Along similar lines, the

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European Commission (EC) in 2020 adopted a new CEAP, as one of the main building blocks of the European Green Deal, Europe's new agenda for sustainable growth. This plan, *inter alia*, enforces measures along the entire life cycle of plastic products to uncover the potential for circularity [8].

Plastic waste does not decompose easily, therefore, leads to long-lasting environmental damage. There are several types of plastic materials, each with specific characteristics. Plastics can also be categorized depending on the raw materials used for their production, and whether these raw materials are sustainable or not. The main distinctions of plastic materials are whether they can be reversibly melted and hardened through heating, which is a property of thermoplastic materials, or if the process is irreversible, the material is classified as a thermoset plastic [9]. Furthermore, the lifecycle of plastic materials is complex and involves several application sectors, process and practices, rendering horizontal measures difficult to implement. Polyolefins are the leading plastic type (OE and PP), followed by PVC, PET and PUR. Packaging is the lead application in terms of demand for each of these plastics, with the exception of PVC and EPS that are mainly used in construction, in applications such as pipes and insulation.

Given that there are many different types of plastic in use, there are challenges in terms of their recycling. For instance, the UK local authorities quite often face challenges such as finding recycling solutions for certain types of black plastic and low-grade plastic [10]. Currently, unrecyclable plastics are usually landfilled or incinerated. But the UK is not alone in this practice, as 2450 waste to energy plants are active globally [11]. This has, therefore, caused an immense concern to investigate the environmental disadvantages offered by these methods [12] and discover the alternative state of the art technologies that can be employed to illuminate and minimise this problem.

For instance, a comprehensive review of nonbiodegradable plastic waste (NPW) treatment technologies has been carried out by Zhang et al. [13]. Fig. 1 presents classification of currently reported advanced NPW disposal methods. The division was based on the standard D5033-(2000) - of the American Society for Testing and Materials (ASTM). Four types of recycling are specified: primary and secondary recycling (ASTM I, II - mechanical reprocessing), tertiary and quaternary recycling (ASTM III, IV- recovery of valuable intermediates or energy recovery).

Yang et al. [14] reported that municipal solid waste (MSW) incineration does not remove 100% of the plastic. The results of the analysis indicated the presence of microplastics (MP) in the bottom ash from 12 mass burn incinerators and 4 fluidized bed incinerators. It was stated that waste segregation is crucial, because the amount of microplastic was significantly less for those sites where waste was segregated before incineration. The solution to this problem could be upcycling. Horodytska et al. [15] compared upcycling of printed plastic film (considered to be part of a closed material loop) with incineration and conventional recycling. The application of this solution would allow the market for recycled plastics to develop.

There are a number of strategies used to prevent global warming, which have been discussed by Olabi et al. [16–18]. However, it does not focus on assessing the processes to which plastic waste is subjected. Building on the previous studies and the EU and the UK's current emphasis on circularity and the necessity to reflect on the entire life cycle of products, this paper will employ a Life Cycle Assessment (LCA) to evaluate the chosen technologies: incineration and landfill.

A review on plastic waste management was first prepared. The LCA methodology was then discussed including the indicator units for impact categories among the investigated mid-point methodologies as well as the conversion factors for the metrics harmonization being provided. This is followed by a detailed analysis of 2.2 tonnes of general plastic waste including non-recyclable material delivered to a recycling facility. The study will focus on the environmental effects of recycling processes. This paper aims to identify and demonstrate the importance of in-depth LCA analysis for local authorities to choose the most suitable waste management option.

2. Overview of plastic waste management

Globally, the landscape of the plastics recycling industry shifted irrevocably after China's implementation of the National Sword policy in 2018 [19]. Since 1992, China had imported a cumulative 45% of plastic waste globally generated. This policy is projected to create a surplus of plastic, textile and paper waste reaching 111 million tonnes by 2030 [20], forcing the rest of the world to adapt quickly and decisively or risk overwhelming storage, recycling and ultimately waste management infrastructure. Moreover, the scale of the global plastic waste crisis was revealed, as China imported over 60% of plastic waste

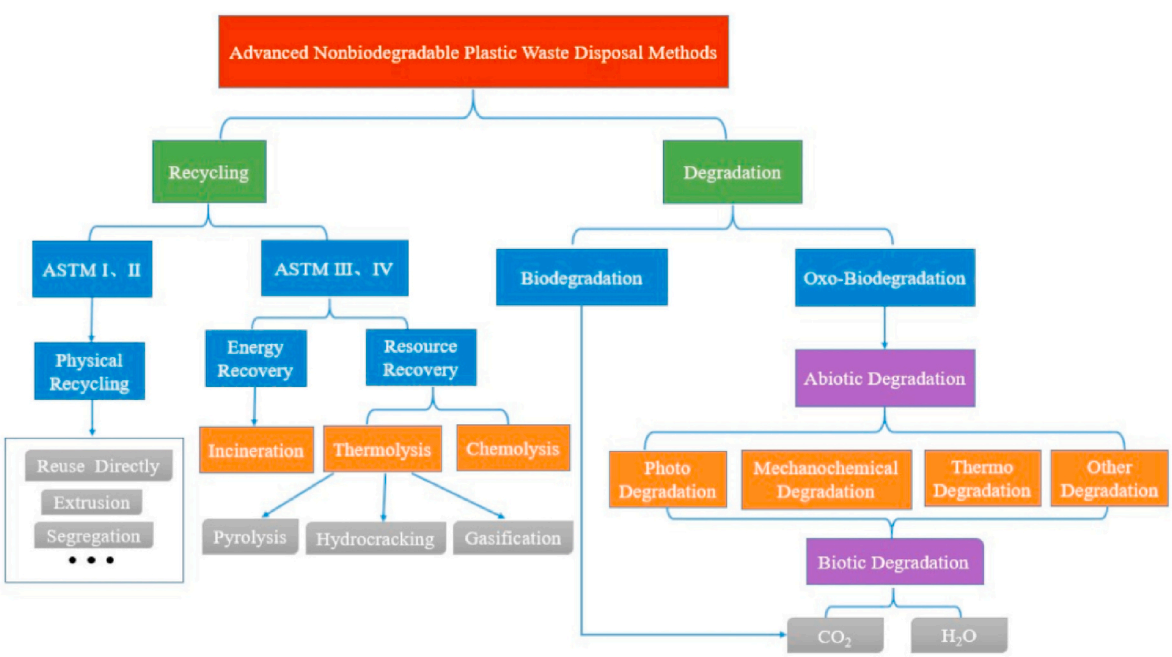

Fig. 1. Classification of advanced NPW disposal methods [13].

generated in the UK, plastic that is now exported to other countries such as Malaysia or simply incinerated [21]. Europe, in 2016, exported about 1.4 million tonnes of plastic waste to China, about half of recyclable waste collected, which is less than 30% of total plastic waste [22]. Following the ban, a mere 14,000 tonnes were exported in 2019 [23]. However, the EU in 2019 still exported 1.5 million tonnes of plastic waste, mostly to Turkey and other Asian countries, such as Malaysia or Indonesia [24]. In the US, over 70% of plastic waste collected was shipped to China [25] in 2018, dropping by 92% following the ban, whereas exports to Thailand shot up by almost 2000% [26]. In 2020 the UK exported approximately 537,000 tonnes of plastic waste, predominantly to Turkey (39%), Malaysia (12%) and Poland (7%), countries with very low recycling rates [27]. Most recently, however, there has been a 13% drop in plastic exports from the UK in 2021 [28].

As far as local recycling in the EU is concerned, the plastic recycling rates vary across different Member States. It has been estimated that approximately 38% of plastic packaging waste was recycled in 2020 in the whole of the EU (Fig. 2), which is lower than in 2019 (41%). However, this is mainly due to the stricter rules for reporting recycling (including stricter accounting of composite packaging material fractions) [29]. More than half the plastic packaging waste generated was recycled in six EU Member States (as indicated in Fig. 2). Specifically, the highest recycling rate of plastic packaging waste was recorded in the Netherlands (57%), followed by Lithuania, Slovakia, Spain, Bulgaria and Cyprus, whereas Malta, France, Denmark, Hungary, Ireland, Romania, Poland and Austria recycled less than one-third of plastic packaging waste (as demonstrated in Fig. 2). The UK, which left the EU in 2020 recorded higher plastic packaging waste recycling rate in 2018 than the EU average [30].

