THE ECONOMICS AND POLICY OF MUNICIPAL SOLID WASTE MANAGEMENT

submitted by

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To my parents,
Anna and Panos Karousakis,
for all their support.

Infinite growth in material consumption in a finite world is an impossibility.

E. F. Schumacher

ABSTRACT

This thesis contributes to the environmental economics and policy of sustainable municipal solid waste management. Significant market and government failures are present that lead to inefficiently high levels of waste generation and distort the optimal allocation of waste to the various disposal options available. The aims of the thesis are to identify and analyse the socio-economic, policy, spatial, as well as attitudinal determinants of municipal solid waste generation, disposal and recycling, at the international macro-economic level and at the household level. The former is conducted using cross-sectional time-series data from the 30 member countries of the Organisation for Economic Co-operation and Development (OECD) over the period 1980 to 2000, whereas the latter is undertaken using original survey data collected from 188 households in London, UK. Three distinct methods have been adopted to undertake this investigation namely panel data econometrics, spatial econometrics techniques, and the stated preference choice experiment method. Conforming with previous studies, the results from the panel data econometrics indicate that waste generation is income inelastic. However, higher income levels are associated with smaller proportions of municipal solid waste disposed of at landfills and greater proportions of paper/cardboard and glass recycling. The role of urbanisation, population density and waste management policies are also examined. Moreover, spatial interaction is present in waste management and policy-making suggesting that governments may be acting strategically in their decision-making processes. Finally, the results from the choice experiment indicate that households are willing to pay for the number of 'dry' materials collected, and the collection of compost, while textile collection and the frequency of kerbside collection is less important. These insights into municipal solid waste management can assist policy-makers in designing and implementing efficient and cost-effective policies in developed countries, helping to promote sustainable municipal solid waste management.

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TABLE OF CONTENTS

List of Figures

List of Tables

CHAPTER 1

Introduction to the Thesis

1.1 An Introduction to Municipal Solid Waste Management

This dissertation analyses the concept of municipal solid waste management policy and its implementation. The concept of sustainable or integrated waste management aimed at providing the correct incentives for waste disposal has been gaining increased attention in both the US and Europe over the past two decades. Heightened recognition of the issues related to waste has developed as a result of often monotonically increasing waste levels, land scarcity for landfill developments in certain regions, increasing public opposition associated with the 'not-in-my-backyard' (NIMBY) phenomenon in relation to landfill and incinerator siting, as well as global externalities contributing to climate change from landfill emissions.

Waste, as defined by the Basel Convention, is "substances or objects which are disposed of or are intended to be disposed of or are required to be disposed of by the provisions of national law". More specifically, "waste is generated when a product or material begins to be treated as waste, and managed as such. Thus, waste generation includes material that is generated, collected and then recycled, composted, burned with or without energy recovery, or landfilled" (OECD, 2004). Though no single definition of sustainable waste management exists, the concept refers to the efficient use of material resources to reduce the amount of waste produced and, where waste is generated, to managing it in a way that actively contributes to the economic, social and environmental goals of sustainable development[1].

From this perspective, waste generation levels are often excessively high and the allocation of waste to the various disposal options inefficient. This is due to a lack of appropriate pricing signals, as the full social costs of landfilling, incineration, and recycling are not adequately reflected in the market. The underpricing of landfills, for example, makes the waste stream larger than it otherwise would be, since recycling and conservation are rejected in favour of artificially cheap landfilling. Furthermore

[1] The most widely used definition of sustainable development is "development that meets the need of the present without compromising the ability of future generations to meet their own needs" (Brundtland Report, 1987).

the underpricing of landfills represents a subsidy[2] to the landfilling business bestowing the landfill business with an unfair advantage, a competitive edge in the marketplace, which has the effect of discouraging private initiative in the development of disposal alternatives. Related to this are of course the environmental externalities associated with waste disposal.

In contrast to some other environmental problems that initially increase with economic growth and then eventually decrease after reaching a turning point, waste generation levels do not seem to be on the decline (Shafik et al., 1992; Cole et al., 1997). Only a very few number of countries have recently managed to decouple waste generation levels and economic growth[3] and as developing countries begin to follow suit, appropriate waste management practices will become a more significant issue for these nations too.

The issue of waste management is an important one, representing a potentially large source of misallocation of resources, and the environmental externalities associated with waste disposal are both significant and long-term. In the OECD countries[4], over 35% of known public and private sector environmental-related expenditures are directly linked to waste (OECD, 2000). Current expenditure on waste management in the European Union (EU) amounts to approximately 48 billion Euro per year (around 14 per cent of which is related to packaging), which constitutes 0.6-0.7 per cent of GDP and 40 per cent of total environmental expenditure (Linher, 2005). It is evident that concerted policy action will be required to mitigate and reverse the trends in waste generation, and to dispose of the waste stream in the most efficient way. Indeed, the current EU Sixth Environment Action Program identifies waste

[2] i.e., paid for by the government or general public.

[3] Though economic growth and waste generation are closely correlated, a small number of countries have recently managed to decouple economic growth and municipal waste generation (e.g., Denmark, the Netherlands, and Switzerland) (EEA, 2003).

[4] The OECD countries are the 30 member countries of the Organisation for Economic Co-operation and Development.

prevention and management as one of its four top priorities, and in the UK waste policy is arguably the second largest environmental challenge after climate change[5].

The purpose of this chapter is to provide a synopsis of the current state of waste management trends and the waste legislation that has evolved to address the above mentioned issues. This is intended to provide a contextual background leading to the final section in which the aims and overview of this thesis are presented.

The remainder of this chapter is organised as follows: Section 1.2 describes existing trends in the generation and composition of waste in OECD countries. The disposal options and financial costs of waste are addressed in section 1.3 and section 1.4 focuses explicitly on the environmental costs of the three primary methods of waste disposal, namely landfill, incineration, and recycling. Developments in waste legislation related to waste management are discussed in section 1.5 at both the EU level and in the US, along with other notable examples from developed countries. Finally, section 1.6 presents the aims and objectives of this thesis.

1.2 Trends and Composition of Waste

The scale and significance of the waste problem can best be illustrated by considering the development of waste arisings over time. Reported total waste generation in OECD Europe grew by nearly 10 percent between 1990 and 1995 (EEA, 1998). Most recently available data estimate that more than 3,000 million tonnes of waste is generated in Europe every year (EEA, 2003)[6]. The main waste-producing sectors are the manufacturing industry, construction and demolition, mining and quarrying, and municipal waste (see Figure 1.1).

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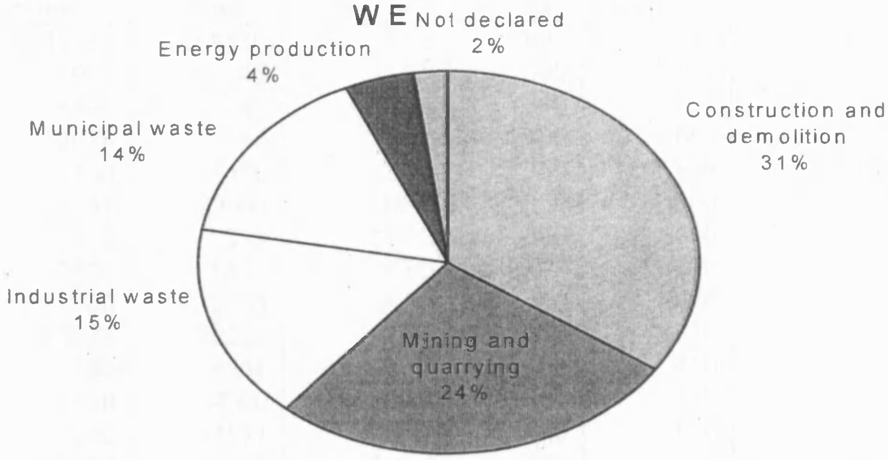
[5] *Waste Not, Want Not – A strategy for tackling the waste problem in England*, Strategy Unit, November 2002.

[6] Comparative figures for the US are not available as the US Environmental Protection Agency (EPA) does not require states to report the total amount of waste generated. Based on information provided by Pennsylvania's Department of Environmental Protection, it is estimated that municipal waste accounts for perhaps less than 20% of the total waste stream.

Figure 1.1 Total waste generation by sector in WE



Source: EEA, 2003.
NB: Figure does not include Belgium, Iceland, Luxembourg, Sweden, Spain, and Switzerland.

Waste from manufacturing industries consists of food, wood, paper, chemicals, non-metallic minerals, basic metal and other waste. This sector accounts for about 740 million tonnes of waste per year and has been on the rise since the mid-1990s in most European countries for which data is available. Waste from construction and demolition, which includes the renovation of old buildings, has generally also been increasing in Western Europe (WE).

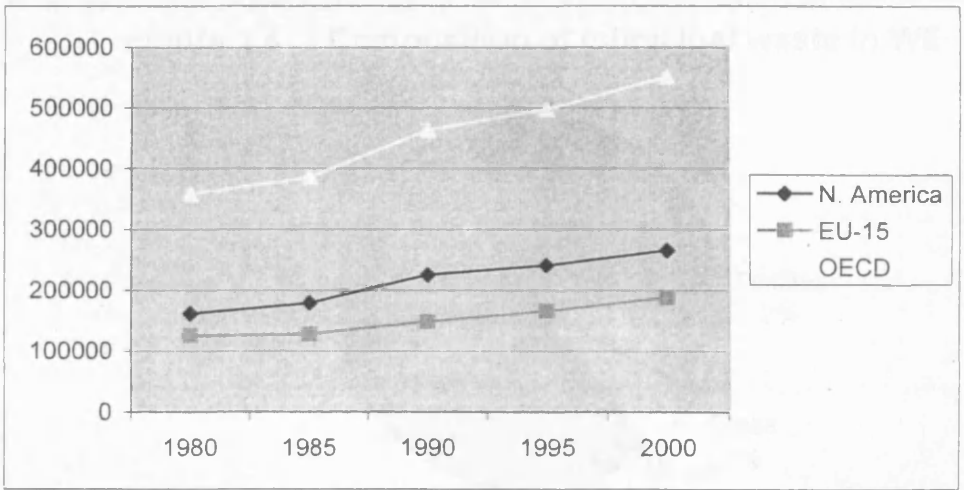
Municipal waste is estimated to account for 14 percent of total waste arisings in Western Europe and 5 percent in Central and Eastern Europe (EEA, 2003). Table 1.1 shows the best available data on municipal waste generation for a number of OECD countries in 1980 and 2000. Between 1980-2000, total municipal waste arisings in the OECD increased by 54 percent. Albeit not as rapidly, municipal waste generation per capita has also increased significantly over the same time period. These trends are depicted in Figures 1.2 and 1.3 below.

Table 1.1 Generation of Municipal Waste in the OECD Countries

	1980 thousand tonnes	**2000** thousand tonnes	**Percent change** 1980-2000	**1980** kg per capita	**2000** kg per capita	**Percent change** 1980-2000
USA	137560	208520	51.6	600	760	26.7
Japan	43995	51446	16.9	380	410	7.9
Belgium	3499	5588	59.7	360	550	52.8
Denmark	2046	3546	73.3	400	660	65.0
Greece	2500	4550	82.0	260	430	65.4
Italy	14041	29000	106.5	250	500	100.0
Luxembourg	128	278	117.2	350	640	82.9
Netherlands	7050	9691	37.5	490	610	24.5
Norway	1700	2755	62.1	550	620	12.7
Poland	10055	12226	21.6	280	320	14.3
Portugal	1980	4531	128.8	200	450	125.0
Sweden	2510	4000	59.4	300	450	50.0
Switzerland	2790	4681	67.8	440	650	47.7
Turkey	12000	24945	107.9	270	390	44.4
EU-15	125000	188000	50.4	370	520	40.5
OECD	358000	551000	53.9	420	560	33.3

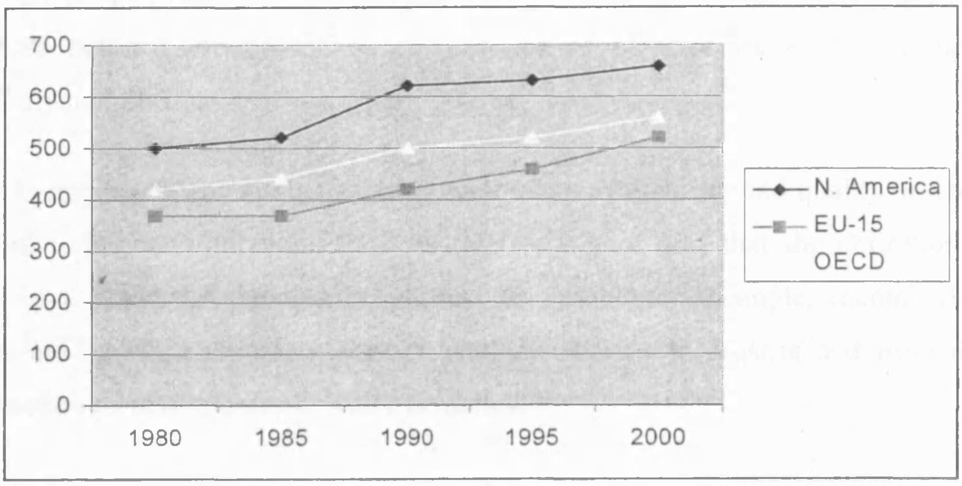
Source: OECD Environmental Data Compendium 2002

Figure 1.2. Total MSW Generation Rates (thousand tonnes)



Source: OECD Environmental Data Compendium 2002

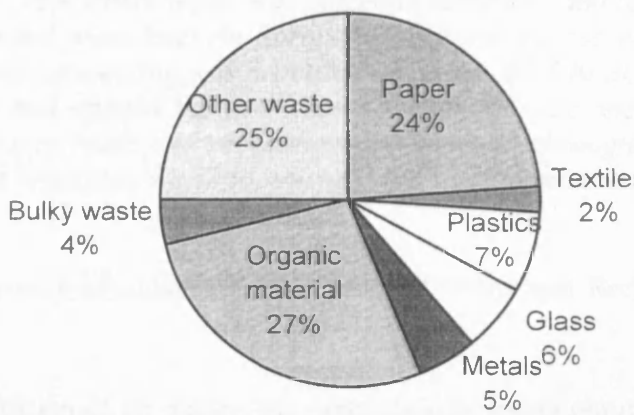
Figure 1.3 Average Per Capita MSW Generation Rates (kg)



Source: OECD Environmental Data Compendium 2002

This amounts to more than 550 million tonnes that are collected each year, or on average, 560 kg/capita. This ranges substantially from 310 kg/capita in Mexico to 760 kg/capita in the US (OECD, 2002). Regarding its composition, municipal waste

Figure 1.4 Composition of municipal waste in WE



Source: Eurostat 2003

consists mainly of organic materials, paper, and other waste. This is followed by plastics and glass, and finally metals, bulky waste and textiles[7] (see Figure 1.4). In most countries, municipal waste consists of 60% or more of household waste (Eurostat, 2003).

It is important to note that existing waste data availability and quality, in comparison with other environmental data, is relatively poor and that the definitions are not always consistent between countries. In France for example, municipal waste is defined as also including sewage sludge, whereas in Austria and Ireland a larger fraction of non-household waste is included.

Despite these data discrepancies, the discernable trend is that waste generation levels are increasing. Moreover, this waste challenge is not limited to OECD countries. Though reliable statistics are hard to come by, the UN Commission for Sustainable Development forecasts that "the amount of waste produced in developing countries will double within just ten years, and that *global* waste generation may increase five-fold by 2025." (OECD, 2000).

1.3 Disposal Options, Costs and Trends

> *The existence of landfill sites dates back to as far as 3000BC in Knossos, the capital of Crete where waste was placed in large pits and layered with earth. One thousand years later, in Europe, bronze was recovered from waste and reused, and composting was practised in China. By 500 BC, the government in Athens had opened the first municipal landfill site one mile outside the city. Forms of reuse and recycling were common throughout the world, as people fed vegetable waste to animals and used manure and green waste as fertiliser.*
>
> *Source:* www.wasteonline.org.uk History of Waste and Recycling Information Sheet.

As with the creation of all matter, the generation of waste requires that it be disposed of in one form or another. In general, waste can either be deposited at landfills,

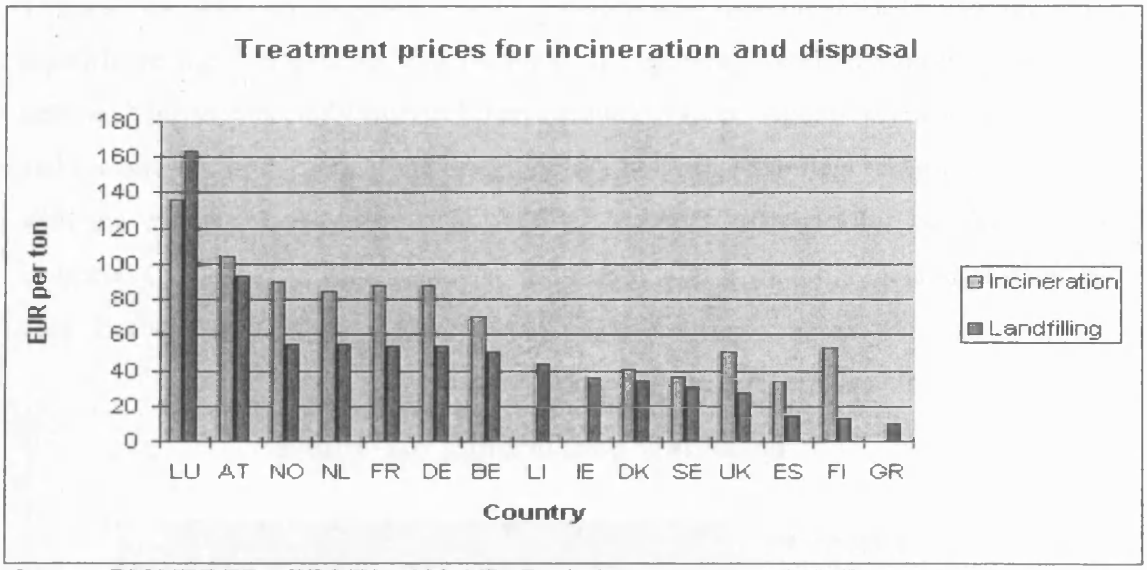
[7] In the US the figures are: Paper: 35.2%; yard trimmings: 12.1%; food scrap: 11.7%; plastic:11.3%; metal: 8.0%; rubber, leather, textiles: 7.4%; glass: 5.3%; wood: 5.8%; other:3.4% (EPA, 2003).

incinerated, or recycled. Landfilling entails the deposition of waste onto or into land (*i.e.*, underground) which has been specified as a waste disposal site. Incineration refers to the thermal treatment of waste with or without recovery of the combustion heat generated. Finally, recycling is a resource recovery method involving the collection and treatment of a waste product for use as a raw material in the manufacture of the same or similar product. Alternatively, the waste can be composted which refers to the controlled decomposition of organic matter such as food and yard wastes. Emerging waste disposal technologies also include hydrolysis and pyrolysis, gasification, and thermolysis.

To date, the disposal methods adopted have been driven primarily by the availability of land space, public opposition to air pollution from incinerators in certain areas (e.g., in California and the UK), and the costs of disposal. For example, in the north-eastern portion of the US, where population densities and land values are high, approximately 40% of generated waste is incinerated. In Northern Europe (e.g., Sweden, Denmark, and Switzerland) a larger fraction of waste is incinerated and recycled, partly a result of land scarcity and related policy. In contrast, other countries such as the UK, Ireland, and Greece rely almost exclusively on landfills.

Costs have been a strong motivating factor for the UK in the popularity of landfills amongst waste management companies, as they are the least expensive option (DTI, 1997). This is partly due to geological reasons but also because of advances in the construction and maintenance of landfill sites. Indeed, in nearly all EEA countries, average treatment prices for landfill use are lower than for incineration (see Figure 1.5).

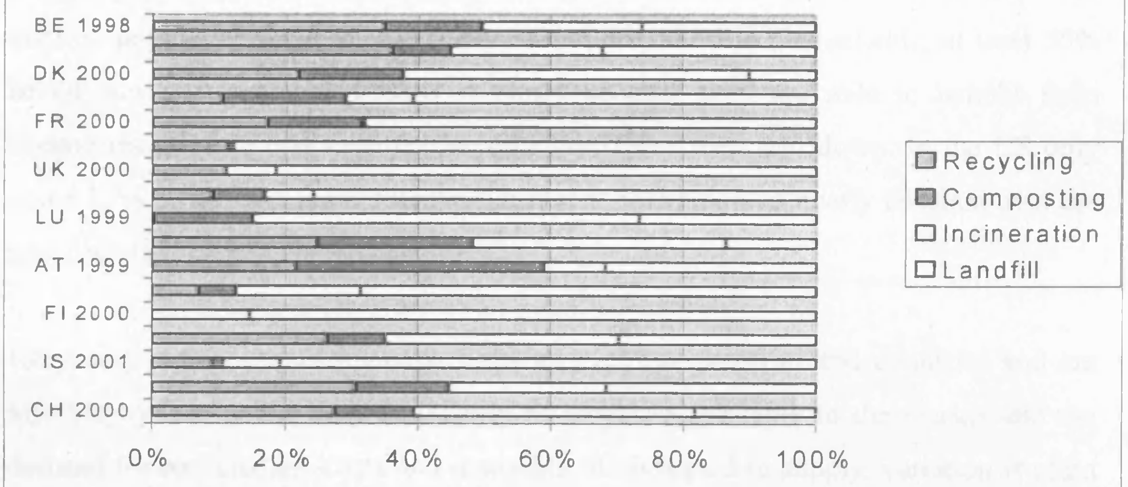
Figure 1.5



Source: EIONET EEA ETC/W, 1998. NB: Excludes waste tax and VAT.

It is worth noting that the generation of revenues from energy production, known as waste-to-energy incineration, can also partially offset the cost of incineration, although there are typically less expensive forms of energy production available. The relative proportion of MSW treated and disposed of in the various alternatives is depicted in Figure 1.6.

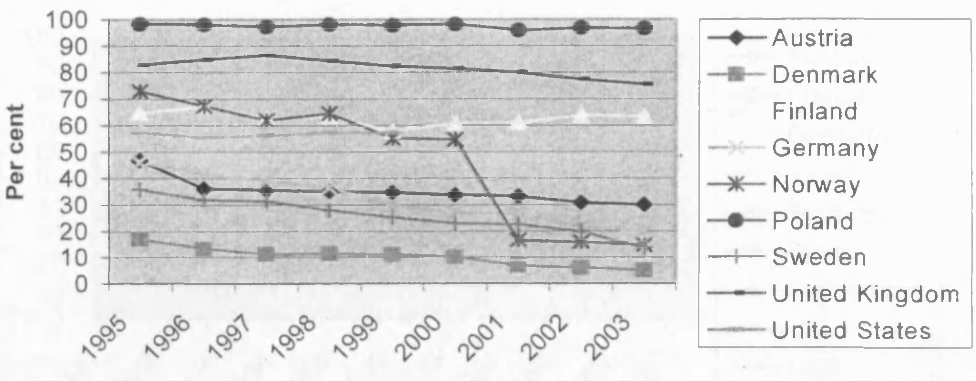
Figure 1.6 Treatment and disposal of municipal waste in WE



Source: Eurostat 2003

Though there has been an increase in the proportion of municipal waste disposed of in landfills in the Netherlands and Portugal, the general trend in Europe is one of decline. This is especially true in Spain (about 20% per annum) whereas in Sweden and Iceland the trend is relatively negligible. Looking at the new EU member states, with the exception of Malta, over 90% of MSW is managed by landfill, and in Bulgaria, Cyprus, Lithuania, Romania and Slovakia, it is the only method of disposal used (Eurostat, 2003).

Figure 1.7 Landfill Disposal Rates

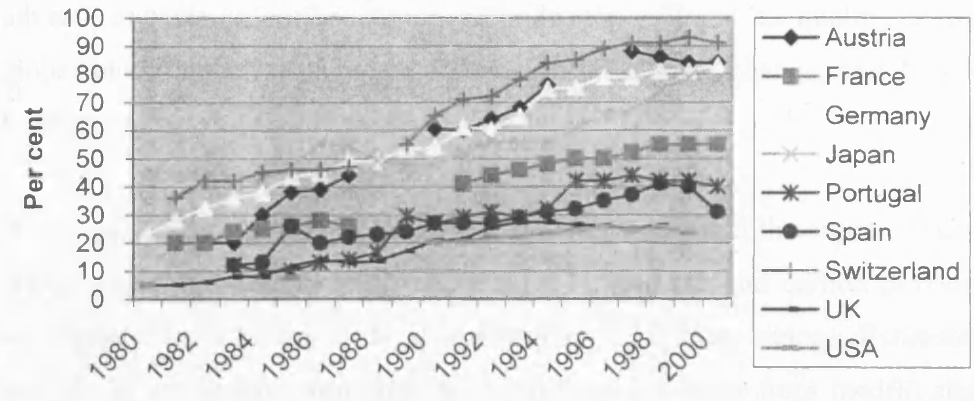


Source: Eurostat 2003

The proportion of waste incinerated also varies considerably between countries. In densely populated countries like Japan, Denmark, and the Netherlands, at least 50% of all waste is incinerated. This is partly because they are able to benefit from economies of scale that keep the average cost of incineration down. In the US only about 11% of waste is incinerated, a figure that has remained nearly constant over the past decade (Fullerton and Raub, 2004).

