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Plastic waste management in the context of a European recycling society: Comparing results and uncertainties in a life cycle perspective

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A B S T R A C T

A number of life cycle assessment (LCA) studies have been undertaken within the last 15 years comparing end-of-life treatment options for post-consumer plastic waste, including techniques such as: mechanical recycling, feedstock recycling, incineration with energy recovery and landfilling. These have attempted to support decisions in the formulation of waste management strategies and policies. In light of the introduction of life cycle thinking into European waste policies, specifically in relation to the waste hierarchy, a literature review of publically available LCA studies evaluating alternative end-of-life treatment options for plastic waste has been conducted. This has been done in order to: establish if a consensus exists as to the environmentally preferable treatment option for plastic waste; identify the methodological considerations and assumptions that have led to these conclusions; and determine the legitimacy of applying the waste hierarchy to the plastic waste stream. The majority of the LCA studies concluded that, when single polymer plastic waste fractions with little organic contamination are recycled and replace virgin plastic at a ratio of close to 1:1, recycling is generally the environmentally preferred treatment option when compared to municipal solid waste incineration. It has been found that assumptions relating to the virgin material substitution ratio and level of organic contamination can have a significant influence upon the results of these studies. Although a limited number of studies addressed feedstock recycling, feedstock recycling and the use of plastic waste as a solid recovered fuel in cement kilns were preferred to municipal solid waste incineration. Landfilling of plastic waste compared to municipal solid waste incineration proved to be the least preferred option for all impact categories except for global warming potential. Due to the uncertainty surrounding some assumptions in the studies, it cannot be said with confidence that the waste hierarchy should be applied to plastic waste management as a general rule.

1. Introduction

1.1. Background – plastic waste management in the EU

In 2007, world production of plastics rose to approximately 260 million tonnes. In Europe, this resulted in the generation of 24.6 million tonnes of post-consumer plastic waste concentrated in the packaging, construction, automotive and electrical and electronic equipment sectors. Half of this waste was disposed of in landfills, whilst 20% was recycled and 30% was recovered as energy (Plastics Europe, 2008).

Municipal solid waste management has been predominantly driven by European waste and natural resource policy, directed by a rationale of protection of human health and the environment and, more recently, sustainability. This is aimed at reducing the negative environmental impact of waste management within economic, technological and social constraints. In 2005 the European Commission's Thematic Strategy on the prevention and recycling of waste established new aims and objectives for EU waste policy. Additionally, it put forward a vision of an EU "recycling society" as its long term goal (European Commission, 2005).

Since as early as 1994, objectives for plastic waste recycling and recovery have been set at the EU level. The directive on packaging and packaging waste 94/62/EC (Council Directive, 1994) set a minimum recycling target of 15% by weight for all packaging waste streams including plastic. This was later amended to 22.5% by weight for plastics counting exclusively material that is recycled back into plastic (Council Directive, 2004), with no

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distinction among the various streams of plastic waste (e.g., industrial, commercial, office, service or household waste). The new Waste Framework Directive (WFD) 2008/98/EC (Council Directive, 2008) has set a 50% recycling target for household waste (including at least paper, metal, plastic and glass streams) and a 70% recycling target for construction and demolition waste. However, the directive does not stipulate if the 50% recycling target should apply to the municipal waste stream as a whole or individual material fractions within this stream.

It also establishes the following waste hierarchy to be applied as a “priority order”: prevention, preparing for reuse, recycling, other recovery – e.g. energy recovery, and disposal.

Directive 2008/98/EC allows specific waste streams to depart from the waste hierarchy when justified by life cycle thinking. The use of life cycle thinking and life cycle assessment (LCA) in EU waste directives is also highlighted by Directive 94/62/EC, “whereas life-cycle assessments should be completed as soon as possible to justify a clear hierarchy between reusable, recyclable and recoverable packaging” (Council Directive, 1994). This places considerable expectations on life cycle thinking and LCA studies to be robust, transparent and deliver reproducible outcomes if the tool is to gain broad acceptance as a legitimate environmental systems analysis tool for waste policy and strategy.

Several reviews of LCA studies have been published on municipal solid waste management systems reviewing studies of individual material fractions and the municipal solid waste stream (Björklund and Finnveden, 2005; Cleary, 2009; Finnveden and Ekvall, 1998; Villanueva and Wenzel, 2007; WRAP, 2006). These studies have been carried out in order to determine whether a common theme exists within the LCA literature in terms of environmentally preferable waste treatment scenarios and the key methodological issues surrounding these studies. This review focuses on post-consumer plastic waste management and the uncertainties that may have an influence upon the results of these studies.

1.2. Aim of the paper

EU waste management policy instruments aim at establishing a trajectory towards waste avoidance and increased material efficiency through re-use and recycling. The purpose of this review is to establish the legitimacy of this trajectory for plastic waste, by determining if a consensus exists on the environmentally preferred waste management options for this stream.

This paper provides an analysis of publicly available studies comparing the environmental impacts of end-of-life treatment options for post-consumer plastic waste in Europe, using an LCA framework. The review aims to determine if there is a consensus as to the preferred plastic waste treatment option from an LCA perspective and identify the main methodological considerations and uncertainties, in order to discuss the application of the waste hierarchy to plastic waste management.

2. Method

2.1. Inventory and selection of existing studies

2.1.1. Selection criteria

More than 50 studies based on an LCA methodology were found to investigate the environmental impacts of post-consumer plastic waste management. Ten high quality studies in line with the aims of this review were selected (see Table 1), using the following criteria.

(1) *Support decisions in plastic waste management systems*: the present review focuses on LCAs of plastic waste management, and has selected studies comparing various end-of-life treatments, i.e. having a functional unit based on the end-of-life treatment of plastic waste.
(2) *Follow the ISO 14040 framework* (ISO, 2006): the studies selected were required to follow the ISO 14040-series (or a compatible framework for the studies that were conducted before the first publication of the ISO 14040-series in 1997).
(3) *Transparency of system boundaries and assumptions*: studies were selected that attempted to be transparent in their assumptions and treatment of system boundaries, background data was also required in order to conduct a comparison of scenarios.

2.1.2. Plastic waste management scenarios

The selected studies included a total of 112 scenarios of plastic waste management, of these scenarios 77 were selected and grouped into four categories: mechanical recycling, feedstock recycling, incineration and landfill. Plastic waste incinerated or sent to landfill was considered to be treated as a part of the municipal solid waste stream. Whilst some of the scenarios for plastic waste management include several treatment technologies, for example material and/or energy recovery and the landfill of residual solid waste from these processes, scenarios have been categorised based on the dominant technology modelled in each scenario.

2.2. Comparison of scenarios

2.2.1. End-of-life technologies considered in the scenarios

Mechanical recycling technologies involved processes including the separation of polymer types, decontamination, size reduction, remelting and extrusion into pellets. Sorting technologies modelled in the LCA studies included near-infra red sorting and float/sink density separation.

Feedstock recycling is defined as the transformation of plastic polymers by heat or chemical agents back to hydrocarbon products that can be used in the production of new polymers or as chemical feedstock (Aguado et al., 2006). Feedstock recycling technologies assessed by the studies included pyrolysis, gasification, hydrocracking, plastic waste as a reducing agent in blast furnaces for steel manufacture and Watech and Stigsnæs processes for PVC waste. These processes produced chemical feedstock products which were assumed to replace virgin chemical feedstock.

Incineration technologies considered in the LCA studies included (1) municipal solid waste (MSW) incineration, and (2) the combustion of plastic waste as part of a solid recovered fuel (SRF) in cement kilns. Recovery of energy was considered by all scenarios as either heat and electricity, only heat or only electricity. The type of energy recovered depended upon assumptions in line with existing national practices and infrastructure.

