UNIVERSITY OF CAMBRIDGE

Electricity Policy Research Group

Hot Issue and Burning Options in Waste Management: A Social Cost Benefit Analysis of Waste-to-Energy in the UK

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Abstract The growing stream of municipal solid waste requires a sustainable waste management strategy. Meanwhile, addressing climate change and security of energy supply concerns require increased use of low-carbon and domestic sources of energy. This paper assesses the economic and policy aspects of waste management options focusing on waste to energy (WtE). We conclude that high levels of WtE and recycling are compatible as waste treatment options. We also present a social cost-benefit analysis of waste management scenarios for the UK focusing on specific waste management targets and carbon price. The results indicate that meeting the waste management targets of the EU Directive are socially more cost effective than the current practice. The cost effectiveness improves substantially with higher carbon prices. The findings show that WtE can be an important part of both waste management strategy and renewable energy policy. However, achieving the full potential of WtE requires development of heat delivery networks.

Keywords Electricity, renewable energy, waste to energy (WtE), waste management, municipal solid waste (MSW).

JEL Classification Q01, Q28, Q42

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1. Introduction

Modern economies produce large quantities of waste as a by-product of economic activity. This tendency is compounded by economic and population growth. In 2005/06 the UK produced almost 29 million tonnes of municipal solid waste (MSW) (DEFRA, 2007). How this waste is managed can have significant economic, environmental, and energy implications. The majority of waste disposal options – including landfilling and recycling – use energy as an input. In contrast, Waste to Energy (WtE) technology can use MSW in order to generate electricity and heat.

The European Union (EU) Directive 1999/31/EF, requires minimisation of the quantity of MSW sent to landfill. The UK has favoured landfill for waste disposal in the past due to its naturally impermeable ground conditions. In April 2004, however, the Landfill Tax rate for active waste was increased by £1 to £15 per tonne and will continue to rise until it reaches £35 per tonne in the medium-long term; bringing the UK in line with other EU countries.

Energy from waste is estimated to increase from the current 9 to around 25 percent by the end of 2020 in the UK (DEFRA, 2006). Waste to energy is a unique source of energy in terms of the cost of fuel. Fuel cost constitutes a significant share of total cost from conventional thermal energy sources. Meanwhile, most renewable energy generation (such as wind, solar, marine, and hydroelectric) is capital intensive, but has no direct fuel cost. A notable exception is biomass energy from crops. MSW is essentially a biomass energy resource; however its use as an input in WtE incurs a negative fuel cost because plants receive gate fees for accepting delivery of the waste. These payments account for the majority of WtE plants' earnings.

Climate change and security of supply pose challenging energy policy issues. Therefore, it is important to assess the potential for, and the significance of, WtE in the context of the UK's energy and environmental policy. The UK has a target of generating 10 and 20 percent of its electricity from renewable resources by 2010 and 2020 respectively (DEFRA, 2006). In 2004, electricity generated from renewable sources amounted to 14,171 GWh – i.e. 3.6 percent of total electricity generation (DEFRA, 2006). Landfill gas and WtE from combustion of biodegradable MSW accounted for 23 and 10 percent of total renewable electricity respectively (DEFRA, 2006).

This paper presents an assessment of the economics, institutions, and policies affecting WtE. It seeks to analyse whether WtE can only be regarded as a waste management option or also a renewable energy source capable of increasing the UK's security of supply and mitigating climate change. Section 2 introduces the concept of the waste hierarchy. Section 3 reviews the UK's waste management policy. Section 4 develops a social cost-benefit analysis (CBA) of selected WtE scenarios. Section 5 presents an analysis of the UK's waste treatment options. Section 6 provides a brief policy discussion. Finally, Section 7 concludes the paper.

2. The Waste Hierarchy

MSW consists of arisings from household, commercial, institutional, and light industrial sources. Waste management decisions have two distinct but related components: how much waste to produce, and how to dispose of this waste. Disposal options include illegally dumping waste at unauthorised sites, using sanitary landfill sites, recycling materials, and incinerating waste with or without energy production.

A definition of waste is important for formulation of appropriate policies. The European Union defines waste as something that is discarded by its owner. MSW mainly consists of waste from households (82 percent of total MSW), small businesses, office buildings and institutions such as schools, hospitals and government buildings (Eurostat, 2003). The generation, separation, collection, transportation and disposal of waste, taking into account parameters such as public health, economics and environment is termed as Municipal Solid Waste Management (MWSM) (Dubois, et al, 2004).

The range of options for managing MSW is often presented in order of preference via the 'waste hierarchy'. The waste hierarchy originates from the 1975 EU Framework Directive on Waste and has since become a key waste management concept in the UK (Figure 1). The waste hierarchy provides a useful framework for formulating waste management regulation and policy. In February 2007, the European Commission (EC) proposed a three-step waste management hierarchy placing prevention first, followed by reuse, recycling and energy recovery on the second level, and disposal at the bottom. The five step hierarchy is preferred in the UK, however, with the understanding that it serves as a flexible principle rather than a rigid requirement for waste policy (WLGA Briefing, 2007).

Breaking the link between economic growth and waste growth is central to a successful waste management policy. This implies reducing the waste intensity of GDP by making products and services with fewer resources. Since 2000, waste in the UK has grown less than GDP. Of the main waste streams, municipal and business waste grew at a slower rate than GDP; municipal waste increased by about 3.5 percent per year up to the millennium and has since slowed to around 1.5 percent per year (DEFRA, 2006).[1] The estimated amount of waste in the UK by 2020 is between 33 and 42 million tonnes, based on annual growth rates of 0.75 to 2.35 percent respectively (DEFRA, 2006).

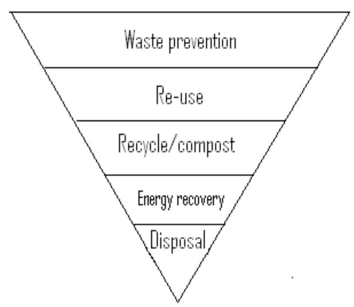


Figure 1: Waste hierarchy in the UK
Source: Adapted from DEFRA (2007)

The waste hierarchy places precedence on reducing the waste stream. In the UK, around 10 tonnes of resources are used to produce 1 tonne of product, signalling 'resource inefficiencies' among the 3.75 million UK companies (Financial Times, 2006). Nevertheless, waste prevention is arguably the most difficult approach to managing waste. Variable charging for rubbish collection has been suggested to reduce the waste stream; this could be based on the volume, weight or the specific material content of waste. A waste prevention strategy that has received attention is that of reducing product packaging. As this does not directly burden the public, it has proved popular. In 2005, the

[1] Business waste here refers to the waste such as office paper, recycling posters, and signs and other waste (not from commercial and industrial waste) not included in MSW.

major UK retailers signed up to the Courtauld Commitment with Waste and Resources Action Programme (WRAP) for finding packaging solutions and technologies to reduce waste. They agreed to stop packaging waste growth by 2008, to deliver reductions in packaging by 2010, and to identify ways to reduce food waste.

Re-use can also decrease waste volumes. This can be applied from households re-using glass milk bottles, to factories employing machinery capable of alteration for a range of functions.

The waste hierarchy suggests that recycling should be undertaken only after reduction and re-use have been exhausted. Many materials can be recycled; helping both to divert waste from landfill and reduce reliance on virgin materials. Organic waste can be composted to return nutrients and minerals to the Earth. However, recycling and composting are not always economically or environmentally viable options. In the UK, recycling and composting of waste has nearly quadrupled since 1996/97, reaching 27 percent of total MSW in 2005/06. Also, since 1998, the recycling of packaging waste has increased from 27 to 56 percent (DEFRA, 2007). In order to comply with the EU Directive, the UK has a target of recycling 40 and 45 percent of household waste by 2010/11 and 2015/16 respectively (DEFRA, 2006).

The UK relies on landfill as its primary method of waste management. This is a result of historical circumstances and geology (Figure 2). The large holes left by mining and quarrying activities were utilised as ready-made landfill sites. The ground conditions, often naturally impermeable, allowed the burial of waste with little liquid seepage and groundwater pollution; making landfilling a cheap option (DEFRA, 2005a). Only a fraction of MSW is incinerated in the UK in contrast to countries such as Sweden and Denmark who rely heavily on WtE. Sweden, for example, has deployed small and efficient WtE plants emphasising pollution control and energy efficiency over plant size. One tonne of waste was seen by Sweden as comparable in value to a barrel of oil and a fuel for power generation from steam (Naanovo Energy, 2007).

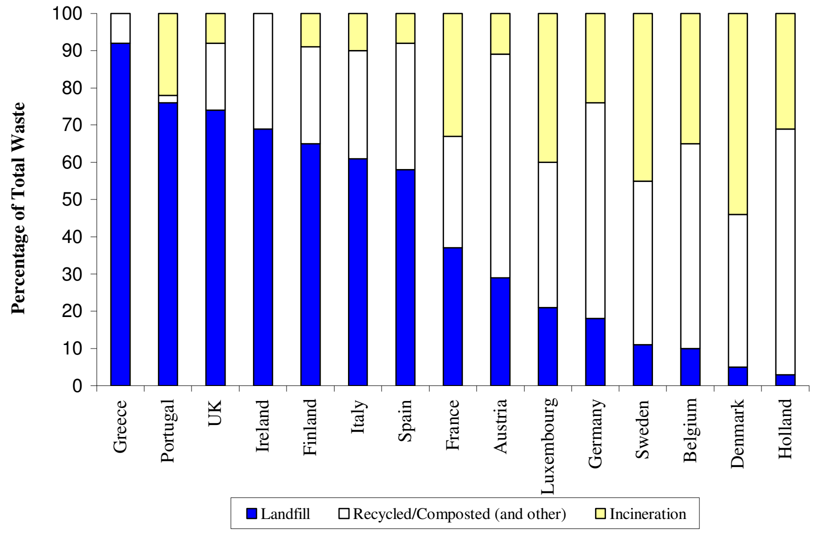


Figure 2: Municipal waste management in the EU 2005/06
Source: Compiled from DEFRA (2006)

Waste-based energy encompasses all technologies that recover energy from waste; the most prevalent examples being landfill gas, WtE, and biomass. Landfill gas plants collect the gas released during the decomposition of waste and use this as fuel. The gas can be used in manufacturing and running engines to produce energy. WtE facilities generate electricity and heat through incineration of MSW. WtE should not be confused with biomass energy which typically burns energy crops, forestry products, and organic agricultural waste. Other forms of waste-based energy generation include anaerobic digestion and gasification of organic waste.[2]

The compatibility of WtE and recycling has been challenged on the grounds that WtE will lead to less recycling. The main argument is that once communities have undertaken the long-term capital investment of building a WtE plant, they will forego recycling in order to ensure adequate supply of waste to the plant (Bahor and Weitz, 2006). However, evidence from progressive European countries suggests that a high level of WtE and recycling can coexist (Figure 2). Furthermore, a survey of all US communities with WtE in 2002 showed that recycling (33 percent) was greater than the national average of 28 percent (Chester et al., 2007).

[2] Anaerobic digestion is a process in which micro-organisms break down biodegradable waste in the absence of oxygen. It reduces the mass and volume of the input material producing methane and carbon dioxide rich biogas suitable for energy production.

3. The UK's Waste Treatment Policy

The disposal of waste through incineration dates back to 1874 when the first fully functional incinerator was constructed in Nottingham. The facility remained in operation for 27 years with the ash from the plant being used as building material. The world's first waste fired electricity generating plant was constructed in Shoreditch, London in 1885. By 1912, there were some 300 waste incinerators in the UK; 76 of which generated electricity (CIWEM, 2007). The early plants emitted ash, dust, and charred paper, which fell over the surrounding neighbourhoods. The resulting local opposition to WtE plants dampened the development of the technology in the UK, and efforts to deploy WtE came to a halt during World War II. Once mining and quarrying opened up large cavities for cheap waste disposal, WtE became a redundant option.

