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Lifecycle assessment and economic evaluation of recycling: a case study

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Abstract

Recycling is widely assumed to be environmentally beneficial, although the collection, sorting and processing of materials into new products also entails significant environmental impacts. This study compares the relative environmental impacts of a recycling system (incorporating the kerbside collection of recyclable materials and their subsequent use by manufacturers), with a waste disposal system (in which the waste is disposed to landfill and primary raw materials are used in manufacture), using the technique of lifecycle assessment. The methodology is then extended to incorporate an economic evaluation of the environmental impacts. This combination of lifecycle assessment and economic evaluation can be termed 'Lifecycle Evaluation'. Lifecycle assessment quantifies and evaluates the environmental impacts of a product from the acquisition of raw materials, through manufacture and use, to final disposal. Lifecycle assessment can also provide a framework for the analysis of environmental impacts from systems such as transport, or waste management, as demonstrated in this paper. The results, for a case study of Milton Keynes in Central England, show that the recycling system generally performs better than the waste disposal system in terms of contribution to global warming, acidification effects and nutrification of surface water. An alternative method of analysis is then used, in which an economic valuation of the environmental impacts is carried out. This produces net benefits for recycling, per tonne of material, of £1769 for aluminium, £238 for steel, £226 for paper and £188 for glass, and net costs of £2.57 for high density polyethylene (HDPE), £4.10 for poly (vinyl chloride) (PVC) and £7.28 for poly (ethylene terephthalate) (PET). It is concluded that lifecycle evaluation, the combination of lifecycle assessment and economic valuation, can be applied to a variety of waste management issues such as the appraisal of alternative methods of collection for recycling or an examination of the UK waste management hierarchy. This technique allows impacts to be expressed

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in homogenous units, and the inclusion of social and environmental impacts that would not normally be addressed within a lifecycle assessment. The approach would also facilitate the evaluation of environmental and social effects at a local level, which are particularly crucial to the success of community recycling schemes. Lifecycle evaluation could provide a powerful tool to aid the development of sustainable waste management and recycling policy.

Keywords: Lifecycle assessment; Recycling; Economic valuation; Waste management

1. Introduction

Recycling is widely regarded to be environmentally beneficial and conducive to sustainable economic development. It mitigates resource scarcity, decreases demand for landfill space and generally involves savings in energy consumption. Nevertheless, the collection of materials for recycling has its own environmental impacts, notably the energy used in collection and sorting, and impacts arising from the use of the recovered materials in new products.

Despite these impacts the UK Government has expressed support for recycling in two recent publications: the Government's White Paper on the Environment [1] which proposed a target of recycling 50% of the recyclable component of municipal solid waste by the year 2000; and the Draft Waste Strategy for England and Wales [2] which represents the Government's policy for achieving sustainable waste management.

The latter document accepts and supports the EU waste hierarchy which ranks recycling high on the list of methods of waste treatment and disposal. However the waste hierarchy does not appear to reflect the actual environmental impacts of waste management techniques, and the ranking appears to be based on intuition rather than on a scientific assessment. An attempt should be made to assess the environmental and social costs and benefits of each waste management option, regardless of their assumed place in the hierarchy [3].

In order to assess the costs and benefits it is necessary to examine both the resources and materials used in each system as well as the emissions generated. One methodology for undertaking this evaluation is lifecycle assessment (LCA), which quantifies the environmental impacts of a product or material over its entire lifecycle [4]. This includes the extraction of raw materials, processing of materials, manufacture of the product, distribution, use and reuse or recycling, and final disposal. The technique can be used to improve the environmental impacts of a single product or to compare the relative impacts of different products. Alternatively, LCA can be used to compare different systems such as recycling and waste disposal.

One of the main problem areas in LCA is the aggregation of the resulting environmental impacts which are usually in non-comparable units. Various methodologies have been developed [5] including economic valuation which is used in this paper. Economic valuation methodologies are concerned with estimating the value that individuals place on non-market goods and services. A 'value' can be revealed by a consumer's behaviour (revealed preferences) or derived from their stated 'willingness to pay' (stated preferences) [6]. Alternatively a 'value' can be revealed by use of dose-response relationships and replacement costs [7].

These techniques can provide an economic valuation of the various emissions which can be used to quantify the 'external' costs and benefits not usually accounted for in the financial assessment of recycling schemes. Externalities are the environmental and social impacts, such as atmospheric pollution or the generation of noise, whose financial implications are not included as 'private' costs.

In this study we evaluate the potential of LCA to determine the environmental and social impacts of recycling, by combining it with economic valuation. The social impacts dealt with are those arising from transport; road traffic accidents and road congestion. A case study of a scheme in Milton Keynes, a town in Central England with approximately 180 000 residents, has been undertaken and a comparison made between two waste management practices; the kerbside collection, sorting and distribution of recyclable materials and their use in the manufacturing system, and the disposal to landfill of household waste and the subsequent use of primary materials in the manufacturing system. 'Primary materials' refer to substances used in a production process for the first time, and 'secondary' materials are those which have been previously used in a similar or different production system, and have been recovered from the waste stream.

2. Lifecycle assessment

Interest in LCA has increased against a background of comprehensive environmental legislation including Integrated Pollution Control (IPC) and Best Available Techniques Not Entailing Excessive Cost (BATNEEC), the growth of the green consumer market and pressure from voluntary bodies. Corporate interest has also been stirred by the introduction of BS 7750 (Environmental Management Systems), and the EC Eco-Management and Audit Regulation in 1993. The criteria of the EC Eco-labelling scheme are based on the results of partial life cycle studies [8].