2.1. Environmental impact of landfills

It is estimated that about 60% of plastic waste globally ends up in either a landfill or the natural environment [31]. Landfilling is an ancient practice, dating back to Crete in 3000 B.C. [32], considered one of the most cost-effective methods [33] and remains a prominent practice, especially outside the EU and the USA. Modern landfills include engineered solutions to eliminate leaching and gas emissions, such as bottom lining [34]. However, in other parts of the world, in developing countries in Asia landfilling remains a significant environmental problem [35,36]. As P. R. Yaashikaa et al. [37] note, the key to an ecological MSW landfill is to provide regular monitoring and thoughtful management aimed at stabilising biowaste. Thus enabling better use of the waste as an alternative fuel for the production of electricity. Currently, it is assumed that landfills should function for more than 20 years [38]. During this time, plastic waste undergoes five stages of stabilisation being influenced by geo and hydro-mechanics, biology and thermology. The results of various studies have confirmed that landfill is a potential source of loss of microplastics which are then distributed to different areas of the environment [35,39]. Q. Huang et al. [38] have shown that microplastic generation increases exponentially with the age of the landfill. The share of MP with dimensions <0.5 mm can be used as indicators to evaluate whether a landfill has reached the booming phase of MP generation. M. Shean et al. [40] focused on the presence of microplastics and nanoplastics in landfill leachates, their interaction with heavy metals and strategies for removal. They stated that the MP detection methods in other types of environmental media are not always suitable for detecting microplastics in leachate. In addition, leachate treatment technology does not take into account the removal of microplastic particles and the implementation of improved technology will increase operating costs. The problem with leachate is even more significant at informal landfills in South China [41] where, as a result of the weathering of plastic waste, the resulting MPs leach into the leachate from the landfill and then into the groundwater and soil. To estimate the amount of plastic released into the environment V. Yadav et al. [35] proposed a conceptual framework for quantifying the loss of plastics from landfills and open dumps. Considering the above, the recovery and recycling of plastic waste is an important environmental protection task in China and for this reason, X. Geng et al. [42] are researching waterless cleaning method for the recovery of plastics raw materials from landfill waste.

Fig. 2. Recycling rate of plastic packaging waste in 2020 [29].

2.2. Waste incineration

Waste to energy processes cover: landfilling with energy recovery, thermal (incineration, pyrolysis and gasification) and biological (anaerobic digestion) conversion [43,44]. Incineration does not eliminate the need for landfill but reduces the amount of waste to be disposed of in landfills. Incineration is the most extensively studied waste to energy technology by virtue of its relatively straightforward application [45]. Bottom ash from incinerated municipal waste can be used as an ingredient in ceramic material that can withstand temperatures of up to 1000 °C [46]. C. Luo et al. [47] additionally focused on identifying a methodology for determining the optimal location for an incineration plant. From the point of view of LCA assessment, however, there are still inconsistencies with regards to the methodologies applied, as highlighted by Astrup et al. [48] and Laurent et al. [49]. The relevant ISO standards such as ISO 2006a and ISO 2006b [50] addressed several of these issues, although to this day there is no unified consensus on the optimal waste to energy pathway. Waste incineration has been found to be one of the most widely studied approaches with alternative options such as pyrolysis, gasification or co-combustion being underrepresented. Thus, in this study, alternative options such as pyrolysis and gasification will also be highlighted, in order to facilitate the inclusion of additional technologies in potential decision making.

Several incineration methods can be identified, as the technique is not restricted to plastic waste [51]: moving grate, fixed grate, rotary-kiln, fluidized bed and specialized incineration, e.g., incineration of flammable material or thermochemical recycling [52]. The environmental impact is not insignificant. The main differences between incineration plants result from different methods of energy recovery and flue gas cleaning processes. In terms of environmental impact, municipal solid waste and resources are inputs for incineration plants and landfill sites, as shown by Figs. 3 and 4.

2.3. Pyrolysis/gasification

Pyrolysis and gasification are recognised as advanced thermal treatment (ATT) methods that provide a solution for resource treatment while meeting emission regulations [54]. The gases from the pyrolysis of the sludge can be burnt to recover part of the heat energy [55] and the waste heat recovery allows industrial plants to be more efficient [56]. Pyrolytic decomposition of plastic waste occurs in temperatures ranging from 200 to 1100 °C in environments devoid of oxygen. Due to the effects of humidity and particle size, pre-treatment such as drying, shredding or use of additives (e.g. lime [57]) are common practice. Three types of products are formed: carbonized solids (char), non-condensable gas and a liquid fraction consisting of gasoline (C4–C12), kerosene (C10–C18), diesel (C12–C23), diesel-like fraction (C11–C22 plastic pyrolysis middle oil) and motor oil (C23–C40) range hydrocarbons [58–60].

Maximum temperature, heating rate [61] and naturally the composition of pyrolyzed material [62] are among the main factors affecting the products of pyrolysis and their value as fuels or feedstocks [63]. Other factors include type of reactor [64], feeding arrangement, type of fluidizing gas and flow rate [65], residence/retention time [66], pressure and presence of catalysts [67]. For instance slower pyrolysis favours gas and char, while short retention times accompanied by lower temperatures result in a high proportion of oil [68].

The composition of gaseous products from the pyrolysis of plastic materials usually contains hydrogen light hydrocarbons such as methane, ethane, ethene, propane and butene. As a result, the gas has significant calorific value (pyrolytic gas from PE and PP in the 42–50 MJ/kg range [69] and reported production of ~13–27% per weight for each kg of feedstock [62]. Other studies have reported 54–66 wt% for PP and 37–59% for PE [70].

Parameters such as process temperature or the presence of a catalyst can enhance the production of the gas due to increased cracking [62].

Likewise, carbonized material from the pyrolysis of plastics can feature significant carbon content (29.3 wt % [71], 46.03 wt% [72]), rendering its potential for solid fuels worth considering. Heating rate, temperature in the reactor and retention time are parameters with significant influence on the structure and quantity of char [73]. The char possesses a microporous or mesoporous structure and contains various inorganic compounds [74]. Possible valorization pathways for chars include effluent capture in the textile industry (replacing activated carbon) [75], gas clean-up adsorbent [76] and other applications for the removal of aqueous contaminants and heavy metals [77].

Finally, pyrolytic oil from plastics shares the favorable properties of gas and char with considerable heating value. Oil yields vary greatly and depend on the parameters of the pyrolytic reaction. Yields of 30–43 wt% for PP and 37–61 wt% for PE were reported in Ref. [70]. There is significant variance in the heating values of various oil fractions produced through pyrolysis of different plastics. For instance, PET is reported at

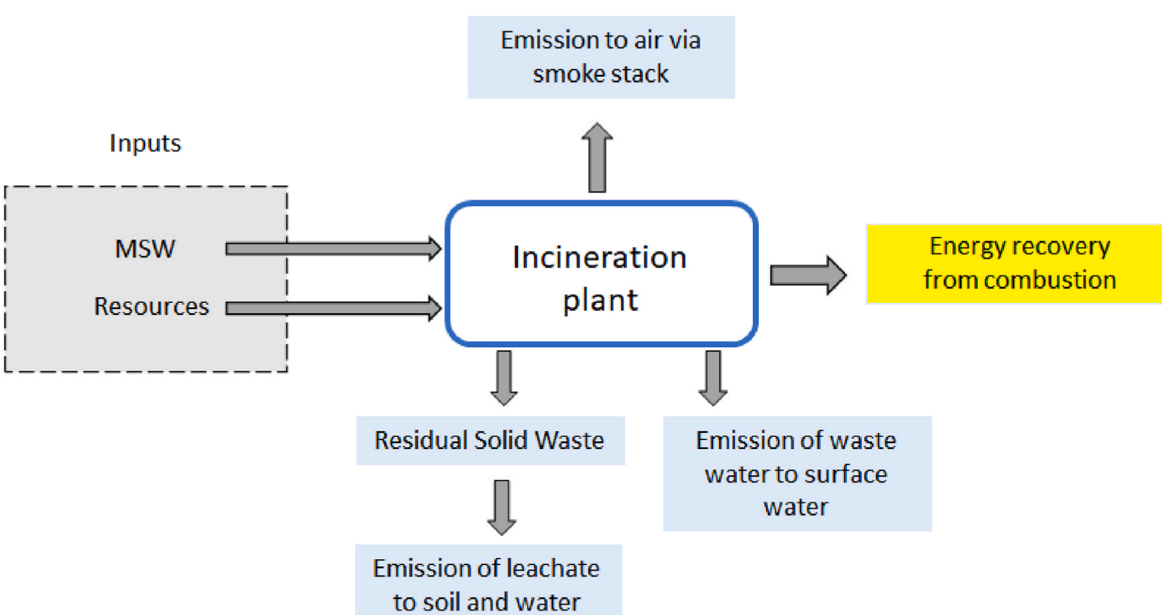

Fig. 3. The inputs and outputs of incineration plants which have an environmental impact [53].