The 1960s and 70s saw a new period of plant construction. About 40 incinerators were built, but because the main objective was to reduce the volume of waste to ease the pressure on landfills, only five were equipped for power generation. Technical knowledge of WtE in the UK had virtually disappeared, and the new firms entering the industry constructed facilities using overseas designs at low cost. Maintenance costs rose above expectations, however, and numerous plant breakdowns made it necessary to provide emergency disposal sites for diverted waste. Landfill proved to be the more reliable alternative (Waste Online, 2007). Furthermore, there was a growing awareness of the invisible environmental and health implications of the largely unregulated emissions from WtE plants with relatively rudimentary emissions control equipment.

By the end of the 1980s, opinion regarding WtE began to change as a result of increased public awareness of the volume of waste sent to landfills. A further 18 plants have since been permitted by the Environmental Agency, with many smaller private projects authorised under Environmental Health powers granted to District and Borough Councils (WS WLP, 2005: 10). Figure 3 shows the highs and lows of WtE plant construction in UK from 1968 till 2008.

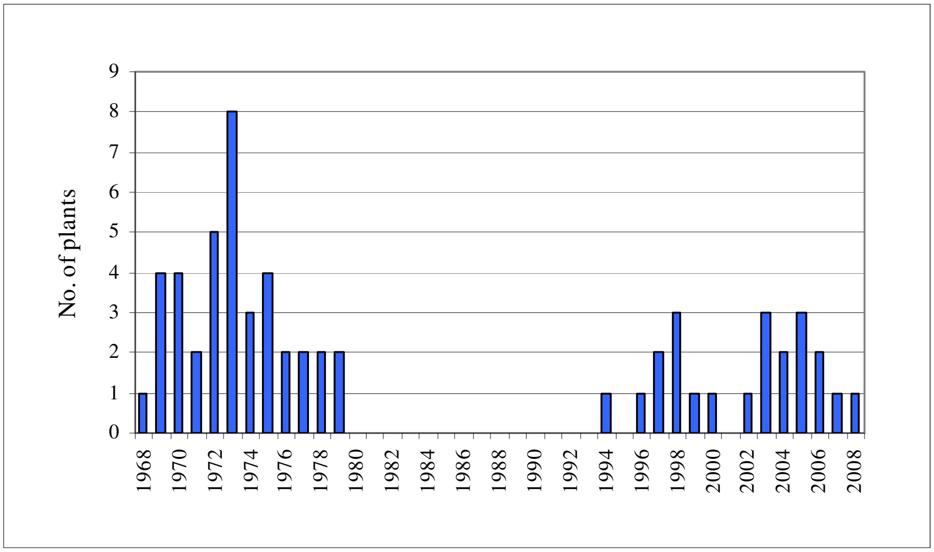


Figure 3: WTE plants commissioned in the UK (1968-2008)
Source: CIWM (2003)

Figure 4 shows MSW management within England by regions during 2005/06. Of the 28.7 million tonnes of waste, 17.9 million (62 percent) were sent to landfill, down from 19.8 million tonnes (67 percent) in 2004/05 (DEFRA, 2006). Around 37 percent of the waste was recycled, composted or incinerated with energy recovery, but with considerable regional variations. In the West Midlands, almost 31 percent of the total waste was incinerated with energy recovery, while the figure was only 9 percent across England.

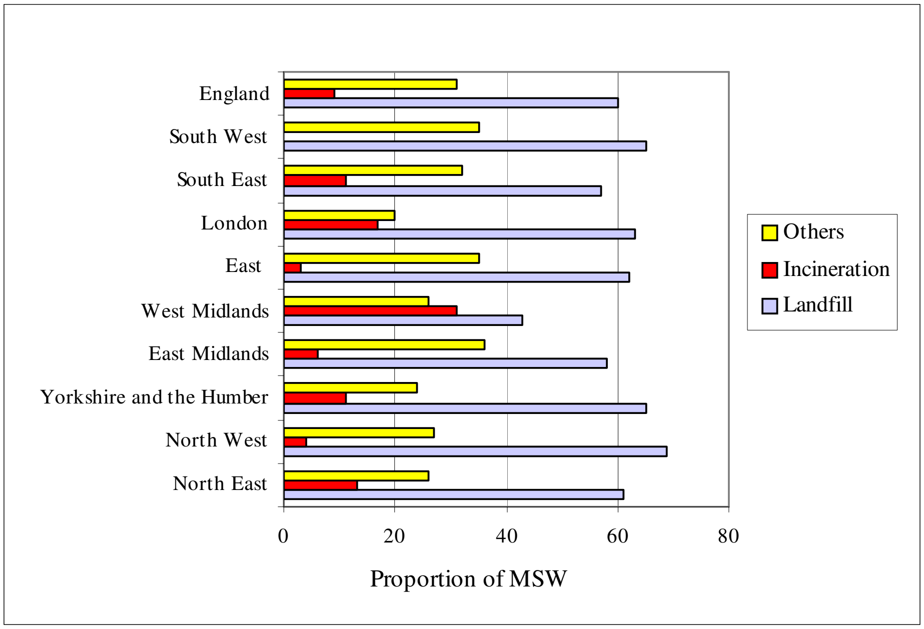
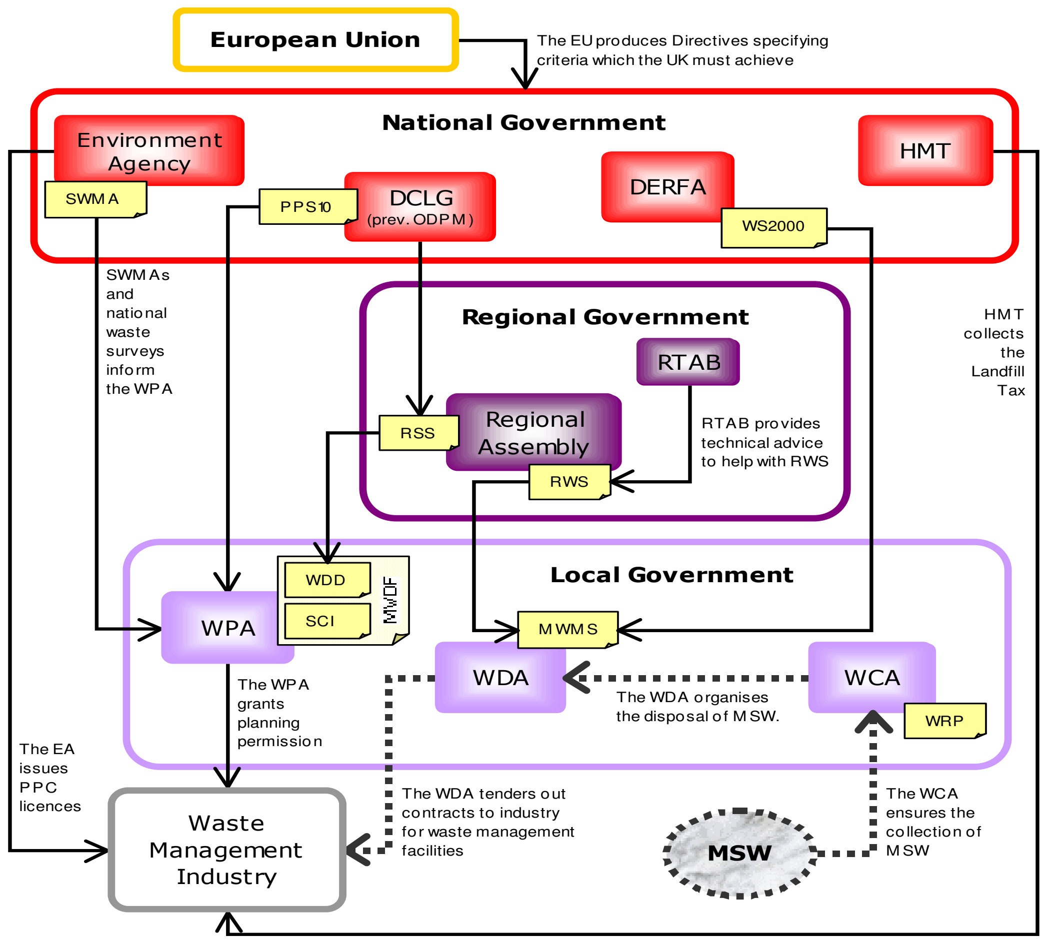


Figure 4: Municipal waste management in England by region
Source: Compiled using DEFRA (2007)

Establishing competitive markets for waste management externalities through the allocation of property rights is inherently difficult. The number of agents involved (on a local, national and global scale) makes defining rights difficult and the accompanying transaction costs prohibitively high. The alternatives are either market-based incentives or command-and-control policies. These instruments are capable of achieving a Pareto optimal outcome under the assumptions of a first-best world, in which government is benevolent and there is perfect competition and perfect information in the market. In the real world, based on the degree to which these assumptions break down, certain policies can be more appropriate than others.

The remainder of this section evaluates the policies that influence WtE decision making, taking into account the circumstances within which they operate. Each policy makes a contribution to the overall framework of MSW management. Figure 5 is a simplified representation of England's waste management decision-making structure. It depicts the relationships between the main bodies and policy documents which can be referred to throughout this section as a guide to how policy and policy making components fit together. While the structures in Scotland, Wales and Northern Ireland are similar to this system, the nature of local government and the names of equivalent bodies differ. We now turn to analysing five important policies affecting WtE decisions.



DCLG	Dept of Communities & Local Government	RTAB	Regional Technical Advisory Board
DEFRA	Dept for Environment, Food and Rural Affairs	RWS	Regional Waste Strategy
EA	Environment Agency	SCI	Statement of Community Involvement
HMT	HM Treasury	SWMA	Strategic Waste Management Assessments
MWDF	Minerals & Waste Development Framework	WCA	Waste Collection Authority
MWMS	Municipal Waste Management Strategy	WDA	Waste Disposal Authority
ODMP	Office of the Deputy Prime Minister	WDD	Waste Development Document
PPC	Pollution Prevention and Control	WRP	Waste Recycling Plan
PPS10	Planning Policy Statement 10	WPA	Waste Planning Authority
RSS	Regional Spatial Strategy		

Figure 5: Waste management policy in the UK
Source: Compiled from DCLG website, Bulkeley (2004), and WS WLP (2005)

3.1 Landfill tax

In principle, a Pigouvian tax is the most efficient instrument to correct market failure from negative externalities (Pasour, 1994). For example, internalising the external effect of a waste management option on global warming would involve taxing the non-carbon neutral GHGs emitted. One tonne of biodegradable municipal waste produces between 200 and 400m^3 of landfill gas as it decomposes. As of 2001, the methane emissions from landfill accounted for 25 and 2 percent of the UK's total methane and GHG emissions respectively (DEFRA, 2005c).

It is difficult, however, to determine whether the emissions are carbon neutral, as this depends on the type of materials in the waste stream and the landfill facilities used. The information requirement and high transaction costs, therefore, make the correct use of Pigouvian tax to internalise the costs of global warming difficult. Such a tax also involves evaluating the marginal social damage at the optimal (not current) level of emissions. It requires information on the damage functions of individual agents and costs of abatement. Imperfect information leads to second-best options where the benefits of a simple-to-operate landfill tax can outweigh those of a complex system aiming to correct each externality separately and directly.