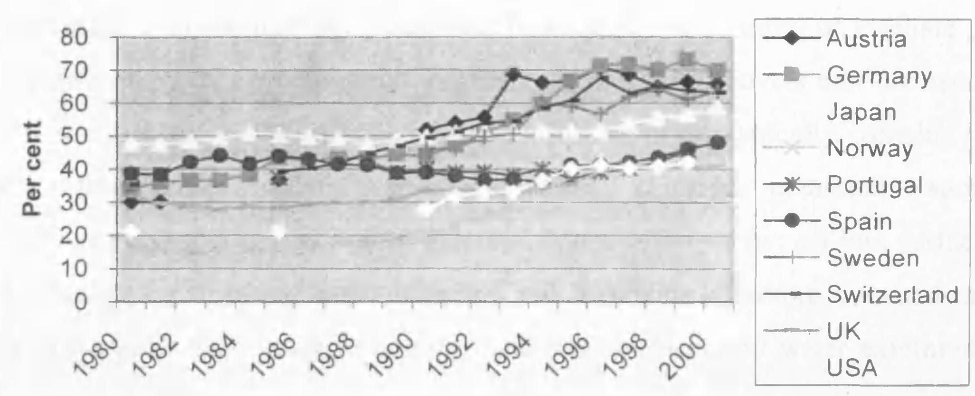
Recycling rates vary substantially between types of material and countries and are affected by barriers to both the supply of recyclate available to the market and the demand for recycled products and materials. With regard to supply, variation is often due to limited collection infrastructure, contamination of supply, low cost of waste disposal, and the high cost of collecting and recycling waste. On the demand side there is competition from low priced virgin materials, limited application to recyclate, and in some sectors, price volatility for recyclate (Environmental Council, 2002). Overall, recycling rates in OECD countries for metals exceed 80 percent, 35-40 % for glass, and 40-55% for paper and cardboard. In Ireland for example, paper recycling is only 10%, whereas in Germany it is 70%. More generally, recycling rates are highest in the Scandinavian countries, and lowest in the OECD Mediterranean (Greece, Portugal, Spain, and Turkey) (de Tilly, 2004).

Figure 1.8 Glass Waste Recycling Rates



Source: OECD Environmental Data Compendium 2002

Figure 1.9 Paper/ Cardboard Waste Recycling Rates



Source: OECD Environmental Data Compendium 2002

The choice of waste disposal and treatment methods adopted has important environmental implications that often are not factored into the market price, thus leading to external effects and social inefficiencies. These issues are reviewed in the next section.

1.4 Environmental Costs of MSW Disposal

In addition to the financial costs associated with solid waste disposal, there are also external environmental costs that must be taken into consideration. These vary according to the type of waste and disposal methods adopted and are a result of adverse impacts on surface water, groundwater, soil and air quality, as well as the global environment with regard to impacts on climate change, and finally, risk to human health.

When waste is deposited at a landfill, the biodegradation of the organic fraction of the waste results in the generation and release of methane and carbon dioxide into the atmosphere, contributing to the global problem of climate change. Estimates suggest that 6% of all methane emissions to the atmosphere occur from landfill sites (Beede and Bloom, 1995)[8]. Trace gases are also present and over 100 types of volatile organic compounds have been identified such as benzene and vinyl chloride. In addition to air emissions, the breakdown of organic matter combined with moisture can result in the formation of leachate. In general, the quantity of leachate generated depends on the net precipitation and the type of landfill covers that are used. During the initial phase in the lifetime of the landfill, leachate typically contains very high concentrations of organic carbon, ammonia, chloride, potassium, sodium and hydrogen carbonate. This in turn can leak into aquifers or run off into surface waters, contaminating drinking water supplies and adversely affecting human health. Few attempts have been made to quantify and evaluate soil and water externalities from

[8] Doorn and Barlaz (1995) estimate a range of 3-19% of global anthropogenic source of methane emissions.

landfills as these tend to be highly site specific and vary depending on the quality of the soil, the location of the landfill and its proximity to groundwater reservoirs and receiving waters. There is also a time dimension in evaluating these effects; eventually all landfills will start to leak and can continue to do so for hundreds of years[9].

There are also disamenity effects associated with landfills such as visual impacts, noise, smell and litter, as well as those caused by the transportation of waste to the sites. Finally, there are risks to human health. Eschenroeder and Stackelback (1999) compare the health risks of landfills and incinerators and conclude that the cancer-causing risk factor associated with landfills is approximately 100 times higher than that for waste incineration. Bridges et al. (2000) find similar results with respect to airborne pollutants. In contrast, Elliot et al. (2001) find no increase in the rates of cancer in populations living close to landfill sites. They do however find a one percent increase in the rate of congenital anomalies in populations living within 2 km of a landfill site, and a seven percent increase for those living within a 2km radius of a landfill site containing hazardous waste.

In addition to the financial and external environmental costs associated with landfill, there is also the opportunity cost of land and the user cost which should also be incorporated to adequately reflect the full social cost of landfill disposal.

In the case of incineration, though the weight and volume of waste is significantly reduced (up to 75 and 90 per cent respectively), the process of burning results in emissions of carbon dioxide, metals such as mercury, lead and cadmium, acidic pollutants (sulphur dioxide and hydrogen chloride), and particulate matter, among others, resulting in adverse effects to human health and the natural environment. Incineration also involves a flue gas cleaning process which may contaminate wastewater. Residual products (bottom ash, fly ash, and air pollution control residues) are disposed of at landfills (COWI, 2000). There are also external costs associated

[9] The issue of discount rates in evaluating these costs is therefore a relevant one (COWI, 2000).

with disamenity effects. Table 1.2 provides some estimates of total landfill disposal and incineration costs.

Table 1.2 Private cost, external cost and energy gain per ton of waste from landfilling and incineration in five OECD countries (1997 US dollars per ton).

	Private cost	**External cost**	**Energy gain**	**Total cost**
Germany				
Landfill	51	3-15	not estimated	53-66
Incineration	104-192	5-14	58-106	52-100
Sweden				
Landfill	16-24	3-15	not estimated	19-39
Incineration	57-65	7-15	35-42	29-37
United Kingdom				
Landfill	8-51	3-15	not estimated	11-66
Incineration	84-96	24-33	63-77	46-62
USA				
Landfill	15-57	3-15	not estimated	18-72
Incineration	69-137	11-20	49-66	31-91
Netherlands				
Landfill	49	36	13	74
Incineration	155	56	57	153

Source: Porter, 2002

The estimates cited above are from Dijkgraaf and Vollebergh (1997) for the Netherlands and from Miranda and Hale (1997) for the remaining four countries. In the latter study, the landfill external costs are composed of air emissions and leachate. Given that landfill emissions were not available for each of the countries, the emissions estimate is based on data available from only one of the countries, the US. The resulting cost from leachate is relatively low and thus the overall result should not be significantly affected by this assumption. The air emissions estimate is based on the assumption that 84% of the landfills do not practice methane energy recovery and 16% do. Looking at the UK in particular, incineration presents much larger external costs than landfills and is due to the less stringent air emissions standards for sulphur dioxide and nitrous oxide emissions compared with the other countries.

It is therefore important to bear in mind that though the results provide an indication of the magnitude of the external costs of landfill (and incineration), that these may not accurately represent the true external costs for each of the countries. Moreover, the results may be somewhat outdated given that new landfills may be required to practice methane energy recovery.

In general, the external costs associated with landfills are likely to vary based on the landfill characteristics and the characteristics in which the landfill is sited. Factors such as facility size, type, design, operational parameters as well as the host community characteristics such as the adjacent land uses, local hydrology, population density, local infrastructure, among others will play a role in the site-specific landfill under consideration.

For incinerators, the external costs represent a small fraction of the production cost. Landfilling is clearly a less expensive option than incineration when only the production cost is considered. Miranda and Hale (1997) caveat that there are uncertainties about the value of external impacts, especially about the impact of methane on global climate as well as the human health impact from air toxics from incineration that could affect the total social costs of each option.

More recently, Davies and Doble (2004) use data from the United Kingdom to obtain an external marginal cost of landfill disposal of £5 per metric ton. The disamenity values i.e., the nuisance value of landfill sites from noise, odour, visual intrusion etc, are estimated at approximately £2/tonne of waste. These estimates are derived from values in the US where most of the recent estimates were available. The estimates of non-disamenity value are approximately £3/tonne of waste (from CSERGE et al. 1993). The robustness of the disamenity estimates were later confirmed in 1999 when the UK Government commissioned a study to estimate the disamenity costs of landfill in Great Britain. These were between £1.52 to £2.18 per tonne of landfill (DEFRA, 2003).

The final option for waste management is recycling and/or composting, regarded as the most environmentally benign. The main external costs are transport related. Furthermore, there may be substantial pollution and energy-use externalities associated with recycling. In particular, reprocessing of glass, paper and metals can involve significant energy requirements, although these are lower than those involved in processing virgin materials and in the case of aluminium are substantially lower (OECD, 2005). This method is hampered by some limitations due to the feasibility of recycling certain materials and the prohibitive costs associated with the separation, transportation and reprocessing of the waste.

Today, the concept and adoption of 'integrated waste management' has become synonymous with the acceptance of a waste hierarchy in which disposal options are ranked according to their environmental costs as follows:

1. Source reduction
2. Re-use
3. Recycling
4. Incineration (and energy recovery)
5. Landfill

In the US, though the Environmental Protection Agency (EPA) also follows a hierarchical approach in its waste policy, the EPA explicitly mentions indifference between the final waste disposal methods of landfill and incineration. Thus, the results on the external costs of landfill versus incineration summarized in Porter (2002) above tend to be in contradiction with the waste hierarchy in the case of the UK and the Netherlands. For Sweden and Germany, this is less clear.

Notwithstanding some controversy regarding the waste hierarchy with respect to the ranking of incineration and landfills (see also Brisson, 1997), the past two decades have witnessed an expansion of waste management policies implemented in OECD countries with the aim of diverting waste streams higher up on the hierarchy. The next section describes the main developments in waste legislation and policy with a focus on OECD countries.

1.5 Developments in Waste Legislation and Policy

The earliest example of a regulation pertaining to waste management seems to be the 1848 Public Health Act in the UK. The Act made provision for waste to be stored in heaps next to properties, called midden heaps: "*They were not actually heaps but large holes where rubbish and sewage was left until full, then dug out and taken away by horse and cart for disposal. ...The major change in waste collection came soon after in the form of the 1875 Public Health Act, a result of a cholera outbreak in London, which claimed many lives. The main thrust of the 1875 Act was to charge Local Authorities with the responsibility to remove and dispose of waste. Scavenging was replaced by a regular collection of waste from each household*".[10]

Most of today's relevant waste legislation and policy was established primarily after World War II when the growth of the industrial base and changing lifestyles resulted in a major increase in air, water and solid waste emissions. As the only international convention addressing waste issues is the 1989 Basel Convention, which deals explicitly with the import and export of hazardous wastes, the subsections below discuss the important waste legislation and policies that have emerged at the regional and local context in Europe and the US, with reference to other countries in the OECD[11].

1.5.1 The European Context

At the European Union level, a number of initiatives have been undertaken to more explicitly address the issues of waste disposal and minimisation. These were first formalised under the 1975 Framework Directive on Waste and the subsequent 1978 (amended 1991) Framework Directive on Hazardous Waste. Since then, several other

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[10] Source: http://www.integra.org.uk/. Integrated Waste Management Initiative

[11] Sands (2003) argues that given the massive increase in the generation of all types of waste resulting from industrialisation, the lack of a well-developed area of international law for waste represents a major shortcoming. At the global level, no UN or other body has overall responsibility for waste, which has led to a fragmented, *ad hoc* and piecemeal international response (p. 675).

Directives have been established setting up more specific guidelines and requirements for waste management.

The 1975 *Framework Directive on Waste* (75/442/EEC)[12] sets out general principles, procedures and requirements for legislation regarding waste management and resource use. The starting point for the drawing up of the Directive was the introduction of national waste regulations in the Member States. Prior to the mid-1970s, most Member States regarded waste as a local or regional matter. The different national provisions on waste in place or in preparation at that time were seen as creating unequal conditions of competition that would affect the functioning of the common market. It is also stated that '*the essential objective of all provisions relating to waste disposal must be the protection of human health and the environment against harmful effects caused by the collection, transport, treatment, storage and tipping of waste*'.

As amended in 1991, the Framework Directive incorporates key elements of Community waste management strategy, including the waste hierarchy and what have become known as the principles of proximity and self-sufficiency. These require the disposal of waste in the closest suitable facilities and that waste produced in the Community should not be disposed of elsewhere. It further obliges Member States to establish waste management plans and a procedure for licensing companies involved in waste disposal or recovery (House of Lords, 1998).

To explicitly address the regulation and control of toxic and dangerous waste, another Framework Directive was established in 1978 on *Hazardous Waste* (91/689/EEC).[13] Herein, Member States are asked to encourage the reduction of waste arisings, re-use and recycling activities, and to authorise installations handling toxic and hazardous waste. The annex of the Directive lists 27 different groups of hazardous or toxic

[12] Official Journal L 194 , 25/07/1975 P. 0039 – 0041.
[13] Official Journal L 377 , 31/12/1991 P. 0020 – 0027, amending Council Directive 78/319/EEC.

waste for which special authorisation, control and surveillance procedures are introduced.

The 1994 *Packaging and Packaging Waste Directive* (94/62/EEC)[14] aims to harmonise national measures in order to prevent or reduce the impact of packaging and packaging waste on the environment and to ensure the functioning of the Internal Market. It contains provisions on the prevention of packaging waste, on the re-use of packaging and on the recovery and recycling of packaging waste, and calls for an information and monitoring system of waste packaging. Furthermore, it establishes some quantitative limits, including a requirement that Member States should ensure the recovery of 50-65% of packaging waste by 2001.

The purpose of the 1999 *Landfill Directive* (99/31/EEC)[15] is to harmonise controls relating to the landfilling of waste between all Member States. The directive mainly affects local authorities that are the major drivers for recycling and composting targets, and business and industry in which companies will be required to separate their hazardous and non-hazardous waste. A primary objective of the Landfill Directive is to reduce the landfilling of biodegradable municipal waste to 75% of 1995 levels by 2006, 50% by 2009, and 35% by 2016. Biodegradable waste refers to garden waste, kitchen waste, park waste, as well as scrap paper and cardboard. As mentioned earlier, anearobic decomposition of this type of waste in landfills produces emissions of methane, a greenhouse gas that is associated with climate change and is 8-10 times more potent than carbon dioxide emissions.

The Directive also bans the co-disposal of hazardous and non-hazardous wastes and places bans or restrictions on the landfilling of liquid waste, clinical waste and other materials. Existing landfills need to comply with the Directive eight years after it is implemented in Member States, i.e., by 2009.

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[14] Official Journal L 365 , 31/12/1994 P. 0010 – 0023.
[15] Official Journal L 182 , 16/07/1999 P. 0001 – 0019.

The most recent Directive relating to waste was put forward in 2000 on the *Incineration of Waste* (00/76/EEC)[16]. This calls for reductions in emissions of nitrous oxides (NOx), sulphur dioxide (SO_2), hydrogen chloride, cadmium and mercury as well as for controls on releases into water. The Directive also targets the incineration of non-hazardous waste, which is identified as the largest source of emissions of dioxins and furans into the atmosphere.

Also established in 2000, the *Directive on End of Life Vehicles* (2000/53/EC)[17] lays down measures that aim, as a first priority, at the prevention of waste from vehicles and, in addition, at the reuse, recycling and other forms of recovery of end-of life vehicles and their components. Its purpose is to reduce the disposal of waste and improve the environmental performance of all relevant economic operators involved in the life cycle of vehicles, focusing particularly on the operators directly involved in the treatment of end-of life vehicles.

Though there have been significant developments in the development of legislation and regulations relating to waste, it is interesting to note a more general trend in environmental policy over the past decade or so towards the attainment of 'optimal' or efficient control. This entails the balancing of marginal costs and benefits or alternatively, a trend toward the use of more cost-effective policies (i.e., away from exclusive use of command and control approaches to environmental policy towards the adoption of economic instruments such as environmental taxes and/or emission trading programs).[18] Although this trend seems to be less prevalent in the waste management Directives established within the E.U., some interest in the use of such instruments has nevertheless ensued. A recent Communication from the Commission states that:

[16] Official Journal L 145 , 31/05/2001 P. 0052 – 0052.
[17] Official Journal L 269 , 21/10/2000 P. 0034 - 0043.
[18] Examples include explicit language regarding the use of economic instruments in many of the protocols of the Long Range Transboundary Air Pollution convention and the Kyoto Protocol under the UN Framework Convention on Climate Change, *inter alia*.

At EU level the development of frameworks for the use of waste taxes or charges, re-use or recovery systems, financial and voluntary instruments should be examined; Possibilities to support the creation and efficient functioning of markets for recycled products should be examined at EU and national level; The concept of producer responsibility specifically addressed in some Member States should be further explored at EU level.[19]

Indeed, a number of European countries have introduced economic instruments as a means of attaining their waste and related environmental objectives at a lower total economic cost. Several countries have introduced landfill and incineration taxes for example, and a few communities have introduced pay-as-you-throw (PAYT) programs (*e.g.*, in Sweden and Germany). The purpose of a landfill tax is to increase the unit price paid for landfill disposal, thus providing municipalities with economic incentives to reduce the amount of waste they deliver to landfills and to stimulate recycling programs. In the UK, the landfill tax, implemented in 1996, was the first tax specifically directed at equating the level of the tax with the marginal external cost i.e., a true Pigouvian tax. Pay-as-you-throw, or unit pricing programs, aim to provide households with direct economic incentives to reduce the amount of waste generated for disposal and encourage recycling.

Interestingly, the case of the UK illustrates the extent to which waste management policy is driven by economic optimality or other considerations. Though the initial landfill tax in the UK was set on an assessment of externalities, an HMCE review published in 1998 found that the existing (optimal) tax had little impact on reducing the volume of active waste to landfills. This was becoming a policy imperative due to the targets established in the Packaging Directive and the forthcoming EU Landfill Directive. There was thus a shift in policy to setting rates to achieve environmental targets. The landfill tax on active waste was therefore increased from £7 to £10 per

[19] Source: Progress Report on Implementation of the European Community Programme of Policy and Action in Relation to the Environment and Sustainable Development "Towards Sustainability" [COM(95) 624] (www.europa.eu.int/comm/environment)

tonne from March 1, 1999, with the possibility to further increase the tax in the future.

The aforementioned concept of extended producer responsibility (EPR) refers to the extension of the responsibility of producers to include the social costs of waste management for their products. An example of this is the deposit refund system (DRS) whereby a deposit is levied on the production or sale of goods, and the refund is given to the household or to the producers that use recycled materials in production. Examples of these are currently in place in Austria (e.g., refrigerators), Belgium, Denmark (e.g., glass bottles), Finland (e.g., beverage cans), Germany (e.g., car batteries), Netherlands, Norway, Sweden, Switzerland, as well as Australia and South Korea (e.g., tires and washing machines).

Another form of extended producer responsibility is a manufacturer take-back program. The German Packaging Ordinance of 1991 is one example wherein manufacturers are required to pay to recycle their post-consumer packaging. Originally, firms were required to recycle 80% of all packaging. This was amended to 50% in 1996 and then 60% in 1998 (OECD, 1998). In order to benefit from economies of scale, the Duales System Deutschland (DSD) was formed, whereby local waste management firms collect all recyclable bottles of member organisations in exchange for payment from the DSD. The Green Dot on packaging identifies DSD members. (On economic criticism of EPR programs see Runkel, 2003 and Fullerton and Raub, 2004).

As of April of 2005, the UK has also established a Landfill Allowance Trading Scheme (LATS), designed to implement Article 5(2) of the Landfill Directive. The UK's landfill Directive targets have been divided between the four constituent countries: England, Wales, Scotland, and Northern Ireland. Trading of allowances is permitted in England, and in Scotland it is currently subject to discretion of the Scottish Executive until 2008, whereupon it will be permitted freely. As such, it represents the first example of a permit trading program applied in the field of waste management.

1.5.2 The US Context

In the US, federal legislation has largely vested responsibility for waste management with states and localities. The first federal law on the disposal of household, municipal, commercial and industrial waste, the *Solid Waste Disposal Act* of 1965[20], initiated a small program of technical and financial assistance to be given to state and local governments for MSW disposal demonstration projects (Macauley and Walls, 2000). The *Resource Recovery Act* of 1970[21] established federal authority to issue general guidelines for waste management. It is in the 1976 *Resource Conservation and Recovery Act* (RCRA)[22] and 1984 *Hazardous and Solid Waste Amendments*[23] that the federal government takes a more direct, though limited, role in MSW management. The primary goals of the RCRA are to protect human health and the environment from the potential hazards of waste disposal, to reduce the amount of waste generated, to ensure that wastes are managed in an environmentally sound manner, and to conserve energy and natural resources. Subtitle D, which deals with MSW, sets forth criteria that restrict the location of landfills, establish guidelines for their design and operation, require the monitoring of groundwater near landfills, and establish rules for opening and closing landfills. As a result, about 900 landfills closed that year.

RCRA also assigned to the states the responsibility of regulating the market for household solid waste collection and recycling. The reason behind this was the inherent differences in industry practices and environmental conditions across the states (Callan and Thomas, 1997). It is thus at the state and local levels that waste legislation in the US becomes more interesting, and indeed a wide variety of policy

[20] Public Law (Pub. L) 89 - 272, Oct. 20, 1965, 79 Stat. 997, as added.
[21] Pub. L. 91-512.
[22] Pub. L. 94-580.
[23] Pub. L. 98-616

approaches have been adopted. The most common is to set a goal for recycling as a percentage of the solid waste stream (Fullerton and Kinnaman, 1995) where more than forty states have legislatively mandated specific, quantified recycling and/or waste reduction goals[24].

Moreover, a number of states have passed laws that require all municipalities to implement curbside recycling programs[25] and to pass local ordinances making household participation in the recycling program mandatory. Today there are more than 9,000 communities in the US with curbside recycling programs[26]. More than 23 states have also banned certain wastes, such as yard waste, from being disposed of at landfills (Kinnaman, 2005). Other materials banned from landfill disposal include automobile tires and batteries. In addition, at least 13 states and the District of Columbia have adopted minimum recycled content standards for newsprint (Palmer, Sigman, and Walls 1997), and California and Oregon have recycled content standards for glass and plastic containers (Macauley and Walls, 2000). Furthermore, 47 states provide some form of tax credits, low-income interest loans, or grants for recycling facilities (Kinnaman, 2005).

In 1971, the State of Oregon was the first to pass legislation for a Deposit Refund System (DRS) for empty beverage containers. This was followed by nine other states[27] in the 1970's and 80's and more recently Hawaii in 2002 (Fullerton and Raub, 2004). California and Florida adopted advance disposal fees (Macauley and Walls, 2000).

Though the US has not embraced the concept of landfill taxes for MSW, several states have introduced taxes on either the generation or disposal of hazardous waste (Sigman, 1996). The US was also the first to introduce unit-based pricing or pay-as-you-throw (PAYT) programs on the West Coast in the mid-1980's. The number of

[24] Kinnaman and Fullerton (1997) find no significant impact of these goals on recycling quantities.
[25] *i.e.*, 22 states as of 1998 (Kinnaman, 2005).
[26] www.epa.gov/epaoswer

these has expanded dramatically and is currently implemented in approximately 6,000 communities[28].

With regard to EPR, the closest the US has come to instituting a similar program is the Mercury-Containing and Rechargeable Battery Management Act, passed by Congress in 1996, facilitating a national voluntary take-back system for nickel-cadmium rechargeable batteries (Macauley and Walls, 2000).

To summarise, in Europe much of the waste policy is influenced by requirements specified at the EU level. Individual Member States are then allowed the flexibility to adopt their own policies and measures to implement the requirements and attain the targets imposed. In contrast, in the US most of the responsibility is vested with states and local authorities and there is no effective federal plan in place to minimise waste and maximise recycling.

Despite these legislative and policy developments, significant government and market failures continue to exist in the field of waste management and need to be addressed in order to mitigate and reverse the trends in waste generation, its inefficient disposal, and the externalities they cause.

[27] California, Connecticut, Delaware, Iowa, Maine, Massachusetts, Michigan, New York, and Vermont.
[28] www.epa.gov/epaoswer

1.6 Aim and Overview of the Thesis

The focus of this thesis is exclusively on municipal solid waste management and policy. Though MSW does not form the largest fraction of waste, its characteristics are such that much of it is biodegradable and thus associated with emissions from landfill sites, and some of it is recyclable and hence subject to recovering and recycling policies. The thesis examines the issues of waste generation, disposal and recycling at the macroeconomic and household level. The main purpose of this thesis is to shed some light on the determinants of municipal solid waste generation and disposal, on recycling behaviour and preferences, and on policy implications for effective MSW management.

To begin, Chapter 2 presents a broad overview of the current state of the literature on sustainable MSW policy incentives and their implementation. The efficient or 'optimal' levels of waste generation and disposal are defined and the use of economic instruments to attain the optimal allocation of waste to the various waste streams (i.e., landfill, incineration, recycling) is reviewed. Finally, existing gaps in the literature are identified and discussed.

Chapter 3 examines the determinants of MSW generation, analysing macro-economic OECD country data to identify the driving forces between inter-country differences. Using recently available cross-sectional time-series data, a reduced structural equation is estimated to establish whether MSW generation levels continue to increase monotonically, as has been found in a handful of studies conducted in the 1990s. Subsequently, the effect of additional economic, demographic and policy variables on MSW generation is examined, and the implications for sustainable MSW management are discussed.

Chapter 4 follows directly from chapter 3, providing a panel data analysis of the determinants of MSW disposal and recycling. With regard to disposal, the focus is on the proportion of MSW that is disposed of at landfills. For recycling, the dependent variables examined are the proportion of paper and cardboard that is recycled (as a percentage of apparent consumption), and similarly, the proportion of glass that is recycled. The results of the analysis provide insights into the economic, demographic, and public policy characteristics that have an important impact on MSW disposal and recycling rates.