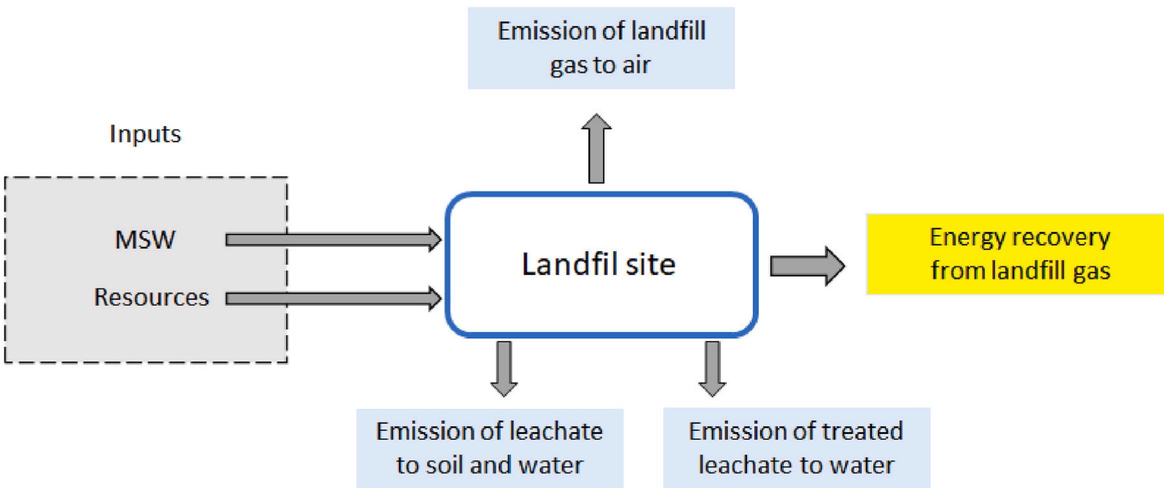

Fig. 4. The inputs and outputs of landfill which have an environmental impact [53].

28.2 MJ/kg [78], PE at 41.45 MJ/kg [79], HDPE at 45.86 MJ/kg [80], LDPE 38–39 MJ/kg [81], PVC 43.22 MJ/kg [82] and PP at 40.8 MJ/kg [83].

A significant disadvantage of pyrolysis is the composition of the exhaust gas that needs to be controlled. A review of the main risks is listed in Ref. [84]: for instance, gas from the pyrolysis of tires contains high concentrations of H_2S, that can be oxidized to SO_2 [85]. Methods to reduce the H_2S concentration in the gas were discussed in Ref. [86]. Furthermore PVC materials produce significant quantities of HCl [87], while food waste can be a source of nitrogen compounds [88].

The consideration of pyrolysis deserves higher mention if the scope is expanded from the end of life management of plastics to the wider municipal solid waste area. This is due to the composition of plastics in MSW, that are predominantly PE, PET, PP, PS and PVC.

Overall, pyrolysis of plastic waste is characterised by a lower climate change impact compared to energy recovery via incineration, similar climate change impact and energy used compared to mechanical recycling but higher than other impacts. As is common with LCA studies, assumptions play a pivotal role: location, energy mix, process efficiency and recyclate quality all have significant effects on the outcomes [89].

Pyrolysis can be a considerable alternative to both landfilling and incineration. On one hand the disadvantages of landfilling have already been reported, but from an LCA perspective there exist cases where it is preferable to incineration [90]. This is also due to the release of toxic gases and GHG released during incineration, that necessitates exceptionally high temperatures in order to be prevented [91]. For instance, in Ref. [92], PET recycling via pyrolysis is shown to have significantly lower CO_2 emissions compared to incineration, as a result of higher energy efficiency and value retention, as well as the avoidance of combustion emissions. In the same study, pyrolysis to waxes was shown to be the second best option for ABS WEEE plastics (6.5 kg CO_2 eq/kg input compared to 6.0 kg CO_2 eq/kg input for dissolution) and presented very positive results for polymers with high polyolefin content such as PE and PP. J. Fox and N. Stacey in Ref. [93] found that pyrolysis of waste polyethylene is a more efficient method of chemical downcycling; nevertheless, it is gasification that provides a product suitable for further production of, for example, synthetic fuels. The possibility of integrating the gasification of waste plastic into a multigenerational waste to energy system was presented by M. Ismail and I. Dincer [94]. Thus confirming that gasification is an environmentally sustainable and efficient method.

To conclude, pyrolysis and gasification are established processes for the thermochemical conversion of plastic waste primarily to energy and fuel. Industrial-scale plants have been operating across the world since 2000 [95] in Japan, 2018 in the US [96–98] as well as other areas as referred in Refs. [90,99,100].

2.4. EU waste management policies

Regulatory frameworks and policies are regarded as key drivers for change. The EU is acting on plastic pollution to accelerate the transition to a circular and resource-efficient plastics economy with the newly issued strategy for plastic. This strategy aims to transform the way plastic products are designed, produced, used and recycled in the EU. For instance, the EU in its Directive on Single-Use Plastics (known as the SUP Directive) [101], has also banned some single-use plastic products such as cotton bud sticks, cutlery, plates, straws, stirrers, sticks for balloons, food containers made of expanded polystyrene, and products made from oxo-degradable plastic. The SUP Directive also introduced new binding targets, such as a 77% separate collection target for plastic bottles by 2025 (90% by 2029); 25% of recycled plastic in PET beverage bottles from 2025, to 30% in all plastic beverage bottles from 2030. These are in addition to the minimum recycling targets of plastic packaging waste of 55% by 2030 as set under the revised Packaging Waste Directive [102].

In terms of waste management, the EU Waste Framework Directive (WFD) [103] defines the framework under which waste management policy is implemented throughout the EU. It is built on the waste hierarchy, where priority is given to waste prevention, followed by preparing for reuse, recycling, then recovery, for example, energy recovery and the least favourite option being disposal (which includes landfilling and incineration without energy recovery). Landfilling is the least preferable option and should be limited to the necessary minimum due to its detrimental effects on the environment and human health.

Waste sent to landfills must comply with the requirements defined by the Landfill Directive (LD) [104]. The amended LD [105] imposes that the Member States must ensure that by 2035 the amount of municipal waste landfilled is reduced to 10% or less of the total amount of municipal waste generated (by weight).

The EU's plastic management strategy has also been depending on exports to cover recycling needs. Until 2020, Europe has been exporting significant quantities of plastic waste to third countries. At the end of 2020, new rules governing the import, export and intra-EU shipment of waste were adopted [106], limiting or outright banning shipping hazardous plastic waste to non-OECD countries. This legislation is expected to have huge implications on the wider European plastics industry. The new rules, amending the Waste Shipment regulations of 2006, are a result of the amendments of the Basel Convention, that sought to establish a global regime governing the international trade of plastic waste [107].

Finally, as part of the Single Market for Green Products Initiative [108], the European Commission is promoting the Product Environmental Footprint (PEF) and Organisation Environmental Footprint

(OEF) methods as a common way of measuring environmental performance. The approach was tested in a pilot phase from 2013 to 2018, followed by a consultation phase and was included in the 2020 Circular Economy Action Plan [8]. While there are concerns about the shortcomings of the methodology [109], the harmonization of LCA methods through the PEF and OEF is expected to eventually evolve into a powerful regulatory tool.