2.1. LCA methodology

An LCA comprises four major stages: goal definition, inventory, impact assessment and improvement assessment [9]. The goal definition stage defines the purpose, scope and boundaries of the study, the functional unit, (based on a fixed quantity of products as supplied to their end use, for example), key assumptions to be made and likely limitations of the work [9].

The inventory stage constitutes a detailed compilation of all direct and indirect environmental inputs and outputs to each stage of the life cycle, including raw materials and energy consumed, emissions to air and water, and solid waste produced. One shortcoming of this technique is that it is difficult to accommodate, at the inventory stage, qualitative information such as the ecological resilience of the receiving environment, the renewability of resources, public perception, occupational safety and risks, and other social impacts [9]. The possibility of including social and economic impacts into LCA has been explored by Assies [4], who argues that social and other economic assessments of a product lifecycle would be useful in the policy-making arena. A small number of valuation methodologies indirectly include political targets and health impacts [10]. Generally, however, the inventory is restricted in scope to quantitative environmental emissions. There is scope for including a measure of social factors at the inventory stage, for example by recording the number of kilometres involved in transportation.

Although the inventory stage is regarded as being more objective than the impact assessment stage, some uncertainty remains concerning emission coefficients for industrial processes, as is evident from the wide range of values which are reported in the LCA literature [5]. Furthermore, it is not the emissions themselves, but their resulting impact upon the environment with which we are concerned. Therefore, lifecycle assessments which are limited to data from the inventory stage are incomplete for policy purposes, and need to be supplemented by the impact assessment stage.

Lifecycle impact assessment is a process whereby environmental impacts from the inventory are assessed, and the overall environmental performance of the product is determined. Impact assessment incorporates three stages; classification, characterisation and valuation. In the first stage, the data are classified according to an environmental problem (for example, global warming or ozone depletion), scale (local, regional or global), and media (air, water or land) [11]. Characterisation quantifies the relative contribution of each input or output to each environmental problem. This is achieved by using, for example, global warming potentials (GWP) or ozone depletion potentials (ODP).

This paper addresses the issues of global warming, acidification and nutrification of surface water. The concept of Global Warming Potential (GWP) has evolved as a result of work carried out by, among others, the Intergovernmental Panel on Climate Change (IPCC). GWPs are a measure of the possible warming effect on the atmosphere from the emission of each gas, relative to carbon dioxide (CO_2). They account for effects over the whole globe and for changes in concentration over time [12]. In this study, the GWPs on a 100-year basis (Table 1) have been used to quan-

Table 1
Relative global warming potentials of greenhouse gases

Greenhouse gas	Relative global warming potential (direct)[a]	
	20-year basis	100-year basis
CO_2	1	1
CH_4	35	11
N_2O	260	270
CFC 11	4500	3400
CFC 12	7100	7100

Source: Houghton et al. [12].
[a]One tonne CO_2 = 1.

tify the carbon dioxide equivalent of emissions of CO_2, CH_4 and N_2O produced for the recycling and waste disposal systems for each material.

Acid gas emissions (SO_2, NO_x, HF and HCl) result in acid deposition when atmospheric waste vapour condenses and precipitation occurs. This results in decreased pH levels in soils and water and increases the mobility of toxins in the soil, with consequent detrimental impact to aquatic life and forests [13]. To aggregate the emissions of acid gases they are expressed in terms of the mass of hydrogen ion equivalents.

Finally, surface water nutrification (also known as eutrophication) is the addition of nutrients to water, which results in enhanced primary productivity. Emission coefficients are available for oxides of nitrogen (NO_x) and chemical oxygen demand (COD) [13], expressed as phosphate ion equivalents.

In order to allow a comparison of environmental impacts which have been quantified in heterogeneous units, some method of weighting these impacts is desirable, and this is carried out in the *valuation* stage. Relative weights are assigned to each problem and an overall value can be obtained for the environmental impact of the system being considered. This is the most contentious stage of the lifecycle assessment, and the most subjective [14], because it involves trade-offs between different environmental problems. In Europe and the US different research groups have developed alternative valuation methodologies. These tend to be based on, for example, social opinion, political decisions, expert rankings, sustainability indicators, or economic valuation [5], but no set of valuation factors has yet been established that is widely acceptable.

For example, regulatory standards rarely take account of threshold effects, where a small increase in pollutant has a disproportionately large effect above a certain 'safe' or stable level. In some cases, legislation is based less on scientific grounds, and more on political, economic and technical possibilities [15]. Multicriteria evaluation (MCE), as used by Fawcett et al. [16] enables a number of options to be evaluated against a range of criteria, and qualitative data can be incorporated by way of a ranking system. MCE unfortunately does not eradicate the need to assign relative weights to the criteria, so a subjective element remains. Although several of the valuation techniques are based on 'scientific' impact categories, they are not necessarily objective or scientifically accurate.

A relatively new alternative is to use economic valuation methodologies to value environmental and social costs and benefits. Valuation methodologies include dose-response relationships, contingent valuation, the hedonic property price approach and stated preferences. Economic valuation approaches are described in more detail in CSERGE et al. [17], and in this paper economic valuation is used to assign weights to environmental and social impacts. This is dealt with in more detail in the methodology section.