This analysis is then augmented in Chapter 5 with the use of spatial econometric techniques. Using this approach, it is possible to examine whether national governments are influenced by waste management trends and policy decisions in countries located nearby. With the use of a spatial weights matrix, spatially weighted values of the dependent variables are created and included in the regressions. Perhaps more interestingly, the chapter also examines whether OECD countries are engaged in strategic environmental policymaking by investigating the determinants of landfill tax rates. The evidence suggests that spatial interaction does exist in certain cases, an element that has not previously been examined in the waste management literature.

As described above, recent developments in national waste management policy has prompted considerable interest into alternative waste management programs that would divert a portion of the MSW stream from landfills. Chapter 6 examines household preferences for kerbside recycling services and uses a stated preference choice experiment to estimate the magnitude of these in monetary terms. Using a sample of 188 households in the London area, the empirical analysis yields estimates of the willingness to pay for the number of 'dry' materials collected, the collection of compost, textile collection and the frequency of collection.

Finally, chapter 7 draws together the main conclusions of the thesis and discusses the implications for sustainable MSW policy. Contributions to literature on MSW management are also discussed. The thesis closes with suggestions for future research to further assist decision-makers in designing policies and programs that can achieve more efficient and sustainable municipal solid waste management.

CHAPTER 2

Municipal Solid Waste Policy and Implementation: A Literature Review

2.1 Introduction

The issue of MSW management remains a practical concern in most regions throughout the world, while waste managers and policy-makers continue to search for appropriate methods to manage this issue more efficiently. Implicit in the concern about MSW management is that in the absence of government intervention there will continue to be excess production of waste, and that this waste will be misallocated between each of the possible disposal methods i.e., landfill, incineration, and recycling. The past two decades have witnessed dramatic changes in the way MSW is managed, and the number of waste related policies that have been implemented continues to increase. Despite these efforts, serious market and government failures still remain and MSW generation levels continue to rise.

The purpose of this chapter is to provide an up-to-date review of the key concepts and issues in the literature on efficient waste management and policy. The chapter discusses the inefficiency of waste generation and disposal in the absence of government intervention and presents the various regulatory and economic incentive methods that can be used to address the waste issue at different stages of the life-cycle. Efficient policies for waste management are presented and their effectiveness is reviewed. The available evidence on the costs and benefits of waste disposal options and household preferences for recycling are also discussed.

The chapter is organised as follows: Section 2.2 begins by defining the socially efficient level of waste generation and disposal, and briefly introduces the available regulatory and economic instruments that aim to foster the correct incentives for sustainable MSW management. These include *inter alia* unit pricing programs, virgin materials taxes, and deposit-refund mechanisms. These are presented in more detail in section 2.3, and section 2.4 reviews their implementation and their effectiveness along with some other related issues. Finally, section 2.5 concludes and identifies existing gaps in the waste literature.

2.2 Solid Waste Generation and Disposal

The overarching goal of waste management is to minimize the generation of waste while maximizing the ability to re-use and recycle it, in a way that is in line with the basic principles of environmental effectiveness, social equity, and economic efficiency.

Several waste management evaluation and assessment tools have evolved over time and have been applied to waste management. Morrissey and Brown (2004) divide these into three categories: Cost Benefit Analysis (CBA), Life Cycle Assessment (LCA), and Multi-Criteria Decision-making Analysis (MCDA). CBA enables decision-makers to assess the positive and negative effects of a set of scenarios by translating all impacts into a common monetary measurement e.g., by estimating how much individuals are willing to pay for an environmental improvement. The scenario with the greatest net benefits is the preferred option.

LCA is a tool that studies the environmental aspects and potential impacts throughout the product's life, from raw material acquisition through production, use and final disposal (i.e. from cradle to grave). This takes a holistic approach for comparing different products or waste management systems so that environmental improvements can be made. McDougall et al. (2001) link the concept of integrated waste management (IWM) with that of LCA. IWM systems combine waste streams, waste collection and treatment, and disposal methods with the objective of achieving environmental benefits, economic optimization, and social acceptability. Though LCA allows the trade-offs of different options to be assessed and comparisons to be made, it cannot guarantee which choice is optimal because it cannot assess the actual environmental effects of the product, package or service system[1].

Finally, MCDA is a technique for comparing impacts in ways that do not involve giving all impacts monetary values. MCDA often involves combinations of some

[1] For examples of LCA studies see Powell (2000) and Bovea and Powell (2006).

criteria measured via monetary terms and others for which monetary evaluations do not exist. It takes individual and often conflicting criteria into account in a multidimensional way. The criteria chosen could include a risk assessment or an environmental impact assessment. The result is a ranking of alternatives. However, the allocation of weights to different criteria is a subjective decision, and changing these can lead to different preferred options or ranking of alternatives. The subjectivity that pervades this can be a matter of concern. Its foundation, in principle, is the decision makers' own choices for objectives, criteria, weights and assessments of achieving the objectives, although 'objective' data such as observed prices can also be included. One limitation of MCA is that it cannot show that an action adds more to welfare than it detracts. Unlike CBA, there is no explicit rationale or necessity for a Pareto Improvement rule that benefits should exceed costs. Thus in MCA, as is also the case with cost effectiveness analysis, the 'best' option can be inconsistent with improving welfare, so doing nothing could in principle be preferable.

Moreover, these models take into account waste once generated and do not generally consider waste prevention, waste minimisation, or product design for the environment which would eliminate the production of materials which cannot be reused, recycled, or naturally biodegraded[2] (Morrissey and Brown, 2004).

CBA aims to maximise, as far as possible, the aggregate social values in the decision-making process and is thus the approach taken here for organising the framework of analysis. It takes both private and social stakeholders into account, where the latter includes a wider social and environmental perspective and the jurisdiction may be local, national, regional or global.

From this perspective, the demand for waste services can be described as a derived demand, arising from the consumption of commodities or from the production of products and services. As these economic activities inevitably lead to the generation of residuals, a demand for waste services is created. In most communities today municipal solid waste services are paid for using general revenues or monthly fees that do not vary per unit of garbage collected. Households thus behave as if more garbage is free. This public provision might be warranted if the service were non-rival[3], but in fact the marginal cost of collecting and disposing of another unit of waste is decidedly nonzero. The community must pay for additional labour, truck space, and tipping fees at regional landfills and incinerators. Additionally, free public provision would be warranted if the service were non-excludable[4], but it is indeed possible to charge a price per unit of waste collected (Kinnaman and Fullerton, 2000).

Figure 2.1 illustrates the demand curve for waste collection services (DWS). As the price of these services declines, their demand increases. If the price is independent of the quantity of waste that is disposed, then households are in effect faced with a zero price of waste disposal. This results in an over-consumption of waste services equivalent to W inducing a welfare loss to society equal to the shaded area L. In contrast, if a household faced a unit price for waste, equal to the marginal social cost (MSC) of waste services, P*, then the quantity of waste requiring disposal would decline to W*, the optimal quantity of waste generation.

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[2] For examples of MCDA studies, see Vaillancourt and Waaub (2002) and Higgs (2006). For a discussion of this approach see Fawcett et al. (1992).

[3] A non-rival good or service is one whereby the consumption by one individual does not reduce the amount of the good or service available for consumption by others.

[4] A non-excludable good or service is one whereby it is not possible to exclude individuals from the goods or services' consumption.

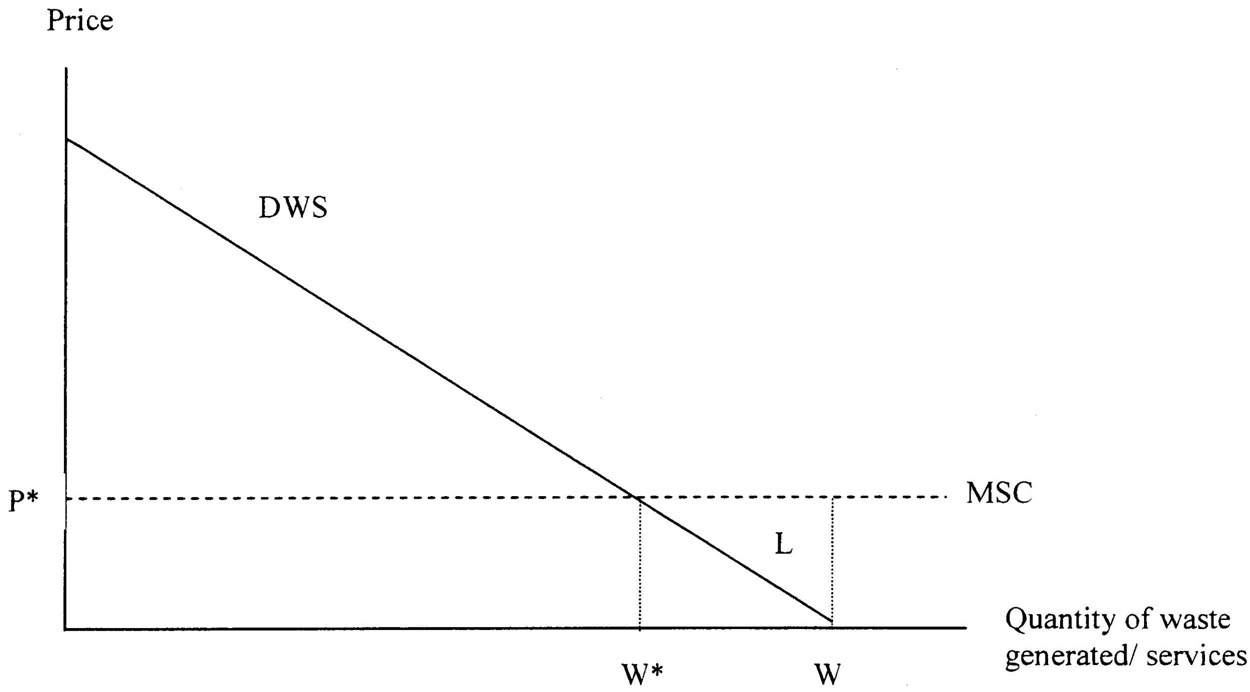

Figure 2.1 Demand for MSW Services
Source: Jenkins (1993)

The marginal social cost of waste services reflects the full cost to society of disposing/treating an additional unit of waste and is composed of two terms: the financial or private marginal cost (PMC) and the marginal external cost (MEC). In the case of landfills for example, the latter consists of the opportunity cost of the land (OP), the marginal user cost (MUC) which reflects the use of a finite resource, and the marginal external environmental cost (MEEC).

$$MSC = PMC + MEC$$

where for landfills,

$$MEC = OP + MUC + MEEC$$

Thus the optimal level of waste generation can be defined. This level occurs at the point where the marginal costs of source reduction equals the marginal benefit of source reduction, which is also equal to the avoided marginal social cost of waste collection and disposal.

Once the optimal level of waste generation (and demand for waste services) has been ascertained, the next issue is to determine how to optimally dispose of and treat the waste. Since reuse is only feasible for a very small fraction of the MSW stream, the management options are assumed to be landfilling, incineration, and recycling. Using a simple optimisation model, Brisson (1997) shows how this can be done. Formally, the objective is to minimise the net social costs (NSC) of waste management for all waste, subject to the total amount of waste, W, to be disposed of, where

$$W = W_L + W_I + W_R$$

and W_L, W_I, and W_R refer to the waste disposed of at landfills, incinerated and recycled respectively. Note that the net social costs consist of the financial costs of waste disposal as well as the external costs of waste disposal, and that waste management (or treatment) can also provide some benefits to society. In the case of recycling for example, benefits are derived from selling the part of the material recovered; for landfills and incineration, benefits are derived from selling the recovered energy[5]. The Lagrangian for this problem is therefore to minimise the NSC of each, subject to the constraint:

$$L = NSC(W_L) + NSC(W_I) + NSC(W_R) + \lambda\,(W - W_L - W_I - W_R)$$

The first order conditions are:

$$MNSC_L = MNSC_I = MNSC_R$$

where MNSC is the marginal net social cost. The solution is shown graphically in Figure 2.2 below.

[5] Nakamura (1999) shows that in the case of paper recycling the benefits may be substantial.

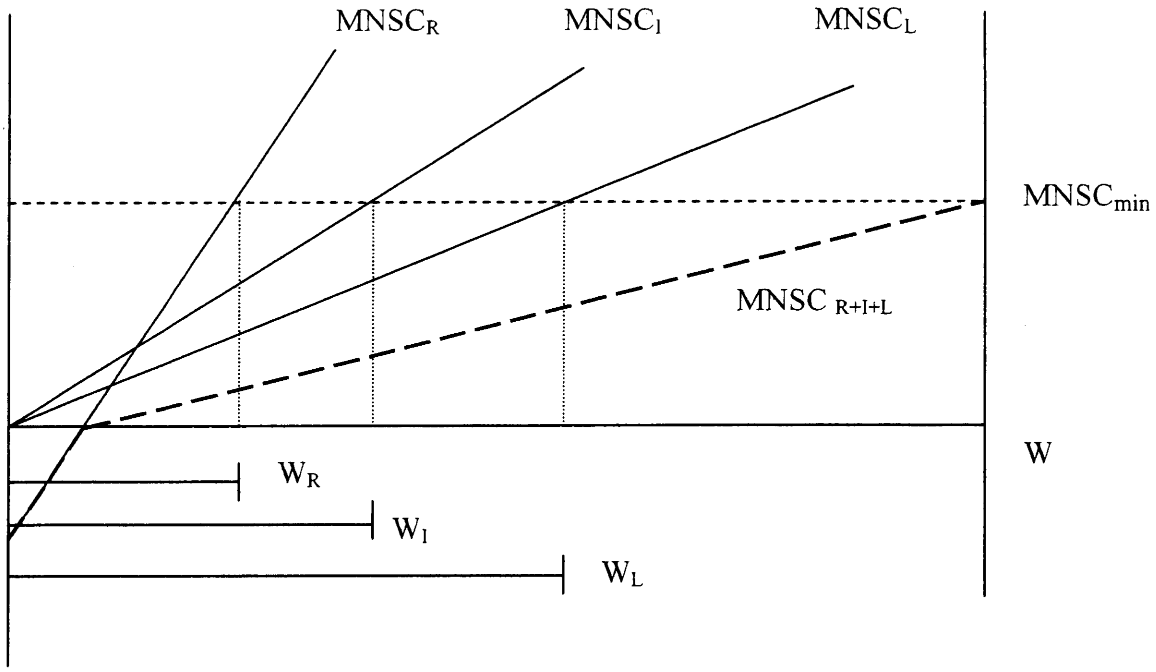

Figure 2.2 Optimal Waste Management Levels

Source: Brisson, 1997

$MNSC_{R+I+L}$ represents the total amount of waste that can be managed at any given marginal net social cost. The minimum marginal net social costs $MNSC_{min}$ at which all waste can be managed is also shown in the figure. The optimal level of each of the disposal options is read off the individual MNSC curves by following the dotted lines.

In comparing this figure to the waste hierarchy discussed in chapter 1, it is worth noting that the hierarchy seems to assume constant ranking at all levels of pollution. In fact, the waste hierarchy appears to be based on some form of 'green intuition' and presents a ranking order through which waste disposal should rise. It should not be interpreted as implying that all waste should be recycled. In reality, we do not know what the optimal allocation of waste to the different disposal/treatment routes is. In fact, optimal levels of landfill, incineration, and recycling may depend on other factors as well. Highfill and McAsey (2001) for example examine the decision between landfilling and recycling under the assumption that income is growing, and find that municipalities with low incomes should rely less on recycling than those with high incomes. The general consensus however is that, due to special tax treatment for extraction of virgin materials, energy subsidies, and other legal and regulatory measures, waste generation is excessive, and that the existing markets tend to encourage the disposal of waste at landfills and incinerators.

A number of regulatory and economic instruments are available which can be implemented at various stages of the life-cycle of products to provide the necessary incentives for the optimal management of waste. These can affect the design, production, packaging, sale, use and disposal (Fullerton and Wu, 1998). Regulatory instruments include, *inter alia*, mandates a) on the disclosure of toxic materials used, and b) on consumer separation of materials for recycling or diversion rates for various materials; the establishment and tightening of existing regulations; and bans or phase outs of hazardous chemicals. The economic instruments that are available are listed in Table 2.1.

Table 2.1 Economic Policy Instruments for Waste Management

Life-cycle stage	Economic instruments
A. Raw material extraction and processing	1. Eliminate special tax treatment for extraction of virgin materials, and subsidies for agriculture. 2. Tax the production of virgin materials.
B. Manufacturing	1. Tax industrial emissions, effluents, and hazardous wastes. 2. Establish tradable emissions permits. 3. Tax the carbon content of fuels. 4. Establish tradable recycling credits. 5. Tax the use of virgin toxic materials. 6. Create tax credits for use of recycled materials. 7. Establish a grant fund for clean technology research.
C. Purchase, use, and disposal	1. Establish weight/volume-based waste disposal fees. 2. Tax hazardous or hard-to-dispose products. 3. Establish deposit-refund system for packaging, hazardous products. 4. Establish a fee/rebate system based on product energy efficiency. 5. Tax gasoline
D. Waste management	1. Tax emissions or effluents from waste management facilities. 2. Establish surcharges on wastes delivered to landfills or incinerators

Source: Office of Technology Assessment, cited in Fullerton and Wu, 1998

An expanding literature has developed that analyses many of these instruments to determine which are most able to achieve first-best (and in some cases, second-best) outcomes. This literature is reviewed in the next section.

2.3 Efficient Policy Incentives for Waste Management

By far, most of the theoretical literature on efficient policy instruments for MSW management has examined the appropriate design of a tax and/or subsidy policy to achieve the efficient allocation of waste to the disposal options available. In general these papers develop models in which households maximize utility subject to a budget constraint that incorporates a unit price for waste collection. The models form the basis for solid waste disposal and recycling demand equations (Jenkins et al.

2003). Kinnaman and Fullerton (1999) and Fullerton and Raub (2004) provide a skeletal model to frame the subsequent discussion of optimal policy design. Assume *n* identical consumers each maximising utility subject to a budget constraint and a mass-balance equation given by c=c(g, r) where *c* is consumption which produces waste that must either be disposed of as garbage *g* or recycled *r*. The total amount of solid waste disposed is given by G = ng, and utility is a function of all of these: U = u[c(g, r)]. Note that consumption *c* has a positive effect on utility and garbage *G* has a negative effect on utility. The household budget constraint is therefore given by:

$$y = (p_c+t_c)\, c(g, r) + (p_g + t_g)\, g + (p_r + t_r)\, r$$

where *y* is income, *p* is price, and each *t* is a tax rate. The price p_r may be negative if a private firm pays consumers for recycled material, and any tax can be positive or negative. With no government intervention and hence all taxes set to zero, households will fail to internalise the full social costs of their disposal decisions, resulting in too much garbage and too little treatment in a decentralised economy. Instead, households can be taxed on each unit of garbage disposed (at rate t_g), or subsidised for their recycling effort (at rate - t_r). Alternatively they may be required to pay an advance disposal fee at the time of purchase (t_c). Producers in the model will produce c according to the production function c = f(v, r) where *v* is virgin material inputs and *r* recycled inputs. Given input prices p_r and p_v the producers chooses output to maximise profits:

$$\pi = p_c f(v, r) - (p_v + t_v)v - (p_r - s_{fr})r$$

where the producer's use of virgin materials could be taxed (at rate t_v), or the use of recycled materials could be subsidised (s_{fr}).

2.3.1 Unit-Based Pricing

Unit-based pricing refers to the imposition of a tax (t_g) on each unit of garbage deposited, which can be levied either by volume or by weight. It has been shown that taxing garbage directly is sufficient to achieve the efficient allocation of resources as long as households face the full social costs of their disposal decisions (Fullerton and Kinnaman, 1995; Palmer and Walls, 1994; Fullerton and Wu, 1998; Calcott and Walls, 2000). Households are charged with a tax per unit of garbage disposed and are thus provided with appropriate incentives for reducing disposal and participating in recycling activities. Furthermore, such unit pricing can also induce firms to produce the optimal amount of packaging per unit and to engage in the optimal amount of green design (Fullerton and Wu, 1998; Eichner and Pethig, 2001). Several potential problems have however been identified with the use of such a mechanism, namely:

- Illegal dumping
- Administrative costs of implementing pricing garbage by the bag
- Different social costs associated with different waste streams
- Prohibitive monitoring and enforcement costs
- Absence of functioning markets for recyclables
- Effect of household reduction effort.

Higher prices on the disposal of waste may induce households to partake in illicit or illegal dumping[6]. Illegal disposal of waste is associated with high external costs, such that these, or the additional monitoring, enforcement and collection costs associated with illegal waste, may outweigh the benefits of reducing legally disposed of waste. Under this scenario, Fullerton and Kinnaman (1995) have found that the optimal tax on legal garbage disposal may well be negative. In another study conducted in Charlottesville, Virginia, they find that the administrative and enforcement costs of a unit based program exceed the $3 per person social benefits of recycling (Fullerton and Kinnaman, 1996). This may not be a universal phenomenon however; in a study of the Dutch municipality of Oostzaan, Linderhof et al. (2001) find that the net costs of waste collection and processing did not increase as a result of a unit pricing program, as any cost increases were offset by lower waste-treatment costs.

[6] Fullerton and Kinnaman (1999) list several empirical studies that examine the evidence for illegal dumping. The results are mixed.

Dinan (1993) raises the issue that a uniform tax on all types of garbage may be inefficient if materials within the waste stream produce different social costs. For example, the social costs associated with the disposing of flashlight batteries are likely to be significantly higher than those associated with newspaper disposal. Again however, administering a differentiated tax may be exceedingly costly.

Calcott and Walls (2000) argue that the efficiency of the tax depends on the assumption that households are being paid for recycling. They find that in the absence of a functioning recycling market, policy instruments need to be targeted at both disposal and recycling. To attain a feasible constrained optimum they argue for a deposit system that entails two rates, one applying to recyclable products and the other to non-recyclable products (see discussion on the modified deposit refund scheme below). Fullerton and Wu (1998) also find that this problem can be corrected with the use of a subsidy on 'recyclability' combined with an output tax and a tax on packaging.

Choe and Fraser (1999) extend these models and incorporate the concept of household waste reduction effort with illegal waste disposal. They focus on waste reduction effort by firms and households, and on illegal waste disposal by households. When household reduction effort is significant, the monitoring of illegal waste disposal becomes necessary and they find that a Pigovian tax is sub-optimal. Instead, the second best policy requires a positive waste collection charge on the household, explicit monitoring of illegal waste disposal, and a positive environmental tax on the firm. This is because, with household reduction effort, the waste collection charge will induce households to reduce their waste. This may also result in illegal waste disposal, which needs to be monitored at an additional cost. Higher waste collection charges aimed at inducing the efficient amount of waste disposal will necessitate higher monitoring costs that may be prohibitive. Instead, the regulator can indirectly tax household waste through an environmental tax on the firm.

These results indicate that though unit-based pricing can be implemented with a relatively simple mechanism to achieve efficient outcomes, it will be necessary to evaluate whether the administrative and enforcement costs outweigh the social benefits of recycling and will need to assess the likelihood of illegal dumping as this can undermine the efficiency of the program.

2.3.2 Virgin Materials Tax

In response to these issues, economists have examined alternative policies that may result in a more efficient waste disposal outcome, including a tax on virgin materials. The intended purpose of such a tax is to increase producer demand for recycled inputs, raise the price paid for recycled materials and increase the economic benefits to households that deliver recyclable materials to secondary markets (Fullerton and Kinnaman, 1999).

Miedema (1983) finds that a tax on virgin materials set equal to the social marginal costs of disposing any resulting waste material produces greater welfare gains than other instruments such as a recycling subsidy to producers or an advanced disposal fee. Sigman (1995) compares the use of a) taxes on the use of virgin materials, b) deposit/refund programs, c) subsidies to recycled material production, and d) recycled content standards, specifically looking at cost-effectiveness in achieving reductions in lead from automobile batteries. She finds that the virgin materials tax and the deposit-refund are the best (i.e., least-cost) policies and equivalent in the incentives they create.

However, subsequent studies have found that virgin material taxes may not be able to attain the efficient outcome. For example, Dinan (1993) argues that a tax on virgin materials does not provide an incentive to increase the use of, say, old newspapers in products where it does not displace virgin materials (e.g., exports and animal bedding). Fullerton and Kinnaman (1995) find that virgin materials should only be taxed if their extraction has a negative externality (e.g., strip mining). Alternatively, if a tax on virgin materials is in place, it needs to be implemented alongside a tax on all other inputs except recycling. Palmer and Walls (1994) find that a virgin materials tax discourages production and consumption in the economy, thus leading to an inefficiently low quantity of waste. They find therefore that a virgin materials tax needs to be combined with a subsidy on the sales of final goods.

Thus, the overall weight of the evidence on virgin materials taxes suggests that the efficient introduction of such a tax requires it to be combined with a number of additional economic incentives, thereby complicating the administrative aspects of its' implementation.

2.3.3 Recycling Subsidy

In lieu of taxing garbage disposal or virgin materials, studies have also investigated the possibility of subsidising recycling. A recycling subsidy has the effect of lowering the cost of waste disposal and subsidising consumption (Palmer and Walls, 1994; Palmer et al. 1997). This instrument cannot therefore attain the optimum level unless it is coupled with a tax on consumption (or advance disposal fee, see below).

2.3.4 Advance Disposal Fee

An advance disposal fee (ADF) assesses a charge on the final product based on the implied disposal cost for the associated packaging, i.e., it is a charge on all consumption of the final material. Palmer et al. (1997) examine the use of an ADF at the producer level rather than the household level, which increases the price of final material to all demanders, including recyclers and nonrecyclers. They find that though the ADF (or upstream tax) has an output effect i.e., it decreases output because of the increase in the production costs brought about by the fee, the ADF does not have an input substitution effect, i.e., of recycled for virgin material inputs. Thus it is not able to attain the optimum on its own.