The landfill tax was introduced in 1996, at a rate of £7 per tonne of MSW based on an assessment of the external cost of landfill. The tax aimed to account for all the external costs of landfill using a single instrument. The tax is a 'green tax' because it is not levied directly on emissions but on the tonnage of waste produced; a quantity which is correlated with the externalities of landfill.

The introduction of a "Landfill Tax Escalator" in 1999 first raised the tax by £1 per year, then by £3 per year from 2005. From 2008, the tax will rise by £8 per year. These increases were initially justified because the original research for deciding the landfill tax was a lower-bound estimate of the cost of landfill, having excluded the disamenity consequences from the calculation (Turner, 1998). The latter tax increases have been justified as a method of achieving targets for the diversion of waste from landfill as specified in the EC Landfill Directive.

The Landfill Tax and Landfill Tax Credit Scheme (LTCS) move in tandem and can influence each other.[3] The LTCS enables waste operators to provide funding to organisations through tax credits for qualifying environmental projects. It also enables landfill operators to claim a credit against their landfill tax payment if they make voluntary contributions to an approved environmental body for an approved project (Morris and Read, 2001).

Landfill tax can encourage the use of WtE and recycling as the cross price elasticity of demand for the different waste management options is positive. However, the increase in landfill tax has also led to an increase in fly-tipping (Morris and Read, 2001). The environmental and health impact of this waste can only be addressed when it is found and moved to an authorised place of disposal.

3.2 Landfill emissions trading scheme

Article 5 of the EC Landfill Directive (1999/31/EC) sets caps the quantities of biodegradable MSW that can be sent to landfill based on three target dates. The UK must achieve reductions of 75, 50 and 35 percent (from the 1995-tonnage) by 2010, 2013 and 2020 respectively. The Directive aims to minimise the impact of landfilling biodegradable waste on health and the environment, particularly with regard to methane emissions.

The targets of the Directive have been translated into local authority allowances and have been grandfathered on the basis of past landfilling activity. In England, these allowances have been tradable since April 2005 under the Landfill Allowance Trading Scheme (LATS). Under this scheme, each Waste Disposal Authority (WDA) is assigned a limited number of allowances for landfilling BMW in England. LATS is aimed at reducing the effect of waste management on global warming, as well as reducing local pollution and improving the use of raw materials. Tradable allowances seek to achieve an aggregate quota at lowest cost. A market for the permits establishes one price for a tonne of waste landfilled and ensures that the marginal cost of abatement is equalised across local authorities. Authorities that can divert waste from landfill at low cost will do so, while those that find reducing landfill expensive

[3] The LTCS was introduced on 1 October 1996 with subsequent reforms made on 1 October 2003.

can purchase allowances instead. However, the value of the allowances is unknown and the Government does not set price floors or ceilings. In theory, the price will be determined by demand and supply for allowances which could be £0 per tonne in case of excess supply or rise to the level of the penalty (£150 per tonne) for exceeding the allocation. In case of a WDA missing its target for any year, the government has indicated that it will fine the authority at a rate of £150 for each additional tonne (LATS, 2005).

Achieving the correct allowance price requires a competitive market, which does not exist. Most local authorities are operating at their allocated allowances leaving only a small number of them in the market. As the targets become more stringent over time, the differences in costs of landfill will become more apparent. Consequently, new entrants encourage the emergence of a competitive market for allowances. The level of allowances will be reduced from year to year to ensure that the EU Directive's overall limits are met. LATS will drive MSW away from landfills and will result in a greater amount of waste being incinerated, rewarding councils who incinerate waste in the UK (House of Commons, 2005). The policy could be improved by making the allowances for Wales, Scotland and Northern Ireland, tradable and thus increasing the number of potential entrants to the market.

3.3 Renewable obligations

The Renewable Directive (2001/77/EC) aims to increase the share of electricity coming from renewable sources in order to achieve sustainable development, increase security of energy supply, and reduce emissions of GHGs. This was translated into UK law in April 2002 in the form of the Renewable Obligation Order, replacing the Non-Fossil Fuel Obligation (NFFO) as the instrument for supporting the development of renewable energy. Suppliers must purchase annually increasing percentages of their electricity from accredited renewable sources to 15.4 percent by 2015/16.

The licensed renewable electricity generators are issued Renewable Obligation Certificates (ROCs) by Ofgem. The certificates are tradable and there is a buyout price for each MWh of the statutory requirement not met to ensure that the renewable electricity price will not reach unacceptably high levels. Figure 6 shows the breakdown by technology of ROCs issued in England as of February

2006. Landfill gas attracted 29 percent of the total ROCs issued in 2005/06 and biomass attracted 7 percent.

Pyrolysis, gasification, and landfill gas are all eligible waste-based energy sources. Although biodegradable waste processed in WtE plants qualifies as a renewable energy source in the Directive, it is not eligible for ROCs. WtE can be source of renewable energy, and the policy of not issuing ROCs to WtE plants may, therefore, need to be revisited. In March 2006, it was decided that a WtE plant would qualify for ROCs only as a combined heat and power (CHP) plant. A report by ILEX Energy for the DTI found no environmental basis for differentiating between technologies for energy recovery from waste, as all plants have to meet the same emissions targets as specified by the Waste Incineration Directive (2005a).

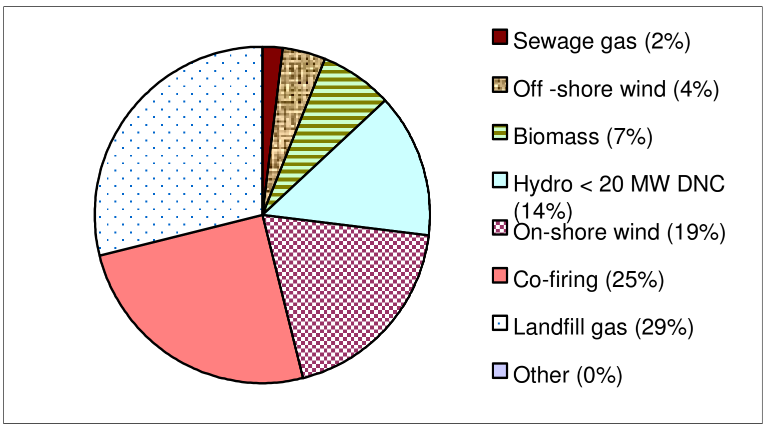


Figure 6: ROCs by technology type in the UK
Source: Compiled from Ofgem (2007)

The decision to exclude WtE (and large hydro) from ROCs is justified by the Government on the grounds that the technology is already capable of competing with electricity from fossil fuels without additional support (DTI, 2000). By distinguishing between renewable technologies, ROCs takes on a second policy aim of encouraging advancement of newer technologies that are not currently commercially viable. Using ROCs for this additional purpose is likely to compromise the efficiency with which it can achieve is original goal of increasing renewable energy generation.

3.4 Pollution prevention and control licences

The Integrated Pollution Prevention and Control Directive (96/61/EC) provides the basis for the UK's waste licensing system by requiring the existence of a waste regulation authority and setting limits for pollution to air, water and soil. More recently, tighter controls have been specified for WtE emissions in the Waste Incineration Directive (2000/76/EC). The limits were chosen using the Best Practicable Environment Option (BPEO) principle; a procedure aiming to minimise health and environmental damage at an acceptable cost.

In England and Wales, the Environment Agency is responsible for issuing Pollution Prevention and Control (PPC) licenses to plants meeting the criteria (EA, 2007). The introduction of PPC licences has led to significant cuts in the emissions of pollutants from WtE plants. Between 1993 and 2003, sulphur dioxide emissions fell by 99.38 percent, lead emissions by 99.5 percent and dioxin emissions by 99.99 percent (ESA, 2006). Cost of compliance has also resulted in the closure of some WtE plants.

Some interest groups dispute the government's findings, and health concerns remain a sticking point during many WtE plant applications due to the perceptions of local residents. In some cases, plants which have secured a PPC licence have been refused planning permission on the grounds that the *perception* of effects would negatively affect the use of the surrounding land (CIWM, 2003). Government reports indicate that the impact on human health from WtE emissions is, at most, minor (DEFRA, 2004) and the emission limits are far stricter than for other forms of electricity generation (ILEX, 2005a).

3.5 Planning permission process

In addition to a PPC licence, new WtE facilities must obtain planning permission from the local Waste Planning Authority (WPA). The planning permission process ensures that firms consider the impact of the plant on the local community and internalise and minimise local concerns about disamenity, congestion and health. To help speed up the planning permission process, each WPA is required to produce a Waste Development Document (WDD) setting out the criteria upon which planning permission requests will be judged. These documents also list specific sites that are well-suited to development and hence most likely to be granted planning permission.

The planning permission process has been criticised for its separation from the management side of waste facility provision. The Waste Disposal Authority (WDA) is responsible for negotiating contracts with the waste management industry for MSW plants. The authority produces a Municipal Waste Management Strategy (MWMS), which details its programme for sustainable waste management, including the types of facilities needed to achieve targets set at the national and regional level. Industry is then invited to outline proposals for achieving the MWMS and to choose the most preferred contract. Industry therefore needs to find a way to meet the requirements of both the MWMS and the planning permission process. The co-ordination between the WDD and the MWMS is insufficient due to their different processes and timetables (Bulkeley, 2004) leading to tensions between the two authorities.

The division of planning and management policies is a recognised problem, and there are initiatives to improve co-ordination. Planning Policy Statement 10 (PPS10) made the production of a Regional Spatial Strategy (RSS) mandatory for each regional assembly (SITA UK, 2007). The strategy provides guidance from the regional level on land that is acceptable for planning permission. The Regional Assembly also produces a Regional Waste Strategy (RWS) and is therefore in a good position to co-ordinate planning and management. Regional government is also expected to encourage co-ordination on waste management between adjoining local authorities. This is beneficial given that waste often crosses local authority boundaries for disposal; and joint management gives authorities greater flexibility in the size and type of WtE facilities they can provide.

4. Cost-Benefit Analysis of WtE

In order to make overall assessments of waste management options such as landfill, WtE, recycling or composting, it is important to estimate and aggregate the costs and benefits associated with the different options while taking into account their key determinants. CBA is an established applied welfare economics approach to estimate and compare the total costs and benefits of alternative policies and scenarios. In this section we develop the main parameters of a social CBA for assessing the socio-economic implications of WtE and alternative waste management options.

4.1 Private costs and benefits

Construction and operation of waste treatment facilities involve a range of direct and indirect private costs. Direct costs include the operating and maintenance (O&M) costs that vary with output. O&M costs include raw materials, labour cost, maintenance of facilities/equipment and training programmes. Indirect or fixed costs do not vary with output. The facilities require substantial capital costs in predevelopment and construction. Land is required to locate the plant and store waste for processing. Capital costs constitute substantial sunk costs. In addition, there are interest costs or lost earnings associated with delays in the planning permission and licensing process, and a tipping fee for disposing of unwanted residual material from the combustion process (Tsilemou and Panagiotakopoulos, 2006).

Plant size has implications for profitability and feasibility of WtE plants. A doubling of plant size can raise the capital costs by just 70 percent (CIWM, 2003) and achieve even larger economies in labour costs (Jeral, 2007). An increase in the amount of waste processed increases all revenue sources proportionately, thus making larger plants more economic. Figure 7 shows the reduction in total cost of treating MSW plant capacity in the UK. Despite the economies in WtE plant capacity, profitability has also been demonstrated at low capacities in Europe and subsequently in the UK where four new builds process less than 100,000 tonnes per annum. Furthermore, the social benefits of smaller plants, in terms of lower (perceived) negative health, congestion and visual effects, tend to be lower.