LCA remains a developing technique, particularly the impact assessment stage. In addition, conventional LCA does not address wider social impacts such as disamenity or health and safety. By supplementing the impact assessment stage with economic valuation of environmental and social impacts, we create 'Lifecycle Evaluation' (LCE). This paper demonstrates how the broader scope of LCE as opposed to LCA, can provide a more complete social and economic assessment of alternative waste management strategies.

2.2. LCA and recycling

Currently there is much debate over the relative merits of kerbside collection of recyclable household waste versus intensified 'bring' systems [18–20]. Both systems require the source separation of recyclable materials by the householder. In bring systems the clean, sorted wastes are taken to central sites where there are glass, can, textile and paper banks or some combination of these. Collect systems involve the kerbside collection of the separated recyclables either at the same time as the remaining household waste or at a different time.

Recovery rates for kerbside collection schemes are higher than those for bring sites, and are thus believed to be the only way in which the Government's target will be reached. At present, only 5% (of 20 million tonnes annually) of household waste is recycled, compared with 85% disposed of to landfill and 10% incinerated [2]. It has been estimated [21] that a weight reduction of 30% of municipal solid waste (MSW) is feasible for individual kerbside collection schemes. However, there is no real consensus as to which is the best way to reach the target. The debate can become more informed by taking into account the external costs and benefits of different recycling schemes, and examining them within a lifecycle assessment.

LCA has been used historically as a tool for examining the environmental impacts of specific products. For example, several studies have compared alternative packaging for beverages and other products [22]. Various authors have attempted to compare the relative environmental impacts of alternative waste management options including recycling, but few have included a valuation of the results. A useful summary of the environmental impacts of recycling within the manufacturing system has been produced by Ogilvie [23], but it does not include impacts which arise during the collection, sorting and distribution of recyclable materials to manufacturers, or impacts from final disposal. Kirkpatrick [24] examined the selection of waste management options in the UK using LCA techniques, and SETAC [9,25] provide some theoretical guidance on dealing with recycling within a lifecycle assessment.

Most studies [26–28] have been confined to the inventory stage, although some authors, such as Møgaard and Atling [29] and Johnson [13], have taken the analysis one stage further, classifying the data into environmental impacts such as global warming and acidification. However, no weighting of these impacts was attempted. Weighting systems using panels of experts were applied by Fawcett et al. [16], in a multicriteria evaluation of waste management options, and by Wilson and Jones [30], to weight environmental stressors in a comparison of washing detergents.

The valuation stage of an LCA was included in an examination of the environmental impacts of diapers by Bovy and Wrisberg [31], however, the only study to apply a valuation methodology to waste management techniques was that by Powell et al. [32] which applies economic valuation to environmental costs arising from recycling schemes. Thus, little has been done in the way of applying full lifecycle assessments to recycling schemes, or to evaluate the external costs of these schemes in comparison with alternative methods of waste disposal.

3. Case study methodology

The case study for the kerbside collection and sorting of recyclable household waste is the scheme run by CROP, the Community Recycling Opportunities Programme, in Milton Keynes. Established in 1982, CROP became the official recycling company for the Borough Council in 1989, and the scheme now services 73 000 households. A large number of different materials are collected, including newspapers and magazines, aluminium and steel cans, and glass and plastic containers. It is the largest kerbside collection recycling scheme in the UK and achieves a reduction in weight of 26% of the MSW stream. CROP also acts as a coordinating centre for recycling activity in the region and provides a local education service.

The recyclable materials are source separated by the householder into two 45-l boxes. Each week the materials are collected at the kerbside and sorted on the vehicle into 11 categories. The remaining household waste is collected separately. On returning to the materials reclamation facility (MRF), the nine collection vehicles are weighed and the glass is emptied into colour-segregated bays. The remaining materials are placed on a conveyor belt, from which an overhead magnet removes steel cans and an eddy-current (opposing electromagnetic field) extracts aluminium cans. Poly (ethylene terephthalate) (PET) and high density polyethylene (HDPE) bottles are removed by hand in the plastic sorting cabin, and poly (vinyl chloride) (PVC) bottles are removed by an X-ray machine, which identifies chloride ions within the plastic. Cans, plastics, newspapers and magazines are baled and stored outside to await distribution to reprocessing sites. Washing and granulating plant for HDPE have been installed on site, the cost of which has been met by Plysu (a local plastics company). The clean pellets are then sent to Plysu to be made into new detergent and oil containers.

3.1. Lifecycle inventory

3.1.1. Boundaries of the comparison

A comparison is made between the management of 1 tonne of waste by the 'recycling system' (the recovery of materials and their subsequent use in new products), and its management by the 'waste disposal system' (the landfill disposal of the waste and use of primary materials). This analysis has been carried out separately for 1 tonne of each material (paper, glass, steel, aluminium, HDPE, PET and PVC plastics), rather than for 1 tonne of mixed household refuse. Empirical data were obtained from the case study for the lifecycle inventory, and this was combined with published data covering primary and secondary production processes.

The lifecycle stages of each waste management option used in this comparison are defined in Fig. 1. Environmental burdens of parts of the life-cycle common to both systems have not been quantified because of the comparative nature of this exercise. These include the manufacture of contents of containers, the distribution by manufacturers to wholesalers and retailers, and the use of materials by the householder. There remain important differences between the transport and manufacturing stages of each system.