2.3.5 Deposit Refund Scheme

An alternative waste policy that has received strong support in the literature is a deposit refund scheme (Fullerton and Kinnaman 1995; Dinan 1993; Sigman 1995;

Palmer et al. 1997). This is in effect a combination of an ADF (or a tax on output) and a recycling subsidy. Under such a scheme, the consumer only bears a cost if the product is discarded. Goods that are produced and then recycled avoid disposal cost charges. It is therefore equivalent to unit-pricing in which households pay for the amount of disposal, but avoids the problem of illegal dumping.

Dinan (1993) compares a deposit refund scheme with unit-based pricing and makes the following conclusions: (1) A deposit refund policy may be more appropriate for items with higher than average disposal costs. This is because unit-based pricing programs usually charge households a constant fee per unit of garbage, irrespective of their contents; (2) A deposit refund policy is better than unit-based pricing in communities where illegal disposal is prevalent or for goods that pose high environmental costs when illegally disposed of; (3) To address administrative costs a deposit refund policy should be targeted for selected items in waste stream e.g., old newspapers, old tires, and lead acid batteries.

Palmer, Sigman and Walls (1997) suggest the deposit refund scheme be placed upstream to avoid the transaction costs of dealing with households[7]. They compare three price-based policies, namely deposit refunds, ADFs, and recycling subsidies and find that the deposit refund is the least costly, and the recycling subsidy the most costly option for attaining a specified percentage reduction of disposal. Sigman (1995) finds similar results with respect to deposit refund schemes in her analysis of lead recycling from automobile batteries.

A generalisation of the deposit refund scheme is suggested by Fullerton and Wolverton (2000) whereby the scheme is applied to *any* waste from production or consumption, including solid, liquid, or gaseous wastes. The tax is on a purchased commodity – a normal excise tax on output, paid by the seller or consumer; the subsidy to clean activity (e.g., abatement, recycling, landfill disposal), and is paid to the household or to the waste-processing firm. To minimise administrative costs, the subsidy could be paid per ton of waste at the sanitary landfill or per ton of recycled material such as aluminium or glass. The subsidy is to be passed on to consumers through market prices, e.g., for a recycling firm to receive higher subsidy payments, they will be willing to offer incentives to consumers such as the free collection of recyclable waste. This approach achieves the same equilibrium as a pigovian tax on 'dirty' activity but does not require the measurement of emissions or dumping. Several implementational issues are considered including the inducement of theft of waste to earn a subsidy; the generation of variable amounts of waste with different marginal external damage costs; and open economy issues.

Despite the theoretical superiority of DRS in comparison to most other policy incentives, in practice, DRS is only feasible for a limited number of products, e.g., glass bottles and batteries, and is therefore not suitable for a large fraction of the MSW stream.

2.3.6 Modified Deposit Refund

Calcott and Walls (2000) examine a scenario under which a recycling subsidy is not feasible. They argue that in most communities, garbage is collected for free and that the cost of implementing a subsidy to recycling may be prohibitively expensive. Thus, in the absence of functioning markets for recyclables, they solve for a second best instrument, namely a modified deposit-refund program. The deposit depends on whether or not a product is eligible for recycling i.e., attaining the threshold level of recyclability necessary for recyclers to be willing to collect the product from households. Producers of products that are recyclable pay a tax up-front that is equivalent to the refund received by recyclers; if not recyclable, producers pay an 'advance disposal fee' which is a tax equal to the marginal social cost of disposal.

[7] Thus, beverage can producers would pay the deposit (or ADF) when they purchase aluminium sheet, and the refund (or recycling subsidy) would be granted to collectors of the used beverage cans who subsequently sell them for reprocessing.

A somewhat similar scenario has also been examined by Fullerton and Wu (1998) who argue that if illegal disposal is an issue, free garbage collection may be necessary. In such a case, manufacturers must be provided with the correct incentives, consisting of a tax on packaging and a subsidy to designs that improve recyclability. This involves three instruments, but suffers from substantial informational requirements.

2.3.7 Landfill and Incineration Taxes

A landfill or incinerator tax is a unit tax on each ton of waste disposed of at the site, and can alter the configuration of waste disposal depending on the relative positions and steepness of the marginal financial costs curves of each. In the case of industrial or commercial waste, a fee linked to the quantity of waste is normally charged for disposal. A tax increase would therefore provide industry or commerce with an economic incentive to reduce the amount of waste they deliver to landfill (or incinerators), which could be achievable via source reduction, recycling, or illegal dumping. Households will not face these incentives unless there is a unit-based pricing scheme. Sigman (1996) argues that a tax system directed at environmental releases such as air emissions from incineration and ground water contamination from landfills would more accurately reflect environmental costs, especially if these varied with geographic factors such as hydrology and population density. More recently, Kinnaman (2004) argues that a Pigouvian landfill tax set equal to the external marginal cost of garbage collection, transportation, and disposal, will induce municipalities to adopt individual solid waste management policies efficiently, and that central governments should abstain from mandating kerbside recycling or user fees for all of their municipalities.

2.3.8 Recycled Contents Standards

Finally, a small literature has developed examining how regulatory approaches to waste policy can generate the optimal amount of disposal, but these studies generally conclude that economic instruments tend to be preferable. Palmer and Walls (1997) examine the use of recycled contents standards which require that products be manufactured with a certain minimum amount of recycled materials as a fraction of total virgin plus recycled materials. They find that, in order to generate the optimal amount of disposal, these must be combined with additional taxes on both the final product and other inputs to production. However, the informational requirements of implementation are high, and the authors conclude that the deposit-refund approach is generally preferable. In a subsequent paper (Walls and Palmer, 2001), they find that regulatory standards with taxes can also attain first best. If the standard is set per unit of polluting input, then a tax on that input is also necessary. If the standard is set per unit of output, an output tax is necessary. They also find that there may be a role for ADF to correct for life-cycle externalities, but only when these are in conjunction with pollution standards per unit of output.

Sigman (1995) also looks at recycled content standards but assumes trading between firms is allowed, thus enabling cost minimisation if the permit market is competitive.

2.3.9 Manufacturer Take-Back Requirements

Manufacturer take-back requirements are another form of regulatory instrument. The rationale for take-back requirements is that firms would have the correct incentives to reduce packaging and to design for recyclability if they were made responsible for the disposal of their own packaging and products. Fullerton and Wu (1998) find however that the take-back requirement in itself is not sufficient to attain an efficient outcome, and it needs to be complemented with a tax on garbage.

In a more recent paper, Shinkuma (2003) argues that when the first-best policy is not attainable (due to the potential for illegal disposal and the existence of transaction costs associated with a recycling subsidy, i.e., a refund, or deposit refund system), then when the price of a recycled good is negative and the marginal transaction cost is relatively high, a producer take-back requirement is the second-best policy.

To summarise briefly, the overall weight of the evidence supports the use of unit-based pricing (and deposit refund schemes on the number of products where this is feasible). However, local authorities will need to assess the likelihood of residents partaking in illegal dumping. This assessment will need to consider/identify the conditions under which illegal dumping is most prevalent (e.g. population density characteristics) and the attitudinal/cultural characteristics of the cohort in question (i.e. that some cultures may be more prone to illegal dumping than others, similar to e.g. tax evasion). Pilot programs would be a useful way to investigate these issues, along with obtaining implementation experience and real data on the administrative and enforcement costs. The following section turns to examine the empirical evidence on the effectiveness of these instruments.

2.4 Implementation and Effectiveness of Solid Waste Policies

Some of the earliest empirical papers on solid waste examined the impact of socio-economic factors on waste generation. With regard to the effect of household income, the results indicate that this has an inelastic impact on the household demand for solid waste management services. For example, in his study of two Detroit suburbs, Wertz (1976) finds an income elasticity of demand of 0.27. Richardson and Havlicek (1978), in their study in Indianapolis, report estimates of 0.24. The income elasticities from a number of other studies are summarised in Table 2.2.

Table 2.2 Estimated Income Elasticities

Study	Data	Estimate
Wertz (1976)	Households in two Detroit suburbs	0.27
Richardson and Havlicek (1978)	Neighbourhoods in Indianapolis	0.24
Hong et al. (1993)	2300 households in Portland, Oregon	0.05
Jenkins (1993)	American municipalities	0.41
Reschovsky and Stone (1994)	3040 households in upstate New York	0.22
Kinnaman and Fullerton (1997)	756 municipalities in U.S.	0.31
Podolsky and Spiegel (1998)	149 municipalities in New Jersey	0.55
Hong (1999)	3017 households from 20 cities in Korea	0.10
Johnstone and Labonne (2004)	30 OECD countries	0.15-0.69

Other socio-demographic factors of waste generation that have been examined include average household size, age composition, urban versus rural households, and the effect of education on waste generation levels. Increases in household size tend to decrease the per capita quantity of waste disposal (Jenkins, 1993; Kinnaman, 1994; Podolsky and Spiegel, 1998) as do education levels (Kinnaman and Fullerton, 1997; Van Houtven and Morris, 1999). Jenkins (1993) finds that an increase in the proportion of population aged 18 to 49 increases waste arisings. The effect of urban versus rural households on waste generation is more ambiguous. Some studies show that urban households generate less solid waste (Podolsky and Spiegel, 1998; Van Houtven and Morris, 1999), whereas others indicate that rural communities tend to have lower waste generation levels (U.S. EPA, 1994; Johnstone and Labonne, 2004). In support of the latter, it is argued that this is perhaps because rural households grow and prepare a greater portion of their food at home, reducing the generation of packaging waste (U.S. EPA, 1994), and because there may be a number of waste management alternatives (e.g., composting, burning, illegal disposal) (Beede and Bloom, 1995).

2.4.1 Unit-Based Pricing

The success of market based policies described in section 2.3 depends on the elasticity of demand for waste disposal services. For example, a unit pricing program will only affect the disposal of waste if the demand for disposal services is sensitive to the price of disposal services. The wide-spread proliferation of unit-based pricing programs in the U.S. in the mid-1980's and 1990's resulted in a number of studies that empirically investigate the effectiveness of these programs in (a) reducing the amount of waste disposed of, and (b) encouraging recycling. The earliest study is conducted by Wertz (1976) who compares the average quantity of garbage collected in San Francisco, a town with a user fee, with the average town in the United States and finds a price elasticity of demand equal to –0.15. In a more comprehensive study, Jenkins (1993) gathered monthly data from 14 towns in the U.S., 10 of which had unit-pricing programs and also found inelastic demand for waste collection services.

A 1% increase in the user fee is estimated to lead to a 0.12 % decrease in the quantity of garbage.

Hong et al. (1993) investigate the role of price incentives and other socio-economic factors in household solid waste recycling using self-reported household data. 2298 households were surveyed in the Portland metropolitan area where a variable service fee based on volume (per additional 32-gallon can), i.e., a block payment system, was in place. The results suggest an increase in the frequency of household participation in kerbside recycling but that such a system did not significantly reduce demand for garbage collection services.

Reschovsky and Stone (1994) use a dummy for the presence of unit pricing programs in upstate New York and find that the price of garbage has no significant impact on the probability that a household recycles. Instead, when user fees are combined with a kerbside recycling program, recycling rates increase by 27 to 58% depending on the type of material.

Miranda et al. (1994) also use self-reported household data from 21 cities throughout the U.S. over an 18 month period. They find that introducing unit pricing and recycling programs have a dramatic effect on the quantity of MSW generated. Towns reduce garbage by between 17 % and 74 % and increase recycling by 128 %.

In contrast to the above mentioned studies, Fullerton and Kinnaman (1996) use data from Charlottesville, VA, where waste has been physically measured for volume and weight at 75 households and where a \$0.80 fee per 32-gallon bag or can was introduced. They find that a household's actual weight of wastes generated fall by 14 %, the volume falls by 37 %, and the weight of recycling increases by 16 %. Note that the impact on weight is more important than that of volume since waste is compacted by collectors and at landfills anyway. The change in space used in the landfill is better measured by the change in the weight at the curb. Furthermore, their results indicate that illegal dumping of waste may account for between 28 to 43 % of the reduction in garbage. They conclude therefore that the incremental benefit of unit pricing is small.

Morris and Holthausen (1994) use data from Perkasie, PA, to calibrate a household production function model and find that unit pricing does have an effect on disposal with a price elasticity of 0.51 to 0.60. Callan and Thomas (1997) look at the percent of total waste stream recycled using community-level data in Massachusetts. By including a dummy for the presence of unit pricing programs, they predict that the portion of recycling increases substantially with a unit fee, and especially so when there is also a kerbside recycling program in place.

In 1995, South Korea implemented the first nation-wide unit pricing program. In a sample of households, Hong (1999) finds that unit-based pricing has a significant positive effect on the recycling rate and that the price elasticity of recycling is 0.46. The price elasticity of demand for solid waste collection services is very low (–0.15).

In contrast, Podolsky and Spiegel (1998) find a very large price elasticity of demand (-0.39) in a large data set of the U.S., and estimate the economic benefits of charging per unit of garbage to be as high as $12.80 per person per year. In another U.S. study of 959 communities, 114 of which have user fees, Kinnaman and Fullerton (2000) estimate the demand for garbage collection as a function of the price of garbage, the presence of kerbside recycling, and other relevant variables. They allow for the possibility of endogenous policy choices (e.g., regional tipping fees, population density, state policy variables, demographic characteristics) and find that correcting for endogenous policy increases the effect of the user fee on garbage and the effect of kerbside recycling collection on recycling. A $1 fee per bag is estimated to reduce garbage by 412 pounds per person per year (44%) but only to increase recycling by 30 pounds per person per year.

Linderhof et al. (2001) are the first to estimate short and long run price elasticities and they are also the first study to examine unit-based pricing in a European municipality, namely that of Oostzaan in the Netherlands. They use actual waste data for both compostable and non-recyclable waste from 4080 households over a period of 42 months. The elasticities are reported in Table 2.3 below.

Jenkins et al. (2003) analyse the determinants of household recycling by examining (a) a kerbside recycling program and (b) a unit pricing program. They also examine the impact of these two programs on different recyclable materials as these have different costs of recycling as well as different values on the open market. They look at glass bottles, plastic bottles, aluminium, newspaper, and yard waste, using a large household-level data set representing 20 metropolitan statistical areas in the U.S. The data set used therefore also facilitates the identification of policies and demographic variables that are significant across regions. Results indicate that access to kerbside recycling has a significant positive effect on the percentage recycled of all five materials. Furthermore, they find that the price of disposal is not a significant determinant of the intensity of household recycling effort for any of the materials.

Another material-specific study is that by Halvorsen and Kipperberg (2003) who examine household recycling in Norway. They use information on the recycling of six materials, namely carton, paper, plastics, metals, glass, and food, and find that both differentiated disposal fees and convenient recycling programs such as kerbside recycling and local drop-off centres positively affect recycling levels (in contrast to Jenkins above).

Interestingly, Klein and Robison (1993) are the only ones who estimate the impact of disposal fees on commercial behaviour and find that firms reduce solid waste generation when faced with higher disposal rates.

Table 2.3 Estimated Price Elasticities

Study	Data	Model	Estimate
Wertz (1976)	San Francisco	Comparison of means	ε = -0.15
Jenkins (1993)	Panel of 14 cities over 1980-88 (10 with user fees)		ε = -0.12
Hong, Adams, and Love (1993)	4306 households in Portland, Oregon	Ordered probit and 2SLS	No significant impact
Miranda et al. (1994)	21 cities in Unites States over 18 months		Significant impact
Morris and Holthausen (1994)	Perkasie, Bucks County, PA		ε = -0.51 to –0.60
Stratham et al. (1995)	Portland, Oregon metropolitan area	OLS	ε = -0.11
Fullerton and Kinnaman (1996)	75 households in Charlottesville, VA	OLS	ε = -0.076 (weight) ε = -0.226 (volume)
Fullerton and Kinnaman (1997)		OLS 2SLS	ε = -0.23 ε = -0.28
Podolsky and Spiegel (1998)	159 towns in New Jersey (12 with user fees)	OLS	ε = -0.39
Van Houtven and Morris (1999)	Marietta, Georgia	Tobit	ε = -0.26
Hong (1999)	3017 households in 20 cities in Korea	3SLS	ε = -0.15
Kinnaman and Fullerton (2000)	959 communities in the United States (114 with user fees)	2SLS	ε = -0.28
Linderhof et al. (2001)	4080 households in Dutch municipality over 42 months	LSDV	ε = -1.10 short run[a] ε = -0.26 short run[b] ε = -1.39 long run[a] ε = -0.34 long run[b]
Jenkins et al. (2003)	Household data in the United States	Ordered logit	No significant impact
Halvorsen and Kipperberg (2003)	Norway	Ordered logit	Significant impact

Source: Adapted from Kinnaman and Fullerton (1999) and Jenkins et al. (2003).
[a] = for compostable waste; [b] = non-recyclable waste.

On looking at the evidence as a whole, the findings are somewhat ambiguous with regard to the effects of unit-based pricing versus the introduction of a kerbside recycling program, and the effects these have on waste generation and recycling rates. The demand for waste disposal services is clearly inelastic. We turn now to examine the empirical evidence for alternative economic incentive waste policies.

2.4.2 Virgin Materials Tax

There are few examples of actual virgin materials taxes in place, and as such, little empirical evidence on their performance. The only available study seems to be a simulation model by Bruvoll (1998) who finds that a hypothetical tax of 15 % on plastic and paper virgin materials in Norway would result in an 11 % reduction in the use of these materials.

2.4.3 Recycling Subsidies and Advance Disposal Fees

Evidence on these instruments is also scant. One exception is Kinnaman (2005) who provides indirect effects of subsidies by looking at the availability of recycling programs. He finds that state subsidies for industries that recycle materials, as well as state recycling goals and bans on materials from landfills have no statistically significant impact on the availability of recycling programs.

Palmer et al. (1997) conduct empirical analysis on supply and demand elasticities for waste reduction using several price-based policy interventions for solid waste reduction namely deposit/refunds; advance disposal fees; and recycling subsidies. They develop a simple partial equilibrium model of waste generation and recycling to evaluate the relative cost-effectiveness of these policies and include the following components of the waste stream: paper, glass, plastic, aluminium, and steel. They find that a deposit/refund mechanism is the most cost-effective and would achieve a 10% reduction in all wastes with a $45 per ton fee. In contrast, the same reduction would be attained with an $85 per ton ADF or a recycling subsidy of $98 per ton. Furthermore, from a cost-benefit perspective, they find that only a modest reduction in MSW would be efficient if it could be accomplished without large administration and transaction costs[8].

2.4.4 Deposit Refund Schemes

Porter (1983) analyses the effects of a deposit refund scheme that was introduced in Michigan in 1978 on containers of packaged beer and carbonated soft drinks (*i.e.*, beverage containers). Beverage-related litter fell by some 85 % and the rate of return of containers for the refund was approximately 95 %. The study estimates the costs and benefits of the mandatory deposit refund scheme and an overall welfare assessment of the program is conducted. The results indicate that the program does not necessarily pass the cost-benefit ratio.

In a study examining the impact of the California Beverage Recycling and Litter Reduction Act on consumers, Naughton et al. (1990) find that the Act will significantly reduce beverage container solid waste and litter, but that the net benefits of the Act depend critically on consumers' valuations of intangible benefits.

Interestingly, Kinnaman (2005) finds that a deposit-refund program decreases the availability of kerbside recycling by 4.6% (though the coefficient is not statistically significant). He argues that municipalities may avoid implementing municipal recycling programs in these states if they believe that consumers would take aluminum (the most valuable recycled material) and glass beverage containers directly to outlets for a return on their deposit.

2.4.5 Landfill Taxes

Martin and Scott (2003) provide a qualitative analysis of the effectiveness of the U.K. landfill tax and argue that the tax has not been effective in diverting waste away from landfills. Only inert waste has decreased as a result of the tax, and it seems that the recycling of construction and demolition waste has been stimulated. There is also anecdotal evidence for illegal waste disposal. In Denmark, a 225% rise in the landfill tax in 1990 shows a 15% reduction in waste deliveries, demonstrating a very low elasticity (Sedee et al. 2000). Given the low elasticity, the ability of the tax to divert waste will be small, but the revenues earned may be substantial and could be earmarked for sustainable waste management programs.

[8] This is based on marginal avoided social waste disposal cost estimate of $30 to $33 per ton of waste disposed of at landfill

Empirical analyses on the impacts of landfill or incinerator taxes are few and far between. A study on the effects of hazardous waste taxes on waste generation and disposal is provided by Sigman (1996). Using plant-level data from U.S. EPA's 1987-1990 Toxic Release Inventories, she examines the impact of variation in state taxes on chlorinated solvent waste from metal cleaning. The econometric analysis suggests that firms' generation of chlorinated solvent waste is very sensitive to waste management costs but that due to the existing low level of taxes, the effect on waste generation is small. The analysis also suggests that high taxes on disposal encourage generators to choose treatment over land disposal.

The potential effects of landfill taxes may be discerned from examining tipping fees. Strathman, Rufolo, and Mildner (1995) estimate the elasticity of demand for landfill disposal of municipal solid waste. In estimating demand for solid waste services, they distinguish between point of generation and point of disposal. They take the latter approach using information on tipping fees and the quantity of waste that is landfilled. Using data from the Portland, Oregon metropolitan area, they specify tons of landfilled waste per thousand residents as a function of tipping fees, average weekly income of manufacturing workers (as a proxy for income in the region), and construction employment (proxy for local business cycle). They find that a 10 % increase in the tipping fee decreases garbage disposal at the landfill by 1.1 % - though costs may not have been passed on to households.

Interestingly, Kinnaman and Fullerton (2000) also find that higher landfill tipping fees in the U.S. increase the likelihood of implementing a recycling program. Specifically, a $1 increase in the tipping fee (from the average tipping fee of $26) increases the likelihood by 0.78%. A more recent study however does not find the tipping fee to be statistically significant (Kinnaman, 2005).

Another strand of literature on waste models optimal tipping fees for landfills. It is possible to model optimal tipping fees based on Hotellings rule if landfill space is characterised as a depletable resource that should be used efficiently over time.

Harold Hotelling (1931) examined the rate at which an exhaustible resource should be depleted and postulated that in continuous time, the rate of change of royalty must equal the social discount rate for there to be optimal depletion of a natural resource. It is thus possible to examine the optimal time path of extraction of landfill space, given a backstop technology of either incineration or recycling.

The earliest study involving tipping fees is probably by Berkman and Dunbar (1987) who discuss the effects of underpricing of landfills when the tipping fees fail to cover the full costs of disposal (i.e., the opportunity cost of land, the depletion of older landfills, and potential environmental damage). Tipping fees should increase over time as marginal costs rise with increases in the annual waste stream.

In a more formal paper, Ready and Ready (1995) model the waste reduction decision in two different ways. First they consider the problem of optimal waste reduction by waste generators and haulers. An increase in the tipping fee induces more waste reduction, resulting in a decrease in the flow of waste into the landfill. They find that the optimal tipping fee equals the variable cost of handling the waste plus a user fee that reflects the scarcity of the landfill, whereby the fee grows at the real interest rate. The optimal tipping fee for a regional landfill is based on the problem of optimal pricing of depletable resource with an added component that the depletable resource can be replaced at some cost, i.e., when a landfill becomes full, a new landfill can be constructed. They find that after a landfill is depleted and a new one is built, the optimal price falls. They also consider the regional government's problem of whether and when to invest in a large-scale waste reduction technology such as kerbside collection of recyclables, centralised composting of yard waste, or waste sorting facility. Using data from a Michigan landfill study, they are able to estimate optimal pricing policies and compare these to the best constant and break even price. Interestingly, they find that the optimal prices fall well below the break-even price of \$23.35 suggesting that "many municipalities may be overpricing their landfill space, which could mean too much effort is spent on reducing the volume of waste flowing to the landfill" (p. 316).

In Huhtala (1997), additional variables are included that are under the planner's control. Huhtala uses recycling efforts as an upper bound on the costs of using a landfill and includes set-up costs. The need for landfill space is implicitly determined given the amount of waste generated and the costs and constraints on recycling. Optimal recycling and landfill disposal paths over time are derived in a theoretical model. A simulation is then undertaken of an optimal waste management plan using data from the Helsinki region in Finland. She finds that the optimal recycling rate lies in the range of 31-51% under different scenarios suggesting that the existing mandates for achieving 50% recycling in municipalities are not unrealistic and are both economically and environmentally justified.

Interestingly, Aadland and Caplan (2004) conduct a study to examine the social net benefits of recycling. The benefits are estimated using 4000 household surveys from across 40 western U.S. cities, and costs are obtained from previous U.S. Environmental Protection Agency studies and interviews. They find that the estimated mean social net benefit of kerbside recycling is almost exactly zero. In contrast however, another study estimating WTP for large-scale recycling and incineration in Finland finds that recycling is the preferred method of waste disposal, and that the benefits of recycling exceed the costs (Huhtala, 1999).

And finally, Kinnaman (2005) finds that these preferences or local tastes for recycling have a significant impact on the probability that a municipality will adopt a recycling program. The second important contribution is that of state legislature and policies. Recycling mandates increase the population with access to recycling programs by roughly 10 % and therefore the recycling rate by roughly 2 %. In contrast, state recycling goals, bans on materials from landfills and subsidies for industries that recycle materials have no statistically significant impact on the availability of recycling programs.

2.5 Conclusions and Gaps in the Literature

Attention to waste management issues from both policymakers and academics has increased substantially over the past two decades and the solid waste collection and disposal industry has undergone significant changes. During this period, a number of policy instruments have been implemented at national and community levels to more adequately address the inefficient generation and disposal of MSW. The extent to which some of these programs produce positive net benefits is debated.