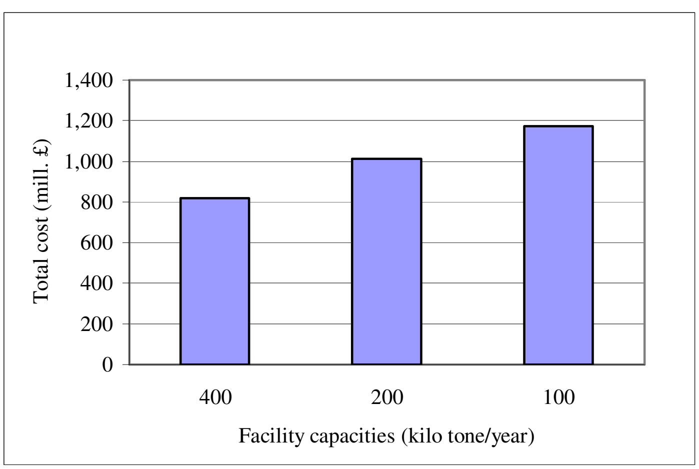


Figure 7: Costs of different WtE capacities

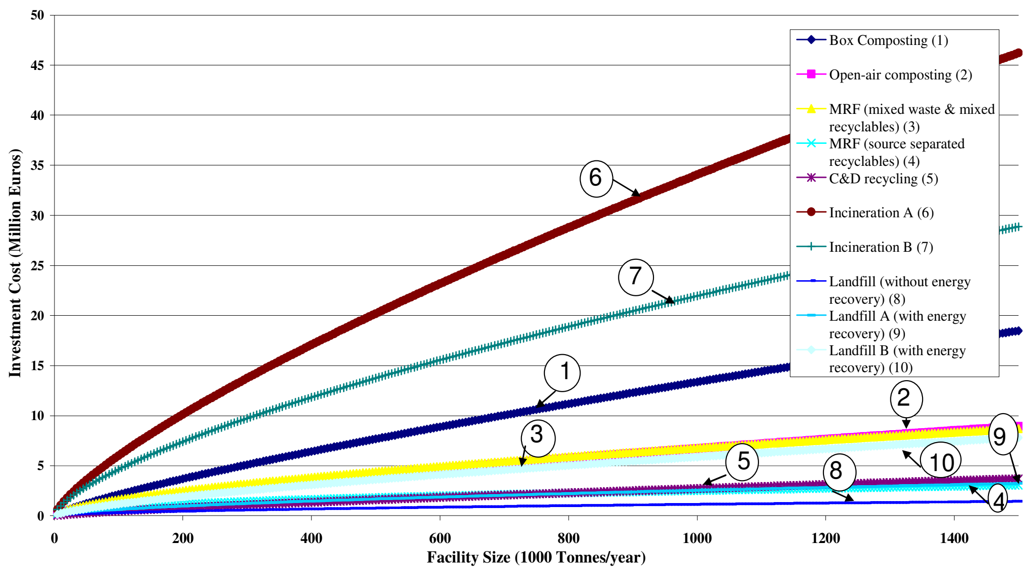
Source: Compiled from DEFRA (2007)

The private cost of WtE plants also depends on the incineration technology. One tonne of waste is capable of generating about 2 MWh of heat and 0.65 MWh of electricity (Rand, 2007). Revenues from heat production can, at least partially, offset the higher costs of a modern WtE plant. The treatment cost per tonne of waste using WtE in Denmark, for instance, is about the same as the price of an empty rubbish sack which is $0.4 a week per household (Rand, 2007).

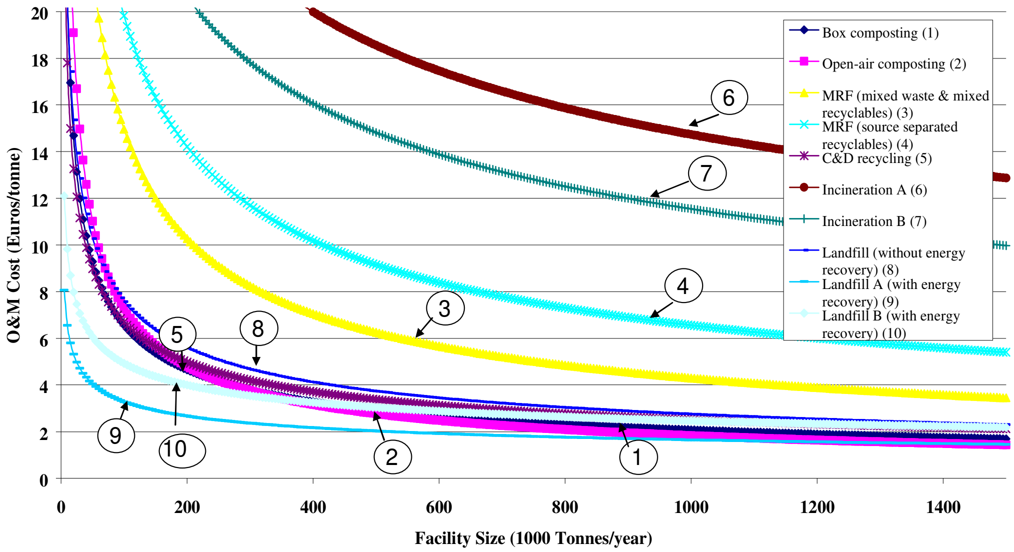
Table 1 shows cost functions for different treatment facilities in terms of capacities. The cost functions for incineration with heat and electricity (Incineration H&E hereafter), incineration with electricity (Incineration E hereafter) and landfill without energy recovery are adopted from COWI/S (2002). Landfill A (for small facilities hereafter) and Landfill B (for large facilities hereafter) both recover energy. The capacity and the amount of waste sent to the facility per year is denoted by (x1) and (x2) respectively. The cost functions for the remaining types of facilities are based on observation of actual plants in Europe including the UK and are adapted from Tsilemou and Panagiotakopoulos (2006). See Figures 8 a-c for comparison of O&M, capital, and total costs of the main different waste treatment options for different facility sizes.

Facility	Cost Functions		Facility size (x in 1000 tonnes per year)
	Investment (y)	O & M (y)	
Box composting	Y= 2000*(x1)^0.8	Y= 2000*(x2)^-0.5	2-120
Open-air composting	Y=4000*(x1)^0.7	Y=7000*(x2)^-0.6	2-100
MRF (mixed waste & mixed recyclables)	Y=11411*(x1)^0.6226	Y= 7160*(x2)^-0.5418	2.5-20
MRF (source separated recyclables)	Y=33322*(x1)^0.4734	Y= 4681*(x2)^-0.4804	2.5-20
C&D recycling	Y=624*(x1)^0.7668	Y=834*(x2)^-0.4249	25-200
Incineration H&E	Y=9346*(x1)^0.754	Y=1372*(x2)^-0.333	120-380
Incineration E	Y=17778*(x1)^0.6757	Y=1572*(x2)^-0.3605	120-380
Landfill (without energy recovery)	Y=3863*(x1)^0.5719	Y=1121*(x2)^-0.4399	10-300
Landfill A (with energy recovery)	Y=6000*(x1)^0.6	Y=100*(x2)^-0.3	0.5-60
Landfill B (with energy recovery)	Y= 3500*(x1)^0.7	Y=150*(x2)^-0.3	60-1500

Table 9: Approximate cost functions for waste treatment options
Source: Tsilemou and Panagiotakopoulos (2006) and COWI (2002)



(a)



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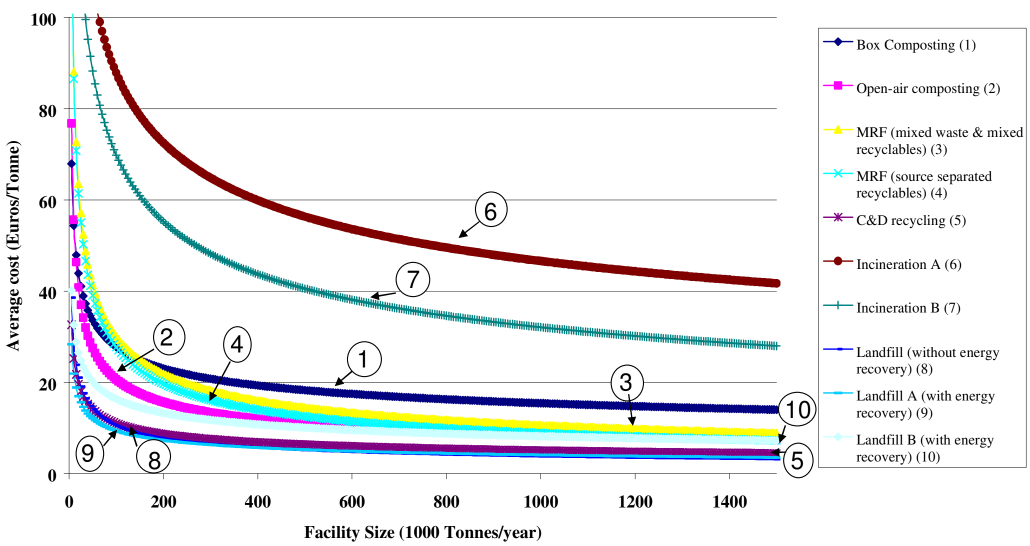


(c)

Figure 8: Cost curves for waste management options
Source: Based on Tsilemou and Panagiotakopoulos (2006) and COWI (2002) (Annualised costs with 20 years plant life and 8 percent discount rate).

There are four sources of revenues for WtE plants: gate fees, energy sales, recycling metal post incineration, and combustion residuals. The gate fees and the sale of energy (electricity and heat) are the main sources of revenue, although the proportions may vary across countries. About 70-80 percent of revenue is from the gate fee (the charge for each tonne of waste accepted) with 20-30 percent generated from the sale of electricity in the case of the UK (Jeral, 2007). There is some evidence that the level of gate fees tend to decrease as the energy produced increases. Some concerns have been raised about the true nature of gate fees and its mirror effect upon the externality cost. The experience of Denmark which has the lowest gate fees in Europe shows an inverse relationship between the gate fee and energy production (Figure 9). Increasing the landfill gate fees will help to divert waste from landfill towards other waste treatment options.

The other revenue sources are rather smaller but nonetheless can be important. These revenues are from recycling of the metal collected after combustion and selling non-landfilled combustion residuals (e.g. ash) as aggregate materials to the construction industry.

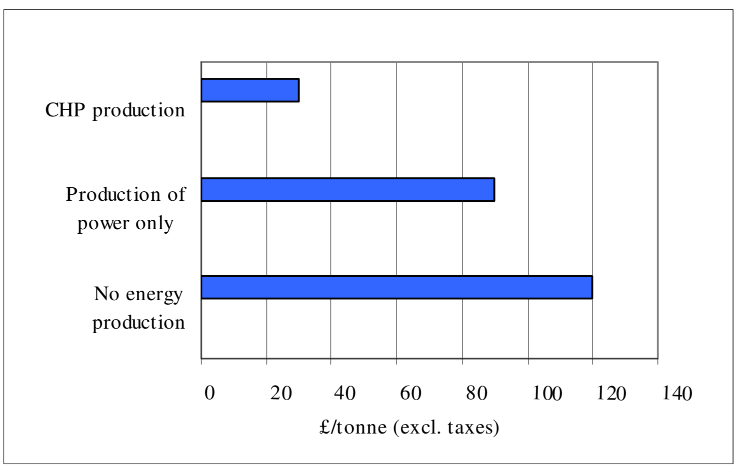


Figure 9: Waste treatment and gate fees in Denmark
Source: Adapted from Anderson (2006b)

4.2 Other factors affecting cost

Other factors that can influence costs include (i) plant efficiency, (ii) composition of the waste stream, and (iii) alternatives for both waste management and electricity generation. Efficiency of plants depends on the technology and design specifications. Over time, the learning effect from research and development (R&D) and learning-by-doing from capacity deployment plants leads to technical progress and cost reductions (Jamasb, 2007). A new WtE plant can operate at 25 percent efficiency, while the oldest plants in the UK achieve 18 percent (Jeral, 2007). More efficient plants have higher capital costs, although they generate more revenue through increased electricity generation and have higher positive externalities by improving energy security and reducing net GHGs. By employing combined heat and power (CHP) technology, efficiency can be raised up to 80 percent. However, heat delivery networks to utilise both electricity and heat including those from WtE require substantial investments. The investment requirements in heat delivery networks are uncertain and hence have not been included in the analysis in this paper.