Landfill technologies modelled consisted of engineered landfills with both leachate and landfill gas extraction. Impacts from the degradation of plastic waste depended upon the modelling assumption in each study (see Section 5.5).

For incineration and landfill scenarios, plastic waste is treated as a part of the MSW stream and only the environmental impacts associated for treatment of the plastic waste stream were considered.

2.2.2. Comparison

To determine which scenarios generate fewer environmental impacts, scenarios were compared in the following categories which reflect the order of the waste hierarchy: (1) mechanical recycling compared to MSW incineration, (2) mechanical recycling compared to feedstock recycling, (3) feedstock recycling compared to MSW incineration and combustion as an SRF and (4) MSW incineration compared to landfill.

Table 1
Overview of LCA studies.

Reference	Waste stream	Polymers	Treatment technologies	No. of scenarios	Commissioner	Geographical scope
Carlsson (2002)	Household plastic packaging waste	HDPE, LDPE	MR MSWI	3	Government (European Commission)	Sweden (National)
Eriksson and Finnveden (2009)	(1) Non recyclable plastic (2) Mixed plastic	Not clear	MSWI LF	36	n.a.	Sweden and European (n.a.)
Frees (2002)	(1) Transport packaging (2) Plastic bottles and cans from households and businesses	HDPE, LDPE, PP	MR MSWI LF	19	Industry (Danish EPA)	Denmark (National)
Jenseit et al. (2003)	ELV plastic components	PP, PE, PC, PU, ABS, PA	MR MSWI SRF in a cement kiln FR – gasification FR – reduction agent in a blast furnace LF	6	Industry (Association of Plastics Manufacturers Europe)	European (Data: DE)
Kreißig et al. (2003)	PVC cable waste	PVC	MR – Vinyloop FR – Stigsneas FR – Watech PVC MSWI LF	5	Industry (Vinyl2010)	European (Data: DE/DK)
Perugini et al. (2005)	Household plastic packaging waste	PET, PE	MR MSWI LF FR – low temp. pyrolysis & MR FR– hydrocracking & MR	5	Industry (CONAI & CoRePla)	Italy (National)
Raadal et al. (2008)	Household plastic packaging waste	HDPE, LDPE, PP, PET, PS	MR MSWI LF	3	Industry (Grønt Punkt Norge)	Norway (National)
RDC and Coopers & Lybrand (1997)	Plastic packaging waste	PET, PVC, HDPE, LPDE	MR MSWI	16	Government (European Commission)	European (Data: CH/DE)
Shonfield (2008)	Mixed post-consumer plastic waste	HDPE, LDPE, PP, PET, PVC, PS	MR MSWI SRF in cement a kiln FR – pyrolysis FR– reduction agent in a blast furnace LF	16	Government (Waste Action Resources Programme)	UK (National)
Von Krogh et al. (2001)	Plastic bottles for foodstuffs	HDPE	MR MSWI LF	3	Industry (Stabburet AS and Plastretur AS)	Norway (Regional)

MR – mechanical, FR – feedstock recycling, MSWI – municipal solid waste incineration, SRF – incinerated as a solid recovered fuel, LF – landfill.

Among previous reviews of LCA studies, several methods for comparison have been used. These include the overall comparison of scenarios, assessing the order from highest to lowest, of environmental impacts for various scenarios (Björklund and Finnveden, 2005), sometimes in greater depth (Cleary, 2009) and the comparison of the relative difference between treatment scenarios (Villanueva and Wenzel, 2007; WRAP, 2006).

In this paper the relative difference between scenarios is assessed, using the method described in Villanueva and Wenzel (2007) and WRAP (2006). This has been done for impact categories relevant to the goal and scope of the studies (see Section 3.4). In order to obtain a preference between waste treatment scenarios within the same study, the relative comparison of the scenarios was calculated. For example, in the comparison of mechanical recycling and MSW incineration the following formula was used for the assessment of impact categories:

$$\frac{(\text{Environmental impacts of mechanical recycling} - \text{Environmental impacts of incineration}) \times 100}{\text{Environmental impacts of incineration}}$$

In this case, a negative value signifies that recycling provides less environmental impacts than incineration and vice-versa. This was repeated for all scenario comparisons.

3. Description of selected studies

The results of LCA studies are dependent upon their goal and scope, the establishment of their system boundaries, as well as some key assumptions, such as the replaced material and/or energy from the recovered material and/or energy. The following section outlines the context in which the studies were carried out, including the goals of the studies, functional units used, impact categories considered, and use of sensitivity analysis. Table 2 details some key assumptions of the LCA studies.

3.1. Goals

The goal definition phase in LCA is important as it should unambiguously state the intended application, the purpose for carrying out the study, the intended audience and whether the results are to be disclosed to the public (ISO, 2006). These requirements were included to different extents in the studies. Goal definition was evident in all the studies and, in addition to the *environmental assessment of different treatment scenarios for plastic waste management*, they included; identification/mapping of recycling activities at a national level (Carlsson, 2002), improvement of recycling processes in LCA software (Carlsson, 2002), investigation of collection and end-use market potential of recycled material (Frees, 2002), and economic assessment of plastic waste management with the integration of these results into an eco-efficiency assessment (Jenseit et al., 2003; Kreißig et al., 2003). Whilst each study clearly stated their goals, the intended application and purpose of the studies were not always as clearly defined.

3.2. Functional units

The functional unit is the way in which a system's functional benefit is measured. It is fundamental to LCA as it allows the comparative assessment of two or more different systems that provide the same function (Rebitzer et al., 2004). ISO (2006) requires functional units to be "clearly defined and measurable".

The functional units of the selected studies can be classified as follows:

(1) the waste management or treatment of a certain quantity of plastic waste, for example: "...the recycling, reprocessing or disposal of 1 tonne of mixed plastic (and other residual materials) arising as waste from a typical UK materials recycling facility" (Shonfield, 2008),
(2) the management of plastic waste that leads to the production of a certain quantity of recycled material, for example: "the management of postconsumer PE and PET liquid containers ... which leads to the production of 1 kg of flakes of (recycled or virgin) PET" (Perugini et al., 2005),
(3) the end-of-life management of a plastic object, for example: "treatment of a discrete plastic component in end-of-life vehicles" (Jenseit et al., 2003).

In effect, scenarios using these different functional units can be compared because all are related to a mass of plastic waste treated. The difference in mass does not pose a problem to the comparison in this review, as this is subjugated by the comparative analysis (Section 2.2). Whilst the primary function of the all studies was the same, the treatment of plastic waste, scenarios also produced additional products, such as material and/or energy. All studies address this issue by undertaking either the system expansion or substitution (avoided burdens) to make the scenarios comparable. For example, when comparing MSW incineration to landfill, the MSW incineration scenario not only treats plastic waste but generates energy whereas the landfill scenario only treats the waste. To make these systems comparable the system boundaries of the landfill scenario are expanded to included the production of an equivalent amount of energy (average or marginal production based upon study assumptions). Hence equivalent systems can be compared.

3.3. Geographical scope and scale

National and transnational geographical scales were considered in the selected studies. Five of the studies (see Table 1) were commissioned in a national context. Jenseit et al. (2003), Kreißig et al. (2003) and RDC and Coopers & Lybrand (1997) were commissioned to support decisions in a European context, however data were sourced from Germany and Denmark. Carlsson (2002), Raadal et al. (2008) and Frees (2002) assessed the transnational transportation and treatment of plastic waste, an increasingly common occurrence as plastic waste for recycling is traded on scrap markets at a global scale.