2.5. UK waste management policies

Given that the UK had left the EU and was not obliged to transpose the SUP directive, the UK government, nevertheless, has announced that it would commit to the vast majority of the EU's Circular Economy Package. The UK is aiming to achieve a 65% municipal recycling rate by 2035 and a maximum of 10% municipal waste going to landfill in the same timeframe [5]. Waste management is a devolved area within the UK, where each devolved administration can decide whether to follow the Directive's provisions. For instance, the Scottish government banned plastic-stemmed cotton buds in 2019 and further bans were introduced in 2021 under the Scottish Government's 2021 Regulations for certain problematic single use plastic items, such as cutlery, plates and expanded polystyrene food and drink containers. The Welsh Government is behind with its new Environmental Protection (Single-use Plastic Products) (Wales) legislation to be introduced soon. England also aims to prevent and reduce the impact of certain plastic products on the environment (i.e. the aquatic environment), and on human health, as well as to promote the transition to a circular economy with innovative and sustainable business models, products and materials, thus also contributing to the efficient functioning of the internal market [110]. However, England in its Environmental Protection Regulations 2020 (Plastic Straws, Cotton Buds and Stirrers) [111] banned only some single use plastic items, such as the distribution and/or sale of plastic straws, stirrers and plastic-stemmed cotton buds in England effective from October 2020. The UK has also taken other steps to eliminate all avoidable plastic waste by 2042, as set out in its 25 Year Environment Plan [112]. For instance, the UK has also introduced a world-leading Plastic Packaging Tax for packaging with less than 30% recycled plastic (effective from April 2022) [113].

While the new measures should reduce the generation of plastics in the future, it does not necessary mean that there will be no plastic waste management problems. Given that China has banned the import of certain types of plastic waste, local authorities in the UK struggle to find alternative destinations, as it is challenging to locate recycling solutions for certain types of black plastic and low-grade plastic [10]. Therefore, most plastic waste is still sent to landfills or to incineration plants. This does not sit neatly with the WFD and its waste hierarchy, which was transposed in the UK in each national devolved authority, for instance, in England and Wales through the Waste Regulations 2011 [114]; in Scotland through the Waste Regulations 2012 [115]; and finally, in Northern Ireland through the Waste Regulations 2011 [116]. Therefore, local authorities should incorporate LCA in their waste management in order to divert its waste to other treatments higher up the waste hierarchy.

3. LCA methodology

Life Cycle Assessment is defined by the ISO 14040 [50]. It is a technique to quantify the environmental aspects and the potential impacts associated with a product. LCA is based on four main steps (Fig. 5) [117]:

1) goal and scope definition,
2) life cycle inventory (LCI)- includes data collection and a calculation procedure for quantification of the inputs and outputs of the system under study,
3) life cycle impact assessment (LCIA)- LCI results are related to environmental impact indicators and categories,
4) interpretation-this phase consists of checking for completeness, consistency and sensitivity as well as accuracy and uncertainty of the results obtained.

This approach can be applied in different areas, such as: comparing two systems in terms of energy usage, total life cycle cost and GHG emissions [118]; two products and their environmental impact [119] or role in achieving the United Nations Sustainable Development Goals [120].

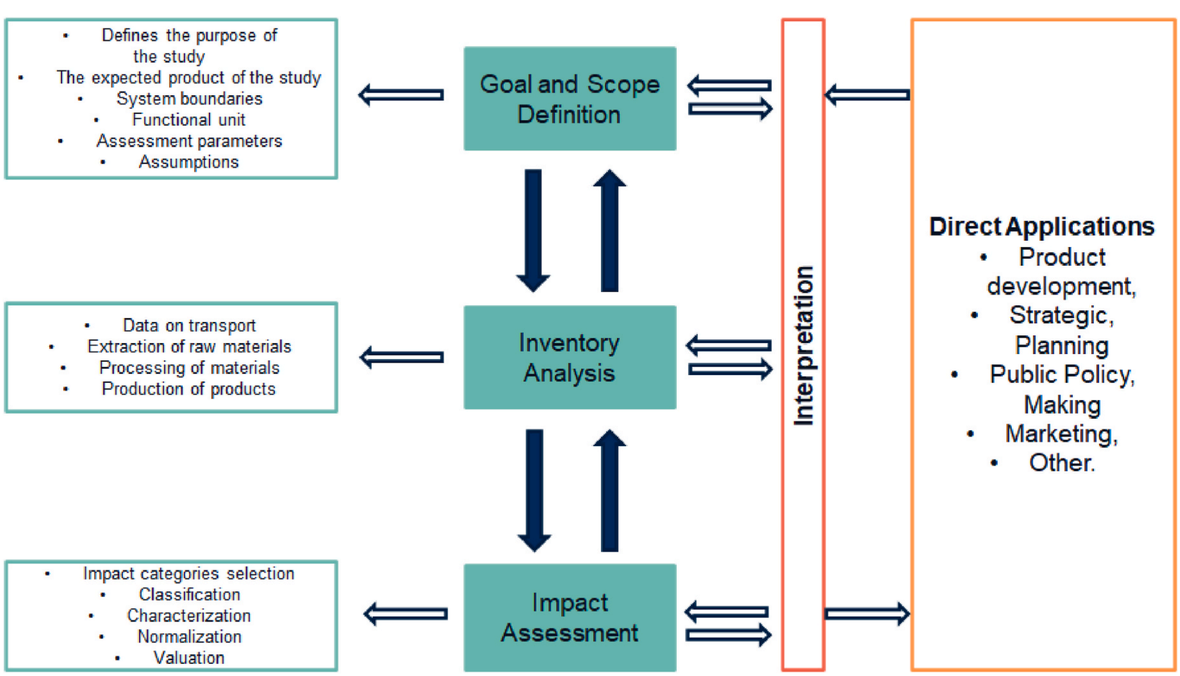

Fig. 5. LCA assessment framework (per ISO 97–06).

3.1. The midpoint impact categories

The authors carried out an analysis of publications from the last 25 years, taking into account several criteria. First, the focus was on keywords: Plastic waste management, LCA or Life Cycle Analysis, plastics and polymers, landfill, incineration. The initial search resulted in 5432 entries ranging from 1999 to 2023, with a steady upward trend in papers published with these keywords each year. To narrow down the results, the following criteria were implemented:

1) Date of publication, deciding on 2015 as the year that the Circular Economy Action Plan was published by the European Commission.
2) The second criterion is whether the publication follows or refers to results yielded in adherence to the ISO 14040:1997–2006 standard [50].
3) The third criterion is whether the publication considers energy recovery and chemical/energy recycling processes.
4) This criterion refers to whether a defined mix of multiple plastics is the object of the research work, rather than a specified item (e.g. plastic bags).
5) the last criterion refers to whether the paper has applicability or refers to EU27 and the United Kingdom.

The impact of the criteria set on the number of publication results is shown in Fig. 6.

In the following Table 1, a summary of the findings of the review work is consolidated. In column (A), references are provided. Column (B) lists the type of articles addressed in each publication, (C) lists the different scenarios and processes that are relevant to this research work. The remaining columns show the midpoint indicators of the ReCiPe impact assessment that comprise the most popular assessment method. Midpoint indicators describe singular environmental problems, such as acidification, eutrophication and resource depletion. In contrast, endpoint indicators identify the environmental impact at three higher levels of aggregation, namely the effects on: 1) human health, 2) biodiversity and 3) resource scarcity [121].

Converting midpoints to endpoints simplifies the interpretation of the LCIA results. Fig. 7 provides an overview of the structure of ReCiPe [121].

ISO/TR 14,047:2012 [132] describes the assessment of seven impact categories: Climate Change, Acidification, Eutrophication, Human Health Toxicity, Photochemical Oxidation, Ecotoxicity and Ozone Layer Depletion. However, the majority of reviewed articles do not comply with this suggestion, usually adopting additional indicators.

The midpoint impact categories addressed by the publications reviewed are the following [133]:

3.1.1. **Particulate matter** (kg PM2.5-eq)

Fine particulate matter formation, quantified based on reference intake of PM2.5 by humans.

3.1.2. Photochemical ozone formation (kg NOx-eq)

Comprising both human health ozone formation potential (HOFP) and ecosystem ozone formation potential (EOFP).

3.1.3. Ionizing radiation; stratospheric ozone depletion (kg CFC-11-eq)

Ozone depletion potential (ODP), referring to a time-integrated decrease in stratospheric ozone concentration over an infinite time horizon [133,134].

3.1.4. **Toxicity** (kg 1,4DCB-eq)

The effects of chemical emissions for: human, freshwater (FW), marine and terrestrial ecotoxicity. The human-toxicological effect factors were derived individually for both carcinogenic and non-carcinogenic effects.