The composition of the waste stream is important in determining the costs and benefits of WtE. MSW is made up of a large number of materials with different Calorific Values (CV).[4] For example, textiles and plastics have high CVs, while those for metals and glass are negligible. The composition of the waste stream will depend on the level and type of recycling and the consumption choices of the locality. Urban waste contains a relatively high proportion of plastics and WtE in these areas could generate more energy (Porteous, 2005). Table 2 shows examples of the energy required by various waste treatment options for a tonne of different waste materials.

Waste element	Energy supplied if burned	Energy for virgin manufacture	Energy for recycle manufacture	Energy saved if recycled
Newsprint	8	27	22	5
Corrugated	7	17	17	0
Tissue paper	8	12	14	-2
Aluminium	0	100	5	95
Steel	0	48	23	25
Glass	0	10	7	3

Table 2: Energy Value Index (million BTUs per tonne)
Source: Compiled from US Department of Energy (1992)

4 CV measures the amount of energy released when a material is combusted.

We have so far assumed that WtE plants divert waste from landfill and that their electricity replaces energy from fossil fuels, however landfill diverted waste could alternatively be recycled. Although the compatibility of WtE with recycling has been established, the profitability of recycling is less clear. The picture is complicated for materials like paper and plastics, which have high calorific value and established recycling markets. The quality of paper is rapidly degraded through the process of recycling. Plastics vary widely, and some can be more successfully and cheaply recycled than others. Plastics used in food or medical packaging are often not suitable for recycling (Miranda and Hale, 1997).

Recycling is electricity intensive during the extruding process and the expensive equipment used increases the overhead costs. Low economic margins insufficient to remunerate small businesses have lowered interest among private operators in recycling (Massarutto, 2007). Analysing the costs and benefits of each material separately does not necessarily capture the nature of the problem. The decision to recycle has an additional cost of sorting the materials. It can increase pollution and congestion if separate vehicles are needed for collecting the separated waste (CIWM, 2006).

Another issue is whether WtE replaces energy from fossil fuels, or some other energy supply, e.g. nuclear power or a renewable source. Moreover, and notwithstanding the debate over air toxics, WtE must be cost competitive with fossil fuel energy when private and social external costs are taken into account (Miranda and Hale, 1997).

4.3 External Costs and Benefits

WtE plants emit some pollutants, which include sulphur dioxide, lead, and dioxins which are linked with damage to health and the environment if they occur in high enough concentrations (Ares and Bolton, 2002). Also, the local population will experience health effects or congestion from vehicles transporting the waste to the plant site. Local residents may also experience some disamenity consequences from having a WtE plants in the neighbourhood as these are often perceived as ‘unsightly’ or produces odours.

WtE also has positive environmental externalities. According to the Waste-to-Energy Research and Technology Council founded by EEC, WtE plants have significant environmental benefits and for each tonne of MSW used they reduce consumption of oil and coal by about one barrel of oil and 0.26 tonnes respectively (ISWA, 2007). A positive global externality from WtE is reduction in emissions of GHGs. The net change in greenhouse gas emissions from WtE can be assessed by considering the energy generation and waste management alternatives that it would replace. The gas produced in a landfill consists mainly of methane; a potent GHG with a warming potential that is 21 times that of carbon dioxide (Franklin Associates, 1994).

As the landfilled material degrades it produces leachate (a liquid containing toxic organic compounds, heavy metals, ammonia and pathogens) which collects at the bottom of landfills and causes groundwater pollution if it escapes. When biodegradable waste is combusted in a WtE plant, the amount of carbon dioxide released is equal to that removed from the environment during the production of the original material. This has no impact on the environment over the product life cycle and has been termed as 'carbon neutral'. However, when biodegradable waste is placed in the oxygen deprived environment of landfill pit, the 'landfill gas' that is formed from biodegraded waste is 65 percent methane and 35 percent carbon dioxide (DEFRA, 2000). Thus, the lifecycle impact of biodegradable products that are sent to landfill is a net increase of GHGs. Figure 10 shows the social (private and externality costs) for landfill and incineration facilities at different scales.

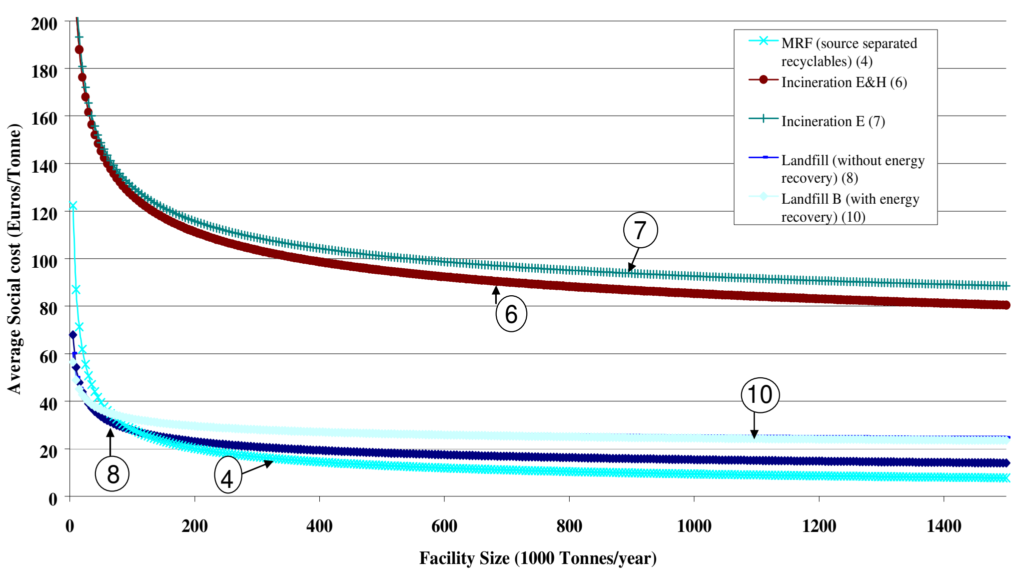


Figure 10: Comparison of social cost functions

Table 3 presents a social CBA framework for main waste management options. The table uses the external costs of waste management options (as in Figure 10) as well as the private costs (investment plus O&M costs) of the four main waste management options. The use of WtE as a waste management option also has positive externalities. WtE reduces the space required by landfill by about 90 percent with an added benefit of avoiding the aqueous emissions from landfills. Recovery of ferrous and non-ferrous metals is also possible from WtE. Moreover, as the UK is gradually becoming a net importer of petroleum, WtE can play a role in improving energy security.

Facilities	Private costs (per tonne of waste)	External Costs (per tonne of waste)
a) Incineration E (with electricity recovery only)	Investment and O&M = €51.23 Facility size = 250000 tonnes	Damage from emissions to the air (mainly NO_x and SO_2) = €50[1] CO_2 = €2.55 (low) - €12.03(high)[1] Disamenity impacts = €8[1] Total = €60.55 - €69.67
b) Incineration E&H (with heat and electricity recovery)	Investment and O&M = €68.18 Facility size = 250000 tonnes	Damage from emissions to air (mainly NO_x and SO_2)= €28.18[1] CO_2 = €2.55 (low) - €12.03 (high)[1] Disamenity impacts = €8[1] Total = €38.73- €48.21
c) Landfill (without. any form of energy recovery)	Investment and O&M = €9.12 Facility size = 200000 tonnes	Global warming (mainly consist of CH_4) = €8[1] CO_2 = €2.13 (low) - €10.04 (high)[1] Damage from leachate = € 1.5[1] Disamenity impacts = €10[1] Total = €21.63 - €29.54
d) Landfill B (with energy recovery)	Investment and O&M = €7.7 Facility size = 700000 tonnes	Global warming (mainly consist of CH_4) = €5[1] CO_2 = €1.27 (low) - €6.01(high)[1] Disamenity impacts = €10[1] Total = € 16.27.- €21.01

e) Recycling/Composting (MRF source separated)	Investment and O&M = €19.7 Facility size = 200000 tonnes	CO_2 = €0.31 - €1.49[3] Pollution from transportation = €0.16[3] Total = €0.47 - €2.65
f) Coal fired plant generating electricity	Investment and O&M = €25.64[2]	CO_2 = €9.5 (low) - €44.89 (high)[4] Damage from other pollutants = €13.74[4] Total = €23.24 - €58.63
Notes: 1. European Commission (2000). 2. Anderson (2006a). 3. European Commission (2001). 4. Final Report ExterneE-Pol (2005, pp.35).		
Assumptions: i. 1 GBP£ = €1.4717 and 1 US$ = €0.729 in this study. ii. 2005 is assumed to be the base year for the costs calculation with RPI=185.2. iii. One tonne of waste can produce 2 MWh of heat and 2/3 MWh of electricity if incinerated. iv. The low and high costs of CO_2 are €13.12 t/CO_2 and €62 t/$C0_2$ respectively (Hope and Newberry, 2008). v. The private costs have been estimated from the graphs (see Figure 8) and annualised using 2005 prices. vi. The disamenity impacts from a coal fired plant are not included in this study.		

Table 3: A comparative cost analysis (€/tonne of waste)

WtE also has a positive emissions displacement effect in terms of climate change targets. One tonne of combusted rather than landfilled MSW reduces emissions of GHGs by 1.2 tonnes of carbon dioxide. Although the methane production from landfill is counterbalanced by the subsequent carbon dioxide production from WtE, the greenhouse effect of methane is significantly more damaging.

5. Analysis of the UK's Waste Options

On the basis of the CBA framework established in Section 4, we assess alternative waste management options for the UK in a time span of 25 years. The analysis uses actual data for 2005/06 as the basis for evaluating alternative scenarios until 2030/31. We first present a business-as-usual (BAU) scenario which assumes a continuation of the UK's current waste management practice. We then assess two scenarios that assume achieving the EU Directive targets.

As carbon prices are important for the future adoption of renewable technologies we assess a high and a low carbon price scenario.

5.1 Scenario 1 – Business-as-Usual (BAU)

Waste arisings

Assuming a waste growth rate at 2 percent per annum, Figure 11 shows the projections for MSW arisings from 2006/07 till 2030/31. The 2 percent growth rate is in line with estimates of growth in waste volume at 1.5 and 2.5 percent by DEFRA.[5] The amount of MSW produced in the UK during 2005/06 is estimated to be 28.7 million tonnes. Using a 2 percent waste growth, the MSW produced will reach 43.05 million tonnes in 2015, 50.23 million tonnes in 2020, and 64.58 million tonnes in 2030.