Global impacts are included in the assessment, such as the contribution to global

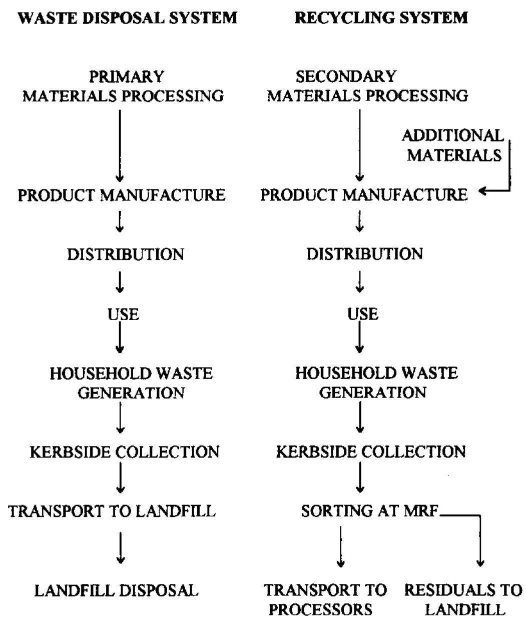

Fig. 1. Lifecycle stages of the two systems used for comparison.

warming, and the export of recovered materials. As with any other system, there is a hierarchy of environmental inputs and outputs. Those of the first order arise as a direct result of a process in the lifecycle, for example energy use or process emissions. Second order inputs and outputs are those connected with the manufacture of capital equipment such as buildings, roads and machinery. In this study, due to the high throughput of material in the system, it has been assumed that the second order impacts contribute an insignificant amount to each incremental tonne of material, and for this reason only first order inputs and outputs are included.

To facilitate comparison, all energy used in the primary and secondary production processes has been assumed to be electrical energy from the current UK fuel mix. The energy produced from landfill gas is assumed to replace electricity generated by the least efficient means, which in the UK are old coal-fired power stations.

3.1.2. Transport

Transport impacts in the recycling system arise from the transport of materials from the consumer to the MRF (kerbside collection), and the distribution of the recovered materials from the MRF to the manufacturers. The fuel consumption from the kerbside collection of recyclable materials is 8.9 l/t. This lies within the range of figures quoted by White et al. [28], of 14.3 l/t for the Adur scheme in West Sussex, and 7.2 l/t for Milton Keynes. The materials are sent to a variety of locations in Britain, from just 10 km away in the case of HDPE (to Plysu), to over 350 km in the case of steel (to Hartlepool). Some materials are exported for recycling, for example PET, which is sent to The Netherlands and then on to Ireland, and aluminium, which is shipped to German rolling mills. In the waste disposal system, transport impacts arise in the kerbside collection and transport of household waste to landfill.

Transport stages make a large contribution to the environmental and social impacts of recycling schemes, involving gaseous emissions, road accident casualties and road congestion. Congestion costs for a range of roads have been estimated by Newbery [33] but vary widely with road category (from 0.05 pence per passenger car unit (PCU) kilometre for 'other rural roads' to 36.37 pence per PCU km for 'urban central peak'). Therefore, the values assigned to road congestion are highly dependent on the road category chosen. In this paper, the kerbside collection is assumed to occur on suburban, non-central roads (an average of peak and off-peak) and the distribution stage involves non-central roads (average) and motorways. As suggested by Newbery [33] one passenger car unit kilometre is equal to two HGV unit kilometres.

Each transport stage is assigned a 'utility' which reflects whether the vehicles make their journeys fully laden or empty. For example, in this study transport by rail and sea is assigned a utility of 100%, which assumes that both the outward and return journey will be made fully laden. For transport by road, a 75% utility is chosen, which assumes that all outward journeys are made laden, and that half the return journeys will be made empty, and half laden. A recent Government estimate suggests that 26% of journeys made by HGVs are empty [34].

3.1.3. Savings within the manufacturing industry

Comparative lifecycle assessments are made easier if it is assumed that the recycled materials perform exactly the same function as primary materials, but this not always the case. In particular several types of plastics cannot be recycled into identical new products because of the need for specific strength and quality of material. For example, PET recovered in the form of plastic bottles is flaked and made into fibre, which is then used as a filling for sleeping bags and jackets (Wellman International Ltd., personal communication). As a result, the savings occur within the manufacture of this fibre from secondary rather than primary materials, and not in the manufacture of PET bottles.

A second problem that occurs at the manufacturing stage is the scarcity of data, particularly with regards to the manufacture of products using secondary plastic materials. Whilst information concerning energy use is sometimes available, data for the remaining environmental inputs and outputs are either commercially sensitive or simply unreported. In this exercise, the process data (emissions arising directly from the process) have been taken as being identical for both primary and secondary plastics, but an *energy* saving (and thus the associated emissions of generation) of 77% is obtained by using secondary materials, as suggested by White et al. [28]. The production of newspapers with secondary fibre is another area for which little data could be found. However process data for the production of bleached sulphite paper from both primary and from secondary materials are available and were used in this study.

3.1.4. Landfill disposal

In the Milton Keynes area, household waste which is not recycled is sent to local landfill sites such as Brogborough and Newton Longville, and no waste is incinerated in this area. This study includes the environmental inputs and outputs of collection and transport of the waste to landfill, which for the waste disposal system is 1 tonne of each material being studied. In the recycling system, for each tonne of recyclable material which reaches the MRF, about 0.025 tonnes are not recyclable and they are sent to landfill. This includes non-targeted materials and those which are heavily contaminated.