3.1.5. **Climate change** (kg CO_2-eq)

Global warming potential (GWP), quantified as the integrated infrared radiative resulting in greenhouse gas (GHG) increase [135].

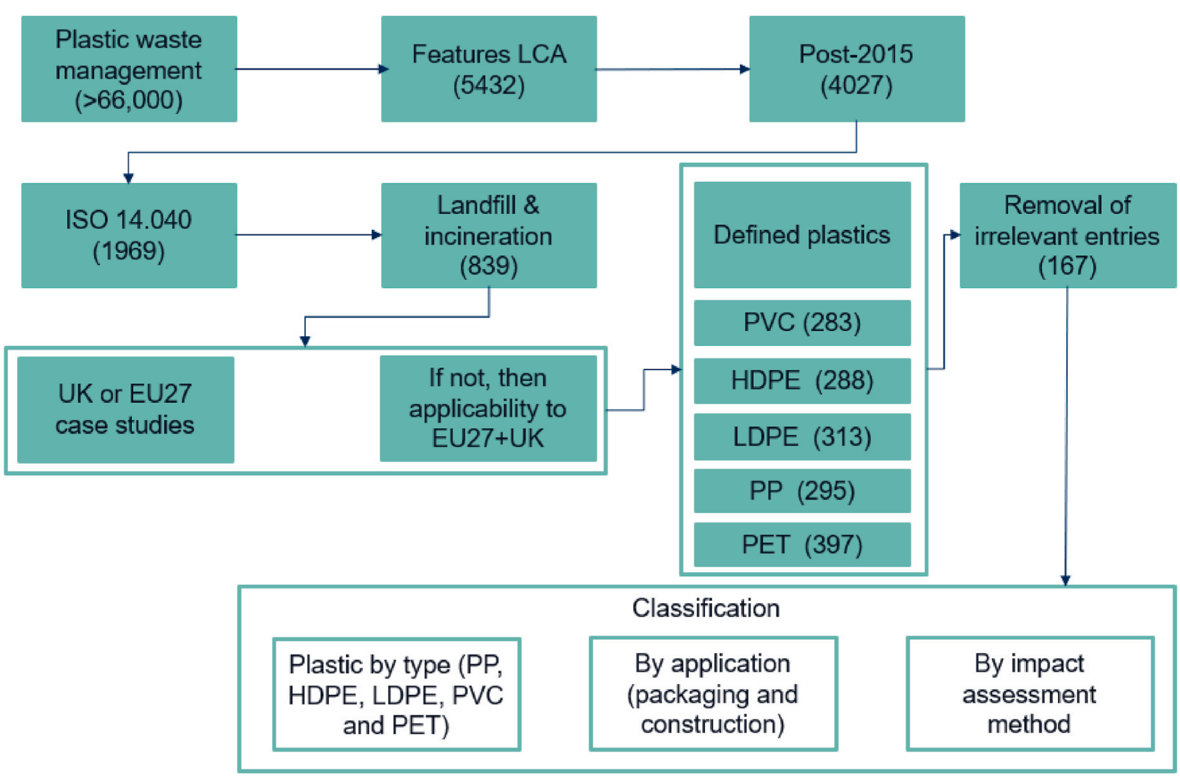

Fig. 6. The impact of the criteria set on the number of publication results.

Table 1
Summary of the findings of the review papers.

(A)REF	(B)Itemtype	(C)Scenarios	ofmidpoints	GWP	Particulatematter	Phot.Ozoneformation	Ionizingradiation	Stratos.Ozonedep	Human(cancer)	HumanToxicity	FWEcotoxicity	Marineecotoxicity	Terrestrialecotoxicity	Wateruse	Eutrophication(aquatic, freshwater)	Terrestrialeutrophication	TropOzone(eco)	Acidification	Landuse/transform	Mineralresources	Fossilresources	EnergyUse	ISO	Database
[89]	Technology comparison; Mixed plastic waste	Pyrolysis, incineration, mechanical recycling and/or energy recovery		X		x				X		X			X		X	X				X	14040/44	ReCiPe
[122]	Pet bottles		15	X	X		X	X	X	X		X			X		X	X	X	X	X		14040, LCC 15686-5	CPLCID, Ecoinvent
[123]	Medical plaste waste	(1) Incineration, (2) Landfill																						
[124]	MWCNT synthesis from pyrolysis oil	Scenario A (S-A), conversion of PET-12 to pyrolysis oil and MWCNTs; Scenario B (S–B), conversion of PET-28 to pyrolysis oil and MWCNTs; Scenario C (S–C), conversion of MVP to pyrolysis oil and MWCNTs.		X			X		X	X	X	X	X								X		14040, 14044	
[125]	Grocery bags	HDPE plastic bag (HPB), kraft paper bag (KPB), cotton woven bag (CWB), biodegradable polymer bag (BPB), and polypropylene non-woven bag (PNB) were the five variants of grocery bags studied.	9	X		X		X				X	X		X						X		14040	Ecoinvent
[126]		PP, PVC, PET, PS, PC, PE, PPA, PUR investigated in (1) recycling, (2) energy recovery, (3) industrial incineration, (4) construction aggregates, (5) landfill		X	X	X	X		X	X		X	X		X		X	X	X	X	X	X		IMPACT 2002 +
[127]		LDPE, LLDPE agricultural plastic waste in (1) once a year collection and (2) for twice a year collection																						
[128]	Food containers	1: Collection & reprocess, 2: landfill of EPS containers	16					X							X			X	X					ILCD
[129]	Water bottles	Mechanical recycling, landfilling, incineration of PET, PLA, aluminium		X		X		X		X	X				X			X	X		X			Ecoinvent
[130]	Thermoplastics	Closed loop and open loop recycling of FRCP		X				X		X	X	X	X		X		X	X				X		CML, Ecoinvent
[15]				x	x	x	x	x		x			x		x				x	x				
[92]	Polymer selection of 25 samples	Closed loop, open loop, feedstock and energy recovery for commodity plastics, engineering plastics and high performance thermoplastics		x	x	x	x	x		x			x		x				x	x			14040	R Studio
[131]		Landfill and composting of LDPE with bio-LDPE and PLA		x							x				x	x		x		x	x			ILCD 2.0