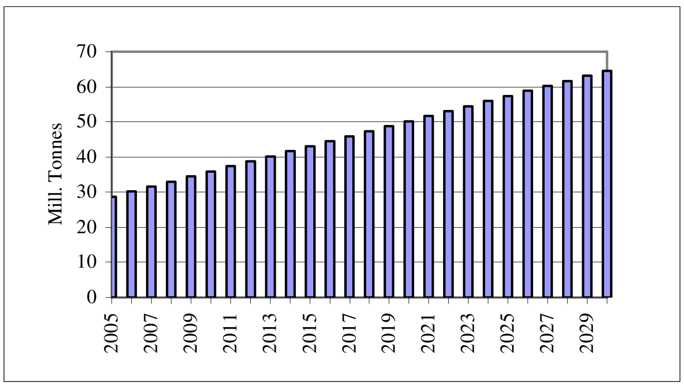


Figure 11: Estimated MSW arisings (2005-30)

Waste allocations

There are currently 19 WtE plants in the UK with 4 plants producing heat and electricity and one plant generating heat only (Longden et al., 2007).We assume that the amount of waste allocated to incineration with heat and electricity is proportionate to the number of EfW plants producing heat and electricity against the total number of EfW plants. Table 16 shows the estimated amount of waste sent to treatment facilities as per the assumptions made. The waste allocation for the year 2030 is projected accordingly.

Year	Estimated amount of MSW treated (million tonnes)

[5] See DEFRA (2007, Annexes).

	Incineration E&H	Incineration E	Landfill	Landfill B	Recycling/ Composting
2005/06	0.7	1.9	5.5	12.4	7.7
2015/16	0.85	2.5	7.1	15.9	9.9
2020/21	0.93	2.7	7.8	17.6	11.0
2030/31	1.14	3.3	9.5	21.4	13.4

Table 4: Estimated amount of MSW treated

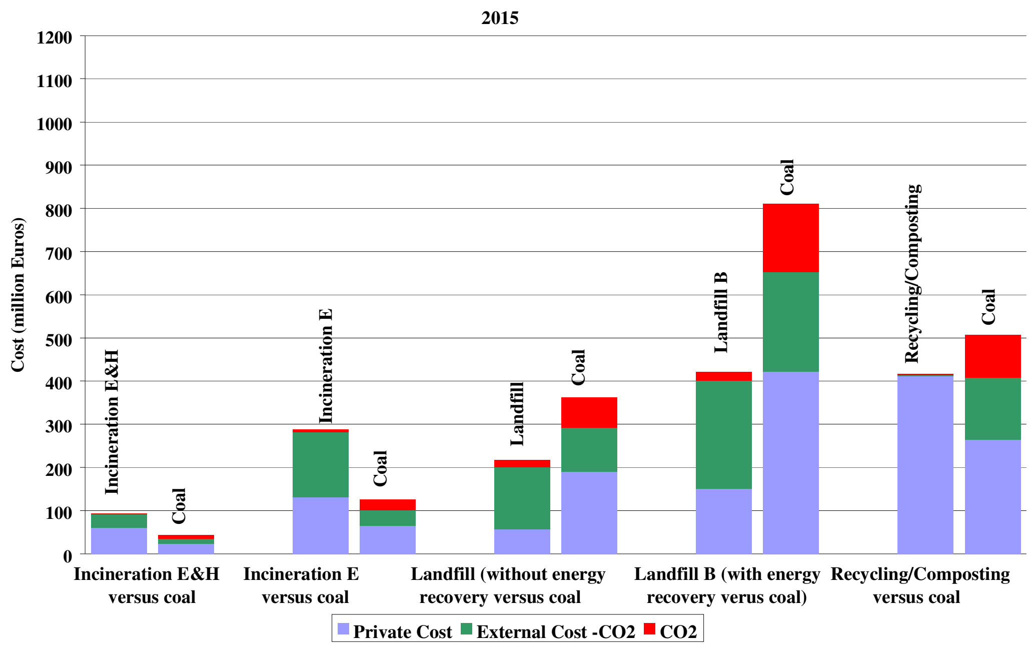
Capacity and number of plants required

The following assumptions are made with regard to the capacity of typical facilities per year. Incineration E&H: 250,000 tonnes, Incineration H: 250,000 tonnes, Landfill (without energy recovery): 200,000 tonnes, Landfill B (with energy recovery): 700,000 tonnes, and Recycling/Composting: 200,000 tonnes. It is assumed that 5 percent of the existing capacities will be replaced each year. Technological progress is assumed to reduce the total private cost of the facilities by 1.5 percent annually. Table 5 shows the estimated number of plants required for different technologies from 2005/06 to 2030/31. These estimates are based on a 2 percent annual growth in MSW with 9 percent incineration, 62 percent landfill, and 27 percent recycling/composting.

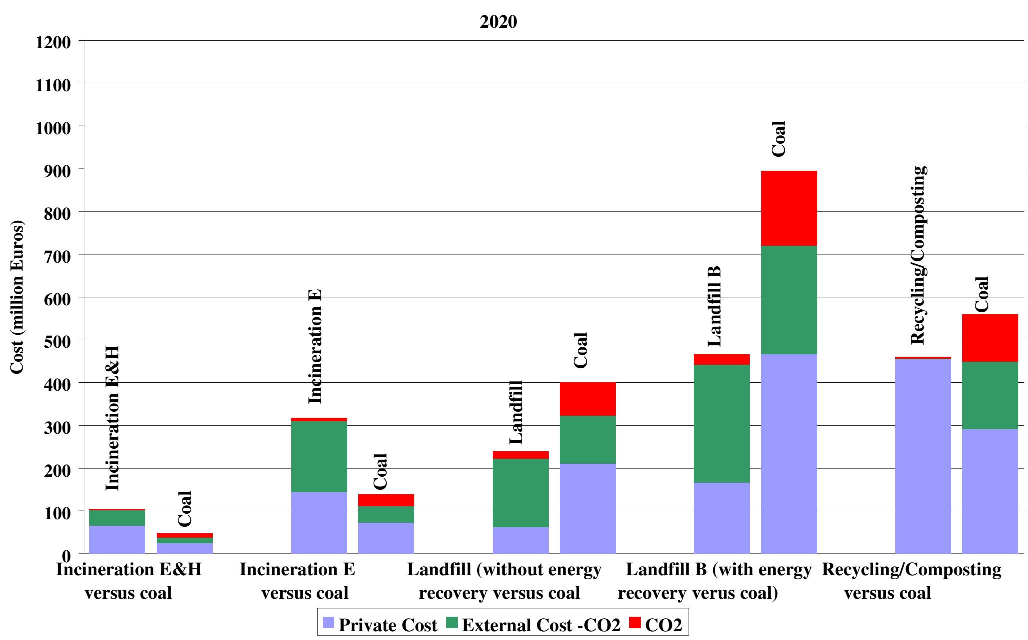
Year	Estimated number of additional plants required				
	Incineration A (heat and electricity)	Recycling/Composting (MRF source separated)	Incineration B (electricity only)	Landfill without energy recovery	Landfill B (with energy recovery)
2015/16	3.33	99.33	9.86	29.82	26.49
2020/21	3.68	109.67	10.88	32.92	29.24
2030/31	4.49	133.68	13.27	40.13	35.65

Table 5: Estimated number of additional plants required

Figures 12 a-c shows that provided the price of coal does not increase, coal power is cheaper than the incineration plants in terms of private costs. However, the contribution of a coal fired plant towards global warming is higher than those of waste treatment options and is certain to increase as the amount of MSW being treated increases. External costs from a coal fired plant may be higher than that of the incineration plants, as the damage from disamenity impacts from coal power is not included in this study.



(a)



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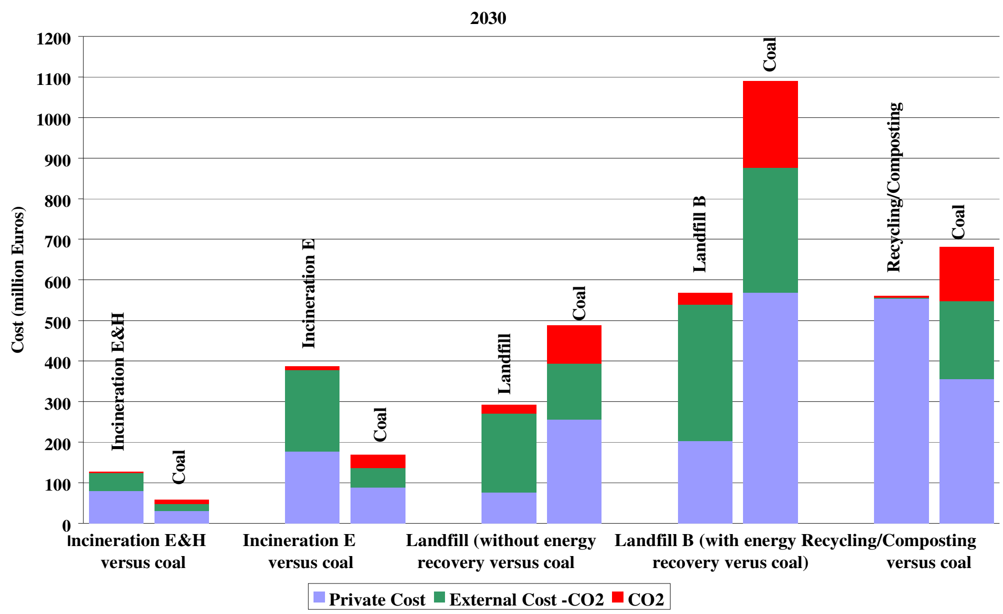


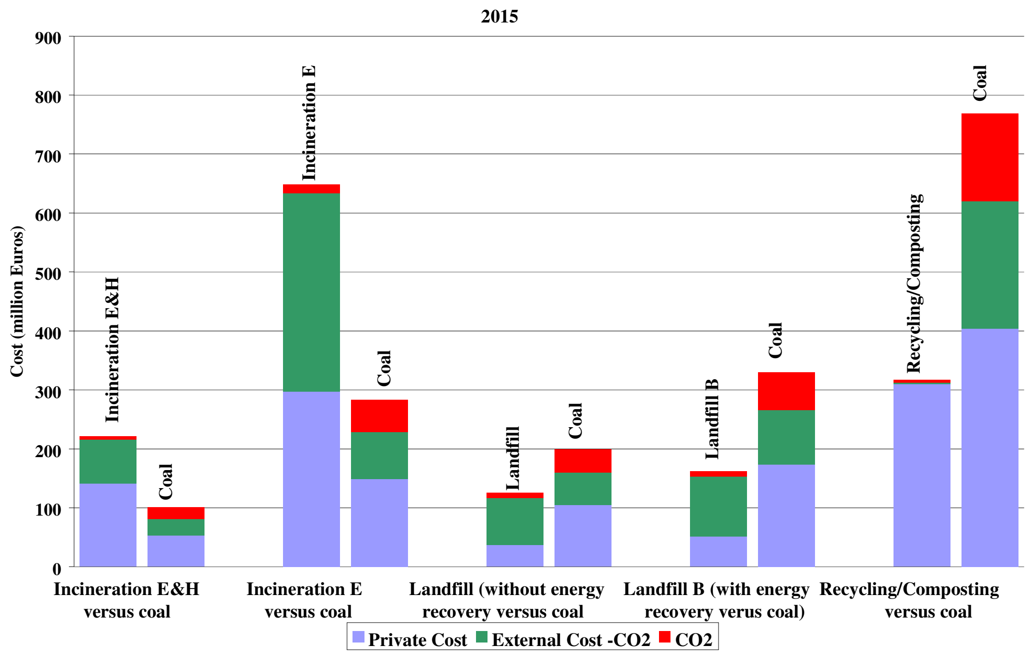
(c)