Some of the economic predictions have been confirmed by empirical work: Higher incomes are found to increase waste for disposal and, to a lesser degree, a higher price per unit of garbage is found to reduce demand for waste services, though the availability of kerbside recycling is also significant. Gaps in the literature on waste remain. Several of these are identified below.

(a) Given the most recently available data, what are the current trends in MSW generation and disposal today? Is there a decoupling with economic growth or is waste generation likely to continue to be an issue of growing concern? Though there have been a number of studies that have empirically examined the determinants of waste generation and recycling rates at the household or community level, there is a distinct lack of available literature examining this at the international level. Indeed, Beede and Bloom (1995) and Johnstone and Labonne (2004) provide the only two examples that examine the determinants of waste generation. It is therefore of interest to extend these studies, and to also examine the determinants of waste disposal and recycling at the macroeconomic level.

(b) Though there is some evidence indicating that households' behaviour with regard to recycling is influenced by the behaviour of their neighbours (Gamba and Oskamp, 1994; Werner and Makela, 1998), there have been no studies to date to test this type of hypothesis at the national level, i.e., to assess for the possibility of so-called spatial interaction between countries with regard to their waste management performance and policy-making. There is today a small but rapidly expanding literature that evaluates the degree of strategic behaviour in policy-making across regions and countries. This has focused primarily on income tax policies, and to a lesser degree on environmental policy stringency. However, there have been no applications of this approach (called spatial econometrics) to the case of waste management and landfill taxes.

(c) Though studies have examined WTP for recycling, these have focused primarily on U.S. data. The existing studies use contingent valuation and contingent ranking techniques but no study to date has employed the choice experiment method to investigate household preferences for recycling and composting. Specific countries such as the U.K. and Greece could benefit from these studies greatly, given their low recycling performance, and the fact that they are required to increase these substantially in the near future.

CHAPTER 3

The Determinants of MSW Generation: An Analysis of OECD Inter-Country Differences

3.1 Introduction

As trends in municipal solid waste (MSW) generation continue to increase, policy-makers proceed to grapple with the issue of increasing landfill scarcity for landfill developments in certain regions, the public opposition associated with the 'not-in-my-backyard' (NIMBY) phenomenon in relation to landfill and incinerator siting, as well as global externalities contributing to climate change from landfill emissions. These phenomena raise important public policy issues such as environmental justice and intragenerational and intergenerational equity.

An understanding of the driving forces of waste generation is important in determining the circumstances under which these are likely to change. This has direct policy implications for identifying the role of public policy and government intervention in promoting more sustainable MSW management and for the choice and implementation of different policy instruments.

As awareness of these issues have increased in the public and political arena, so too has the theoretical and empirical literature devoted to efficient and sustainable solid waste management and policy. Most studies have focused on either theoretical models or empirical analyses at the household or community level. Very few studies however examine municipal solid waste (MSW) generation at the country, or macroeconomic, level. Using cross-sectional time-series data from OECD countries over the period 1980-2000, the purpose of this chapter is to add to this scant literature by examining the determinants of MSW generation and to assess the policy implications of these inter-country differences.

The chapter begins by exploring one of the most dominant themes in the economy-environment debate of the 1990's, namely the so-called environmental Kuznet curve (EKC). The EKC refers to the relationship between income per capita and environmental quality where, in the initial stages of development as economic activity increases, environmental quality deteriorates. Eventually, continued development leads to improvements in environmental quality – hence the inverted U shape, similar to the Kuznet curve for economic development and income inequality. This relationship is tested for in the context of MSW generation.

Though some of the earlier macroeconomic studies on waste generation emerged as a result of the debate on the EKC curve (Shafik et al. 1992; Cole et al. 1997; Lim, 1997), very little empirical analysis has been conducted to examine how additional variables may affect MSW generation at the macroeconomic level. Demographic and policy factors that may influence MSW per capita generation rates include population density, geographic location, household size, waste legislation, public attitudes, source reduction and recycling initiatives, and the frequency of garbage collection (Reinhart, 2004). Two exceptions are those by Beede and Bloom (1995) and Johnstone and Labonne (2004) and are reviewed below. The chapter therefore proceeds by examining the effects of additional economic, demographic and policy variables in MSW generation.

This chapter is organised as follows: Section 3.2 presents an overview of the economic growth and environment debate, and reviews the existing macroeconomic literature on the determinants of MSW generation. In section 3.3, the methods used for the analysis are presented along with a description of the data, and the results are presented. Conclusions and policy implications are discussed in section 3.4.

3.2 Economic Growth and the Environment

The EKC has received much attention in the literature over the past decade and a half. The renewed interest between economic growth and the implications this has for environmental quality initiated, in part, as a result of the debate on trade liberalisation. Environmental groups have argued that the expansion of markets and economic activity leads to increased pollution levels and a faster depletion of scarce natural resources. The existence of the EKC relationship was first suggested by Grossman and Kruger (1991) who examined sulphur dioxide and "smoke" concentrations and their relationship with economic growth, using a cross-country sample of comparable measures of pollution in various urban areas. They found an inverted U shape for sulphur dioxide and dark matter suspended in the air, and a monotonically decreasing curve for the mass of suspended particles in a given volume of air.

Since then, a number of empirical studies have examined the relationship between environmental quality (either through levels of emissions or ambient concentrations in the air) and economic growth. The environmental pollutants and natural resources that have been studied to date include: Sulphur dioxide, total suspended particles (TSPs), nitrogen oxides (NO_x), carbon monoxide, carbon dioxide, methane, CFC emissions, automotive lead emissions, rates of deforestation, drinking water, urban sanitation, as well as the state of oxygen regime, fecal contamination, and contamination by heavy metals of river basins (i.e., measures for river quality), and finally, municipal waste. These studies typically employ panel data and regress a flexible functional form of income per capita on the measure of environmental quality:

$$E_{it} = \alpha + \beta_1 Y_{it} + \beta_2 Y^2_{it} + \beta_3 Y^3_{it} + \beta_4 t + \beta_5 V_{it} + \varepsilon_{it} \qquad (1)$$

where E is emissions, α is a scalar, Y is income, t is a time trend to account for technological change, V reflects other explanatory variables, and ε is a stochastic error term. The subscripts *i* and *t* denote a country and a time index respectively.

Unlike structural models, these reduced-form models do not require *a priori* information on numerous parameters and enable the influence of income on environmental quality to be directly estimated. Reduced-forms however do not provide information on the underlying causes of changes in environmental quality (i.e., whether reductions in pollution levels are achieved due to stricter environmental regulations or due to autonomous structural and technological changes), and are therefore not well-suited for policy analysis. In an attempt to obtain a better understanding of the underlying factors of the EKC, several authors have included explanatory variables such as trade-related measures (Suri and Chapman, 1998) and population density, as well as policy variables such as 'contract enforceability' to proxy for quality of institutions (Panayotou, 1997) or GINI coefficients to proxy for inequality and power (Torras and Boyce, 1998). Other studies have formulated structural models that disaggregate the growth-environment relationship into a scale effect, a structural or compositional effect, and an abatement effect (de Bruyn, 1997; Antweiler et al. 1998). Potential explanations that have been offered for the existence of an EKC can be categorised under behavioural changes and preferences, institutional changes, technological and organisational changes, and international relocation of consumption and production (de Bruyn and Heintz, 1999). Theoretical models include those by Lopez (1994), the adaptation of the Forster model to the EKC by Selden and Song (1995), a trade and environment model by Copeland and Taylor (1999), a growth model by Chaudhari and Pfaff (1999), and a consumption-based model by Gawande et al. (2001), among others.

Just as the causes underlying the EKC have remained largely undetermined, the empirical findings of panel data studies have been somewhat ambiguous. The general consensus is that the EKC exists for local air pollutants, while more global or indirect impacts tend to increase monotonically with income (Cole et al. 1997; Ekins, 1997).

In general, the use of different data sets, functional forms (e.g. logarithmic vs. levels), and estimation methods can lead to very different results (Ekins, 1997; Cole, 2003). There has also been econometric criticism of the EKC, namely that studies ignore the issue of heteroskedasticity which is likely to be present in cross-section data and that studies that use only OECD data may estimate turning points at lower per capita income levels than those using data from the world as a whole. Furthermore, most EKC studies estimate a quadratic relationship between pollution and income and therefore fail to allow for the possibility of emissions beginning to increase again at high income levels (Cole, 2003). More recently therefore, emphasis and analysis has been placed on a more systematic and rigorous application of econometric models and the econometric techniques applied to test for the EKC have thus developed in sophistication and consistency (Stern, 2003).

Despite the fact that some of the earliest studies on waste generation emerged as a result of the debate on the EKC, only a very limited number of studies have in fact explicitly examined the existence of an EKC for MSW. These are by Shafik et al. (1992), Cole et al. (1997), and Lim (1997) (see Table 3.1 for a summary of results). Using city level information for 39 countries compiled for the year 1985, Shafik et al. (1992) find that municipal waste per capita unambiguously rises with increasing GDP. The log linear specification performed best. Cole et al. (1997) use data from 13 OECD countries over the period 1975-1990 and adopt generalized least squares (GLS) to estimate the relationship between municipal waste per capita and income[1]. They also find that waste increases monotonically throughout the observed income range. In Lim (1997), time series data is used from South Korea over an 11-year period. Wastes are divided in two categories, domestic and industrial. With regard to the daily disposal of domestic wastes, the estimated result follows the inverted-U shape curve, and the regressions show that the double-log and quadratic specification has the strongest explanatory power. In contrast, industrial waste increases

[1] GLS was undertaken following Kmenta (1986), *Elements of Econometrics*, London: Collier Macmillan., to account for heteroskedasticity and autocorrelation. The Hausman test statistic indicated that the fixed effects estimation is favoured to random effects. The data on income is from Penn World Tables Mark 5.6; data on waste is from the OECD Environment Data Compendium 1995.

unambiguously with rising per capita GDP. Note that none of these studies include a time-effect in their analysis to proxy for technological development[2].

Table 3.1 Previous Empirical EKC Results on MSW

	Constant	Y	Y^2	Y^3	Adj R^2	Obs
Shafik et al.1992 Cross-sectional						
Log linear	2.41 (5.51)	0.38 (7.69)	-	-	0.6	39
Quadratic	11.02 (2.50)	-1.7 (-1.60)	0.13 (1.96)	-	0.63	39
Cubic	-33.96 (-0.99)	15.08 (1.08)	-1.95 (-1.10)	0.08 (1.17)	0.64	39
Cole et al.1997 Panel data						
Logs quadratic	-	-17.46 (-5.50)	0.96 (5.69)	-	0.93 Buse	52
Levels quadratic	-	-35.04 (-5.40)	0.0022 (8.89)	-	0.99 Buse	52
Lim 1997 Time series						
Linear-Log	-751.4 (-2.342)	101.51 (2.342)	-3.420 (-2.336)	-	0.603	11
Double-log	-531.57 (-3.541)	71.694 (3.535)	-2.415 (-3.525)	-	0.552	11
Linear-Log	43786 (2.944)	-8927 (-2.963)	606.57 (2.982)	-13.735 (-3.001)	0.746	11
Double-log	25443 (3.182)	-5190.3 (-3.204)	352.87 (3.226)	-7.995 (-3.248)	0.796	11

t-statistics in parenthesis

The studies on municipal solid waste generation thus consist of cross-sectional, panel, and time series approaches across developed and developing countries. Only the results of Lim (1997) exhibit an EKC relationship, though MSW in absolute levels rather than per capita levels are used. Shafik et al. (1992) conclude that because solid waste disposal can be transformed into a localised problem, particularly in areas that are not densely populated or are low-income communities, higher incomes are not associated with reductions in waste generation. Cole et al. (1997) who also observe monotonic increases in waste generation throughout the observed income range suggest that the lack of an EKC is because municipal waste only indirectly harms the

[2] Another study by De Groot et al. (2001) examines *industrial* solid waste generation. Using panel data from thirty regions in China over the period 1982-1997, they examine the relationship between gross regional product (GRP) and solid waste in levels, in per capita terms, and in per unit of GRP. They find

environment by representing an increased use of resources and generating methane when disposed of at landfill sites, which are global air pollutants and therefore do not create sufficient incentives to reduce emissions unilaterally.

Recently available data from the OECD (2002) however suggests that there has been a relative decoupling[3] of waste going to final disposal from private final consumption (PFC) since 1995. Moreover, nine OECD Europe countries (Austria, Belgium, Denmark, Germany, Italy, Luxembourg, Netherlands, Norway and Switzerland) recorded a significant absolute decoupling[4] with the amounts of waste going to final disposal. Given this information, a current re-examination of the EKC for municipal solid waste generation seems timely and warranted.

Albeit providing a useful first step towards answering the question of how economic growth affects the environment, the EKC has on occasion been referred to as a 'black box' in that it does not provide any information on the method in which the income-environment relationship works (Panayotou, 1997). Moreover, though there are a number of studies that examine the determinants of MSW generation using household-level or community-level data within a single region or country[5], very few studies have been undertaken to examine the determinants of MSW generation at the macroeconomic level. One exception is a study by Beede and Bloom (1995) who investigate the relative importance of growth in real per capita income and population in MSW generation rates. Using data from a cross-section of 36 countries they find that income elasticity is 0.34 and that population elasticity is 1.04. They also conduct time-series analysis for the U.S. (1970-1988) and Taiwan (1980-1991) and find that income elasticity is 0.86 and 0.59 respectively, and that population elasticity is 0.63 and 1.63 (not statistically significant). The second exception is by Johnstone and Labonne (2004) who apply a model based on household utility maximisation proposed by Kinnaman and Fullerton (1997). Each household is assumed to derive

an N-shaped curve for the first, statistically insignificant coefficients on the second, and a statistically significant but negative relationship for the third variable.

[3] Relative decoupling when MSW increases at a lower rate than GDP

[4] Absolute decoupling when MSW decreases as the GDP rises.

utility from a single aggregate consumption good and household municipal solid waste collection services whereby the use of household MSW collection services is considered to be dependent upon a vector of demographic characteristics such as household size and the degree of urbanisation. The demand for the use of MSW services is therefore a function of these variables as well as the cost of provision of such services which depend on the density of residential development. As such, they regress household solid waste generation on final consumption expenditures per capita, the degree of urbanisation, population density, and the percentage of children in the population. They find that household MSW generation rates are relatively inelastic with respect to household final consumption expenditures (0.15 – 0.69), that population density and more ambiguously the degree of urbanisation have a positive effect on MSW generation, and finally the proportion of children has a significant and negative influence on MSW generation.

This type of analysis can help to assess the relative importance of a number of potentially significant factors that have an impact on the rate of MSW generations and can provide insights into which, if any, of these can be influenced by government policy.

3.3 The Determinants of MSW Generation: Methods and Results

3.3.1 Econometric Methods and Description of the Data

The data set used is a combination of cross-sectional and time-series data, suggesting the appropriateness of a panel data analysis. Panel data analysis has the advantage of improving the reliability of the estimates and can control for individual heterogeneity and unobservable or missing values (Baltagi, 2003). As before, denoting the cross-section dimension i where $i = 1, \ldots, N$ and the time-series dimension t, where $t = 1, \ldots, T$ the model is the following:

[5] Refer to Chapter 2 for a review of these.

$$y_{it} = \alpha + \beta' x_{it} + \varepsilon_{it} \tag{2}$$

$$\varepsilon_{it} = \mu_i + v_{it} \tag{3}$$

The term μ_i denotes the unobservable individual-specific time-invariant effect that takes account of any individual specific effect not in the regression. The v_{it} denotes the disturbance. Assume that μ_i are fixed parameters to be estimated and the remainder disturbance is stochastic with v_{it} independently and identically distributed, iid $(0, \sigma_v^2)$. If the set of regressors x_{it} are assumed to be independent from the v_{it} for all i and t, then Fixed Effects (FE) regression is the appropriate model specification, in which ordinary least squares (OLS) is applied to:

$$y_{it} - \bar{y}_i = \left(X_{it} - \bar{X}_i\right)\beta + \left(v_{it} - \bar{v}_i\right)$$

where for instance:

$$\bar{y}_i = \frac{1}{T}\sum_{t=1}^{T} y_{it}$$

The fact that the FE estimator can be interpreted as a simple OLS regression of means-differenced variables explains why this estimator is often referred to as the within-groups estimator. That is, it only uses the variation *within* an individual's set of observations. Random Effects (RE) assumes μ_i is not correlated with the regressors and is a (matrix) weighted average of the estimates produced by the between and within estimators. It applies generalised least squares (GLS) to estimate the coefficients (Wooldridge, 2002; Hsiao, 2003; Baltagi, 2005).

The generally accepted way of choosing between fixed and random effects is running a Hausman test (1978). Statistically, fixed effects are always reasonable with panel data as they always give consistent results. However, they may not be the most efficient model to run. Random effects will provide better P-values as they are a more efficient estimator, so RE should be run if it is statistically justifiable to do so. The Hausman test checks a more efficient model against a less efficient but consistent model to make sure that the more efficient model also gives consistent results. The Hausman test tests the null hypothesis that the coefficients estimated by the efficient random effects estimator are the same as the ones estimated by the consistent fixed effects estimator. If they are (insignificant P-value, Prob>χ^2 larger than 0.05) then it is safe to use random effects. If the P-value is significant however, fixed effects should be used.

The description and sources of the data used in the EKC analysis as well as the subsequent analysis where additional variables are included are described in Table 3.2 below. The analysis is restricted to OECD countries because good quality internationally comparable data on municipal solid waste generation is not available.

Table 3.2 Description and Sources of the Data

MWPC	Waste generated per capita (municipal and household). 1980-2000 in 5 year intervals. Source: OECD Environmental Data Compendium 2002.
GDPPC	Gross Domestic Product (GDP) per capita, in 1995 prices and purchasing power parities (PPP) in U.S. dollars. 1980-2004. Source: World Development Indicators, 2004
POPD	Population density, defined as people per square kilometre. 1980-2004 Source: World Development Indicators, 2004.
URB	Urban population, defined as percentage of total. 1980-2004 Source: World Development Indicators, 2004.
POLDX	Waste legislation and policy index. 1995 Source: Adapted from Guerin et al. 2001; European Environment Agency 1998.

Due to some missing observations, the data constitutes an unbalanced panel. The analysis is restricted to OECD countries because of the existence of higher quality waste data for these countries and the fact that definitions and survey methods for MSW and data collection vary more substantially if non-OECD countries are included. The OECD dataset is more consistent and reflects existing and ongoing work on waste classification at the international level[6].

[6] Furthermore, studies have found that single global EKC models may be a misspecification (Islam 1997, List and Gallet 1999, Stern and Common, 2001).

The OECD defines municipal waste as waste that is collected by or on the order of municipalities. It includes waste originating from households, commercial activities, office buildings, institutions such as school and government buildings, and small business that dispose of waste at the same facilities used for municipally collected waste. Note that there are differences between waste classification used by different countries. The waste legislation and policy index (POLDX) assigns scores based on national government policy to implement *inter alia* waste management plans, packaging eco-taxes, producer responsibility, prevention, and recovery/recycling programs, and whether the government has ratified the Basel Convention on the control of transboundary movements of hazardous wastes. Due to limited data availability, this is an aggregated index with a single score for each of the countries. This implies that though the index can provide an indication of the effects of waste policies on waste management performance *in general*, it will not be possible to discern the *individual effects* of these policies on waste management performance levels[7]. The full list of countries with further information on the waste legislation and policy index is reported in Appendix 3.1. The descriptive statistics of the data are reported in Appendix 3.2. The correlation matrix of all variables indicates that none of the independent variables are highly correlated.

For each of the variables, total variation is decomposed into between and within class variation and an F test is conducted to test the hypothesis that between classes variation is large relative to within class variation. For all of the variables, Prob > F = 0.000, indicating that between class variation is large relative to within (Table 3.3).

[7] Note that despite this weakness in using aggregate indices, there are a number of examples in the literature that do indeed use them. These include Guerin et al. (2002), Dasgupta et al. (1995), Eliste and Fredriksson (2002), and Pellegrini and Gerlach (2005), amongst others.

Table 3.3. Analysis of Variance for the Data

Variable	Between	Within	Total	F test
GDPPC	3.074e+10	1.000e+10	4.074e+10	76.32
POPD	11168657	47867.975	11216525	5792.82
URB	114956.95	6401.52	121358.47	445.85
MWPC	1511445	497260	2008705.5	8.79
POLDX	2420	0	2420	-

3.3.2 Investigating Income Per Capita and MSW Generation

To examine the relationship between income per capita and MSW generation, both quadratic and cubic functional forms are postulated in both levels and logs for comparative purposes. All econometric analysis is undertaken using the statistical software package STATA 8.0. The models thus take the following form:

$$MWPC_{it} = \alpha + \beta_1 GDPPC_{it} + \beta_2 GDPPC_{it}^2 \; (+ \beta_3 GDPPC_{it}^3) + \varepsilon_{it} \qquad (5)$$

$$\log MWPC_{it} = \alpha + \beta_1 \log GDPPC_{it} + \beta_2 \log GDPPC_{it}^2 \; (+ \beta_3 \log GDPPC_{it}^3) + \varepsilon_{it} \qquad (6)$$

The estimates from equations 5 and 6 using fixed effects (FE) and random effects (RE) are presented in Table 3.4 below.

Table 3.4 EKC for MSW Generation.

	Constant	GDPPC [a]	$GDPPC^2$	$GDPPC^3$	R^2	Obs/ Groups[b]
Logs Quadratic RE	23.08 (4.00) ***	-4.091 (-3.39) ***	0.241 (3.80) ***	-	W/n = 0.455 B/w = 0.480 O/a = 0.513	110 29
Logs Quadratic FE	21.01 (3.23) ***	-3.701 (-2.74) ***	0.222 (3.17) ***	-	W/n = 0.456 B/w = 0.477 O/a = 0.506	110 29
Levels Quadratic RE	145.57 (3.15) ***	0.021 (5.79) ***	-1.61e-07 (-2.40) ***	-	W/n = 0.519 B/w = 0.438 O/a = 0.497	110 29
Levels Quadratic FE	101.534 (1.93) ***	0.024 (5.33) ***	-1.94e-07 (-2.53) ***	-	W/n = 0.519 B/w = 0.439 O/a = 0.496	110 29
Logs Cubic RE	168.748 (2.78) ***	-50.184 (-2.62) ***	5.086 (2.53) ***	-0.169 (-2.78) ***	W/n = 0.490 B/w = 0.506 O/a = 0.539	110 29
Logs	179.77	-53.961	5.505	-0.184	W/n = 0.493	110

Cubic FE	(2.69) ***	(-2.54) ***	(2.47) ***	(-2.37) ***	B/w = 0.504 O/a = 0.532	29
Levels Cubic RE	352.73 (3.39) ***	-0.013 (-1.34)	1.32e-06 (3.39) ***	-1.80e-11 (-3.86) ***	W/n = 0.586 B/w = 0.453 O/a = 0.565	110 29
Levels Cubic FE	327.39 (4.06) ***	-0.012 (-1.07)	1.29e-06 (3.03) ***	-1.77e-11 (-3.54) ***	W/n = 0.586 B/w = 0.453 O/a = 0.563	110 29

*** significant at the 1% level, ** 5% level, * 10% level
Notes: [a] The minimum and maximum levels of per capita income are $3,657 and $51,637, respectively. [b] MWPC data not available for New Zealand.

The nature of the relationships are determined by the signs of the coefficients. In the quadratic case, if $\beta_1 > 0$, $\beta_2 < 0$, and $\beta_3 = 0$, this implies the inverted U shape of the EKC. In the cubic case, if $\beta_1 > 0$, $\beta_2 < 0$, and $\beta_3 > 0$, the relationship is N-shaped, implying that there is a first increasing, then declining but finally again increasing relationship between emissions and per capita income[8]. Alternatively, if $\beta_1 < 0$ $\beta_2 > 0$ and $\beta_3 < 0$ then there is a sideway-mirror-S-shape (Ekins, 1997).

As can be seen from Table 3.4, the results are somewhat ambiguous. For the logs quadratic, the relationship is monotonically increasing whereas for the levels quadratic there seems to be an EKC. In comparison, the results in Cole et al. (1997) indicate consistent signs for both the log and levels quadratic case. Quadratic in levels imposes a symmetric shape on the EKC however. Stern (1998; 2004) argues that logarithmic specification is preferred. The log-linear specification implies non-negativity restrictions upon the variables which the linear model does not. The restriction that emissions cannot be negative is not unreasonable in this case. For the cubic form, the cubic terms are always statistically significant and in both levels and logs the signs are consistent, indicating a sideway-mirror-S-shape[9]. This implies two turning points (as opposed to one turning point in the inverted U curve). Overall, the measure of goodness of fit, R^2, is adequate, and the log-linear specification performs

[8] Ekins (1997) points out that the N shape holds only if the absolute values of $\beta3<\beta2<\beta1$.
[9] Household final consumption (HFC) was also used as the explanatory variable in order to test for decoupling as this is where some trend seems to be apparent (OECD, ibid). Data on private final consumption was not available, therefore household final consumption is used instead. These regressions did not exhibit an EKC and are not reported here.

better than levels in the quadratic model whereas the levels performs better than the log-linear specification in the cubic model.

To test whether the random effects or the fixed effects model is preferred the Hausman statistic is used which is a test of H_0: that random effects is consistent and efficient, versus H_1: that random effects is inconsistent. (Note that fixed effects would certainly be consistent.) The result of the test is a vector of dimension k (dim(β)) which will be distributed $\chi^2(k)$. If the Hausman test statistic is larger than the critical χ^2 (k), one must use FE; if the statistic is smaller, one can use RE. The preferred regression model varies according to the model used[10]. In the logs quadratic case, the RE model is preferred [crit. χ^2 (2) = 5.99 > 0.79], whereas in the logs cubic case, the FE model is preferred [crit. χ^2 (2) = 5.99 < 19.28] (Greene, 2000. p. 577).