Environmental impacts from landfill include landfill gas generation, leachate and disamenity (such as increased volumes of traffic, odour, perceived health risks, noise and loss of visual amenity) [17]. Some waste components can be assumed to be inert (glass, steel and aluminium), whilst others will decompose, so it is necessary to calculate the landfill impacts for each waste component individually. The landfill gas impacts for individual waste components have been estimated (the authors, unpublished) and are included in the calculations for this paper, but it has not proved possible to do the same for the other landfill impacts owing to a lack of available data.

The principal gaseous emissions from the anaerobic decomposition of waste in landfill sites are methane and carbon dioxide. Both are greenhouse gases, but methane has a far greater radiative forcing potential than carbon dioxide. It can be argued that as these emissions are from 'new carbon' sources such as wood and agricultural crops, they are part of the carbon cycle and should not be included as an environmental impact. However if the waste was to decompose in an aerobic situation, that is, not in a landfill site, methane would not be produced, only carbon dioxide. For these reasons in this study the carbon dioxide from landfill sites has not been included as an externality, but the methane has.

For each average tonne of waste which is disposed of to landfill in the UK, 81% by volume of the gaseous emissions are released to the atmosphere, 13% are flared, and 6% are used in landfill gas generating schemes [35,36]. This paper uses this average data when calculating the amount of electricity recovered. The electricity generated will displace emissions from old coal-fired power stations, and this study gives credit for these.

3.2. Lifecycle impact assessment

The lifecycle impact assessment was carried out in two ways; initially the traditional classification and characterisation stages were undertaken and a comparison made between the two waste management options. An alternative approach was then used, in the form of an economic valuation of the inventory results.

3.2.1. Classification and characterisation

The inventory results for each material for each of the two systems were classified as contributing to either global warming, acidification or nutrification of surface water. After characterisation, a tentative appraisal is made as to which alternative is the less damaging to the environment.

3.3. Economic valuation

In addition to the standard LCA stages of classification and characterisation, an economic valuation of the inventory results has been undertaken using monetary estimates for the damage done by specific gaseous emissions, casualties from road traffic accidents, and road congestion. Monetary estimates for gaseous emissions have been taken from Fankhauser [37] and the Commission for the European Communities [38]; the UK Government's Department of Transport publishes estimates of the external cost of road traffic accident casualties of different severities [39]; and Newbery [33] has calculated the external cost of road congestion on different roads and at different times of day.

When using economic valuation there is no need to aggregate the inventory data by classification, and the resulting figures appear in homogenous units. From this exercise we have derived a first approximation of the external cost of managing 1 tonne of each material by each of the two systems. The economic values used in this paper are presented in Table 2. The valuation procedure is to multiply the relevant economic value by the physical parameter, for example the amount of CO_2 emitted or the number of casualties expected.

The physical impacts of gaseous emissions have been derived from dose-response functions of damage to crops and forests. For human health, the damage value is based on the value of lost productivity, medical costs, the value of a statistical life and willingness to pay to avoid symptoms [38]. The number of expected casualties from road traffic accidents can be calculated from the Department of Transport's published data concerning the number of incidents and distances travelled annually. The external cost of casualties per kilometre can then be derived from this, using the economic valuation of casualties as given in the Department of Transport Valuation of Road Accidents [39]. These values are based on the willingness-to-pay approach, which encompasses both 'human' and direct economic costs (including pain, grief and suffering, as well as lost output and medical costs). Newbery's estimates of congestion costs [33] represent the opportunity cost of productive time wasted due to congestion. All monetary estimates are subject to varying degrees of uncertainty.

Table 2
Economic parameter values for external costs

Emission	(Pence/kg)	Road casualties	(£/casualty)	
CO_2	0.40	Mortality	744 060	
CO	0.60	Serious injury	84 260	
CH_4	7.20	Minor injury	6540	
SO_2	258.40	Road congestion	(pence/PCUkm	/HGVUkm)
NO_x	127.00	Motorway	0.26	0.52
N_2O	61.40	Non central	12.30	24.60
PM10[a]	898.00	Rural	0.07	0.14

Sources: CO_2, CO, CH_4 and N_2O: Fankhauser [37]; SO_2, PM10 and NO_x: Commission for the European Communities [38]; Road casualties: Department of Transport [39]; Road congestion: Newbery [33].
[a]Particulates of less than 10 μm diameter.

By applying the monetary estimates above of various external cost components to each lifecycle stage, and summing them, we arrive at the total external cost of each system. This can be represented as follows:

$$E = G + C_{rta} + R$$

where E = total external costs; G = external costs of gaseous emissions; C_{rta} = external costs of casualties from road traffic accidents; R = external costs of road congestion.

Table 3
Transport distances for collection and distribution, and energy use associated with sorting at Milton Keynes

Material	Collection (tonnes per week)	Collection distance (km per tonne of mixed waste)	Energy used in sorting (MJ/t)	Distance to manufacturer (km/t)[a]		
				Road	Rail	Sea
Paper	80.0	13.90	14.93	17.87	0.00	0.00
Aluminium	4.0	13.90	14.93	37.91	0.12	25.00
Steel	15.0	13.90	14.93	22.32	0.00	0.00
Glass	60.0	13.90	0.00	14.13	0.00	0.00
PET	6.0	13.90	14.93	51.71	0.00	74.83
HDPE	7.0	13.90	14.93	0.89	0.00	0.00
PVC	2.0	13.90	14.93	21.89	0.00	0.00

[a]Road transport assumes 75% utility, rail and sea transport assume 100% utility.