Figure 12: BAU scenario –CBA Cost of waste management options vs. coal power

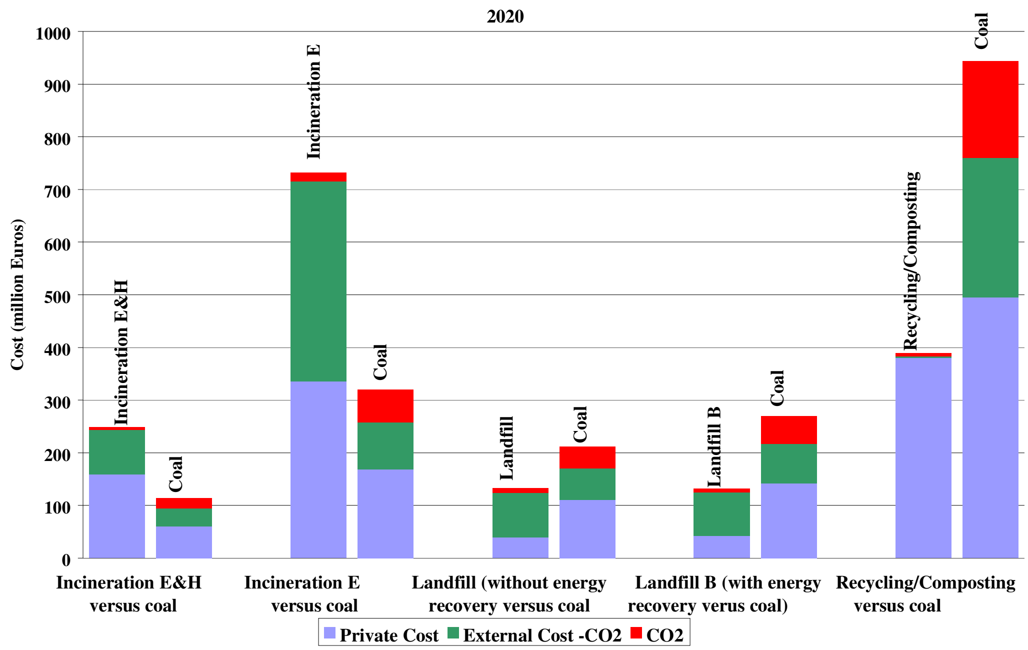
5.2 Scenarios 2 and 3 – Meeting EU Directive Targets

In order to comply with the EU Directive, the UK has adopted targets for waste landfilled, incinerated and recycled/composted. The aim is to landfill 62.4%, 55.6%, and 50% of the total MSW landfilled in landfill with energy recovery in 2015, 2020, and 2030 respectively. The targets are set in the waste strategy in DEFRA (2007). The recycling/composting rate is assumed to increase from 40 percent in 2010 to 45 percent in 2015, 50 percent in 2020 and reaching 60 percent in 2030. It is assumed that the UK will incinerate 22.5, 23 and 30 percent of total MSW in 2015, 2020 and 2030 respectively. The waste to be landfilled without energy recovery is assumed to be 37.6% in 2015, 44.4% in 2020 and 50% in 2030 of the total amount of MSW to be landfilled in the respective years. The estimates for 2030 are based on the UK's targets for 2020.

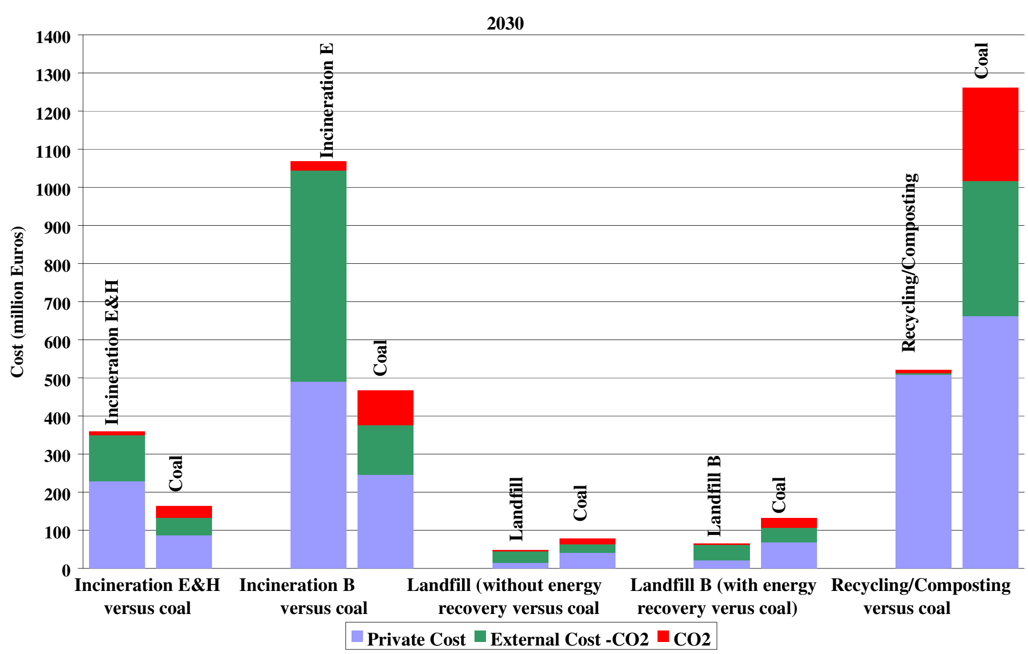
The costs of meeting the EU Directive target can be affected by the cost of CO_2 emissions. Therefore, we analyse the costs of meeting the EU Directive targets under a low and a high CO_2 cost scenario – i.e. €13.12 t/$C0_2$ and €62 t/$C0_2$ respectively (Table 3). As shown in Figures 13 and14 a-c, the price of carbon is important for assessing the social cost of waste management options. The benefits of the options in terms of global warming and external costs is significant in relation to that of coal power as shown by the differences in the CO_2 and total external costs.



(a)

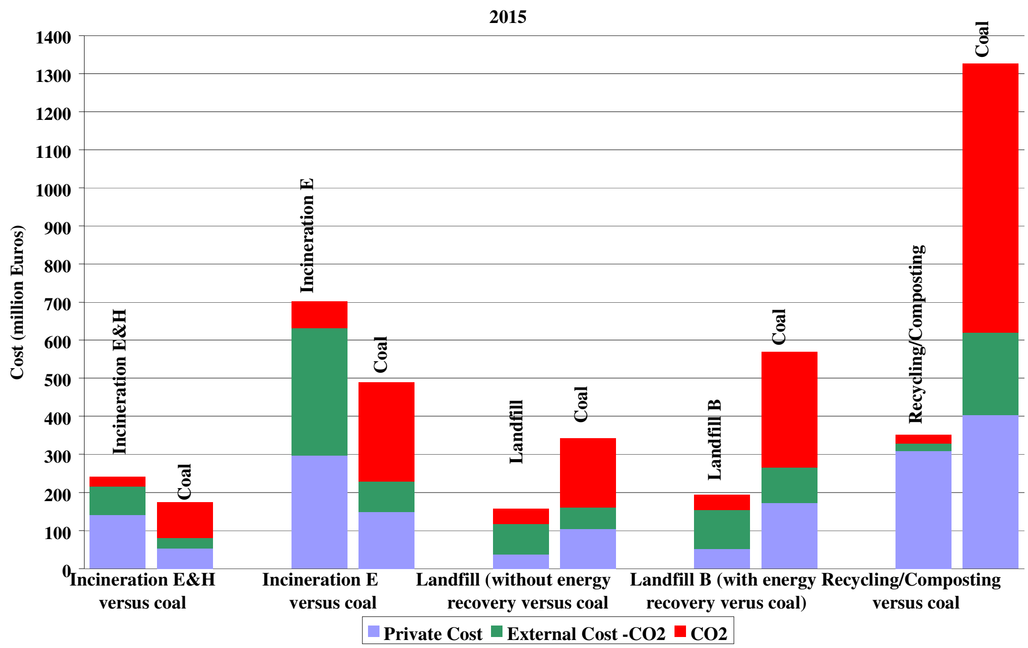

2020
Cost (million Euros)
1000
900
800
700
600
500
400
300
200
100
0
Incineration E&H
Coal
Incineration E
Coal
Landfill
Coal
Landfill B
Coal
Recycling/Composting
Coal
Incineration E&H versus coal
Incineration E versus coal
Landfill (without energy recovery versus coal
Landfill B (with energy recovery verus coal)
Recycling/Composting versus coal
Private Cost
External Cost -CO2
CO2

(b)

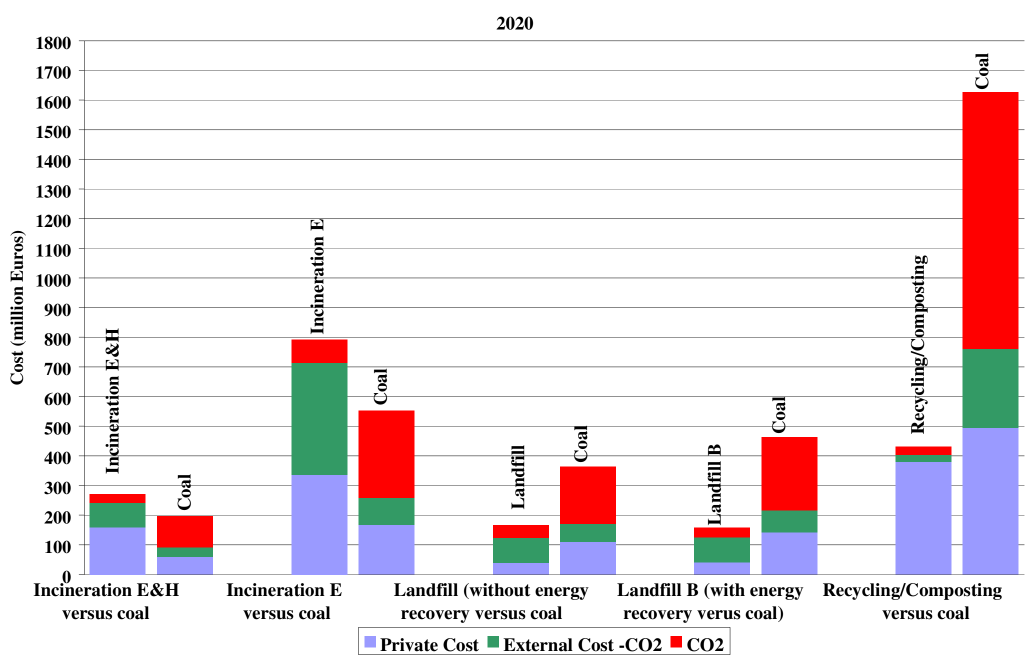

2030
Cost (million Euros)
1400
1300
1200
1100
1000
900
800
700
600
500
400
300
200
100
0
Incineration E&H
Coal
Incineration E
Coal
Landfill
Coal
Landfill B
Coal
Recycling/Composting
Coal
Incineration E&H versus coal
Incineration B versus coal
Landfill (without energy recovery versus coal
Landfill B (with energy recovery verus coal)
Recycling/Composting versus coal
Private Cost
External Cost -CO2
CO2

(c)

Figure 13: Comparative cost analysis under low carbon price



(a)