Diagnostic tests are conducted to test for the presence of heteroskedasticity and serial correlation, indicating the presence of both of these in the data. The χ^2 (1) test in a Breusch-Pagan / Cook-Weisberg test is equal to 6.27 with Prob > χ^2 = 0.0123. For panel data, the Wooldridge test for serial correlation can be used which is a Wald test of no first-order serial correlation (Wooldridge, 2002). The test indicates the presence of autocorrelation with F(1, 19) = 26.215, Prob > F =0.0001. (Note that in this particular case, because the MWPC data is in 5-year intervals and therefore not consecutive, the interpretation of the test is not identical).

Unfortunately, it is not possible to correct for both of these problems in the presence of a fixed effects model with a large number of missing observations. STATA does allow for estimation of panel data with missing variables, where the error terms are heteroskedastic and/or serially correlated in the generalized least-squares procedure. More specifically, it assumes that the error terms across panels are heteroskedastic

[10] STATA 8.0 suggested scaling the variables when conducting the Hausman test on the levels specification for both the quadratic and cubic case and that the results of the Hausman test may otherwise be misleading. These test results are therefore not reported.

and that there is uniform AR(1) serial correlation within the individual panels (STATA, 2003). The results are presented in Table 3.5.

Table 3.5 Feasible Generalized Least-Squares Estimates of MSW Generation

	Co-efficient	Std. Error	Z	P>|z|
GDPPC	-4.895	1.226	-3.99	0.000
GDPPC2	0.283	0.064	4.41	0.000
Constant	26.854	5.843	4.60	0.000
Wald $\chi^2(2) = 143.60$, Prob > χ^2 =0.000 Log likelihood = 74.6126				
	Co-efficient	Std. Error	Z	P>|z|
GDPPC	-65.264	20.609	-3.17	0.002
GDPPC2	6.648	2.154	3.09	0.002
GDPPC3	-0.223	0.075	-2.98	0.003
Constant	217.168	65.621	3.31	0.001
Wald $\chi^2(3) = 183.25$, Prob > χ^2 =0.000 Log likelihood = 73.3248				

The results from the FGLS procedure are similar to those above. The cubic term remains statistically significant but the relationship between income and per capita municipal waste generation is monotonically increasing over the observed income range[11]. Hence, as a first investigation of the trends in MSW generation, the results from the RE, FE, and FGLS models do not provide any robust evidence for the existence of an EKC relationship between income and waste levels in the 29 OECD countries for which data is available. These results therefore conform to earlier findings of the 1990's, and more recently available data does not seem to make a difference.

These results are in line with alternative viewpoints regarding the nature of emissions and the income relationship as illustrated in Figure 1.

[11] The turning point for the quadratic expression is (x= 8.648, y= 5.668). The turning points for the cubic expressions are: (x= 8.86, y= 6.45) and (x= 10.9, y= 6.79).

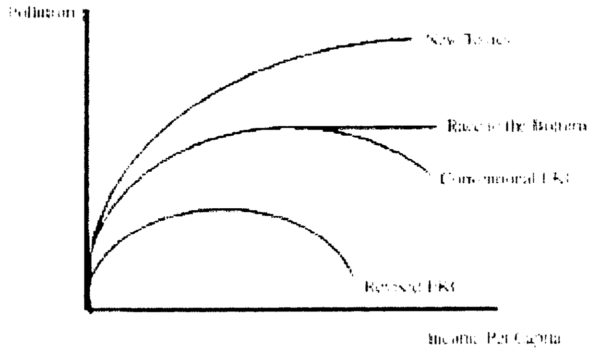

Figure 3.1. EKC: Different Scenarios.
Source: Dasgupta et al. (2002)

Stern (2004) neatly summarises these alternative viewpoints:

> *'The "new toxics" scenario claims that while some traditional pollutants might have an inverted U-shape curve, the new pollutants that are replacing them do not. As the older pollutants are cleaned up, new ones emerge, so that overall environmental impact is not reduced. The "race to the bottom" scenario posits that emissions were reduced in developed countries by outsourcing dirty production to developing countries, which will find it harder to reduce emissions. Moreover, the pressure of globalisation may also preclude further tightening of environmental regulation in developed countries and may even result in it's loosening in the name of competitiveness'.*

The issue of the "race to the bottom" in environmental waste policy is examined in more detail in Chapter 5 of this thesis. Finally the revised EKC scenario suggests that the U-shape curve is shifting downwards and to the left over time due to technological change. As stated earlier, the EKC has been criticised because it does not provide any information on the method in which the income-environment relationship works (Panayotou, 1997). As such, the next section examines how additional economic, demographic and policy variables may affect MSW generation.

3.3.3 A Macroeconomic Analysis of the Determinants of MSW Generation

In contrast to Johnstone and Labonne, 2004, the analysis here examines *total* MSW generation, rather than household solid waste generation[12]. This provides a more comprehensive and comparable picture for the subsequent analysis in Chapter 4 when MSW disposal is examined (defined as the waste disposed of at landfills as a percent of total MSW), rather than just household solid waste. Similarly, the percentage of paper/cardboard and glass recycled is from apparent consumption from *all* MSW sources, not just household sources of waste.

It is considered that the use of total MSW collection services is considered to be dependent upon a vector of economic, demographic as well as policy characteristics of a country such as per capita income levels, the degree of urbanisation, the cost of provision of such services, and the public policies that have been introduced to address waste generation. Waste generation levels are expected to rise with increases in per capita income levels. The variable URB is expected to be important, as residents in rural areas for example may be more likely to grow and prepare their own food, thus reducing the generation of packaging waste. Some evidence to this effect is apparent at least in the U.S. (U.S.EPA, 1994). As such, URB is anticipated to be positive. The anticipated sign on the demographic variable POPD is somewhat ambiguous. Some argue that it is likely to be positive because there are significant economies of scale in the provision of waste collection services, thus reducing the cost of service provision (Johnstone and Labonne, 2004). In contrast, others argue that high population densities imply scarce land resources and thus more pressure to preserve land and environmental quality, and thus improve waste management (Matsunaga and Themelis, 2002). Finally, higher waste legislation and policy indices (POLDX) are expected to be inversely related to MWPC levels as higher indices indicate greater national commitments towards the sustainable management of the

[12] Household solid waste accounts for 60 percent of the total MSW generated.

generation and disposal of waste[13]. Thus, the regression model which attempts to explain inter-country differences in per capita MSW generation rates can be formulated as:

$$MWPC_{it} = \alpha + \beta_1 GDPPC_{it} + \beta_2 POPD_{it} + \beta_3 URB_{it} + \beta_4 POLDX_{it} + \varepsilon_{it} \qquad [4]$$

All of the variables are expressed in log form to allow for ease of interpretation of the coefficients as elasticities. Note that the argument related to POPD from Matsunaga and Themelis (2002) implies correlation with POLDX. As such, there is likely to be ambiguity in the interpretation of the results as it is difficult to disentangle what the different variables are capturing. Furthermore, the within class variation of POPD is very small indicating that there is not much variation across time. This implies that much of the same information is captured in the fixed effects. The results from the random effects and fixed effects models are reported in Table 3.6.

[13] Note that an important, but rarely addressed, issue affecting the empirical analysis of public policies is the potential endogeneity of the policies under study. Policy-making however would need to be responsive to economic and political considerations *within* the country for endogeneity to be an issue. As described in Appendix 3.2, POLDX represents the number of initiatives a country has taken to sustainably address waste management. Nearly all categories however are based on and reflect top-down *international* and *regional* mandates, such as the Basel Convention and other EU Directives (e.g. on packaging laws, recycling mandates, and the Landfill Directive) and thus endogeneity issues are likely to be less important. The usual test for endogeneity is a Wu-Hausman test but this requires the existence of appropriate instruments that are unfortunately not readily available in the waste context. Policy endogeneity, if present, can lead to biased estimates and hence one may not be able to draw firm conclusions regarding the importance of X on Y. It is said however that the majority of studies treat variation in policy variables as exogenous (Besley and Case, 2000).

Table 3.6. Parameter estimates for MSW Generation.

	Random Effects	Fixed Effects
	Co-efficient (Z value)	Co-efficient (t value)
GDPPC	0.422	0.4540
	(6.28)***	(4.31)***
POPD	-0.036	0.861
	(-1.43)	(0.21)
URB	0.477	0.557
	(2.76)***	(1.88)*
POLDX	-0.229	dropped
	(-1.65)*	
Constant	0.5769	-1.08
	(0.82)	(-0.80)
No. observations	110	110
No. groups	29	29
R-squared	Within = 0.4197	Within = 0.5266
	Between = 0.5783	Between = 0.5587
	Overall = 0.5069	Overall = 0.5262
	Wald $\chi^2(5)$=88.21	F(3,78)=18.88
	Prob>χ^2=0.000	Prob>F= 0.000
		F test that all u_i=0: F(28, 78) = 5.44
		Prob > F = 0.0000

*** significant at the 1% level, ** 5% level, * 10% level.

In both the RE and the FE models, GDP per capita and URB are positive and statistically significant indicating that higher income levels and the more urbanised a country is, the higher is the generation of MWPC. In both models, URB reveals a stronger influence than GDPPD. POPD is statistically insignificant in both the RE and FE models. It should be noted that the urbanisation rate and the population density are not highly correlated. In the sample, the correlation coefficient between the two is only –0.2422, thus it does not seem to be the case that a large part of the variation on POPD is explicable by the variation of URB. POLDX shows the intuitively correct negative sign but is statistically significant at the 10% level. Assessing the overall fit in panel data is undertaken by examining the overall R^2 for a random effects model and the within R^2 for a fixed effects model. The Hausman test statistic with $\chi^2(3)$ = 2.08, Prob>χ^2= 0.5568 suggests that the random effects regression is the appropriate model for this data. The R^2 of 0.51 suggests a relatively good fit. Diagnostic tests are conducted to test for heteroskedasticity and serial correlation in the data. The Breusch-Pagan / Cook-Weisberg test for heteroskedasticity with $\chi^2(1) = 12.14$, Prob > $\chi^2(1) = 0.0005$ which is greater than the critical $\chi^2(1) = 3.84$. The null hypothesis of homoskedasticity is therefore rejected. The Wooldridge test for autocorrelation with F(1, 19) = 15.071, indicates that there is autocorrelation in the data (Prob > F = 0.0010). The method of feasible generalized least squares (FGLS) is therefore used to estimate the model in STATA which allows estimation of panel data with missing variables and allows for the presence of AR(1) autocorrelation within panels, as well as heteroskedasticity across panels (STATA, 2003; Johnstone et al. 2004). The results are presented in Table 3.7.

Table 3.7. FGLS Estimates of MSW Generation

	Co-efficient	Std. Error	z	P>|z|
GDPPC	0.4356	0.0352	12.36	0.000
POPD	-0.0395	0.0067	-5.92	0.000
URB	0.4718	0.0645	7.31	0.000
POLDX	-0.1884	0.0387	-4.86	0.000
Constant	0.3739	0.3505	1.07	0.286
Wald χ^2 (4) = 458.92, Prob > χ^2 = 0.0000				
Log likelihood = 2.2594				

The results are fairly consistent with the RE and FE estimates above, with the exception that in the FGLS model, POPD is now statistically significant. This suggests that an increase in population density does have a significant downward impact on the amount of MWPC generated. All the other variables continue to exhibit the same signs on the coefficients. The positive and significant sign on URB is not encouraging as projections show that the share of total population living in urban areas will continue to grow in the future (WRI, 1996). In addition, the waste legislation and policy index, though negative, is statistically insignificant (P-value = 0.255) suggesting that the national commitments towards sustainable waste management have not had a major impact on reducing the amount of MSW generated[14].

[14] One caveat is that the variable POLDX does not vary over time and the coefficient may therefore not adequately capture all the relationship between MWPC generation. Moreover, POLDX is an aggregated proxy and therefore it is not possible to determine the incremental effects of each policy.

Conforming with the results of Johnstone and Labonne (2004), MWPC is indeed income-inelastic. The strongest impact on MWPC however is urbanization. Furthermore, in this model, the waste legislation and policy index is negative and statistically significant, indicating that greater national commitments to sustainable waste management have had an impact in reducing the amount of MWPC generation.

Neither of these models are ideal and the analysis is restricted by the capacity of STATA to address models in which the panel data is affected by heterskedasticity and autocorrelation. Though the FGLS model addresses this to an extent by making assumptions on the nature of the autocorrelation, the accepted norm is to report the range of estimates provided by the preferred RE/FE model and the FGLS model (see e.g., Johnstone et al.)

Finally, the interpretation of the POLDX may perhaps be ambiguous. For example, if policy stringency also captures enforcement, it may result in a reduction of the waste disposed of illegally and hence in an increase in the generation of waste. Furthermore, in many cases the incentives may not be transmitted back to the generators of waste.

3.4 Conclusions and Policy Implications

An examination the relationship between of the income per capita and MSW generation reveals that the results confirm with previous studies of this nature conducted in the 1990s. There does not seem to be strong evidence to suggest that MSW generation is initially increasing with rising incomes, and after attaining a turning point, that these levels begin to decline with continued economic growth.

The subsequent analysis examines the determinants of MSW generation rates in OECD countries in more detail. It provides evidence on the economic and demographic determinants of generation rates in municipal solid waste and has made a first attempt at including an important potential influence on municipal solid waste generation, namely that of public policy, as proxied by the waste legislation and policy index. From a policy perspective, the results are not particularly encouraging. Conforming with previous studies, the data indicate that per capita MSW generated is increasing monotonically with GDP per capita. Moreover in addition to GDP per capita, the degree of urbanisation also has a positive impact on the generation of municipal waste. This is discouraging given that projections show that the share of total population living in cities will grow at a fast rate in the future (WRI, 1996). There is however some evidence suggesting that national commitments towards sustainable waste management may have a positive effect in reducing the levels of MSW generation. A clearer understanding of the determinants of MSW generation is an important prerequisite for planning and implementing sustainable MSW policies. The results suggest that policy-makers may wish to focus their efforts in addressing waste generation levels in urbanized areas, to promote a shift in the structure of consumption and production so as to reduce the environmental impacts of waste. Greater emphasis needs to be placed on reducing the environmental and resource intensity that is linked to the consumption and production of different goods and services. Waste policies, as proxied by the waste legislation and policy index, seem to have had an impact on the rates of MSW generation across different countries. However, as noted earlier, results from the POLDX variable need to be interpreted with care given that this only provides a rough proxy for waste management policies. Further efforts for future research could focus on obtaining comparable and consistent data on individual waste management policies in OECD countries and their development over time.

Though household level studies are better suited to examining the specific waste policies that are more effective in encouraging a transition to sustainable waste management practices, the analysis conducted here yields some interesting insights into the determinants of MSW generation across OECD countries.

This analysis however does not provide any information on the way that the volume of waste is managed. As noted by Cole et al. (1997), the actual environmental impact of municipal solid waste is masked by the fact that the volume of municipal waste does not indicate just how much of that waste is recycled. Waste disposal management and its determinants is an area that has been negligibly addressed in the literature and can provide additional and important insights into the waste issue. This issue is examined in Chapter 4.

Appendix 3.1. The Waste Legislation and Policy Index

List of countries and the waste legislation and policy index (POLDX).

Country	**Waste policy index (11 point scale)**
Australia	8
Austria	10
Belgium	10
Canada	8
Czech Republic	*5*
Denmark	10
Finland	10
France	10
Germany	9.5
Greece	9
Hungary	*5*
Iceland	*6*
Italy	9
Ireland	8
Japan	8
Korea	7
Luxembourg	7
Mexico	6.5
Netherlands	10
New Zealand	7
Norway	*10*
Portugal	6
Poland	*5*
Slovak Republic	*5*
Spain	6
Sweden	9
Switzerland	*6*
Turkey	6
UK	9
USA	6

Source: Adapted from Guerin, Crete, Mercier, 2001
EEA Europe's Environment: The Second Assessment 1998. Luxembourg.

The waste legislation and policy index is computed by summing each country's scores based on their policy initiatives concerning different aspects of waste management. The scores are based on national government policy in 10 categories: 1) Waste management plans; 2) Priority to prevent and reduce waste harmfulness; 3) Waste eco-taxes; 4) Producer responsibility; 5) Prevention, 6) Recovery/recycling programs; 7) Hazardous waste reduction; 8) Ratification of the Basel Convention on the control of transboundary movements of hazardous wastes; 9) Bans on hazardous waste; 10) Bans on other waste. Each category is assigned one point, with the exception of the category on waste eco-taxes which can score 1 point for one eco-tax or two points for two eco-taxes (e.g. packaging tax, and tax on waste generation).

The 10 countries with regular font is taken from Guerin. Crete, Mercier, (2001) who used data from the EEA (1998) to construct the index. Scores in italics have been created from the EEA (1998) for the remaining countries for which information is directly available. The other scores are estimated based on information from the Secretariat of the Basel Convention, UNEP, at http://www.basel.int/, the OECD Environmental Taxes database, and other sources.

An excerpt of the scores of several countries is illustrated in the matrix table below.

Countries	Waste Mngmt Plans	Priority to prevent and reduce waste harmfulness	Waste eco-taxes	Producer Responsi-bility	Prevent-ion	Recovery/ Recycling	Hazardous waste reduction	Basel Conve ntion	Bans on hazardous waste	Bans on other wa
Austria	1	1	1	1	1	1	1	1	1	1
Belgium	1	1	1	1	1	1	1	1	1	1
Denmark	1	1	2	1	-	1	1	1	1	1
Finland	1	1	2	1	X	1	1	1	1	1
France	1	1	2	1	-	1	1	1	1	1
Germany	1	1	1[1]	1	1	1	1	1	1	1
Greece	1	1	X	1	1	1	1	1	1	1
Ireland	1	1	X	1	X	1	1	1	1	1
Italy	1	X	2	1	X	1	1	1	1	1
Luxem-bourg	1	1	X	1	X	-	1	1	1	1
Nether-lands	1	1	1	1	1	1	1	1	1	1
Portugal	-	-	X	1	X	1	1	1	1	1
Spain	1	X	X	1	X	-	1	1	1	1
Sweden	1	1	X	1	1	1	1	1	1	1
UK	1	1	1	1	X	1	1	1	1	1

1 Only in some laenders or communities.

Appendix 3.2. Descriptive Statistics

Variable	Obs.	Mean	Std. Dev.	Min	Max
GDPPC	750	17935.898	7374.898	1122.97	55102.73
POPD	750	129.716	122.3737	1.91	488.03
URB	750	72.23851	12.729	29.44	97.23
MWPC	110	448.3636	135.7516	190	760
POLDX	750	7.7	1.7975	5	10

NB: The negative minimum for ldtax and rldtax within is not a mistake; the within is showing the variation of (r)ldtax within country around the global mean 4.899

CHAPTER 4

The Determinants of MSW Disposal and Recycling: Examining OECD Inter-Country Differences for Waste Management

4.1 Introduction

Though the general trend in MSW generation in OECD countries has been on the incline over the past 20 years, this does not provide any information on the way the MSW is being managed. It has been argued that the environmental impact of municipal waste is masked by the fact that the volume of municipal waste does not indicate just how much of that waste is recycled (Cole et al., 1997). Indeed, the proportion of waste disposed of at landfills is on the decline and, in terms of recycling for paper/cardboard and glass, the past two decades have witnessed an overall increase in recycling rates[1]. Moreover, there is wide variation in the proportion of MSW disposed of at landfills and in recycling rates across OECD countries. For example several countries such as the UK, Poland, and Greece landfill more than 70 percent of the MSW generated, whereas other countries such as Denmark and Sweden dispose of less than 20 percent of the MSW in this manner (Eurostat, 2003).

This chapter examines the underlying factors that determine the way MSW is managed once it has been generated. More specifically, it examines (a) the proportion of MSW that is disposed of at landfills, (b) the proportion of paper and cardboard that is recycled as a percentage of apparent consumption, and (c) the proportion of glass that is recycled as a percentage of apparent consumption. An analysis of what the main determinants of landfill disposal and recycling rates is important for understanding the degree of policy flexibility in affecting these rates. For example, Berglund et al. (2002) argue that if recycling rates are largely determined by important cost elements (e.g. population density), it may be costly to pursue very ambitious recycling targets and also to implement harmonised policy targets across countries.

In addition to economic and demographic variables, this study analyses the effects of two policy variables on landfill disposal and recycling rates, namely a waste legislation and policy index, and the level of landfill taxes that have been introduced

[1] See Chapter 1, Figures 1.8 and 1.9.

in a number of OECD countries. The results reveal some interesting insights into the nature of future waste trends and the effect that public policy may have on these.

The chapter is organised as follows: Section 4.2 reviews the existing macroeconomic literature on waste management. Section 4.3 presents the data to be used in this analysis and the panel data regression models that are estimated to identify and analyse the main determinants of MSW landfill disposal and recycling rates. Finally, Section 4.4 concludes and discusses implications for policy.

4.2. A Review of the Macroeconomic Waste Literature

Though only a few studies have examined the determinants of MSW generation at the macroeconomic level, notably less empirical analysis has examined how MSW is disposed of between landfill, incineration, and recycling. A small but growing literature has examined the determinants of recycling at the household level. For example, many of the studies reviewed in chapter 2 that examine the effects of economic and regulatory instruments on recycling rates (e.g., unit pricing programs and kerbside recycling programs) have also analysed the effects of economic and demographic characteristics of the households and the impacts these have on recycling rates. Results indicate that socio-economic factors such as income, population density, single or multi-family dwellings, household size, education, and average age of the head of the household influence recycling behaviour (Hong et al. 1993; Jenkins et al. 2003; Halvorsen and Kipperberg, 2003, among others). Specifically, income, education, age and household size have a positive impact on recycling whereas population density is negatively associated with recycling and home composting of organic waste. Ando and Gosselin (2003) find that multi-family dwellings are less likely to recycle than single-family dwellings. Most of these studies however are conducted using datasets from the U.S. and it is not clear whether these results carry over to other countries.

Two studies that do provide some evidence on the determinants of recycling at the macroeconomic level are those by Terry (2002) and Berglund et al. (2002). Terry (2002) uses time-series data from 1960-1990 in the U.S. and regresses the proportion of MSW recovered from generation on income, MSW composition, landfill disposal, and other demographic characteristics. The results indicate that income has a positive coefficient but is not significant at the 5% level. In contrast, the percentage of population between the ages of 25-44, landfill disposal, durable and packaging waste and the time trend are statistically significant. Berglund et al. (2002) examine the determinants of paper recycling and regress the paper recovery rate on GDP per capita, population density, the percentage of total population living in urban areas, and waste paper prices. Using cross-sectional data from 89 countries, they find that the coefficients on GDP per capita and population density are statistically significant with a positive sign. The adjusted R-squared value however is only 0.24, and they argue that the study might benefit from the use of panel data and the inclusion of policy variables.

The purpose of this analysis is to analyse the determinants of MSW landfill disposal and recycling at the macroeconomic level. Specifically, this chapter:

(i) Analyses MSW landfill disposal and recycling using similar datasets.

(ii) Uses cross-sectional time-series data and therefore can provide more information than the studies by Berglund et al. (2002) and Terry (2002). This enables an examination of the main determinants that account for the different rates of waste management performance both across countries and over time, and to assess whether country-specific findings carry over to the OECD countries as a whole.

(iii) Includes two variables to proxy for policy, namely the waste legislation and policy index, and the landfill taxes that have been introduced in a number of countries to divert waste away from landfill disposal.

4.3 The Determinants of Waste Disposal and Recycling: Methods and Results

4.3.1 Description of the Data

The data used in this analysis and their sources are described in Table 4.1 below. Definitions of the waste data are provided in Appendix 4.1.

Table 4.1 Description and Sources of the Data

%LDFL	Proportion of MSW disposed of at landfills. 1995-2003 for the EU-25 countries. Source: Eurostat, 2004.
PAPER	Paper and cardboard recycled, defined as percentage of apparent consumption. 1980-2000. Source: OECD Environmental Data Compendium 2002.
GLASS	Glass recycled, defined as percentage of apparent consumption. 1980-2000. Source: OECD Environmental Data Compendium 2002.
GDPPC	Gross Domestic Product (GDP) per capita, in 1995 prices and purchasing power parities (PPP) in U.S. dollars. 1980-2004. Source: World Development Indicators, 2004
POPD	Population density, defined as people per square kilometre. 1980-2004 Source: World Development Indicators, 2004.
URB	Urban population, defined as percentage of total. 1980-2004 Source: World Development Indicators, 2004.
LDTX	Landfill taxes. 1980-2004. Source: OECD/EEA Environmentally Related Taxes database 2006.
POLDX	Waste legislation and policy index. 1995 Source: Adapted from Guerin et al. 2001; European Environment Agency 1998.

Due to some missing observations, the data constitutes an unbalanced panel. The descriptive statistics of the all the variables are reported in Appendix 4.1. The correlation matrix of all variables indicates that none of the independent variables are highly correlated.

For each of the variables, total variation is decomposed into between and within class variation and an F test is conducted to test the hypothesis that between classes variation is large relative to within class variation. For all of the variables, Prob > F = 0.000, indicating that between class variation is large relative to within (Table 4.2).