3.4. Collection and transport related emissions

All studies included transport related emissions, however these were considered to different extents within the studies. Carlsson (2002), Frees (2002), Perugini et al. (2005), Raadal et al. (2008) and Von Krogh et al. (2001) included collection related emissions. Jenseit et al. (2003) included transport emissions from end-of-life vehicle pre-treatment to shredding but did not consider emissions associated with end-of-life vehicle collection. Eriksson and Finnveden (2009), Kreißig et al. (2003), Shonfield (2008) excluded emissions related to plastic waste collection. All studies included the transport of waste from sorting processes to either; mechanical recycling, feedstock recycling, incineration with energy recovery or landfill processes. No further consideration of transport was considered after the manufacture of recycled plastic pellets.

3.5. Impact assessment

Environmental impact categories considered by the LCAs studies are shown in Table 3 .

Table 2
Key LCA parameters and assumptions.

Reference	Plastic waste polymers	Avoided material	Virgin material substitution ratio considered	Avoided electricity	Avoided heat	Transport	Time horizon LF emissions to air	Time horizon LF emissions to ground water
Carlsson (2002)	HDPE, LDPE	Virgin PE, PET (wood)	1:1 1:0.8 for HDPE 1:0.7 for LDPE	n.a.	Biomass coal	Collection excluded	n.a.	n.a.
Eriksson and Finnveden (2009)	Not clear	n.a.	n.a.	Wind coal	Oil Biomass CHP	Collection excluded	100 years	n.a.
Frees (2002)	HDPE, LDPE, PP	Virgin HDPE, LDPE, PP	1:0.9	Danish average natural gas	Danish average (natural gas) (oil and gas)	Collection included	Not clear	Not clear
Jenseit et al. (2003)	PP, PE, PC, PU, ABS, PA	Virgin PP	1:1	Not clear	Not clear coal: SRF – cement kiln	Collection excluded	Not clear	Not clear
Kreißig et al. (2003)	PVC	Virgin PVC	1:1	Not clear	Not clear	Collection excluded	Not clear	Not clear
Perugini et al. (2005)	PET, PE	Virgin PET, PE	1:1	Italian average	n.a.	Collection included	Not considered	Not considered
Raadal et al. (2008)	HDPE, LDPE, PP, PET, PS	Virgin HDPE, LDPE, PP, PET, PS	1:0.95	n.a.	Oil and electricity	Collection included	Not considered	Not considered
RDC and Coopers & Lybrand (1997)	PET, PVC, HDPE, LPDE	Virgin PET, PVC, HDPE, LPDE	1:1 1:0.5	Not clear	n.a.	Collection included	n.a.	n.a.
Shonfield (2008)	HDPE, LDPE, PP, PET, PVC, PS	Virgin HDPE, LDPE, PP, PET, PVC, PS (wood) (concrete)	1:1 1:0.2	Natural gas UK average	Coal: SRF – cement kiln	Collection excluded	100 years	60 000 years
Von Krogh et al. (2001)	HDPE	HDPE	1:1	n.a.	Oil	Collection included	Not clear	Not clear

ISO (2006) states the selection of impact categories should to be consistent with the goal and scope of the study, and reflect the main environmental issues related to the product/service system. This review has compared impact categories of global warming potential (GWP), acidification potential (AP), eutrophication potential (EP), energy use (EN) and residual solid waste destined for landfill (SW), as these impact categories were assessed in half or more of the studies. Abiotic resource depletion potential (ADP) was also assessed considering that material and energy recovery from plastic waste replaces the need for fossil feedstock.

A number of different impact assessment methods were used including, CML (Guinée, 2001) in Shonfield (2008) and Kreißig et al. (2003), EDIP (Wenzel et al., 1997) in Frees (2002), IPCC, 2007 (IPCC, 2007) in Eriksson and Finnveden (2009) and weighting factors developed for ORWARE (Sundqvist et al., 1999) in Carlsson (2002). Other studies did not specify clearly which impact assessment

Table 3
Environmental impact categories included in reviewed studies.

Impact category	Carlsson (2002)	Eriksson and Finnveden (2009)	Frees (2002)	Jenseit et al. (2003)	Kreißig et al. (2003)	Perugini et al. (2005)	Raadal et al. (2008)	RDC and Coopers & Lybrand (1997)	Shonfield (2008)	Von Krogh et al. (2001)
Abiotic resource depletion			X	X					X	
Global warming potential	X	X	X	X	X	X	X	X	X	X
Human toxicity								X		
Ecotoxicity								X		
Acidification potential	X		X		X			X	X	X
Eutrophication potential	X		X					X	X	X
Photochemical oxidant formation			X	X					X	X
Stratospheric ozone depletion									X	
Air emissions						X				
VOC emissions to air	X									
NO_x emissions to air	X									
Water emissions				X		X				
Water consumption						X				
Solid waste (residual solid waste generation)			X	X	X	X		X	X	
Waste water generation					X					
Energy use	X			X	X	X	X	X	X	X

methods were used. The influence of impact assessment methodology is beyond the scope of this paper. However, for the impact categories compared in this paper, the differences between characterisation factors for impacts such as GWP, AP, and EP in these established methodologies are not expected to cause any significant uncertainty to our assessment.

3.6. Sensitivity analysis

Sensitivity analysis can be undertaken as a part of the interpretation phase in order to assess the robustness of the overall LCA results with respect to variations and uncertainties in data, methods and assumptions (Guinée, 2001). Sensitivity analysis was undertaken to varying degrees in the majority of studies reviewed and included; sensitivity to energy replaced by MSW incineration (Carlsson, 2002; Frees, 2002; Shonfield, 2008), sorting efficiency (Frees, 2002; Raadal et al., 2008), polymer composition of plastic waste streams (Shonfield, 2008) and virgin substitution ratio (Carlsson, 2002; Shonfield, 2008).

4. Results of scenarios comparison

Figs. 1–7 illustrate the comparison of scenarios for the impact categories GWP, AP, EP, ADP, EN and SW. The relative difference between the two scenarios is shown on the *x*-axis. The results of the relative comparisons are shown within 25 percentile bands (i.e. 25–50%). The preference of one scenario over another is shown, ranging from greater than 100% to 0% on one side of the *y*-axis and vice versa on the other side. For Fig. 1, the comparison of mechanical recycling scenarios to MSW incineration scenarios, negative values (left of the *y*-axis) indicate an environmental preference for mechanical recycling whereas positive values (right of the *y*-axis) indicate a preference for MSW incineration. The number of scenarios is indicated by the size of the bubble. Hence, in Fig. 1, it can be seen that for GWP, 10 scenarios show mechanical recycling has 25–50% less impact than MSW incineration.

4.1. Mechanical recycling compared to incineration with MSW

4.1.1. All scenarios

Fig. 1 shows all 37 scenarios comparing mechanical recycling to MSW incineration. When all scenarios are considered, there is a clear preference for mechanical recycling for GWP, ADP and EN. However an obvious preference is more difficult to determine for other impact categories. The presence of organic contamination and the choice of virgin material substitution ratio were recognised as two critical parameters influencing the preference for scenarios (WRAP, 2006) and are explored in further detail.

4.1.2. Organic contamination

Fig. 2 shows the comparative difference between mechanical recycling and MSW incineration scenarios in the case of (a) scenarios not considering organic contamination and (b) scenarios including organic contamination, considered in Frees (2002) and Von Krogh et al. (2001).