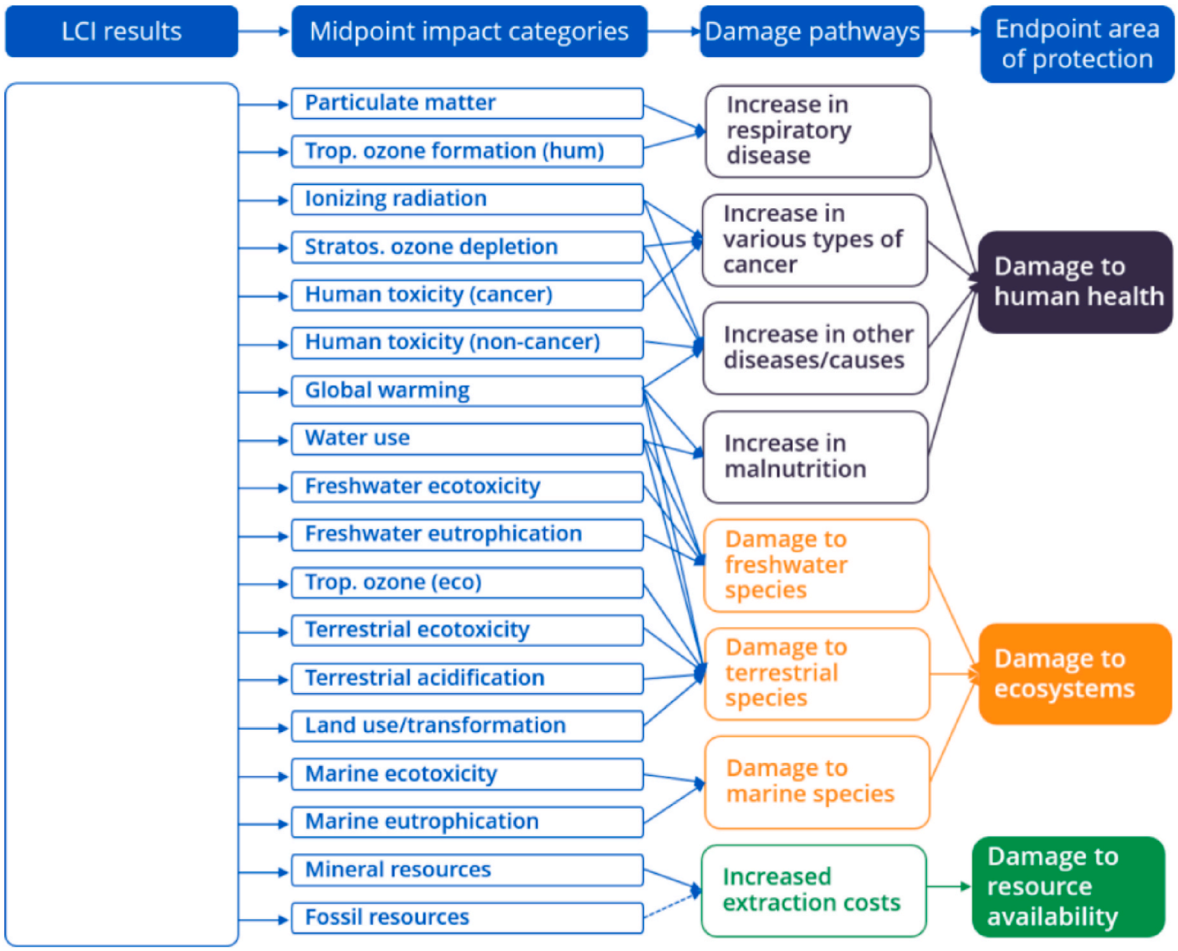

Fig. 7. Overview of the impact categories that are covered in the ReCiPe 2016 method and their relation to the areas of protection. The dotted line means there is no constant mid-to-endpoint factor for fossil resources [121].

3.1.6. *Water use* (m^3 consumption/m^3 extraction)

Different calculations are incorporated for agricultural, industrial and domestic usage.

3.1.7. *Freshwater eutrophication (kg P-eq)*

Freshwater eutrophication is characterised depending on the fate of phosphorus.

3.1.8. *Terrestrial eutrophication; acidification; land use (m^2yr annual crop-eq)*

When land is used for a specific purpose (i.e., annual or permanent crops, mosaic agriculture, urban land, forestry or pasture), the midpoint characterisation factors include relative species loss caused by this use [133]. In the study conducted by Elshout et al. [136] the relative species loss was established by comparing field data on richness of local species in specific types of natural and human-made land covers.

3.1.9. *Mineral resource scarcity (kg Cu-eq)*

For mineral resources scarcity the midpoint characterisation factor is Surplus Ore Potential (SOP), quantifying the additional amount of ore mined per additional unit of resource extracted. Bearing in mind that primary mining by concentrating the resource in ores increases the amount of ore exploited per kilogram of mineral mined [133,137].

3.1.10. *Fossil resource scarcity (kg oil-eq)*

Referred to as Fossil Fuel Potential (FFP), reflects the ratio between the higher calorific value of a fossil resource and the energy content of crude oil [133,138].

Energy use.

3.2. *Assumptions for the LCA analysis*

In the following, the focus will be on two options of the waste management hierarchy, which are landfill and energy recovery. The WFD distinguishes two categories of 'disposal' or 'recovery' based on the level of energy recovered, as incineration can fall under either of them [139]. Waste recovery, for example, is understood as incineration with highly efficient energy recovery. The energy efficiency (EE) of the installation must be $\geq$ 0.65 for facilities in operation since 2009 and $\geq$ 0.60 for facilities in operation before 2009. The EE is calculated following Equation (1)

$$EE = [Ep - (Ef + Ei)][0.97 \times (Ew + Ef)] \tag{1}$$

where,

EE = Energy efficiency.

Ep = Energy produced (electricity or heat) in GJ/year.

Ef = Energy consumption as fuel in GJ/year.

Ew = Energy content of wastes in GJ/year.

Ei = Annual imported energy excluding Ew and Ef in GJ/year [140].

In terms of the methodology employed in this paper, the figures obtained from the published reports and Life Cycle Inventory (LCI) analysis conducted by Shonfield [141], will be used for analysis.

In order to create a balanced evaluation between landfill and waste recovery, two models that incorporate supply chains for the recycling processes will be developed. This means that for each process, the amount of input of materials entering the system will be the same. Also, it is assumed that the major plastic types are separated from the non-recycled materials (such as general waste) and fully recycled. It is indicated that for the purpose of comparison between the recycling methods and techniques mentioned, the disposal, processing and recycling of 2.2 tonnes of waste general plastic (including non-recyclable materials) delivered to a recycling facility will be considered.

The technologies investigated in this study are selected based on the current methods commonly used by recycling facilities in the United Kingdom for recycling general plastic or the state-of-the-art technologies which have been developed and proven through current research and studies and can be used in the future. The study geographically will aim to only consider the current condition in the UK, using the specific data provided for recycling options, transport distances or energy consumption. Specifically, it will use domestic waste that is collected and sorted by MRFs (Material Recycling Facilities) in the UK where, recyclable materials including general mixed plastic is segregated and recycled.

It should be mentioned that the collection process of the plastic materials may not be considered as a benefit of this technique, as such a process does not exist. The results derived from this study and for comparison will therefore only include the output of recycled materials from a MRF and specific plastic types.

This, for instance, means that for a model the system boundaries are defined from the moment when the product of the recycling process leaves the recycling system and a new product as a new bulk material, or energy is produced. Fig. 8 shows an example of the system boundaries of a certain state of the art technology and a MRF.

3.2.1. Technology

This LCA report studies the selected state-of-the-art technologies and

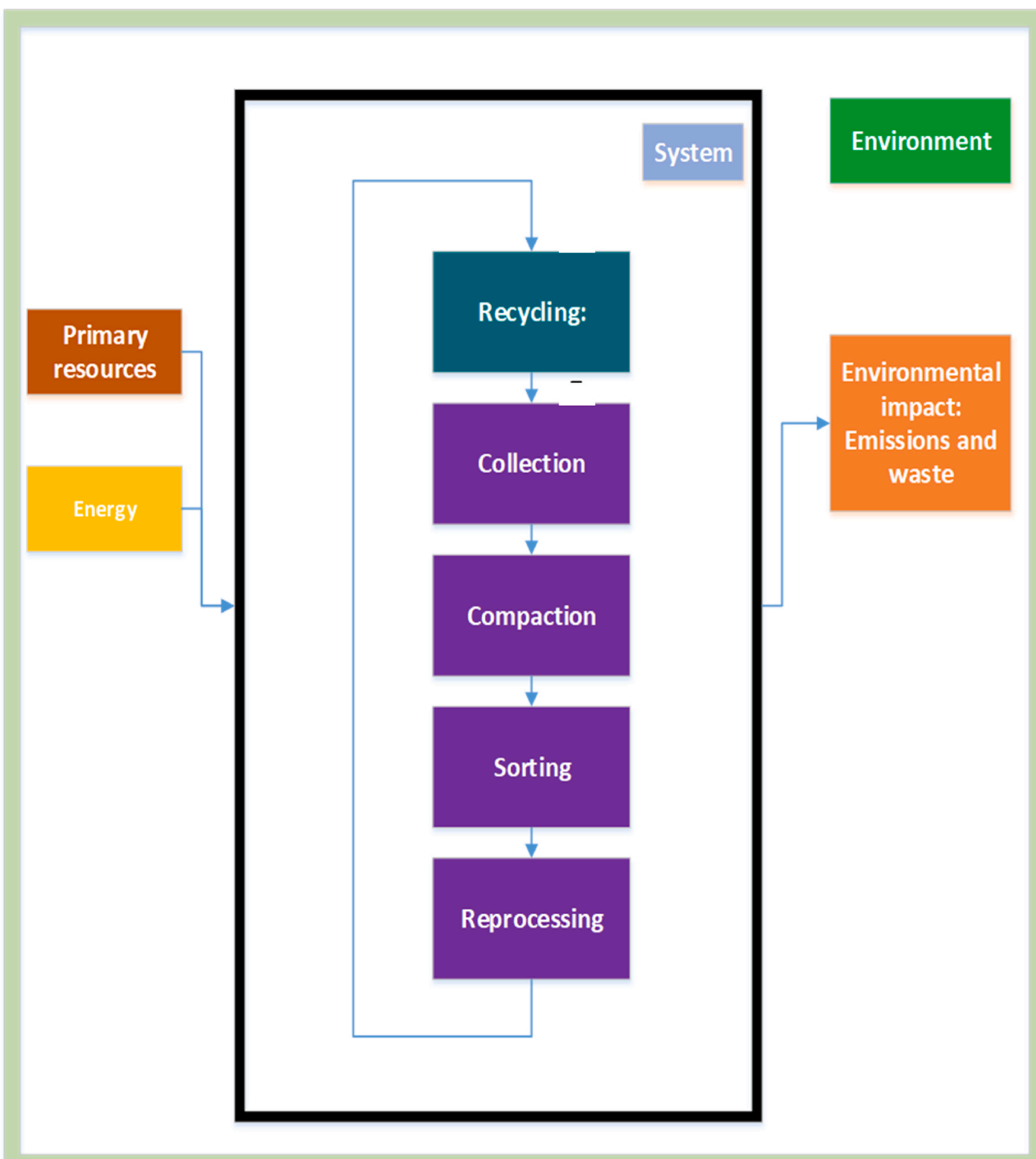

Fig. 8. Example process diagram showing the system boundaries.