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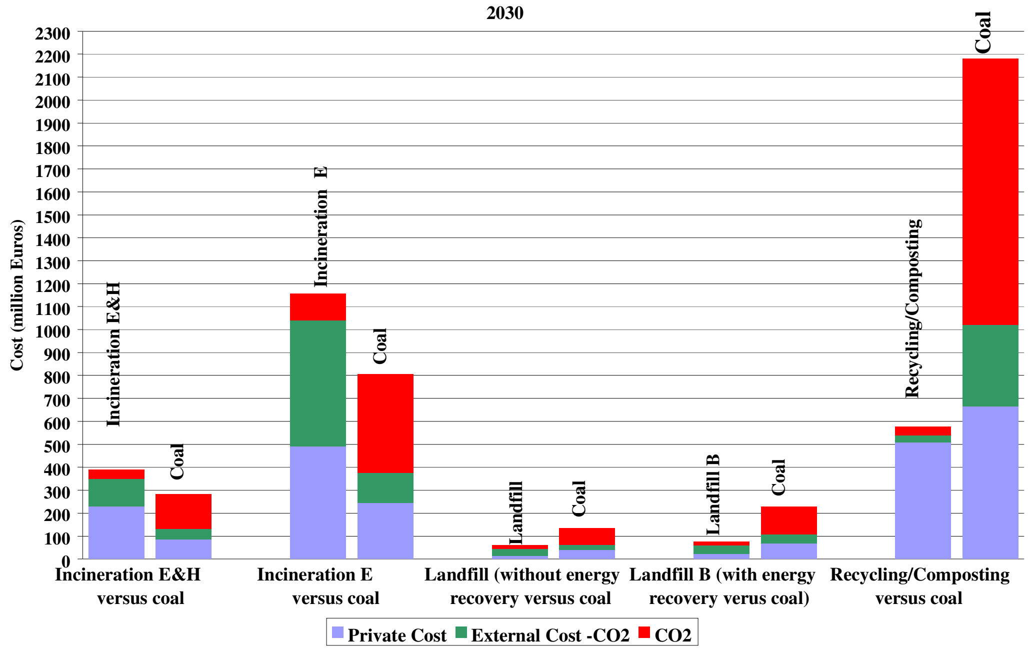

(c)

Figure 14: Comparative cost analysis under high carbon price

5.3 Summary of scenario analysis

The scenarios described in this section shed some light on the extent to which WtE as a low-carbon energy source can contribute towards achieving the UK's climate change and renewable energy targets. Tables 6-7 summarise the cost and energy implications of the waste management options under these scenarios. The estimated demand for electricity is 360, 380 and 381 TWh in 2015, 2020 2030 respectively (Dirks, 2007). The total demand for heat is expected to be 27, 28.5, and 31.5 TWh in 2015, 2020 and 2030 respectively.[6]

The BAU scenario shows that if the UK continues the current waste management path and allocates its MSW as per the year 2005/06 (base year), by 2020, WtE will account for 0.64 percent of total electricity demand and 6.5 percent of the total heat demand in the UK. This implies that, WtE will only

[6] The heat projections are based on heat/electricity ratio for 2004 and are assumed to remain the same for the subsequent years. The demand for electricity and heat in the UK for 2004 is estimated to be 340 TWh and 25.5 TWh respectively (IEA, 2004).

provide 3.2 percent of the total renewable electricity needed to meet the Government's 20 percent renewable electricity target by 2020. By 2030, these figures rise to 0.71 percent and 7.2 percent respectively. Under this scenario the social costs of WtE will be €1,934 million while the social cost of coal power equivalent will be €2,496 million for the year 2030.

The EU Directive scenarios represent a notable progress relative to the BAU scenario. In this scenario, by 2020, 1.43 percent of the total electricity demand and 16.4 percent of the total heat demand will come from WtE. Therefore, WtE will provide 7.2 percent of the total renewable electricity needed in order to meet the Government's 20 percent renewable electricity target by 2020. By 2030, the shares of electricity and heat will increase to 3.1 and 32.4 percent respectively. In other words, by 2030, the electricity from WtE will account for 15.5 percent of the Government's target for 20 percent renewable electricity. Moreover, under the high carbon price scenario, the relative social cost of coal power rises significantly thus increasing the desirability of WtE.

Tables 6 and 7 also show the coal power cost equivalent of the waste management options which allow a comparison of costs across scenarios that are based on different levels of energy generated from the waste. As shown by the results, by 2030, under the BAU scenario, the total social costs of the current waste management practice exceeds its coal power cost equivalent by 562 million euros in annualized terms. In contrast, the annualized total social costs of the EU Directive scenarios under low and high carbon price are 45 and 1,369 millions euros respectively lower than their coal power cost equivalents. The results suggest a marked improvement in cost effectiveness of the waste management option under the EU Directive scenarios. Moreover, the results show that the higher price of carbon has a significant positive effect on the cost effectiveness of the EU Directive waste management targets.

(a) Total costs vs. coal fired power

	Private costs (mill €)			CO_2 costs (mill. €)			External costs (mill. €)			Social costs (mill. €)		
	2015	2020	2030	2015	2020	2030	2015	2020	2030	2015	2020	2030
Waste management options	809	893	1,088	49	54	66	629	694	846	1,438	1,587	1,934
Coal power	964	1,064	1,298	362	399	487	884	976	1,198	1,84	2,04	2,49

										8	0	6

(b) Energy contribution of WtE

Year	Incineration E&H (GWh)		Incineration E (GWh)	Energy produced (GWh)	% of total UK demand	
	Electricity	Heat			Electricity	Heat
2015	567	1,700	1,667	3,934	0.62	6.3
2020	620	1,860	1,800	4,280	0.64	6.5
2030	760	2,280	2,213	5,253	0.71	7.2

Table 6: Scenario 1 - Business as usual – Total costs and energy supplies

(a) Low Carbon Price

	Private costs (mill €)			C0₂ costs (mill. €)			External costs (mill. €)			Social costs (mill. €)		
	2015	2020	2030	2015	2020	2030	2015	2020	2030	2015	2020	2030
Waste management options	837	957	1,263	42	45	48	637	679	800	1,474	1,636	2,063
Coal power	883	975	1,108	327	358	409	800	883	1,000	1,683	1,858	2,108

(b) High Carbon Price

	Private costs (mill €)			C0₂ costs (mill. €)			External costs (mill. €)			Social costs (mill. €)		
	2015	2020	2030	2015	2020	2030	2015	2020	2030	2015	2020	2030
Waste management options	837	957	1,263	199	212	226	808	863	1,000	1,645	1,820	2,263
Coal power	883	975	1,108	1,545	1,706	1,933	2,019	2,228	2,524	2,902	3,203	3,632

(c) Energy contribution of WtE

Year	Incineration E&H (GWh)		Incineration E (GWh)	Energy produced (GWh)	% of total UK demand	
	Electricity	Heat			Electricity	Heat
2015	1,380	4,140	2,340	7,860	1.0	15.3
2020	1,553	4,660	3,867	10,080	1.4	16.4
2030	3,400	10,200	9,520	23,120	3.1	32.4

Table 7: Scenarios 2 and 3 – EU Directive targets – Total costs and energy supplies

6. Policies for Improved Waste Management Options

Achieving the potential benefits of ambitious waste management and WtE options requires a new institutional and policy framework. Our review of the policy framework for WtE decisions in the UK has shown that a range of policies are currently originating from different levels of government.

Conflict of objective can be a major cause of policy failure. For example, the Landfill Directive intends to phase out landfill sites; while the Renewable Obligation encourages landfill and discourages WtE. The landfill Tax and LATS are aiming to internalise the externalities associated with landfilling waste. The weaknesses in the collection of landfill tax and the operation of LTCS have plagued the waste management system. Tax collection has failed on the grounds that not all sites have a weighbridge and non-weight calculations are open to abuse. LTCS is claimed to be ineffective due to lack of transparency and independence (Morris and Read, 2001). These issues have given rise to questions as to what extent the management of waste has improved after the Landfill Tax and to what extent the money being raised through LTCS is used to promote better waste management (Morris et al., 1998).

The government can improve the waste strategy by managing the municipal, commercial and industrial waste together to minimise the number of policies, improve efficiency, and reduce the transaction costs. Also, increased transparency and autonomy would reduce potential conflict of interest. The PPC licence and planning permission process work towards internalising the local costs of WtE, in particular the health and disamenity effects. While the PPC licence sets stringent emissions for WtE plants, there is strong opposition to new plants due to health concerns having an impact on planning permission successes (DEFRA, 2005b). Given that the PPC has, at a minimum, internalised the negative health externality, the externalities associated with blight remain unaddressed.[7]

In order to reduce the influence of local campaigners, policy should be issued from higher levels of government. For example, the Department of Communities and Local Government (DCLG) could use the PPS10 to remove health concerns as a criterion for rejecting planning permission while the role played by the community led approaches such as community volunteerism should not be undermined.

A barrier for renewable is the New Electricity Trading Arrangements (NETA) (Connor, 2003). NETA is a mechanism to balance the electricity supply market in the UK but it has been criticised as unfavourable to generators with less

[7] Landfills can blight an area and causes a fall in house prices and personal wealth (BMBC, 2006).

predictable outputs. The failure to take into account the advantages of the distributed generating technology is a barrier to renewable energy in UK. As a result, some economic advantages of these technologies have been ignored which makes it less cost effective, less desirable and thus less likely to be competitive (Connor, 2003).

A final shortcoming of WtE policy is the absence of a mechanism for internalising the external benefits from WtE in terms of net reductions in GHGs and increased security of energy supply. WtE is currently excluded from eligibility for ROCs on the grounds that it is a commercially viable technology despite having large positive externalities. The eligibility of pyrolysis and gasification and ineligibility of WtE offers comparative advantages to newer thermal treatments.

The proliferation of a technology depends significantly on public acceptance. Public perception of WtE differs from country to country. Denmark has one hundred years of experience with WtE and the public is familiar with the technology. The national energy policy, flow control, fiscal and legislative measures as well as a ban on the landfill of combustible waste have promoted WtE in Denmark meeting the EU Directive (Dalager, 2007). Public involvement in the waste planning process could therefore mitigate local opposition and foster balanced opinions on WtE. The government is currently planning to encourage the WPAs to produce Statement of Community Involvement (SCI) documents specifying how stakeholders will be consulted, and how their views can feed into the WDD process (LDF, 2005).

7. Conclusions

This paper highlights the potential of WtE as an effective waste management option and energy source. WtE can minimize the amount of waste sent to landfill and by virtue of its biomass content can contribute to achieving the UK's renewable energy and electricity targets. It can also improve security of supply by reducing dependence on imported fuels.

WtE is a favorable alternative where the private cost of landfill is high, for example, due to the price of land in densely populated areas. Likewise, high externality costs and diversion of waste from landfill as per the EU Directive make landfill an unattractive waste treatment option. The private costs of WtE decline with increasing size of-the-art facilities that take advantage of economies of scale. The externality costs of producing energy from fossil fuels, such as coal, are high. The benefits of WtE compared to coal power will improve if other externalities are also taken into account. For example, our cost-benefit analysis does not include the positive externalities of WtE from increased security of supply or the negative externalities associated with disamenity of coal fired plants.

The results indicate that WtE is a socially cost-effective waste management option and meeting the EU Directive targets will increase these benefits. Moreover, the cost effectiveness of WtE improves substantially with higher carbon costs. In the future, the cost of landfilling the waste is, due to land scarcity and disamenity, likely to increase further thus making energy recovery from waste more cost effective. By approaching the waste treatment levels in best practice countries, the electricity and heat from WtE can be an important part of waste management strategy as well as energy and environmental policies.

Achieving the full potential of WtE requires developing the delivery networks that will need to be developed to allow the use of electricity and heat from combined heat and power including those from WtE. Moreover, institutional improvements such as removing regulatory barriers in planning permission and policy improvements such as managing municipal, commercial and industrial waste collectively will provide a more favourable framework for the promotion of progressive waste management and WtE options.

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