Table 4
Environmental emissions associated with collection and sorting

Material	CO_2 (g/t)	CO (g/t)	CH_4 (g/t)	SO_2 (g/t)	NO_x (g/t)	N_2O (g/t)	TSP (g/t)
Glass	23 708	92.26	0.1	32	462	0.00	11.04
All others	26 560	92.8	12.1	32	468.87	0.15	11.04

Table 5
Transport distances for collection and transport to landfill, and associated gaseous emissions

Distance (km/t waste)	CO_2 (g/t)	CO (g/t)	CH_4 (g/t)	NO_x (g/t)	N_2O (g/t)	SO_2 (g/t)	TSP (g/t)
Household collection of waste							
3.61	6139.17	24.13	0.04	119.78	0	0	2.83
Transport to landfill							
1.55	2635.93	10.36	0.02	51.43	0	0	1.21

Table 6
Methane emissions from landfill and displaced pollution from energy recovery

Waste type	CH_4 (g/t waste type)	Energy recovery (kWh/t)	Displaced emissions from coal fired power stations (g/t waste component)				
			CO_2	CH_4	SO_2	NO_x	TSP
Paper	50 500	18.36	19 787.36	75.26	256.98	97.28	2.94
Plastics	13 000	4.74	5113.59	19.45	66.41	25.14	0.76

4. Results

4.1. Lifecycle Inventory

4.1.1. Recycling system

The data collected and analysed for the Milton Keynes kerbside collection scheme include vehicle distances travelled to collect the recyclables, the quantity of different materials collected, the energy used in sorting operations and the distance to manufacturers (Table 3). As all the recyclable materials are collected at the same time, the collection distance for each of them is equal. As the materials are sorted simultaneously, it has been assumed that each material consumes an equal amount of energy per unit weight in the sorting procedures (for operation of conveyors, baling machines and magnets, for example). The exception is glass which is unloaded, already colour segregated, from the collection vehicles and stored without further sorting.

The environmental emissions associated with the kerbside collection and sorting operations are presented in Table 4. These include transport emissions and those from energy generation. The transport to landfill of non-recyclable materials which accumulate at the MRF is also included.

Table 7
Aluminium: contribution to global warming

	Waste disposal (kg/t)			Recycling (kg/t)		
	CO_2	CH_4	N_2O	CO_2	CH_4	N_2O
Emission	49 995.79	206.50	2.71	2502.53	10.34	0.14
Coefficient[a]	1	11	270	1	11	270
Impact	49 995.79	2271.54	732.7	2502.53	113.77	36.72
Total CO_2 equivalent	52 998.99			2653.02		

[a]Source: Houghton et al. [12].

Table 8
All materials: contribution to global warming

Material	Waste disposal, CO_2 equivalent (kg/t)	Recycling, CO_2 equivalent (kg/t)
Aluminium	52 998.99	2653.02
Glass	2514.07	1394.64
Paper	548.29	50.46
Steel	122.24	116.23
HDPE	159.50	31.22
PET	162.75	98.25
PVC	156.32	53.86

4.1.2. Waste disposal system

In the waste disposal system, environmental impacts associated with transport will arise from both the kerbside collection of household refuse, and its transport to landfill. The transport distances per tonne of waste collected and disposed are shown in Table 5, along with the associated emissions. Table 6 presents the atmospheric methane emissions from landfill disposal, the amount of energy recovered from a tonne of each waste component, and the gaseous emissions displaced by using this energy to generate electricity in place of old coal-fired power stations. This is the least efficient method of producing electricity in the UK, and thus would be the first to be replaced.

4.2. Lifecycle impact assessment: classification and characterisation

4.2.1. Global warming

Table 7 illustrates the results from the classification and characterisation stages using aluminium as an example, and Table 8 summarises the results for all materials. The waste disposal systems generally make a larger contribution to global warming

Table 9
Aluminium: contribution to acidification

	Waste disposal (kg/t)				Recycling (kg/t)			
	SO_2	NO_x	HF	HCl	SO_2	NO_x	HF	HCl
Emission	580.59	127.47	0.25	0.05	29.62	7.63	0	0.76
Coeff.[a]	31.3	21.7	50	27.4	31.3	21.7	50	27.4
Impact	18 172.47	2766.10	12.5	1.37	927.11	165.57	0	20.82
Total H^+ equivalent		20 952.44				1113.501		

[a]Source: Powell [40].

Table 10
All materials: contribution to acidification

Material	Waste disposal, H^+ equivalent (kg/t)	Recycling, H^+ equivalent (kg/t)
Aluminium	20 952.44	1113.50
Glass	1156.94	677.41
Paper	3231.04	650.48
Steel	327.27	243.26
HDPE	80.92	91.31
PET	131.09	166.45
PVC	46.43	64.79

than the recycling systems. For aluminium the recovery and use of secondary aluminium makes a saving of 95%, which is the largest for any material, both in absolute and percentage terms. There are also large savings involved in recycling glass and paper, 44% and 91%, respectively, although the difference is minimal for steel (5%). The savings for plastics are 80%, 40% and 66% for HDPE, PET and PVC, respectively.