investigates their potential for domestic general mixed plastic waste recycling. The LCA will consider and analyse the results obtained from the studies conducted for recycling information and utilities' requirements of Waste and Resources Action Programme (WRAP) [141, 142].

3.2.2. Geography

The main purpose of this study is to analyse and investigate the plastic waste recycling options for the UK. There was therefore an attempt to gather specific data from the UK recycling facilities and where such information was not available, the data from recycling facilities from Western European countries (mainly France and Germany) have been used, as currently the European regulations are followed for energy consumption and efficiency.

3.2.3. Municipal waste disposal

The UK is a major contributor of plastic waste, generating more plastic waste per person than any other country save the USA (Carrington D (2020)) "US and UK citizens are world's biggest sources of plastic waste" [143]. For instance, in 2018 the UK generated approximately 5.2 million tonnes of plastic waste with nearly half of it belonging to plastic packaging [144]. In 2016, the majority of plastic waste (91%) that went to treatment went to 'recycling and other recovery', with the remainder (9%) going to landfill [10]. Data obtained from Ref. [145] demonstrates that from the waste streams which are not currently recycled, about 16% are incinerated and 84% are landfilled. In this regard, these methods have been selected to be considered for comparison and analysis. It is also indicated that almost 99.8% of the incineration process is used to recover energy that meets the WFD requirements. Therefore, 100% of the incineration process is assumed to be for energy recovery.

3.2.4. Transport

The following assumptions have been made in regard to the model of the transport and the distance travelled by the waste.

Transportation is assumed to be conducted by a 32-tonne lorry with full load on outward and empty load on inward journeys to and from the facility.

The lorry fully complies with the EU emissions rules.

Transport distances for this study are presented in Table 2 below. These figures indicate a best representation of the distance and are obtained from Ref. [141].

3.2.5. Electricity consumption

Electricity use for this study is assumed to be entirely based on the average UK electricity generation produced through a combined cycle power plant. This approach and according to Ref. [146] complies with the UK Government guidelines on evaluation of greenhouse gas policy.

3.2.6. Inventory analysis

The life cycle inventories produced for each case are made from both the input and output of the process in question. The output parameters are based on the study conducted by Ref. [147] for plastics (as input) such as PET, HDPE, PP, PS and PVC and they include:

- Process energy (Energy)
- Abiotic Depletion Potential (ADP)
- Acidification Potential (AP)
- Eutrophication Potential (EP)
- Human Toxicity Potential (HTP)
- Ozone Layer Depletion Potential (OLDP)
- Photochemical Ozone Creation Potential (POCP)
- Global Warming Potential (GWP)

Table 2
Transport distances used in the LCA models.

Route	Distance, km
From MRF to polymer sorting facility	50
Typical distance to landfill site	20
Typical distance to incinerator (energy recovery facility)	50

3.3. Indicator metrics

As a guidance to understand the intrinsic differences among LCIA methodologies and the path applied to make them comparable, Table 3 shows the indicator units for impact categories among the investigated mid-point methodologies and Table 4 provides the conversion factors for the metrics harmonization.

It is also mentioned that these parameters that incorporate the LCI are produced for each individual process and developed for the entire system boundary based on the quantities of energy, material, waste and emission relevance [148]. The LCI shown in Table 5 indicates the environmental impacts relative to the production of 2.2 tonnes of waste plastic.

The following section will analyse the two different case studies, as each waste management option will produce different types and amounts of recycling product and energy. It is important to consider how many advantages and disadvantages each technology offers by scoring each individual case.

4. LCA analysis of thermal treatments for waste management

The model (Fig. 9) illustrates the process diagram for incineration energy recovery. The model as explained by Ref. [149] takes into account several assumptions which are stated below:

- The plastic waste is not sorted and is sent "mixed" from the MRF facility to the incineration plant for processing.
- The waste is burned through a gas-fired power unit and the energy recovered from the process is used to produce electricity.

The model boundaries for the plastic waste defined for this case, as presented in the diagram above, start from the MRF facility to the incineration plant. This means that the results obtained from this study do not take into consideration the energy and material consumption used to produce the waste in the first place.

The main advantage of the incineration process is the recovery of energy and the production of electricity from the waste stream. The energy which is recovered from the mixed waste plastic is derived from the net calorific values shown in Table 6 [150].

Table 3
Indicator metrics for impact categories among the investigated mid-point methodologies [32].

	EDIP 97/2003	CML 2001	Impact 2002+	ReCiPe	ILCD
Global warming/ climate change	kg CO_2-eq	kg CO_2-eq	kg CO_2-eq	kg CO_2-eq	kg CO_2-eq
Ozone depletion	kg R11-eq	kg R11-eq	kg CFC-11-eq	kg CFC-11-eq	kg CFC-11-eq
Acidification	kg SO_2-eq	kg SO_2-eq	kg SO_2-eq	kg SO_2-eq	AE
Eutrophication	kg NO_3-eq	kg PO_4-eq	kg PO_4-eq	kg P-eq	kg P-eq
Photochemical oxidation	kg C_2H_4-eq	kg C_2H_4-eq	kg C_2H_4-eq	kg NMVOC-eq	kg NMVOC-eq
Human toxicity	m^3	kg 1,4-DCB-eq	–	kg 1,4-DCB-eq	Cases
Ecotoxicity	m^3	kg 1,4-DCB-eq	kg TEG-eq	kg 1,4-DCB-eq	–
Water depletion	–	–	m^3	m^3	m^3

Table 4
Indicator metrics – value.

	EDIP 97/2003	CML 2001	Impact 2002+	ReCiPe	ILCD
Global warming/ climate change	1.0				
Ozone depletion	1.0				
Acidification	1.0				–
Eutrophication	1.0	10.44	10.44	32	4.43
Photochemical oxidation	1.7	1.7	1.7	1.0	1.0
Human toxicity	1.40E−5 (air) 1.12E−2 (water) 1.43 (soil)	1.0	–	1.0	–
Ecotoxicity	1.40E−5 (air) 1.12E−2 (water) 1.43 (soil)	1.0	1.62E−3	1.0	–
Water depletion	–	–	–	m^3	m^3

Comparing the above net calorific values of the analysed plastics with the typical most commonly used fuels such as: methane (53 MJ/kg) [151], LPG (46.3 MJ/kg) [152], natural gas (38.1 MJ/kg), diesel (42.6 MJ/kg) [153], hard coal (16.7–29.3 MJ/kg) [154], wood pellets (17.3 MJ/kg) or the calorific value of municipal solid waste in UK (6.7–7.0 MJ/kg) bearing in mind that in other countries this value can vary significantly [155,156], it can be concluded that the potential amount of energy to be recovered from waste plastics is significant. However, it will depend on the proportions of the individual components of the plastic waste. If it is assumed that the 2.2 tonnes analysed contain only PE, the amount of energy recovered will be significantly higher than if the same amount of plastic waste contained mainly PVC.

The dataset obtained for the incineration also takes into account the additives used to prepare the waste before and after the recycling process. Tables 7 and 8 show the impact assessment of the incineration process.

5. LCA analysis of landfills for waste management

The model (Fig. 10) represents the landfill method, which illustrates the effects of transportation of the waste to the landfill site and the impacts of the landfill site on the environment.