Table 4.2. Analysis of Variance for the Data

Variable	Between	Within	Total	F test
GDPPC	3.074e+10	1.000e+10	4.074e+10	76.32
POPD	11168657	47867.975	11216525	5792.82
URB	114956.95	6401.52	121358.47	445.85
LDFILL	16.6544	1.2292	17.8836	92.42
PAPER	60548.703	26675.463	87224.117	31.95
GLASS	94217.602	83488.934	177706.54	14.81
POLDX	2420	0	2420	-
LDTX	62646.854	98329.536	160976.39	15.82
RLDTX	40133.496	76412.595	116546.09	12.82

4.3.2 Landfill Disposal of MSW

Given that landfill deposition is the lowest on the waste hierarchy, and that environmental quality is a normal good, one would intuitively expect that as income levels rise, the percentage of MSW disposed of at landfills will decline[2]. To examine this hypothesis, recently available data on the proportion of MSW generated that is disposed of at landfills (%LDFL) is used as the dependent variable. The variable GDPPC is included as an independent variable to examine its impact on %LDFL, and is expected to be negative. Variables for population density (POPD) and urbanization (URB) are also included in the model and are expected to be negative. This is because in densely populated regions and/or in which people live clustered in highly urbanized areas, the likelihood for high landfill prices will be high, and so will the cost of landfill disposal. Furthermore, higher population density lowers the cost of recycling, thereby indirectly lowering the demand for landfill disposal. The waste policy and legislation index (POLDX) is also included to test the assumption that *ceteris paribus*, the higher the POLDX, the lower the %LDFL is likely to be. The final variable included in the regression is the real landfill tax (RLDTX) in various countries.[3] The purpose of the landfill tax is to raise the costs of landfill disposal

[2] Similarly, it is anticipated that as income levels rise, the proportion of waste recycled will increase. Perhaps the only method of MSW disposal that may exhibit an inverted U EKC shape is incineration whereby at low levels of income, incineration increases as landfill decreases, followed by a decline in incineration as recycling increases. Unfortunately, the lack of consistent panel data on the proportion of MSW sent to incineration makes it impossible to test this hypothesis.

[3] Converted from nominal landfill taxes using the GDP deflator (WDI, 2004).

relative to alternative disposal routes (e.g., incineration and recycling) and thus to divert waste away from landfill. Thus, higher landfill taxes are anticipated to be inversely related to the percentage of MSW generated disposed of at landfill.

The role of prices and/or taxes has not been fully examined in either the EKC debate (De Bruyn et al., 1998) nor in the scant (macroeconomic) literature on waste disposal management. There are to date only a few EKC studies that have included a price variable in the regression analysis. De Bruyn et al. (1998) for example include energy prices in their model to examine CO_2, NO_x and SO_2 emissions in four countries (Netherlands, UK, USA, and western Germany). These turn out to be statistically insignificant in two of the cases. Agras and Chapman (1999) include energy prices (i.e., gasoline) in their analysis of CO_2 emissions and find that income is no longer the most relevant indicator of environmental quality. Lindmark (2002) investigates the relationships among CO_2 emissions and proximate explanatory factors including economic growth, fuel prices, technology and income levels in Sweden during the 19^{th} and 20^{th} centuries and finds that fuel prices are statistically significant. Finally, Culas and Dutta (2002) include an export price index to assess the effect this has on deforestation. Their results indicate that the export price index is only significant for Latin America. Though the effect of prices has been examined in studies using household-level data as a result of an increasing number of pay-as-you-throw programs (see chapter 2), to my knowledge, no existing study has explicitly included landfill prices and/or taxes at the macroeconomic level. This may not be surprising given that landfill prices are set at the local level. However, landfill taxes are set at the national level.

In the case of waste, average national prices for landfill disposal (also known as tipping fees) have been rising over time making it more expensive to dispose of waste at landfills. In the U.S. for example, average tipping fees increased from $10 in 1983 to $50 in 1990. For the analysis undertaken here, ideally data on average national tipping prices at landfills along with data on landfill taxes should be used. Panel data on the former however is not readily available, and to some extent, POPD may serve as a proxy for landfill prices. In addition, any inter-country differences in the proportion of MSW deposited at landfills may show up most clearly as a result of changes in landfill taxes as these are specifically intended to help divert waste away from landfill disposal, i.e., to incineration and recycling[4].

The equation for the proportion of municipal waste disposed of at landfills is written in log-linear form and is formulated as:

$$\%LDFL_{it} = \alpha + \beta_1 GDPPC_{it} + \beta_2 POPD_{it} + \beta_3 URB_{it} + \beta_4 POLDX_{it} + \beta_5 RLDTX_{it} + \varepsilon_{it} \quad [5]$$

As before, Equation 5 is estimated using random and fixed effects models. The results are presented in Table 4.3 below.

Table 4.3. Parameter estimates for the percentage of MSW landfilled

	Random Effects	Fixed Effects
	Co-efficient (*Z* value)	Co-efficient (*t* value)
GDPPC	-0.469 (-2.87)***	-0.024 (-0.10)
POPD	-0.257 (-3.19)***	-0.16 (-0.11)
URB	-0.13 (-1.75)*	-7.55 (-3.44)***
POLDX	-0.017 (-0.04)	dropped
RLDTX	-0.011 (-6.14)***	-0.010 (-5.59)***
Constant	9.99 (3.76)***	32.62 (4.70)***
No. observations No. groups	220 29	220 29
R-square	Within = 0.2489 Between = 0.5107 Overall = 0.4670	Within = 0.2944 Between = 0.0917 Overall = 0.1637
	Wald $\chi^2(5) = 87.76$	$F(4,187) = 19.50$ $Prob>F = 0.0000$

[4] NB. Similar assumptions have been made by Rietveld and van Woudenberg, (2005) in their analysis of why fuel prices differ.

	Prob>χ^2 = 0.0000	F-test that all u_I = 0: F(28, 187) = 37.56, Prob>F= 0.000

*** significant at the 1% level, ** 5% level, * 10% level
NB %Ldfl data not available for Canada

All the signs on the regression coefficients are as intuitively expected. In the RE model, GDPPC has the largest influence on the amount of waste disposed of at landfills, followed by POPD. The POLDX is negative but insignificant, and the RLDTX is negative and statistically significant suggesting that the higher is the real landfill tax in a country, the lower is the amount of waste that is deposited at landfills. In the FE model, only URB and RLDTX are statistically significant with the expected negative signs. The FE (within) estimator ignores the between state variation in the data which explains why POPD is insignificant (see Table 4.2). The Hausman statistic is $\chi^2(3)$= 14.22, Prob > χ^2 = 0.0026, suggesting the FE model is preferred. The within R^2 is not particularly high, indicating that a relatively large proportion of the variation in the dependent variable remains unexplained.

Diagnostic tests for heteroskedasticity and serial correlation indicate that these are both present in the data. The Breusch-Pagan / Cook-Weisberg test for heteroskedasticity is significant with $\chi^2(1)$ = 30.69 therefore the null hypothesis of constant variance is rejected. Similarly, the Wooldridge test for autocorrelation in the data is significant with F (1, 24) = 21.895 therefore the null hypothesis of no first-order autocorrelation is rejected. As such, feasible generalized least squares is used for estimation and the results are presented in Table 4.4.

Table 4.4 Feasible Generalized Least-Squares Estimates of % Landfilled

	Co-efficient	Std. Err.	Z	P > |z|
GDPPC	-0.3524	0.0789	-4.46	0.000
POPD	-0.1913	0.0253	-7.56	0.000
URB	-0.7808	0.2814	-2.77	0.006
POLDX	-0.3844	0.1365	-2.82	0.005
RLDTX	-0.0009	0.0014	-0.66	0.512
Constant	7.7867	1.0473	7.43	0.000

Wald $\chi^2(5)$ = 126.44 , Prob > χ^2 = 0.000
Log likelihood = 202.9489

In the FGLS model, all the variables are significant, with the exception of the RLDTX. The estimated coefficient on GDPPC is –0.3524, whereas in the FE model it is –0.024. POPD and URB are both negative indicating that as these levels increase, the percentage of waste deposited at landfills declines. The magnitude of the co-efficient on POPD is similar in the FE and FGLS model and ranges from -0.16 to –0.19. A 10% increase in the population in urban areas results in a 7.8% decrease in the proportion of MSW generation disposed of at landfills. The sign on the estimated co-efficient on POLDX is negative and statistically significant implying that a higher POLDX is associated with a lower percentage of waste deposited at landfills. The RLDTX is insignificant in the FGLS model but statistically significant in the RE and FE model with the correct sign.

To my knowledge, this is the first study that has empirically investigated the socio-economic and policy determinants of the proportion of MSW generated that is disposed of at landfills. Using panel data on OECD countries, this analysis provides evidence that higher levels of GDP per capita are associated with a smaller fraction of MSW deposited at landfills. Population density and urbanization also seem to have an effect on the proportion of MSW landfilled, with a significant and negative influence. Furthermore, there is some indication that greater national commitments to sustainable waste management, as measured by the waste legislation and policy index, have been effective in reducing the amount of MSW disposed of at landfills. The impact of the other policy variable, RLDTX, seems to be weaker in the FGLS model, though it is significant in the FE and RE model.

4.3.3 Paper/ Cardboard and Glass Recycling

The relationship between recycling and economic growth can provide further useful insights into the dynamics of waste. Ideally one would like to examine the proportion of MSW recycled, as was done in section 4.3.2. for MSW disposed of at landfills. Unfortunately, panel data on the percentage of MSW generated that is recycled is not available for all countries. The OECD does however have annual cross-sectional and time-series data on the waste recycling rates for paper and cardboard as well as glass, as a percentage of apparent consumption, over the period 1980-2000.

As before, it is assumed that economic growth may have an impact on preferences for environmental quality and thus that higher levels of income will be associated with higher recycling rates of both paper/cardboard and glass. In addition, costs of collection and recovery are also likely to be important economic factors in determining recycling rates. The marginal cost of recovery is dependent on the size of the waste stream and hence areas with higher population densities and degree of urbanisation are assumed to be related to collection costs of recycling materials. Finally, these collection costs may not be purely market driven but may also be affected by government policies including landfill taxes and other policies that raise the costs of alternative waste disposal treatments in relation to recycling.

Thus, the recycling equations for each of the two materials is written in log-linear form and are formulated as:

$$\%RCYC_{it} = a + \beta_1 GDPPC_{it} + \beta_2 POPD_{it} + \beta_3 URB_{it} + \beta_4 POLDX_{it} + \beta_5 RLDTX_{it} + \varepsilon_{it} \qquad [6]$$

For both paper and cardboard, and glass, it is anticipated that all of the coefficients on the independent variables are positive: As income levels rise, preferences for environmental quality improvements become stronger as the environment is considered to be a normal good. Population density affects the economics of recycling, as recycling materials becomes more viable in densely populated and urbanized areas where the costs of collecting and separating waste decrease. Further it is expected that higher policy indices indicate greater efforts towards implementing sustainable waste management, and that higher real landfill taxes will divert greater portions of the two materials away from landfill disposal to recycling.

A similar analysis for paper recovery rates has been conducted by Berglund et al. (2002) in which they regress GDP per capita, population density, and the percentage of total population living in urban areas on paper recovery rate. Using cross-sectional data from 89 countries, they find that the coefficients on GDP per capita and population density are statistically significant with a positive sign. The adjusted R-squared value however is only 0.24. Equation 6 above extends their analysis by examining paper recovery in a panel data setting, and by including two waste management policy variables in the regression model, i.e., the waste legislation and policy index and the real landfill tax. Similar analysis is then undertaken to investigate the determinants of glass recycling.

Table 4.5 presents the parameter estimates for the coefficients in the RE and FE models for paper and cardboard. Each of the models is jointly significant as measured by the respective χ^2 and F statistics. In the RE model only GDPPC is statistically significant. The impact of GDP per capita has the expected positive sign, indicating that a 10% increase in GDP per capita will result in a 7.2% increase in the proportion of paper and cardboard recycled. The demographic variable POPD has the expected positive sign whereas URB does not. The variable POLDX does not exhibit the intuitively expected sign and is statistically insignificant. Finally, though higher landfill taxes are positive as expected, this variable is insignificant. In the FE model, POPD is also statistically significant and shows the intuitively correct (positive) sign. The Hausman test statistic with a $\chi^2(4) = 11.24$ indicates that the fixed effects model is preferred over the random effects model. The R2 values are very low indicating that a large proportion of the variation in the %paper/cardboard recycled remains unexplained by the model.

Table 4.5. Parameter estimates for the percentage of paper and cardboard recycled

	Random Effects	Fixed Effects
	Co-efficient (Z value)	Co-efficient (t value)
GDPPC	0.719 (6.72)***	0.576 (4.45)***
POPD	0.106 (1.58)	1.95 (3.39)***
URB	-0.209 (-0.64)	-0.58 (-1.45)
POLDX	-0.310 (-0.91)	dropped
RLDTX	0.002 (1.12)	0.001 (0.75)
Constant	-2.348 (-1.62)	-8.47 (-3.54)***
No. observations No. groups	408 28	408 28
R-squared	Within = 0.1286 Between = 0.2070 Overall = 0.1840	Within = 0.1521 Between = 0.0385 Overall = 0.0633
	Wald $\chi^2(5) = 61.25$ Prob>χ^2 = 0.000	F(4, 376) = 16.86 Prob>F = 0.000 F-test that all u_I = 0: F(27, 376) =29.66, Prob>F= 0.000

*** significant at the 1% level, ** 5% level, * 10% level

Diagnostic tests indicated the presence of both heteroskedasticity and serial correlation in the data. The Breusch-Pagan/ Cook-Weisberg test for heteroskedasticity is significant with a $\chi^2(1) = 106.91$, and the Wooldridge test for first-order autocorrelation in the data is significant with F (1, 25) = 15.644. The estimates from the feasible generalized least-squares estimates are reported in Table 4.6.

Table 4.6. FGLS Estimates of % Paper and Cardboard Recycled

	Co-efficient	Std. Err.	Z	P > |z|
GDPPC	0.4302	0.0714	6.03	0.000
POPD	0.1461	0.0334	4.37	0.000
URB	-0.0592	0.1506	-0.39	0.694
POLDX	-0.2484	0.1208	-2.06	0.040
RLDTX	0.0015	0.0007	2.24	0.025
Constant	-0.4672	0.7906	-0.59	0.555
Wald $\chi^2(5) = 67.20$, Prob > $\chi^2 = 0.000$ Log likelihood = 394.5131				

In the FGLS model, all the variables except URB are statistically significant. As before, GDPPC and POPD have the strongest impact on paper and cardboard recycling, and the RLDTX now also exhibits a statistically significant and positive sign. The coefficient on POLDX exhibits a statistically significant but inverse relationship with recycling which is counter-intuitive and signals the need for further research. One caveat with respect to data quality is that the POLDX variable is time-invariant. This is due to a lack of data availability, and the coefficient is therefore unlikely to capture the full relationship with the dependent variable. Furthermore, the POLDX variable only reflects the degree of national and international commitment towards sustainable waste management, rather than actual effort, and the variable does not account for monitoring or enforcement of these national commitments. This is a weakness in data quality.

These models therefore provide further evidence on the importance of population densities as opposed to urbanization on paper and cardboard recycling rates. There is also some indication that higher landfill taxes are associated with more paper and cardboard recycling. From the RE and FE models, the R^2 values are all quite low indicating that a large proportion of the variation in paper recycling rates remains unexplained, and the inclusion of the policy variables does not make a substantial contribution. Further research efforts should be put in identifying the main determinants of paper and cardboard recycling rates.

Turning to the results of the RE and FE models for the percentage of glass recycled presented in Table 4.7 it can be seen that, as in the case of paper and cardboard recycling, the impact of GDPPC on the percentage of glass recycled also has the expected positive sign and is statistically significant. A 1% increase in GDP per capita will result in a 1.9-2.2% increase in the proportion of glass recycled (for the RE and FE model respectively). With regard to the demographic variables, POPD is positive and statistically significant in the RE model, again suggesting that the lower the cost of glass collection and recovery, in terms of transport etc, the higher the rate of recycling. Similarly, urbanization is positive but insignificant in both models. The policy index is statistically significant but intuitively incorrect with a negative sign implying that countries with a lower waste legislation and policy index recycle more glass. The real landfill tax is statistically significant in both models, suggesting that higher taxes on the waste disposed of at landfills does have a positive impact on the amount of glass that is recycled. The Hausman test statistic of $\chi^2(4) = 99.56$ indicating that the fixed effects model is preferred. The within R^2 in the FE model is represents a good fit.

Table 4.7. Parameter estimates for the percentage of glass recycled

	Random Effects	Fixed Effects
	Co-efficient (*Z* value)	Co-efficient (*t* value)
GDPPC	1.983 (15.59)***	2.217 (13.92)***
POPD	0.210 (2.90)***	0.500 (0.82)
URB	0.323 (0.83)	0.303 (0.63)
POLDX	-0.895 (-2.22)***	dropped
RLDTX	0.005 (3.28)***	0.004 (2.66)***
Constant	-16.208 (-10.09)***	-21.64 (-9.38)***
No. observations No. groups	354 26	354 26
R-squared	Within = 0.6158 Between = 0.1897 Overall = 0.2534	Within = 0.6174 Between = 0.2574 Overall = 0.2598
	Wald $\chi^2(5) = 444.17$ Prob $>\chi^2 = 0.000$	F(4, 324) = 130.70 Prob >F = 0.000 F-test that all u_I = 0: F(25, 324) =38.50, Prob >F= 0.000

Diagnostic tests are undertaken to examine the existence of heteroskedasticity and autocorrelation in the data. The Breusch-Pagan / Cook-Weisberg test for heteroskedasticity with $\chi^2(1) = 17.71$ therefore the null hypothesis of homoskedasticity is rejected. The Wooldridge test for autocorrelation in the data is significant with F (1, 21) = 57.906, indicating the presence of serial correlation. The model is therefore estimated using the FGLS method and the results are reported in Table 4.8.

Table 4.8. Feasible Generalized Least-Squares Estimates of % Glass Recycled

	Co-efficient	Std. Err.	Z	P > |z|
GDPPC	1.0743	0.0788	13.63	0.000
POPD	0.1984	0.0418	4.75	0.000
URB	0.4321	0.2460	1.76	0.079
POLDX	-0.3046	0.1506	-2.02	0.043
RLDTX	0.0013	0.0012	1.16	0.247
Constant	-9.0678	1.1810	-7.68	0.000
Wald $\chi^2(5) = 342.75$, Prob > $\chi^2 = 0.000$ Log likelihood = 153.6321				

In the FGLS model, the estimated co-efficients on GDPPC, POPD, and URB are somewhat smaller than in the FE model but are all statistically significant and positive. POLDX is statistically significant but negative. As before, the possible explanation suggested here is the same as that in the case of the % of paper recycled. The RLDTX remains positive but is now insignificant.

This leads to the conclusion that higher levels of GDPPC unambiguously have a positive influence on the percentage of glass that is recycled, with a range in the estimated coefficients of 1.07 to 2.22. POPD and URB exhibit the strongest influence on glass recycling in the FGLS model that accounts for heteroskedasticity and autocorrelation in the data. From a policy perspective, this is a promising phenomenon suggesting that these trends are likely to increase in the future. With regard to the policy variables POLDX and RLDTX included in the regression however, there does not seem to be consistently statistically significant evidence that existing waste management policy has been effective in achieving its objectives. The POLDX variable has been used as a proxy for national efforts towards sustainable waste management. Perhaps this variable inadequately reflects national *effort* as opposed to simply national *commitments* towards sustainable waste management.

4.4 Conclusions and Policy Implications

With the use of panel data from approximately 30 OECD countries over 20 years, this chapter has attempted to identify and analyse the main trends and determinants in municipal solid waste landfill disposal and the recycling rates of paper/cardboard and glass. The results reveal some interesting insights into the issue of MSW management (see Table 4.9 for a summary of results).

With regard to the disposal of municipal solid waste, in the preferred fixed effects model for the waste deposited at landfills, the results indicate that urbanization and the real landfill tax introduced by national governments both have negative impacts. This implies that though urbanization is associated with higher amounts of generated waste (chapter 3), the waste is managed in a more environmentally friendly way (i.e., either via incineration or via recycling). The negative sign on the real landfill tax indicates that the higher the landfill taxes on waste, the smaller is the proportion of waste that is deposited at the landfills. This is strictly a policy variable and should be very encouraging to governments wishing to divert additional waste away from landfills. With regard to the amount of paper and cardboard, and glass that is recycled, the main determinants of recycling are economic growth, followed by population density. In the case of glass recycled, this is also affected by the real landfill tax. Recycling of these two materials is therefore determined more by market forces rather than by policy forces. Higher population densities are expected to lower the collection and recovery costs of recycling, thus increasing the economic viability of this disposal option.

Thus, waste disposal and recycling are affected by economic as well as demographic variables, and the results reveal that countries with higher GDP per capita perform better in terms of diverting waste away from landfill disposal, and achieve higher recycling rates for both paper and cardboard, and glass.

To the degree to which population density serves as a proxy for landfill prices and/or the cost of collection and recovery for recycling, the results provide evidence suggesting that trends in waste generation and disposal are also market driven.

Overall, the waste policy and legislation index provides weak evidence as a determinant in improving waste generation and disposal. For landfill disposal, the variable is negative and statistically significant, and thus intuitively correct. For paper/ cardboard and glass recycling, the variable is negative, usually statistically significant, and thus intuitively incorrect. One important caveat however is that the index is fixed across time and may not accurately reflect the changes in national policy targets over the 20 year time period examined here. Further, the index may reflect national commitments, but may not adequately capture actual national efforts or concrete measures towards sustainable waste management. Guerin, Crete, and Mercier (2001) offer a similar explanation with regard to their results and refer to Read (1999) who argues that often, the pace of policy making (for waste) has not been matched by an equal effort to provide effective policy implementation. Future research effort should therefore focus on obtaining more accurate indices for this purpose, one that reflects policy change over time. Ideally, data on individual national policies should be collected and included separately as dummy variables in the regression analysis[5]. Public policy variables that reflect the amount of resources spent on waste management per capita at the national level, as well as indices for monitoring and enforcement could also be useful.

Finally, the results provide evidence that real landfill taxes have a significant impact on sustainable waste management. Higher landfill taxes are associated with lower proportions of landfill disposal, and higher rates of paper/cardboard and glass recycling. This implies that governments wishing to divert waste away from landfill

[5] Concerted collaborative international efforts need to be undertaken to ensure that this data is accurate and consistent across countries. An attempt was made here to disaggregate the waste legislation and policy index and to update it over time but due to lack of data availability and inconsistency, this was ad hoc and therefore not feasible.

disposal, which entail the highest external cost to incineration and recycling, are likely to do so successfully via the introduction of landfill taxes.

Table 4.9. Summary of Results

	Random Effects		**Fixed Effects**		**FGLS**	
	Sign	Significance	Sign	Significance	Sign	Significance
LANDFILL DISPOSAL						
GDPPC	-	3	-		-	3
POPD	-	3	-		-	3
URB	-	3	-	3	-	3
POLDX	-		Dropped		-	3
RLDTX	-	3	-	3	-	
PAPER RECYCLING						
GDPPC	+	3	+	3	+	3
POPD	+		+	3	+	3
URB	-		-		-	
POLDX	-		Dropped		-	3
RLDTX	+		+		+	3
GLASS RECYCLING						
GDPPC	+	3	+	3	+	3
POPD	+	3	+		+	3
URB	+		+		+	3
POLDX	-	3	Dropped		-	3
RLDTX	+	3	+	3	+	

Appendix 4.1. Definitions of Landfill and Recycling Data

Municipal Waste Landfilled presents the amount of municipal waste disposed of through landfill. The bulk of this waste stream is from households, though "similar" wastes from sources such as commerce, offices and public institutions are included. Landfill is defined as deposit of waste into or onto land, including specially engineered landfill, and temporary storage of over one year on permanent sites. The definition covers both landfill in internal sites (i.e. where a generator of waste is carrying out its own waste disposal at the place of generation) and in external sites. The quantity of waste landfilled is expressed in kg per capita per year.

Paper and Cardboard Recycling
Recycling is defined as reuse of material in a production process that diverts it from the waste stream,except for recycling within industrial plants and the reuse of material as fuel.The recycling rate presented here is the ratio of the quantity collected for recycling to the apparent consumption (domestic production +imports -exports).It corresponds to the CEPI collection rate.

Glass Recycling
Recycling is defined as reuse of material in a production process that diverts it from the waste stream,except for recycling within industrial plants and the reuse of material as fuel.The recycling rate is the ratio of the quantity collected for recycling to the apparent consumption (domestic production +imports -exports).

Appendix 4.2. Descriptive Statistics

Variable	Obs.	Mean	Std. Dev.	Min	Max
GDPPC	750	17935.898	7374.898	1122.97	55102.73
POPD	750	129.716	122.3737	1.91	488.03
URB	750	72.23851	12.729	29.44	97.23
%LDFL	220	0.5524	0.2857	0.0267	1
PAPER	408	39.083	14.6393	1.6	73.09
GLASS	354	41.7259	22.4370	4.96	93
LDTX	750	4.8999	14.6602	0	83.61
RLDTX	750	3.9813	12.5752	0	83.61
POLDX	750	7.7	1.7975	5	10

NB: The negative minimum for ldtax and rldtax within is not a mistake; the within is showing the variation of (r)ldtax within country around the global mean 4.899

CHAPTER 5

Spatial Interaction in Waste Management and Policy-Making

Everything is related to everything else, but near things are more related than distant things.

Tobler's First Law

5.1 Introduction

Just as individual recycling behaviour may in part be induced by the recycling behaviour of their neighbours (Gamba and Oskamp, 1994; Werner and Makela, 1998), so perhaps may national governments be influenced by waste policies introduced in countries nearby. If this conjecture is true, this would constitute a form of so-called spatial interaction or structure, which refers to the importance of "space" (or geography) in some specified relationship. In a general context, spatial interaction may arise due to social norms, neighbourhood effects, copy-catting, peer group effects, and the strategic nature of government policy making (Anselin, 1999). Though these are likely to be interrelated to some degree, they are nevertheless distinct. Social norms refer to shared beliefs of what is normal and acceptable and contribute to shaping and enforcing the action of people (and government) in a society. Neighbourhood effects occur as a result information spillovers and diffusion effects. Copy-catting concerns the adoption of ideas and policies due to their beneficial effects. Peer group effects pertain to the pressure to conform to the group of peers with whom one interacts e.g. an EU member. Finally, the strategic nature of policy-making refers to games and either co-operative or non-co-operative behaviour taken to improve actions and the economic position of a player.