4.2.2. *Acidification*

The calculations and results from the aluminium example are presented in Table 9, and the results for all materials in Table 10. The majority of acid gases are emitted as sulphur dioxide and nitrous oxides, with small amounts of hydrogen fluoride and hydrogen chloride. The waste disposal systems contribute more to acidification than the recycling systems in the cases of aluminium, glass, paper and steel, again aluminium exhibiting the greatest savings. The savings from the recycling systems are 95% for aluminium, 41% for glass, 80% for paper, and 26% for steel. The recycling of all three plastics however contribute more to acidification than the waste disposal system, the recycling system being higher by 13% for HDPE, 27% for PET and 40% in the case of PVC.

Table 11
Aluminium: nutrification of water

	Waste disposal (kg/t)		Recycling (kg/t)	
	NO_x	COD	NO_x	COD
Total	127.47	19.02	7.63	0
Coefficient[a]	0.13	0.022	0.13	0.022
Impact	16.57	0.42	0.99	0
Total phosphate ion equivalent	16.99		0.99	

[a]Source: Johnson [13].

Table 12
All materials: nutrification of water

Material	Waste disposal, phosphate equivalent (kg/t)	Recycling, phosphate equivalent (kg/t)
Aluminium	16.99	0.99
Glass	1.09	0.74
Paper	4.58	0.97
Steel	0.38	0.45
HDPE	0.18	0.22
PET	0.75	0.96
PVC	0.12	0.23

4.2.3. Nutrification

Table 11 shows the results for the recycling and waste disposal systems for aluminium, and Table 12 for all materials. The greatest benefit again comes from the recycling system for aluminium, and there are large savings from the recycling system for paper. In percentage terms, the savings resulting from recycling amount to 94% for aluminium, 32% for glass, and 79% for paper. However, for some materials, there is a net increase in the nutrification impact with the recycling system. As percentage increases compared with the waste disposal system, these values are 18% for steel, 22% for HDPE, 28% for PET and 91% for PVC.

4.3. Economic valuation

Monetary estimates for the external cost of gaseous emissions, road traffic accident casualties and road congestion were applied to these parameters for each material, for each of the two systems. There were no monetary estimates available for nutrification. As congestion costs remain controversial, the final net results will be given both including and excluding these. For the recycling system, the valuation of external costs arising from the kerbside collection and sorting scheme alone produces values of £0.88/t for emissions, £0.71/t for casualties and £3.40/t for congestion. This is a total weighted average external cost of £4.99 per tonne of mixed

Table 13
Economic valuation of external costs associated with the kerbside scheme (£/tonne of recovered materials)

Material	Emissions (£/t)	Casualties (£/t)	Congestion (£/t)	Total external cost (£/t)
Glass	0.86	0.71	3.40	4.97
Other materials	0.89	0.71	3.40	5.00
Weighted average	0.88	0.71	3.40	4.99

Table 14
Economic valuation of external costs (£/tonne)

Material	Waste disposal (£/t)	Recycling (£/t)	Net benefit from recycling (£/t)	Net benefit excluding congestion (£/t)
Aluminium	1880.27	111.41	1768.86	1771.84
Glass	254.78	67.20	187.58	189.96
Paper	299.85	73.79	226.07	228.42
Steel	269.40	31.64	237.76	240.26
HDPE	9.49	12.07	−2.57	−0.21
PET	13.98	21.25	−7.28	−4.05
PVC	7.46	11.55	−4.10	−1.57

recyclable materials (Table 13). For the complete recycling system, which also includes the distribution and manufacturing stages, the results for the total external costs are £111.41 for aluminium, £67.20 for glass, £73.79 for paper, £31.64 for steel, £12.07 for HDPE, £21.25 for PET and £11.55 for PVC (Table 14).

For the complete waste disposal system, the external costs are £1880.27 for aluminium, £254.78 for glass, £299.85 for paper, £269.40 for steel, £9.49 for HDPE, £13.98 for PET and £7.46 for PVC. When the two systems are compared a net economic benefit is produced for the recycling systems of £1768.86 for aluminium, £187.58 for glass, £226.07 for paper and £237.76 for steel. In the case of plastics, there were net costs from the recycling system of £2.57 for HDPE, £7.28 for PET and £4.10 for PVC (Table 14). The net economic benefits excluding congestion costs are £1771.84 for aluminium, £189.96 for glass, £228.42 for paper, £240.26 for steel, and net costs of £0.21 for HDPE, £4.05 for PET and £1.57 for PVC.

5. Discussion

5.1. Lifecycle impact assessment

For each material, the waste disposal system (using primary materials and disposing of all waste to landfill) is generally a larger contributor to enhanced global warming than the recycling system (using recycled materials and recycling waste). This is mainly due to savings in energy consumption which can be achieved by using secondary materials in place of primary ones, particularly in the case of aluminium. Although there are substantial greenhouse gas emissions generated in the transport of recovered materials to reprocessing facilities, these are usually outweighed by the savings at the manufacturing stage. In the waste disposal system, the contribution to global warming is largely due to methane emissions from landfills, particularly from paper. These can be reduced and offset to some extent by the generation and recovery of energy from landfill gas.

For acidification impacts, the waste disposal system contributes more to the problem than does the recycling system for all materials except plastics. This is partly because the emissions of acid gases from transport stages outweigh the emissions saved at the manufacturing stage (due to decreased energy demand for using secondary materials in manufacturing), and also to the displacement of coal emissions (relatively high in SO_2 and NO_x) when energy is recovered from landfill gas. The results may suggest that the recycling of plastics is not the most environmentally beneficial waste management technique, but given the lack of data in this area, this remains difficult to confirm. In addition, the data do not account for geographic distribution of acidic precipitation, the vulnerability of the receiving environment or for the different solubility of the gases (which affects the distances that they are transported).