Frees (2002) considered seven scenarios including high, medium, low and no organic contamination, measured as chemical oxygen demand (COD), and cleaning of plastic waste in both hot and cold water. COD loads expressed as kg COD/kg plastic waste for high, medium, low and no categories were 1.5, 0.7, 0.2 and 0 respectively. In recycling scenarios, increased COD levels resulted in increased loads to municipal waste water treatment plants, thus increasing electricity demand and related environmental impacts. On the other hand, for MSW incineration scenarios, higher COD levels increased the heating value of the plastic waste, producing more energy without increasing GWP, as the organic contamination is of biogenic origin. In scenarios with warm water washing, GWP, AP and SW impacts categories show greater impacts for recycling than for incineration. As the level of COD increased the environmental benefits from recycling were reduced due to a combination of the increased energy demand from waste water treatment and an increase in heating value of the plastic waste, due to organic contamination.

Von Krogh et al. (2001) considered a low level organic contamination scenario (7% of the product), including the incineration of the biogenic organic contamination but not including the energy use required for COD removal from the wastewater from the washing process. The results show mechanical recycling was preferred to MSW incineration by a ratio of greater than 100%, confirming the significance of energy use in waste water treatment for medium to high levels for organic contamination.

4.1.3. Virgin material substitution ratio

Fig. 3 illustrates the influence of virgin material substitution ratio (the ratio at which recycled plastic substitutes virgin plastic or non-plastic materials) for the comparison between mechanical recycling and MSW incineration. Scenarios were separated as a function of the substitution ratio they considered. Fig. 3a shows scenarios considering a ratio of 1:1, Fig. 3b scenarios considering a ratio of less than 1:1 and greater than 1:0.5 and Fig. 3c scenarios with a ratio less than or equal to 1:0.5.

When a substitution ratio of 1:1 is considered, the majority of environmental impact categories for the 21 scenarios compared, show lower environmental impacts from material recycling when compared to MSW incineration (see Fig. 3a). The outlying SW impact category can be attributed to specific assumption of recycling processes and the waste treatment option used to treat the residual solid waste from recycling processes. For a substitution ratio between 1:1 and 1:0.5, Fig. 3b shows that, for impact categories GWP, ADP, EN and SW, the majority of the five scenarios show a lower environmental impact from material recycling compared to MSW incineration. Fig. 3c shows that, for the nine scenarios, MSW incineration was favoured over mechanical recycling with a ratio less than or equal to 1:0.5 for impact categories AP, EP and EN.

Fig. 3 shows, as the substitution ratio is reduced, the preference between mechanical recycling and MSW incineration becomes more uncertain. It can be seen in Fig. 3a that in all impact categories the clear majority of scenarios favour mechanical recycling. In Fig. 3c the preference among impact categories differs, making a preference between mechanical recycling and MSW incineration harder to define.

4.1.4. Substitution of average mix or marginal energy

Energy use and generation plays a significant role in LCA of waste management systems, as the heat and/or electricity produced by the combustion of plastic waste can be used to replace energy from fossil fuels. Two studies investigated the sensitivity of the results to marginal energy generation technologies instead of the more common attribution approach where the electricity produced is modelled to replace the average grid mix electricity. Frees (2002) and Shonfield (2008) evaluated the influence of electricity produced by plastic waste incineration replacing electricity from combined cycle gas turbines as the long-term marginal electricity production technology (see Fig. 4).

Whilst the use of marginal data does have an effect upon emissions, no change in preference between mechanical recycling and incineration scenarios was apparent. However, testing the sensitivity of the energy source being replaced does allow for greater confidence in the robustness of the results for future decision making purposes.

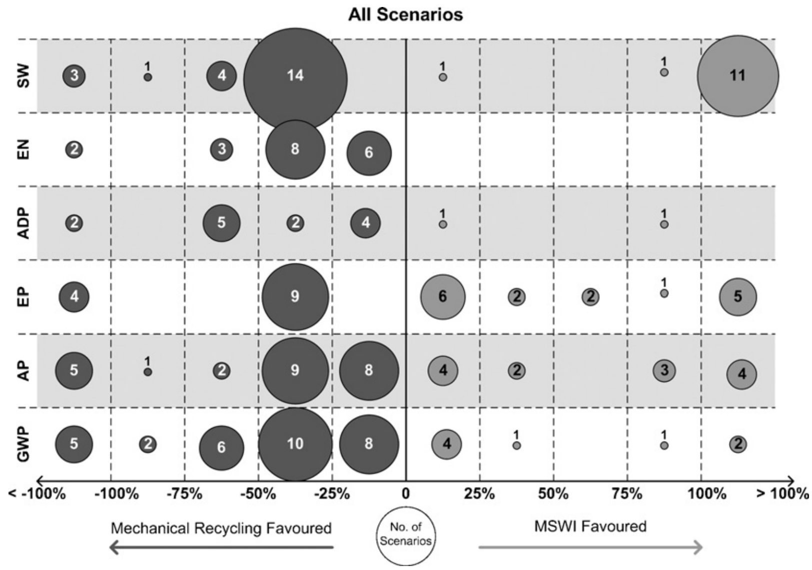

Fig. 1. Comparison of scenarios: mechanical recycling compared incineration – all scenarios.

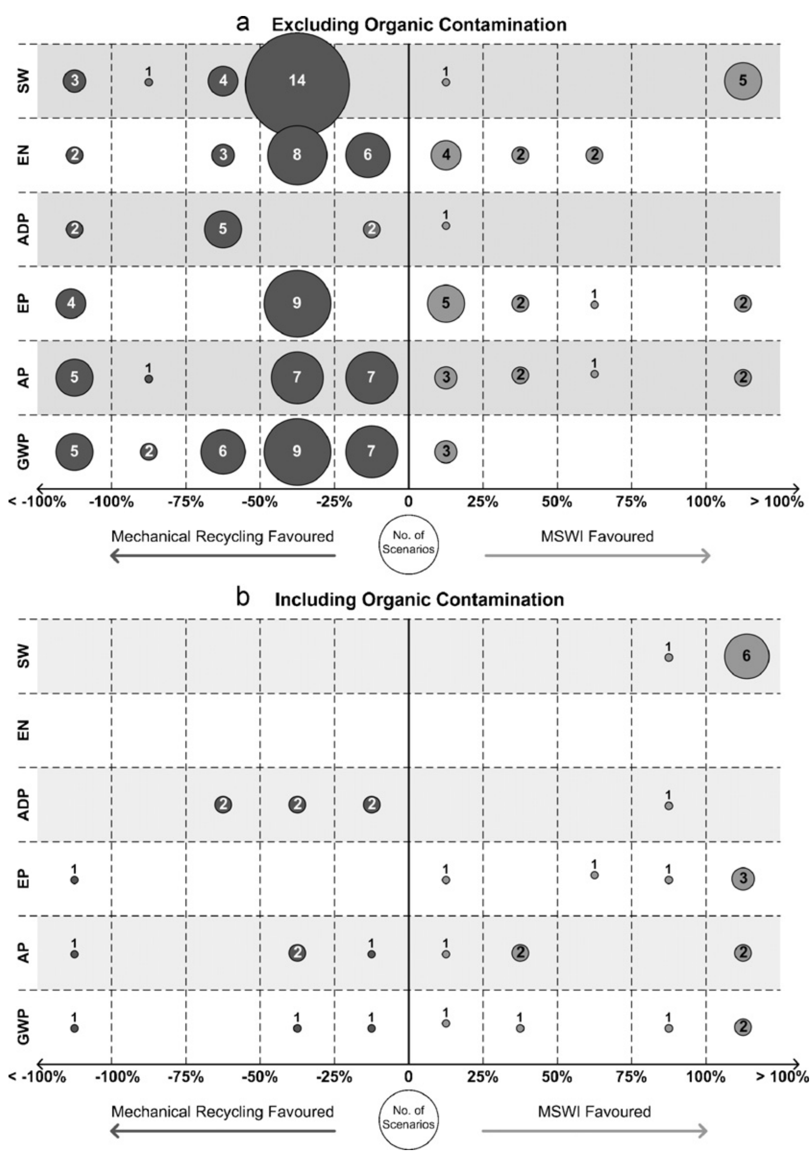

Fig. 2. Comparison of scenarios: mechanical recycling versus incineration (a) excluding all scenarios considering organic contamination and (b) only scenarios including organic contamination.