LCI data used for this case are gathered from Ref. [141] which takes into account the untreated waste for a municipal sanitary landfill. The landfill model in this case considers gases produced from the landfill, leachate (water that passed through the waste) and the wastewater treatment. The data also account for:

- Emission over short and long-term period via landfill gas and leachate with no energy recovery.

Table 6
Net calorific value for different type of plastic.

Plastic Type	Net calorific value (MJ/kg)
PE	42.47
PP	30.78
PET	22.95
PS	38.67
PVC	21.51
PLA	30.79
Paper	14.12
Residuals (as aluminum or steel)	0 (removed as solids waste)

Table 7
Case study I (incineration) impact assessment results.

Impact Category	Unit	Low Polyolefin	Default	High Polyolefin
Energy	MJ	−18574.6	−26582.6	−31112.4
ADP	kg eq. Sb	−9.3016	−12.067	−13.728
AP	kg eq. SO_2	1.2056	0.121	−0.407
EP	kg eq. PO_4^+	0.2178	0.099	0.0374
HTP	kg eq. DCB	2706.99	2970.11	3079.12
OLDP	kg eq. R11	−1.54E-04	−1.91E-04	−2.16E-04
POCP	kg eq. C_2H_4	−0.0264	−0.132	−0.1914
GWP	kg eq. CO_2	3583.8	4023.8	4221.8
Solid Waste	kg	116.6	107.8	105.6

Table 5
Environmental impacts relative with the production of 2.2 tonnes of waste plastic.

Impact Category	Units	PET	HDPE	PP	PS	PVC
Process energy (Energy)	MJ	95339.2	58077.8	53671.2	95891.4	85450.2
Abiotic Depletion Potential (ADP)	kg eq. Sb	72.6	72.6	72.6	83.6	39.6
Acidification Potential (AP)	kg eq. SO_2	26.4	46.2	44	37.4	22
Eutrophication Potential (EP)	kg eq. PO_4^+	6.6	2.2	2.2	4.4	2.2
Human Toxicity Potential (HTP)	kg eq. DCB	1617	147.4	110	121	332.2
Ozone Layer Depletion Potential (OLDP)	kg eq. R11	0	0	0	0	0
Photochemical Ozone Creation Potential (POCP)	kg eq. C_2H_4	4.4	6.6	4.4	4.4	2.2
Global Warming Potential (GWP)	kg eq. CO_2	5429.6	4160.2	4397.8	6107.2	2939.2

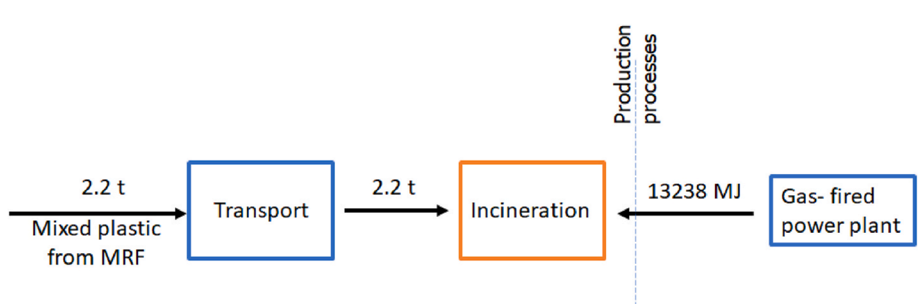

Fig. 9. Process diagram for Case Study I (incineration) - Mixed plastic input (MRF).

Table 8
Case study I (incineration) impact assessment results showing different process stages.

Impact Category	Unit	Incineration	Transport	Avoided Impacts	Total
Energy	MJ	3933.6	422.4	−30938.6	−26582.6
ADP	kg eq. Sb	1.1198	0.187	−13.3738	−12.067
AP	kg eq. SO_2	1.4982	0.1364	−1.5114	0.121
EP	kg eq. PO_4^{+}	0.2486	0.0308	−0.1826	0.099
HTP	kg eq. DCB	2985.818	1.606	−17.314	2970.11
OLDP	kg eq. R11	1.39E-05	3.52E-06	−2.09E-04	−1.91E-04
POCP	kg eq. C_2H_4	0.1672	0.0154	−0.3146	−0.132
GWP	kg eq. CO_2	5704.6	28.6	−1709.4	4023.8

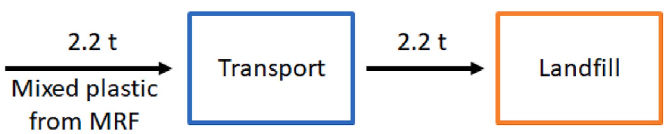

Fig. 10. Process diagram for Case Study II (landfill).

- Treatment of leachate over short and long-term period in a wastewater treatment plant.

The parameters for the input materials take into consideration the effects of the chemical products which are used to process and purify the waste for recycling.

Tables 9 and 10 show the case study impact assessment results.

The results presented for the case study I (incineration) and II (landfill) can be used to rank the impact categories in order of importance: GWP, solid waste, energy, HTP, EP, POCP, AP, OLDP. Specific impact categories can also be analysed at different scales: global-GWP, ADP and OLDP; regional-AP; as well as local-EM, POCP and HTP. Based on the results presented in Tables 7–10, it was demonstrated that landfill presents the worst choice for all different assessment categories, particularly in the context of solid waste. However, when considering the main impact category: GWP, incineration has a more significant environmental impact.

6. Conclusion

Building on the EU's plastic strategy, which is an important element in the EU's move towards a circular economy, there is a clear direction to reduce plastic waste and simultaneously improve the management of plastic waste generated. Despite leaving the EU, the UK government as part of its 25 Year Environmental Plan, has also taken steps to eliminate all avoidable plastic waste by 2042, including introducing a world-leading Plastic Packaging Tax for packaging with less than 30% recycled plastic. At the current rate of recycling, which was found to be below 30% [157]), there was shown to be substantial room to grow before the plastics value chain becomes circular. Thus, the goal of this paper was to highlight the detrimental effects of one of the most utilised options of waste management in the UK when compared to other frequently suboptimal practices of waste disposal. Moreover, this paper also aimed at illuminating the advantages and disadvantages of waste landfilling and incineration through conducting a LCA study. Tables 7–10 for the two scenarios analysed show the environmental effects in each impact category. Furthermore, it was pointed out that the amount of energy that could be recovered as a result of the incineration

Table 9
Case study II (landfill)-impact assessment results.

Impact Category	Unit	Low Polyolefin	Default	High Polyolefin
Energy	MJ	968	1007.6	1020.8
ADP	kg eq. Sb	0.3938	0.4114	0.4158
AP	kg eq. SO_2	0.5786	0.528	0.5082
EP	kg eq. PO_4^{+}	3.2714	2.3166	1.7424
HTP	kg eq. DCB	2495.482	3664.606	4250.026
OLDP	kg eq. R11	8.80E-06	9.02E-06	9.24E-06
POCP	kg eq. C_2H_4	0.0968	0.1056	0.1078
GWP	kg eq. CO_2	323.4	349.8	365.2
Solid Waste	kg	2200	2200	2200

Table 10
Case study II (landfill) -impact assessment results illustrating different process stages.

Impact Category	Unit	Landfill	Transport	Total
Energy	MJ	838.2	169.4	1007.6
ADP	kg eq. Sb	0.3366	0.0748	0.4114
AP	kg eq. SO_2	0.473	0.055	0.528
EP	kg eq. PO_4^{+}	2.3056	0.0132	2.3166
HTP	kg eq. DCB	3663.968	0.638	3664.606
OLDP	kg eq. R11	7.70E-06	1.45E-06	9.02E-06
POCP	kg eq. C_2H_4	0.099	0.0066	0.1056
GWP	kg eq. CO_2	338.8	11	349.8

process is significant as it depends on the net calorific value, which for plastic waste is comparable to that of coal or diesel (depending on the proportion of each type of plastic in the mixed plastic waste).

The results generated from this study can be used to rank the impact categories in order of importance. Based on this, it was demonstrated that landfill presents the worst choice for all different assessment categories.

Authorship contributions

Please indicate the specific contributions made by each author (list the authors' initials followed by their surnames, e.g., Y.L. Cheung). The name of each author must appear at least once in each of the three categories below.

Category 1

Conception and design of study: All Authors

Category 2

Drafting the manuscript: All Authors

Category 3

Approval of the version of the manuscript to be published (the names of all authors must be listed): All Author

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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