The results show that the potential for surface water nutrification is higher for the waste disposal system than the recycling system in the case of aluminium, glass and paper, but lower for steel and plastics. There are, however, some inadequacies with the data for some materials. For example there is incomplete information regarding emissions to water from the paper recycling process. There are many uncertainties concerning the effects of particular emissions to water, which depend significantly on the vulnerability of the receiving environment, and this is reflected by the current debate surrounding the proposed Statutory Water Quality Objectives (SWQOs) [41]. In the light of this there are limitations as to how well the emissions translate into actual nutrification effects.

From the classification and characterisation stages demonstrated above, one could draw the conclusion that recycling is better in environmental terms than waste disposal to landfill for aluminium, glass and paper, that the case is doubtful for steel, and that waste disposal is the better option for plastics if global warming is the primary concern. This illustrates the problem of comparing environmental impacts in heterogeneous units and the inadequacy of the methodology for providing a coherent basis on which decision makers can formulate waste management policy. It is evident that some system of weighting is necessary before any decision can be made regarding which is the 'better' system, waste disposal or recycling, for each material.

5.2. Economic valuation

As an alternative to classification and characterisation, economic valuation of the recycling and waste management systems for each material was carried out, extending the exercise to lifecycle evaluation. This gave a net benefit from recycling for all materials except plastics. The results imply that it is preferable to recycle aluminium, glass, paper and steel, but that recycling is not the environmentally optimal solution for plastics. When the net external benefits are calculated excluding congestion costs, there is a greater benefit for aluminium, glass, paper and steel, and a reduced net cost for plastics. This is because the transport stage, from which congestion results, accounts for a higher proportion of the impacts in the recycling system than the waste disposal system. Thus to ignore congestion costs would favour recycling.

When compared on a tonne for tonne basis, plastics have a greater environmental impact than other materials because of their high volume to weight ratio. As a result, a smaller volume of plastics can be carried on each trip compared with other materials, giving a greater number of kilometres travelled per tonne. As discussed previously, emissions data are particularly scarce for the manufacture of products using secondary polymers and this hinders the search for a conclusive answer on the issue of plastics recycling.

5.3. Limitations of lifecycle evaluation methodologies

Data accuracy, confidentiality, availability and quality are common problems within the lifecycle inventory, which will affect both methods of comparison. Detailed information regarding collection schemes for recycling in the UK is rare, and it is for this reason that this paper presents a case study of a scheme, rather than claiming to be representative of recycling in general. The effect on the overall results, of small changes in the data, could be further investigated by using sensitivity analyses.

There are some social and environmental impacts which are not easily quantified in either physical or monetary terms, but which may determine the success or failure of a particular recycling scheme. For example, separating recyclables requires effort on the part of the householder to clean, sort and store the materials. Space is needed for storage and the less convenient this is, the greater the incentive not to recycle. Social costs may also be incurred in the form of noise from collection vehicles and from the materials reclamation facility. Visual disamenity, arising from vehicles and buildings, also imposes a social cost, but again is difficult to quantify. Occupational health and safety is another important social aspect, but one on which very few studies have been carried out, and none on UK waste management employees [40].

Benefits of a recycling scheme include educational value, and increased environmental awareness. There is also a 'feel good' factor, arising from having the opportunity to contribute towards a scheme which produces environmental benefits. Social surveys can be used to ascertain some of these determinants, but although important, they remain difficult to quantify and include in lifecycle assessment.

Monetary estimates only exist at present for a limited number of environmental and social factors, and this restricts the number of criteria against which the lifecycle can be judged in an economic valuation. However, monetary evaluation of non-market goods is an evolving technique, and there is a continuing expansion of the range of parameters for which monetary estimates are available. Furthermore, unlike the use of classification and characterisation, social and environmental impacts which cannot be quantified in physical units can be analysed by applying monetary estimates. As more estimates become available, a broader range of social and environmental impacts can be incorporated into lifecycle evaluation, which will facilitate the comparison of alternative options on a more comprehensive basis.

6. Conclusion

In this study we have examined and compared the environmental and social impacts of a waste disposal system and a recycling system using lifecycle evaluation, a combination of lifecycle assessment and economic valuation. LCE can be applied to a variety of waste management issues, for example to question and assess the assumed ranking of management alternatives in the EU waste hierarchy. The relative advantages of alternative methods of collection for recycling, such as intensive bring and kerbside schemes, can also be analysed within such a framework. There is much to be gained by the application of lifecycle evaluation to recycling and waste management.

The stages of classification and characterisation in LCA are sufficient to examine the environmental impact of alternative systems in terms of specific environmental problems, and to make a first approximation of which system may be 'better' overall. However, the use of LCE facilitates a comparison in homogenous units, and the inclusion of social and economic impacts other than those which can be quantified in physical terms. Although the number of monetary estimates which are available at present is limited, this is an evolving area of research. Such research would be particularly useful if it focused on social and environmental impacts which occur at a local level, such as noise and disamenity. These are particularly important for recycling schemes, which are operated and administrated at a community or district level, and where local impacts are likely to be most prominent.

Lifecycle evaluation, particularly given its potential to combine external costs with private costs, could prove useful in establishing the relative total economic cost of different recycling schemes, and could prove conducive to the development of sustainable waste management and recycling policy in the UK and elsewhere.

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