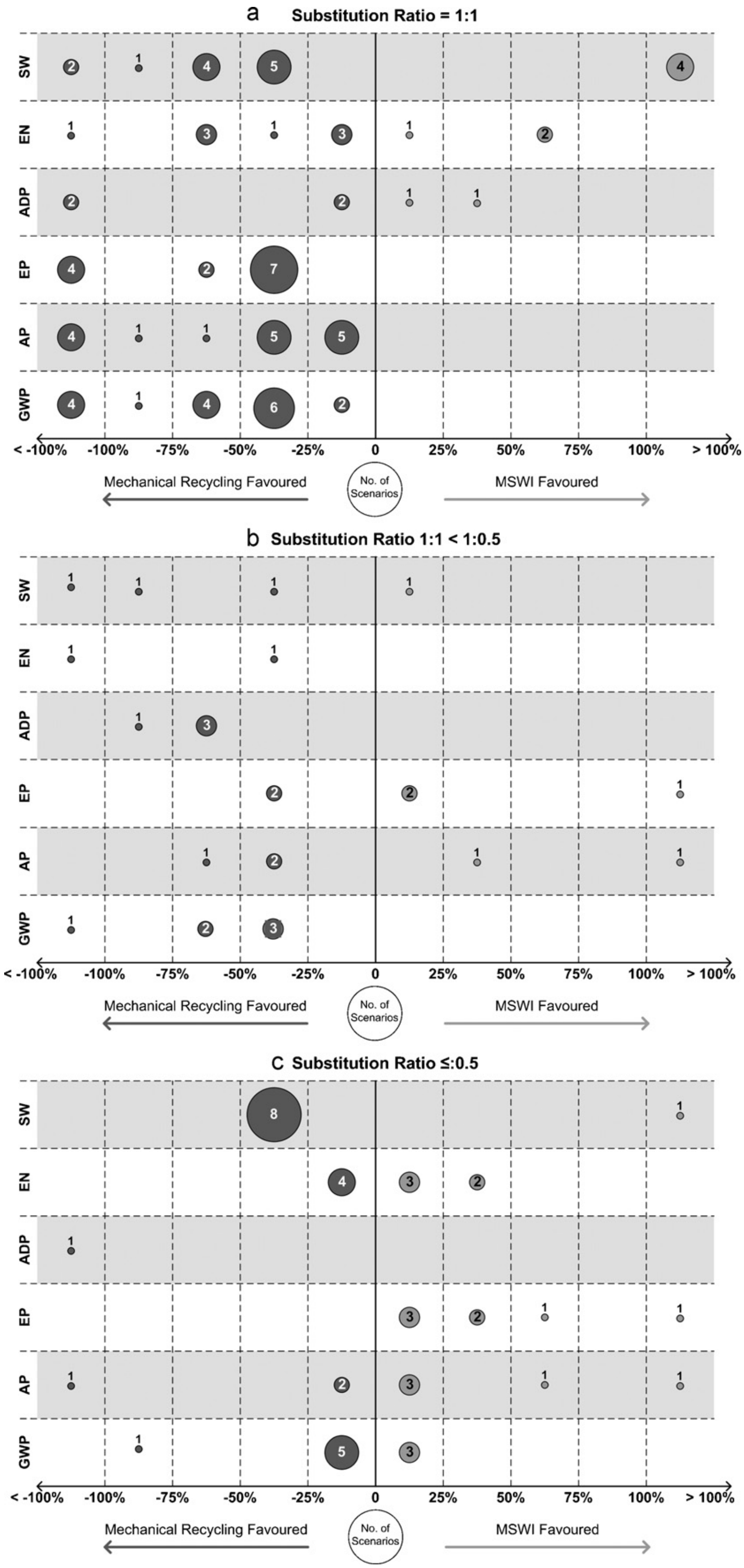

Fig. 3. Comparison of scenarios: mechanical recycling versus incineration, with a virgin material substitution ratio of (a) 1:1; (b) between 1:1 and 1:0.5; and (c) ≤1:0.5.

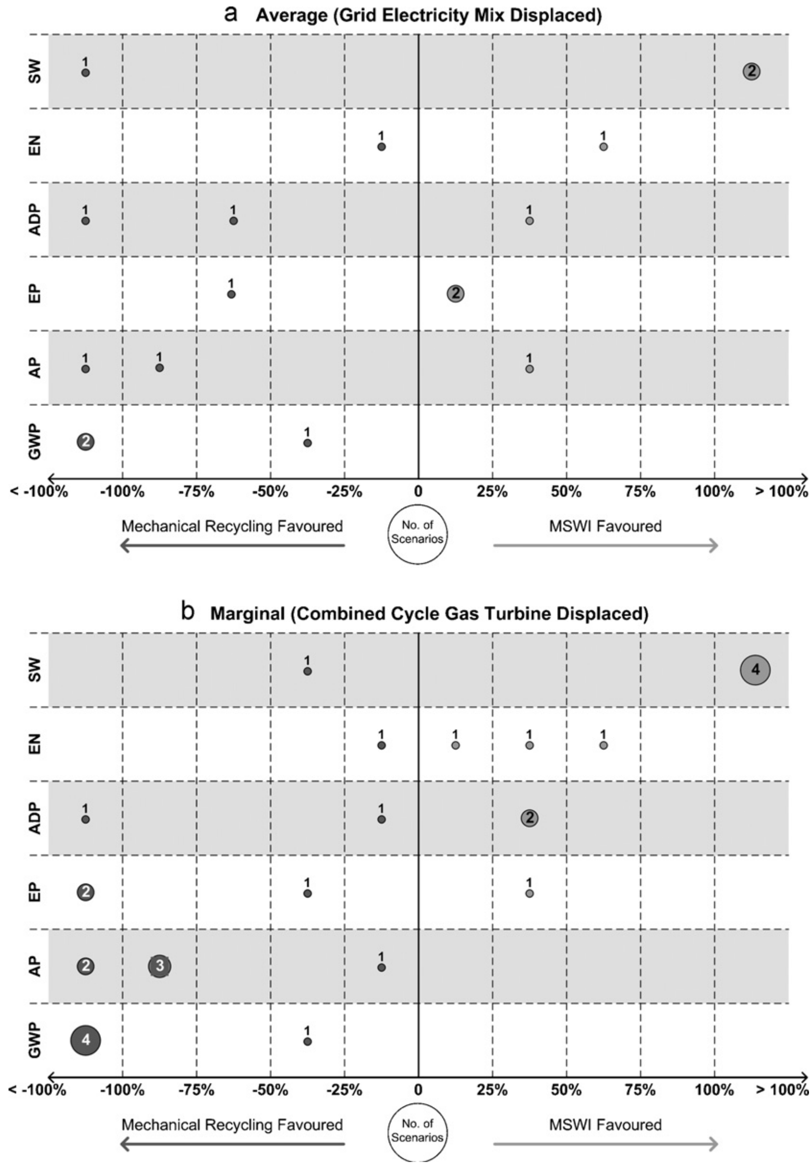

Fig. 4. Comparison of scenarios: mechanical recycling versus incineration with electricity produced replacing (a) the average grid mix electricity displaced and (b) electricity from combined cycle gas turbines displaced.

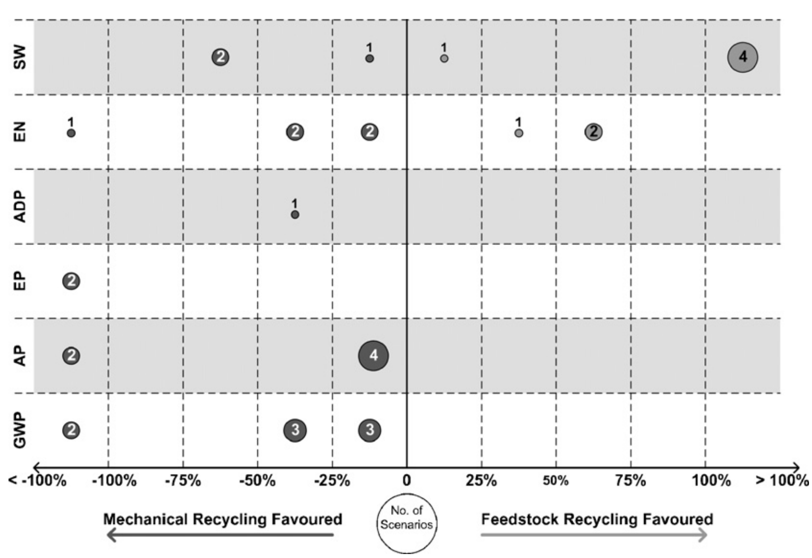

Fig. 5. Comparison of scenarios: mechanical recycling compared to feedstock recycling.

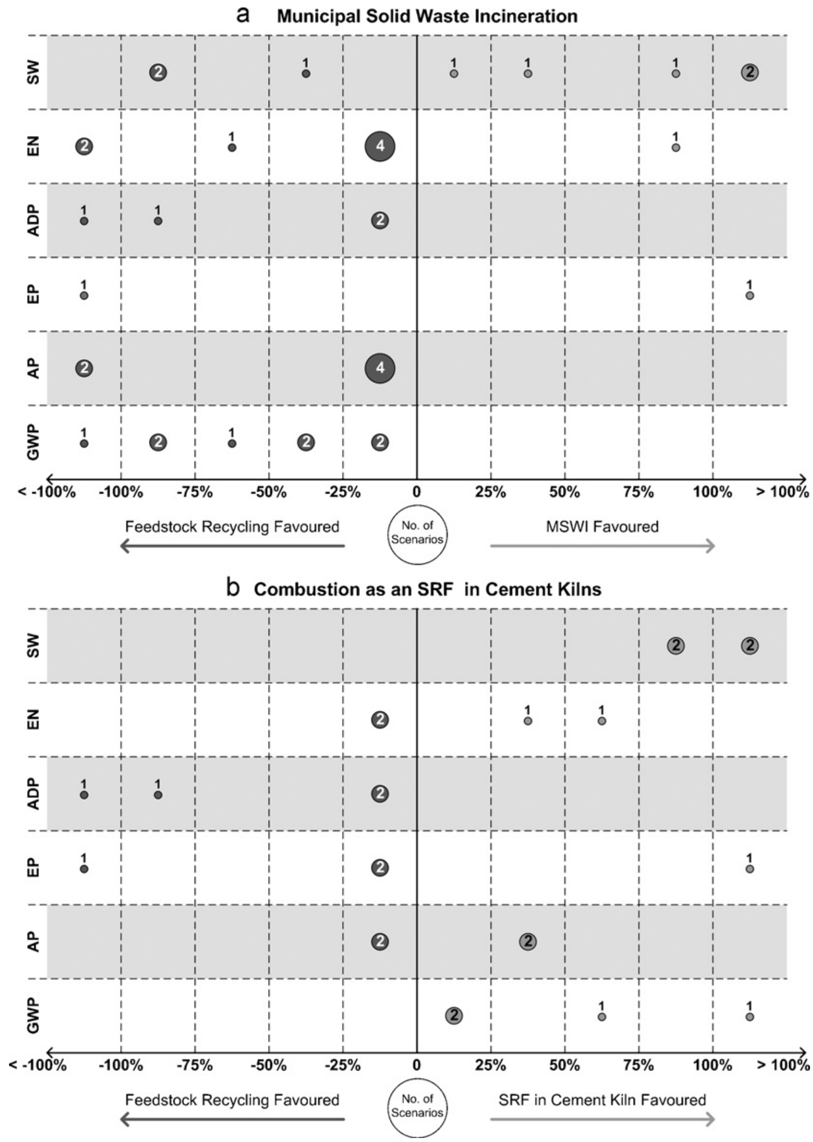

Fig. 6. Comparison of scenarios: feedstock recycling compared to (a) municipal solid waste incineration and (b) combustion as an SRF in cement kilns.

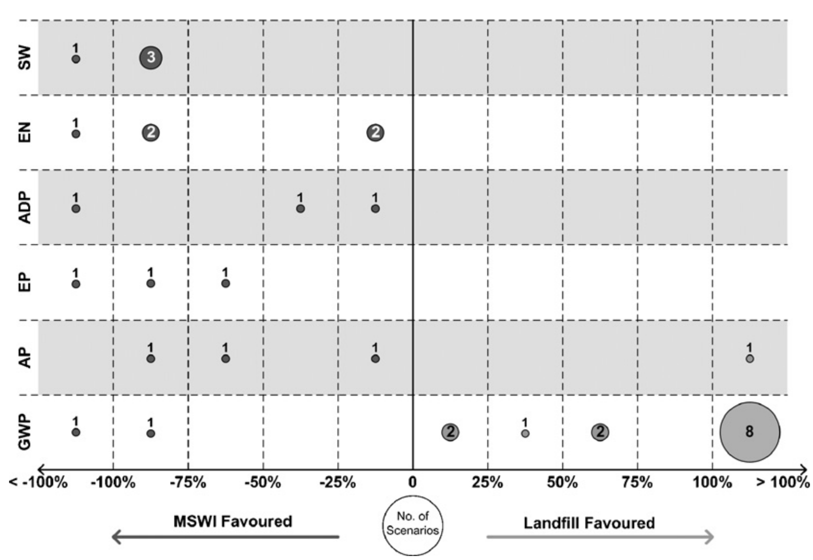

Fig. 7. Comparison of scenarios: MSW incineration compared to landfill.

4.2. Mechanical recycling compared to feedstock recycling

Eight scenarios in four studies, assessed feedstock recycling technologies (see Table 1). A general comparison of mechanical and feedstock recycling scenarios is made difficult as there are several different feedstock recycling technologies considered. Feedstock recycling scenarios were compared as standalone technologies, in the case of single fraction plastic waste inflows (Jenseit et al., 2003; Kreißig et al., 2003), or in combination with mechanical recycling technologies for waste fractions containing polyolefin and non-polyolefin waste (Perugini et al., 2005; Shonfield, 2008).

Fig. 5 illustrates the comparison of feedstock and mechanical recycling scenarios. Mechanical recycling was the preferred scenario for GWP, EP and AP, however feedstock recycling produced less SW in the majority of cases. EN was heavily dependent upon feedstock recycling technology choice.

4.3. Feedstock recycling compared to incineration

Fig. 6 compares feedstock recycling scenarios for the two incineration technologies that were considered, (a) MSW incineration and (b) combustion of SRF in cement kilns.

As only eight scenarios were reviewed and a variety of feedstock recycling technologies considered, it is difficult to make definitive conclusions on the environmental preference between these scenarios. Fig. 6a and b shows feedstock recycling utilises fewer abiotic resources than either MSW incineration or SRF combustion in cement kilns. For GWP, the preference is determined by the incineration technology and fuel source of the substituted energy. When plastic is used as an SRF replacing coal for the generation of heat in cement kilns, GWP was lower than feedstock recycling technologies. However this preference was reversed when MSW incineration replaced heat and/or electricity. This is due to a combination of both the energy efficiency of the technologies and the substituted energy source (see Table 2). Whilst the salience between these two parameters was not investigated, this highlights the importance of these parameters when expanding the system boundaries of waste management systems. As the SW impact category is a function of the material input and fractions accepted by feedstock recycling technologies, feedstock recycling scenarios produced less solid waste where single plastic streams were investigated, conversely incineration was preferred for mixed waste streams.

Whilst feedstock recycling is an interesting future technology, only 0.3% of plastic waste was processed by feedstock recycling in 2007 (Plastics Europe, 2008). Feedstock recycling technologies are predominantly situated in Germany, a vestige of high recycling targets set by the German Packaging Ordinance of 1991 (Fischer and Petchow, 2000). There is a high degree of uncertainty surrounding these technologies, in some cases data has been taken from previous studies which may not reflect the current situation. Additionally comparing pilot scale technologies to established technologies introduces uncertainly into the comparison.

4.4. MSW incineration compared to landfill

A comparison of the 15 scenarios comparing the treatment of plastic waste in MSW incineration and MSW landfill is shown in Fig. 7. The majority of landfill scenarios were associated with a lower GWP than MSW incineration scenarios, due to a combination of incinerator efficiency, electricity-to-heat ratio, fuel source of the substituted power generation and type of power generation. The two scenarios where MSW incineration was favoured, in Eriksson and Finnveden (2009), were due to a high efficiency and high electricity-to-heat ratio where electricity replaces fossil fuels for district heating. All other impact categories favoured MSW incineration.

According to Bez et al. (1998) and Finnveden et al. (1995), the degradation of plastic waste in landfill is approximately 1–5% during a 100 year time period, leading to potential air and groundwater emissions. This indicates that landfill scenarios may pose greater environmental impact if the temporal boundaries are extended, which could further reaffirm the results from this comparison that landfill is the least preferred treatment option for plastic waste management.

5. Discussion

5.1. Impact assessment categories

The selection of environmental impact categories can be attributed to several factors including:

(1) *relevance*: impacts related to energy use (GWP, AP, EP) are assessed in more than half of the studies, indicating the importance of energy use and energy savings from plastic waste management,
(2) *political drive*: the number of studies focusing on climate change reflect the prominence of this issue in the public arena,
(3) *data and impact assessment constraints*: lack of data and high uncertainty for human and eco-toxicological impacts and the need for development of improved impact assessment methodologies for these categories lead to their exclusion from some studies.

The significance of energy use and energy savings from the production of recycled materials, as well as the energy substituted by the incineration of plastic waste in MSW incinerators or cement kilns, is reflected by the dominance of energy related impact categories. Energy use was presented in seven studies, and according to Björklund and Finnveden (2005), is often a good indicator for other environmental impacts. As in many other fields the tendency to focus on climate change is reflected in the studies reviewed. Whilst global warming potential is an important consideration and can be used as a beneficial indicator to draw attention to environmental issues, Fig. 7 shows it is important to consider other environmental impact categories relevant to this field. Whilst different impact assessment methods were used in the studies review, the results suggest that the differences in the preference of scenarios among studies relate to difference in key assumptions, rather than impact assessment methods.

Although human and eco-toxicological impacts were not included in the majority of the studies, this is a significant issue in LCA of plastic waste management systems, as technologies which have the potential to reduce these impacts may be undervalued. Additionally technologies are not subject to the same emission standards at the European level. MSW incineration is addressed by Directive 2000/76/EC on the incineration of waste (Council Directive, 2000), establishing emission limits to both air and water. However, SRF incinerators are not subject to the same emission limit controls, hence a shift to SRF incineration may be advantageous in terms of energy efficiency but not in terms of human and eco-toxicological impacts.

5.2. Geographical boundaries

The current trend in EU material recovery has been the increased intra and extra trade of recyclable materials, especially plastic waste. Trade within the EU-15 for the main commodity polymers of PE, PP, PS and PVC increased by 41%, 150%, 74% and 39% respectively during the period 1995–2005. More remarkable is the extra EU-15 trade of waste plastics, with trade of PE, PS and acrylic polymers rising by 1156%, 2389% and 1363% respectively during the same period. Approximately half of this trade is shipped to Hong Kong and China (Fischer et al., 2008).

Of the studies reviewed, Raadal et al. (2008) and Carlsson (2002) included intra EU trade and recycling of plastics in their studies, whilst Frees (2002) included a scenario of plastic recycling in China. Christensen et al. (2007) have noted several difficulties in modelling material recycling in a global market including the availability of data for recycling facilities (which is predominantly sourced from developed countries and might not be representative of the facilities in developing economies) and the identification of the actual materials that are substituted by recovered materials. This increasing trend in the export of plastic waste for recycling and the uncertainties encountered in modelling this in LCA are an important consideration for decision makers when interpreting the results of LCA studies.

5.3. Organic contamination

Within the selected studies, Frees (2002) and Von Krogh et al. (2001) investigated the influence of organic contamination. Frees (2002) indicated that in the case of medium or high organic contamination, a lower environmental impact was obtained through MSW incineration when compared to plastic waste washing and mechanical recycling. As only one study reviewed included the environmental significance of waste water treatment of organic contamination, no general conclusions can be drawn.

As indicated by Frees (2002), the issue of organic contamination introduces a degree of uncertainty that needs to be addressed. In the context of the new WFD 2008/98/EC (Council Directive, 2008), recycling part of the substantial municipal mixed plastic waste stream is well worth considering. Whilst this stream has been exploited in Germany and some other European countries for many years, countries such as France and Belgium are yet to explore this option. Yet this stream possesses the greatest potential for organic contamination. Additionally if this stream is collected with a greater efficiency, one could imagine a greater volume of COD that would need to be treaded, a potential disadvantage of recycling this stream. Not only would a greater amount of COD need to be treated by waste water treatment plants, it is unknown what effect altering recycling behaviour would have on levels of COD contamination. This should be further investigated through life cycle thinking, in order to determine any potential environmental benefits of increased recycling efforts for this waste stream.

This does however raises the question, whether from an environmental perspective, if cleaning of organic contamination from plastic waste is better done by the product user (in the household) or by industry as a part of the recycling process? The level of organic contamination can be considered a result of the products application and the behaviour of the user (for instance not using all the food in plastic container). The system boundaries in LCAs of waste management systems are typically drawn at the boundary where a product becomes waste. Product users may not utilise all food in the packaging, if packaging is cleaned in the household in order to have a clean waste fraction for recycling, the use of water and energy is not considered in the LCA system boundaries. This would have a consequence as to the overall environmental impact of the recycling system but not reflected in LCA results.

5.4. Virgin material substitution ratio

In LCA, materials are usually modelled in closed loop systems when they are recycled back into the same product system, or when there is no change in inherent material quality. In these cases, recycled material can replace virgin material at a ratio of 1:1. In reality materials are recycled in open loop systems, which often involves a loss in material quality during either; the use phase (e.g., UV degradation), sorting/separation processes as a result of contamination (e.g., acid producing contaminants, water and colouring contaminants), or remelting processes (e.g., loss of molecular weight due to extrusion) (Awaja and Pavel, 2005). Plastic can also substitute non-plastic materials such as aggregates for concrete or timber products. Ekvall and Tillman (1997) note the issue of material quality is complex, as different applications require different quality attributes, thus the definition and measurement of quality can be difficult in LCA.

ISO (2006) states that wherever possible allocation (i.e., the use of substitution ratios) should be avoided by expansion of the system boundaries to identify which materials would be substituted by the recycled material. This removes the need for arbitrary decisions that are required by various allocation procedures. Although all studies utilised a system expansion methodology (in terms of determining the energy systems replaced) no studies expanded the system to determine the substituted material as a result of the decision to recycle. Whilst Carlsson (2002) and Shonfield (2008) went further to look at the sensitivity of recycled plastic replacing non-plastic materials, allocation procedures were used to determine the degree to which virgin plastic was substituted.

When system boundaries are not expanded in open loop recycling systems, ISO (2006) recommends allocation for open-loop recycling based upon “physical properties (e.g. mass); economic value (e.g. market value of the scrap material or recycled material in relation to market value of primary material); or the number of subsequent uses of the recycled material”. Physical properties could include length of polymers, content of impurities, elasticity, etc. However, when the technical functionality of a material is multi-criterial, more than one technical parameter may determine the material’s inherent properties, potentially leading to the selection of an arbitrary allocation parameter for material property (Werner, 2005).

The virgin material substitution ratio is a function of two variables. First, the ratio at which recycled plastic substitutes virgin plastic or non-plastic materials, which can be determined by an analysis of material flows as was done by Carlsson (2002). However, as plastic recycling is becoming ever more embedded in the global economy, mapping the flows and use of recycled materials becomes increasingly difficult. Secondly, “*Changes in the inherent properties of materials shall be taken into account*” (ISO, 2006). In effect, the notion of quality is dependent upon the function that is required to be fulfilled by the material in its subsequent life cycle. Schmidt and Strömberg (2006) recommend a conservative estimate of 1:0.8, a 20% loss in material quality, in order to achieve the same functionality of the original product and for cases where the future user of the recycled plastic is unknown.

Allocation procedures have been developed to treat this issue, such as the value-corrected substitution method (Huppes, 2000; Werner and Richter, 2000), using the price difference between recycled and virgin material as a proxy for material quality. Economic parameters reflect additional aspects such as the supply and demand situation within the socio-technical system (Werner, 2005). Rigamonti et al. (2009) determined a virgin material substitution ratio of 1:0.81 for plastic film in the Italian market using this method. Many of the virgin material substitution ratios used in the LCAs reviewed in this paper relied on assumptions, with little documentation of the justifications behind these assumptions. Carlsson (2002) included a material flow analysis of plastic waste in Sweden which indicated 20% of recycled HDPE was used to replace wood products.

There is an apparent difficulty in determining a realistic substitution ratio which has the consequence of generating uncertainty in the overall results of the LCA studies when comparing mechanical recycling to other treatment options. In light of this uncertainty a detailed sensitivity analysis would be recommended to determine, for each context, the substitution ratio that is required for mechanical recycling to be the preferred treatment option.

5.5. Plastic waste management, the waste hierarchy and a recycling society

The objectives of this review were to determine the presence of a consensus as to the environmentally preferable treatment option for plastic waste management and, consequently, to assess the validity of the application of the waste hierarchy to plastic waste management.

The results from the LCA studies reviewed have indicated that, from a life cycle perspective, mechanical recycling of plastic waste is generally preferred to (1) feedstock recycling and (2) MSW incineration, provided that (a) recycled plastic originates from clean plastic waste fractions with little organic contamination and (b) recycled plastic substitutes virgin plastics at a ratio of close to 1:1. The lack of certainty regarding these two points has been discussed in the preceding sections.

For plastic waste fractions that are not currently mechanically recycled, due to institutional or market parameters, this review indicates that feedstock recycling is preferable to MSW incineration, but no clear preference is evident between feedstock recycling and combustion as an SRF in a cement kiln. The use of plastic waste as a reducing agent in steel manufacture and incineration as an SRF in facilities with a high ratio of energy recovery, have been shown to be preferred to incineration in MSW incinerators with energy recovery. Landfill was the least preferred treatment option for all impact categories except for GWP, highlighting the importance to consider other environmental impacts other than GWP, and also the influence of temporal boundaries in landfill scenarios.

The majority of the scenarios compared seem to fit within the waste hierarchy, suggesting there is legitimacy in applying the waste hierarchy as a general rule to plastic waste management. However, when one focuses on the studies' assumptions (specifically organic contamination and virgin material substitution ratio) and their impact upon the results, the application of the waste hierarchy to plastic waste management becomes hard to justify. Whilst the waste hierarchy remains a guiding principle of EU waste management policy, LCA can provide valuable information as to when plastic waste streams depart from this waste hierarchy, as asserted in the new WFD (Council Directive, 2008) and the packaging and packaging waste directive (Council Directive, 1994). At the same time, the uncertainties surrounding LCA should not be forgotten. These uncertainties raise questions as to the robustness of LCA results, which is sure to be continued in the dialectic between the waste hierarchy and LCA.

For the polymers investigated, listed in Table 1, not all studies presented their results for individual polymers. However, for the studies that assessed scenarios of individual polymer fractions (Frees, 2002; Jenseit et al., 2003; Kreißig et al., 2003; RDC and Coopers & Lybrand, 1997; Von Krogh et al., 2001), there was no evidence that results differed for individual polymers.

Enlarging the analysis framework of our paper, another more global lesson might be highlighted. In the view of establishing a "recycling society" (European Commission, 2005), it would be necessary to close material loops through reuse and recycling, as is advocated by the concept of industrial ecology. Yet, we see that in some impact categories for plastic waste, recycling activities generate greater impacts than other treatment options. If a trajectory of strategies based on mechanical recycling is shown to be of interest for fractions of plastic waste such as mixed household plastics, several limits to recycling and closing material loops may be encountered under certain assumptions. Organic contamination and other forms of contamination (e.g., heavy metals and additives) of plastic waste can also be a driver to investigate upstream changes affecting plastic waste management. If a "recycling society" is a genuine long term goal of the EU, with the aim of closing material loops, issues regarding recycling should be targeted upstream at the level of production. It may be of relevance, rather than focusing on the downstream aspects such as quality of recycled plastics, to investigate the environmental impacts of upstream measures, such as the use of plastic materials in applications where there will be only minimal contamination or downgrade of material quality.

6. Conclusion

The results presented in this paper indicate that for the majority of scenarios previously investigated by LCA, mechanical recycling is generally the environmentally preferred treatment option. This is relevant for environmental impact categories related to energy use, including global warming potential, acidification potential, eutrophication potential, abiotic resource depletion potential and residual solid waste production. However, it has been shown that the virgin material substitution ratio and amount of organic contamination could lead to recycling showing lower environmental benefits than other treatment options such as incineration with energy recovery, confirming the outcomes of WRAP (2006). A degree of uncertainty exists as to what ratio recycled plastic is substituting virgin plastic and the level of organic contamination for mixed plastic waste, this suggests a focus be placed on recycling of high quality plastic waste to realise maximum environmental benefits from recycling. This underlines the need for a greater understanding of material flows related to different plastic resins in different waste streams, as well as plastic material characteristics and its ability to be recycled.

Due to the uncertainty surrounding some of the critical assumptions in LCAs of plastic waste management, a case by case assessment would be required to demonstrate in which situations the waste hierarchy is applicable. In this paper, the comparative analysis has highlighted some of the sensitive key points for plastic waste treatment. If the technological trajectory of the EU, the "recycling society", is to be achieved the uncertainties discussed in this paper require further investigation.

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