The possibility of spatial relationships in environmental policy-making has been gaining increasing attention. The interjurisdictional regulatory literature is now well established and has been amply surveyed by Wilson (1996, 1999) and Oates (2001). It has a history in a parallel literature of fiscal federalism[1] and tax competition wherein the work by Oates and Schwab (1998) and Wilson (1996) have a corollary to the seminal papers by Zodrow and Mieskowski (1986), Wilson (1986) and Wildasin (1989)[2]. On the one hand, interjurisdictional competition is viewed as a beneficent force, compelling public agents to make efficient decisions; on the other hand there is

[1] Fiscal federalism addresses the vertical structure of the public sector. See Oates (1999) for a good introduction to this. The literature on fiscal competition originates from the seminal paper by Tiebout (1956).

the contention that competition is a source of distortion in public choices. It is the latter view that has raised considerable concern and debate on the possibility of a 'race-to-the bottom' in environmental standards. This term reflects the notion that, due to government and business perception of a significant trade-off between economic growth and environmental protection, pursuit of economic development in a competitive setting may drive governments to lower their environmental standards and/ or curtail their environmental enforcement efforts (see Esty, 1996; Engel, 1997; Woods, 2005)[3].

There is also a case for interaction effects for localised pollution problems where governments may strategically manipulate environmental standards in an attempt to attract capital (Markusen et al. 1995). This would result in an under-provision of public goods. Markusen et al. (1995) also demonstrate that a race-to-the-top dynamic can emerge if jurisdictions compete to avoid an undesirable facility, such as a hazardous waste treatment plant or nuclear power plant, by raising their environmental standards. This is the NIMBY effect associated with negative externalities. Additional papers on strategic environmental policy include those by Barrett (1994) who examine the competitiveness of existing industries in the context of international trade, and Ulph (1999). Murdoch et al. (1997) find empirical evidence to suggest that there is the possibility that a country will limit its cleanup efforts as others reduce emissions.

The literature on the conditions under which local government authority would lead to the same Pareto-optimal environmental regulations as a welfare-maximising centralised authority are fairly restrictive and have been neatly summarised by Levinson (2002). These are (i) No cross border externalities; (ii) Many jurisdictions[4]; (iii) All economic rents earned locally by the competing jurisdiction; (iv) Welfare

[2] Their work shows that non-benefit taxation of capital by local governments leads not only to regional misallocation of capital but also to distorted local public finance.

[3] This relates directly to the issue of environmental federalism i.e. the role of different levels of government in environmental management, and is of interest given the recent US devolution of authority from federal level to state governments, and the contrasting shift in the EU to harmonise environmental legislation.

maximising local regulators[5]; (v) No constraints on available policy instruments; and (vi) No redistributive policies (all taxes are benefits taxes[6]). Oates (2001) reminds us however that even if lax local environmental standards is the case, the alternative which is central intervention in the form of standards for environmental quality on a nation-wide/regional scale is also not an efficient, first-best rule. See Ulph (2000) for a detailed discussion on this.

There is also the case however in support of the so-called 'Porter hypothesis' (Porter, 1991). This hypothesis asserts that stringent environmental regulation can lead to a situation whereby both social welfare and the private net benefits of firms can be increased. Thus, governments may act strategically and set policies which are too high relative to the first best rule (marginal abatement costs exceed marginal damage costs) in an effort to provide their producers with incentives to innovate green technologies ahead of their rivals and thus to gain a long-term competitive advantage. Jaffe et al. (1995) who review 16 empirical studies on the effects of environmental regulation on competitiveness in the U.S., as well as other more recent studies, do not find conclusive evidence either for or against this hypothesis. See also Becker and Henderson (2000).

Though the discussion thus far has focused on competition induced by migration of mobile factors (e.g. capital, goods, and wealthy taxpayers), information and ideas can also cross jurisdictional borders. For a seminal paper on policy innovation and diffusion, see Walker (1969). Bennet (1991); Dolowitz and Marsch (2000); and Oates (1999) discuss the dynamics of policy transfer. In addition, jurisdictions may also be interdependent when it comes to setting well established policy instruments -such as taxes, environmental standards, and minimum wages- not only the novel policies typically examined in the context of policy diffusion and transfer. This idea is

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[4] See Markusen et al. 1995.

[5] If instead the objective was tax revenue maximisation, local regulators would ease environmental regulations in order to attract capital and inflate the tax base.

[6] Benefit taxes reflect social marginal cost and therefore lead consumers and firms to choose jurisdictions efficiently.

examined by Breton (1991), Hall (1993) and Besley and Case (1995) using a yardstick competition model[7].

Furthermore, individual regions may not compete with one another by may instead follow one or two innovators. Vogel (1995) for example argues that in the case of the US, increased regulatory stringency in California is matched by the rest of the country (at least with regard to automobile emission standards), a trend which has been coined the 'California effect'. He states further:

> *'The term 'California effect' is meant to connote a much broader phenomenon than the impact of American federalism on federal and state regulatory standards. The general pattern suggested by this term, namely, the ratcheting upward of regulatory standards in competing political jurisdictions, applies to many national regulations as well'.*

Fredriksson and Millimet (2002a) present a simple model of yardstick competition in pollution abatement costs and investigate California's leadership role empirically using state-level panel data across the U.S. Their results indicate at best a minor role for California. In a tax competition context, Altshuler and Goodspeed (2002) test whether European countries set capital tax rates in response to U.S. rates in a Stackleberg model[8]. The empirical evidence suggests that European countries interact strategically with their neighbors to set capital tax rates but not to set labor tax rates and follow the lead of the United States in setting capital tax rates after 1986.

To examine the extent to which states or nations look to others in determining the appropriate composition of taxes or tariffs, levels of expenditure, and public good provision for example, requires the use of spatial econometric methods. This approach is adopted here to examine two particular issues of interest. First, the aim is to account for the effects from the interaction of neighbouring governments in the determination of waste generation and disposal performance. The second, perhaps more interesting objective, is to examine whether countries choices of tax rates on

[7] The yardstick competition model refers to a set-up where agents use the performance of others as a benchmark. For a discussion of the yardstick model of tax competition as a model of information spillovers, see Brueckner (2003).

landfill disposal are affected by choices made in neighbouring countries. Both of these are examined within the context of OECD member countries. Evidence for spatial interaction in the determination of landfill taxes and waste management performance has important policy implications in so much as that changes in the tax rate and performance in one country will imply cascading ramifications into other countries waste management if there is interaction. Thus, this paper is related to the theoretical literature on tax competition (Brueckner, 2000), and on the theoretical literature on capital competition using environmental policy (Oates and Schwab, 1988; Markusen et al. 1995; Ulph, 2000). It is not possible to tie the empirical results to any single strand of economic theory because these may in fact operate in tandem. Instead, empirical analysis can shed some light onto which, if any, of the theoretical motivations of spatial interaction dominate[9].

The chapter is organised as follows. The next section describes the theoretical background of spatial econometric methods. Section 5.3 presents a brief summary of previous applications with an emphasis on studies that have examined the strategic nature of government policy-making. In section 5.4 the case for spatial interdependencies in the case of waste management and policy is presented, along with the econometric model. The results are reported and discussed in section 5.5, and section 5.6 concludes with some policy implications.

5.2 Spatial Econometrics

Spatial econometric methods deal with the incorporation of spatial interaction (spatial autocorrelation) and spatial structure (spatial heterogeneity) into regression analysis (Anselin, 1999). More specifically, spatial autocorrelation in a collection of sample data observations refers to the fact that one observation associated with a location i depends on other observations at locations $j \neq i$. Spatial heterogeneity refers to

[8] The Stackleberg model tries to capture the essence of a market where firms are competing but, for some reason, there is a dominant firm, or leader.
[9] See Brueckner (2003) for a clarification of the theoretical roots of empirical studies of strategic interaction among governments.

variation in relationships over space (LeSage, 1988). These methods are appropriate when the focus of interest is the assessment of the existence and strength of spatial interaction, or when one would like to test for autocorrelation in the error terms.

Spatial econometric methods require the use of a spatial weight or spatial lag operator. Let W denote a *(N* x *N)* spatial weights matrix describing the spatial arrangement of the spatial units and w_{ij} the (i, j)th element of W, where i and j = (1,..., N). W is assumed to be a matrix of known constants and that all elements on the diagonal are equal to zero. A spatial lag for the dependent variable y at location i is then expressed as:

$$[Wy]_i = \sum_{j=1,.N} w_{ij} \times y_j$$

where i, j refer to individual observations (locations). The elements of the weight matrix should be non-stochastic and exogenous to the model and can be formed in a variety ways. Probably the simplest method is to assign a weight of zero to non-contiguous countries (i.e., those that do not share a common border) and equal weights to contiguous countries. Alternative methods include using k nearest neighbours, economic distance, and distance decay (inverse distance or inverse distance squared), among others.

The spatial relationships can be modelled in two distinct ways: Models in which spatially lagged weighted averages of the dependent variable are included as independent variables are referred to as *spatial lag models*. These models assume that the spatially weighted average of waste generation/disposal in a region affects the generation/disposal of waste in each country (indirect effects) in addition to the standard explanatory variables of country characteristics (direct effects). This can formally be expressed as:

$$y = \rho Wy + X\beta + \varepsilon$$

where ρ is a spatial autoregressive coefficient, and ε is a vector of independently and identically distributed (i.i.d.) error terms. Using OLS here would lead to biased and inconsistent results due to simultaneity bias and the model must be estimated using maximum likelihood techniques.

A *spatial error model* is a special case of a regression with a non-spherical error term. This model does not include indirect effects but is based on the assumption that there may be omitted variables in the equation and that these vary spatially. Thus the error term tends to be spatially autocorrelated:

$$y = X\beta + \varepsilon$$

and

$$\varepsilon = \lambda W\varepsilon + u$$

where λ is the spatial autocorrelation coefficient and u is a vector of i.i.d. errors (Anselin 2001, Elhorst, 2003). For example, waste levels at any location would be a function of country characteristics but also of the omitted variables at neighbouring locations. OLS here would remain unbiased, but would lose the efficiency property.

Given the cross-sectional time-series nature of the data set used in this analysis, we are specifically interested in spatial panel data models (see Elhorst, 2003). One potential specification is the so-called *pure space-recursive model* in which dependence pertains to neighbouring locations in a different time period (Anselin, 2001):

$$y_{it} = \gamma\, [Wy_{t-1}]_i + f(z) + \varepsilon_{it}$$

where $[Wy_{t-1}]_i$ is the i-th element of the spatial lag vector applied to the observations on the dependent variable in the previous time period, and f(z) is a generic designation of regressors which may be lagged in time and/or space. Such models are therefore also referred to as *spatio-temporal models*. In contrast to the above models, this model does satisfy the assumptions of the classical linear model and can be estimated by OLS.

Other potential specifications are the *time-space recursive model* in which the dependence relates to the same location as well as the neighbouring locations in another period:

$$y_{it} = \lambda y_{it-1} + \gamma\, [Wy_{t-1}]_i + f(z) + \varepsilon_{it}$$

and the *time-space simultaneous model,* with both a time-wise and a spatially lagged dependent variable:

$$y_{it} = \lambda y_{it-1} + \rho\, [Wy_t]_i + f(z) + \varepsilon_{it}$$

where is the *i*-th element of the spatial lag vector in the same time period. Instrumental variables methods are necessary here due to correlation with the residuals (see Kelejian and Robinson 1993, Kelejian and Prucha, 1998). Others who have used this approach are Buettner (2001), Fredriksson and Millimet (2002b), Levinson (2002) and Hernandez-Murillo (2003). The alternative method is to use maximum likelihood techniques[10].

5.3 Previous Applications

Applications of spatial econometrics were initially found in specialised fields such as regional science, urban and real estate economics and economic geography. More recently however, spatial econometric methods have been applied to studies in labour economics, public economics and local public finance, as well as agricultural and environmental economics. The latter include studies by Benirschka and Binkley (1994), Murdoch et al. (1997), Bell and Bockstael (1999), Fredriksson and Millimet (2002b), Levinson (2002), Fredriksson et al. (2004), Kim et al. (2003), Konisky (2005), Maddison (2006), and Eliste and Fredriksson (forthcoming). Fredriksson and

[10] Maximum likelihood techniques have been used by Besley and Case (1995) and Brueckner and Saavendra (2001). These methods however can be computationally demanding (see Konisky, 2005).

Millimet (2002b) for example analyse whether U.S. states are engaged in strategic environmental policy-making. They regress two measures of environmental policy stringency in state *i* on a number of state *i*'s characteristics (e.g., population, population density, degree of urbanisation, per capita state income), as well as a spatially weighted variable of other states environmental policy stringency. They find that states are influenced by both their contiguous and regional neighbours in that they are 'pulled' to higher levels of abatement costs by improvements in neighbours with regulations that are already relatively stringent. Levinson (2002) extends this study by examining whether regulatory competition became more severe during the Reagan Administration, testing the hypothesis that competition should become more intense during periods of greater state control of environmental policy. He does not find convincing evidence for this. Fredriksson et al. (2004) consider strategic behaviour across multiple interrelated policies, namely environmental regulation, taxation, and infrastructure investment and find that uni-dimensional studies may provide lower bound estimates of strategic interaction. Eliste and Fredriksson (forthcoming) find evidence for strategic interaction across countries in the determination of environmental policy in the agricultural sector. The degree of regulatory interaction is found to depend on geographical distance and the degree of openness to trade between trade partners. Murdoch et al. (1997) derive an econometric specification for the demand of sulphur and nitrogen oxides that adjusts for the spatial dispersion of the pollutant in European countries. They include a spatially lagged variable of voluntary sulphur or NO_x emission reductions as an independent variable and find that strategic behaviour, whereby a country limits its cleanup efforts as others reduce emissions, characterises both problems but appears stronger for NO_x. Konisky (2005) examines strategic interaction in the U.S. with regard to water, air and waste regulation and finds strong evidence in support of this. Maddison (2006) is the first to employ spatial econometric techniques in the estimation of EKCs for sulphur dioxide, nitrogen oxides, VOC's and carbon monoxide emissions. He finds that SO_2 and NO_x emissions are positively influenced by neighbouring countries' emissions and finds no evidence that per capita emissions are affected by proximity to high per capita income neighbours. Finally, Kim et al.

(2003) develop a spatial-econometric hedonic housing price model and estimate the marginal value of improvements in sulphur dioxide and NO_x concentrations for the Seoul metropolitan area. They find that the spatial lag model is valid for their specific housing market.

With regard to the strategic behaviour of governments in the determination of taxes, Case (1993), Besley and Case (1995), and Hernandez-Murillo (2003) examine the strategic interaction in (income) tax policies among U.S. states. See also Brueckner and Saavendra (2001) for property taxes, and Heyndels and Vuchelen (1998) for local income and property taxes among Belgium municipalities. Levinson (2002) examines the interaction in the setting of hazardous waste disposal taxes among U.S. states.

Thus, the majority of the studies have focused primarily on the U.S., and no previous study examines the possible existence of strategic interaction or behaviour of environmental policy in an OECD country context. Furthermore, no previous study has examined this issue in the context of municipal solid waste management. Given the magnitude of the waste problem and the large fraction of total environmental expenditures on this resource (i.e. 40% in the EU – see chapter 1 for more detail), this is an important issue that merits further consideration.

The remainder of the chapter explores the spatial aspects of waste generation and disposal management. Three reasons for spatial autocorrelation in particular seem pertinent in the context of waste disposal management. These are (i) policy-mimicking, (ii) cross-border waste trade, and (iii) recycling technology spillovers and diffusion effects.

(i) Policy-mimicking may be induced by a greater degree of political interactions and information exchange in neighbouring countries. This could be due to historical relationships (e.g. Norway-Sweden-Finland) or a lack of language barriers (e.g. Germany-Austria, France-Switzerland). Policy-mimicking may also be related to similar socio-cultural habits, sense of civic solidarity, and "environmental awareness", as well as to economising on the costs of policy-setting. See Ladd (1992) on the mimicking of local tax burdens among 248 counties in the U.S.

(ii) There may be similar landfill prices in neighbouring countries for competitive reasons, i.e., to prevent cross-boundary dumping of waste from countries with high landfill prices to countries with low landfill prices, and analogously for landfill taxes. This would imply some form of regulatory or tax competition. Indeed, Levinson (1999) has found that in the U.S., waste disposal taxes deter interstate transport of hazardous waste. Furthermore, the cost of transporting waste to far away places implies spatial restrictions[11].

(iii) Neighbouring countries may have similar recycling technologies that lower the cost of recycling. Research and development (R&D) spillovers may be restricted in space implying that geographical proximity matters. Case (1992) for example examines the adoption of technological innovations in a study on new harvesting tool among Indonesian farmers. Likewise, there could be spatial aspects in terms of specializing production and generalizing consumption patterns (global branding and life-styles), called diffusion effects.

These avenues for spatial interaction can fall under a number of the theoretical roots of strategic government interaction, as discussed by Brueckner (2003). Brueckner categorises these into spillover models (environmental and yardstick competition models) and resource flow models (tax competition and welfare competition).

[11] Thus perhaps a neighbouring country will wish to raise landfill taxes to avoid imports of waste but countries further away from the exporter do not need to do so because the resulting transport costs do not make it economically viable to export so far. If this is the case, distance from a country will affect waste policy. Note however that Britain is exporting large amounts of waste to China where the cost from the UK for a 26-tonne container of waste plastic is £500. Germany, Italy, and the Netherlands tend to be net exporters of waste whereas Spain, Great Britain, Denmark and France are net importers. Source: Dopp, J.,1997 http://www.american.edu/TED/mswen.htm

To test for the presence of spatial interactions and strategic environmental policy-making in the waste management context, an adequate measure of national waste regulations is required. Since such a measure does not exist, the data to be used is the same as that in Chapter 4 where the waste data serve as a proxy for waste management performance. Moreover, the data on landfill taxes will is used to test for strategic behaviour in waste management policy-making.

5.4 Spatial Econometric Model and Results

In order to test for spatial effects in the determinants of MSW generation and disposal, the analysis in Chapter 3 and 4 is augmented by spatially weighted values of the dependent and independent variables. To do this, it is first necessary to prepare the data for the spatial-econometric analysis. The first step is to create a weight matrix.

The most popular approach for a weight matrix, which also ensures exogeneity, is for the matrix to reflect the inverse geographical distances between observations (i.e., inverse distance and inverse distance squared). Alternative forms of weighting include contiguity weights, simple average weights, population weights, and gravity weights[12]. All five forms of weight matrices are created and tested for in the data. These are described in turn:

The inverse distance weight: To create the inverse geographical distance weight matrix, denoted W^D, the geographic co-ordinates of OECD country centroids are obtained from the CIA WorldFact Book[13] where data on the centroid in Cartesian space is represented by latitude and longitude. The Haversine formula is used to calculate the distances between latitude and longitude from each country (see Appendix 1 for a description).

[12] For example, inverse distance weights have been used by Hernandez-Murollo (2003), Levinson (2003), Konisky (2005), Madisson (2006); Contiguity weights by Fredriksson and Millimet (2002), Konisky (2005); Population weights by Brueckner and Saavendra (1999) and Fredriksson and Millimet (2002).

When using the Haversine formula, it is important to have positive and negative values correspond appropriately with North (+), South (-), East (-), and West (+), so that the distances are calculated correctly. The leading diagonals should equal to zero. Once the distance matrix D_{ij} is obtained, the inverse distance matrix $1/ D_{ij}$ can be calculated, which is of course time invariant (Appendix 2 provides a subsection of the distance matrix for illustrative purposes). The inverse distance weight is meant to examine whether a country competes with all other countries, and that the degree of competition is a function of proximity. In effect, by using the inverse distance weight, an attempt is made to identify whether, via policy diffusion, transfer, or other, a country adopts policies that have been implemented in other countries in close proximity.

The contiguity weight: The contiguity weight matrix is created by assigning a value of 1 to a contiguous neighbour and a value of zero if otherwise, and is denoted W^C. This implies that $\Sigma_j w_{ijt}Y_{jt}$ simplifies to the mean of the environmental stringency in neighbouring states. As in the inverse distance weight, the weights for each country are time invariant. The contiguity weight is very similar in nature to the inverse distance weight except that it only considers whether a country is influences by policies implemented by their contiguous neighbours.

The population weight: The population weight is created whereby competing country j is given a weight equal to its population in time t. The resulting weight matrix, whose representative element is $w_{ijt} = P_{jt}$ for $j \neq i$ is denoted W^P (Brueckner and Saavendra, 1999). The use of population in the weight matrix aims to elicit whether countries may be more responsive to neighbouring states that are larger or where the generation of pollution is greater. In contrast to the inverse distance and contiguity weight, the population weight is time variant.

[13] http://www.cia.gov/cia/publications/factbook/

The gravity weight: This weight is created by calculating the product of the two populations in country *i* and *j* divided by the square of the distance D_{ij}^2 and is denoted W^G. It therefore reflects a combination of the inverse distance weight and the population weight. It is reasonable to expect that a country may be influenced jointly by the distance to as well as the size of neighbouring countries.

The simple average weight: The simple average weight is created by assigning a weight of ΣY_j / J for each t (and is therefore time variant) and is denoted W^{SA}. For example, in the case of MSW generation, the average MWPC over all j countries is obtained by adding MWPC over all countries (except country *i*) and dividing by *j* (= 29). This weight is used to assess whether the average performance in the OECD countries is affecting country *i*. For example, if the general trend is a relaxation of environmental stringency, a country may be tempted to relax its own environmental policies to prevent adverse competitiveness effects.

With the exception of the simple average weight, all of the weights are row standardised[14] using $(w_{ijt}) / \Sigma\, w_{ijt}$ i.e., the spatial weight matrix is normalised so that the rows sum to unity: Thus for each *i*,

$$\sum_j w_{ijt} = 1$$

This normalisation facilitates the interpretation and makes the parameter estimates of alternative models comparable (see Anselin, 2002).

One additional type of weight was constructed, referred to as the Ybest weight. This weight is created by calculating the difference between y_{it} and the best level of y_{jt} in

[14] The simple average weight is not row-standardised because of the nature of how the weight is constructed.

that year, and is denoted by W^B. The purpose of this weight is to examine whether country *i* is affected by the leader in waste performance and regulation[15].

In the spatial econometrics literature, all of the studies examine more than one type of spatial weight. The nature of this analysis is somewhat exploratory and the purpose is to identify whether there is indeed a spatial relationship, and if so, to identify in what manner it operates most strongly. With the exception of the Ybest weight, each of the weights described above are commonly found in the literature, and have been applied to a number of different contexts.

In spatial econometric analysis, dealing with missing values in the data is not simply a case of dropping the observation in question. This is because, depending on the type of the weight matrix employed, if just one country has a missing value, then some or all of the spatially weighted variables cannot be created (Maddison, 2006). Ignoring the problem leads to error-in-variables bias (Cressie, 1993). Due to missing data in the waste data, it was necessary to linearly interpolate some of the variables so that the spatially weighted variables could be created.

Stationarity was then tested for in the data using the Im, Pesaran, Shin (2003) test. This is because in the absence of stationarity in the data, the regression is subject to the risk of spurious results (see e.g., Perman et al. 2003). Results from the IPSHIN command in STATA 9.0 indicated the presence of non-stationarity which was consequently removed once the data had been first-differenced[16].

$$\Delta y = y_t - y_{t-1}$$

The results from the tests are summarized in Table 5.1.

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[15] The Ybest weight attempts to examine the possibility of the "California" effect. See Fredriksson and Millimet (2002a) for a similar example. Note that it is also not possible to row standardise this weight.
[16] Note that Besley and Case (1995) use first-differenced data for their *y* and *wy* variables as they are interested in state' *changes* in tax liabilities, rather than on states' *levels*. Moreover, Figlio et al. (1999) also choose to estimate their model of state benefits in first-differences rather than levels, to account for the trend that state benefits have been trending downwards over the time period for which they had data.

Table 5.1: Panel Unit Root Test Statistics

	Without time trend	With time trend	
Log(GDPPC)	-1.539	-2.111	Do not reject unit root null
Δlog(GDPPC)	-3.040	-3.253	Reject unit root null
Log(MWPC)	-1.556	-1.726	Do not reject unit root null
Δlog(MWPC)	-1.858	-2.206	Reject unit root null at 5%
Log(LDFL)	-0.650	-2.130	Do not reject unit root null
Δlog(LDFL)	-2.376	-2.574	Reject unit root null
Log(PAPER)	-1.306	-1.746	Do not reject unit root null
Δlog(PAPER)	-3.258	-3.745	Reject unit root null
Log(GLASS)	-1.944	-1.731	Do not reject unit root null
Δlog(GLASS)	-2.882	-3.220	Reject unit root null

Each of the spatial weight matrices were then applied to the first-differenced data to create the spatially weighted variables.

As explained in section 5.2 above, two estimation techniques are used to test for spatial interaction, namely ordinary least squares (OLS) and instrumental variables (IV). OLS may be used for spatio-temporal models. Since influences from neighbouring countries are not assumed to occur instantaneously but rather with a time lag, three spatially weighted temporal lags were created, (t-1), (t-2), (t-3) for each of the dependent variables. These temporal lags also serve to circumvent the problem of potential endogeneity of environmental policies of other countries. More specifically, if there is some form of strategic interaction among the countries with regard to how they select their waste management policies, then this may cause concern regarding the direction of causation. The temporal lags eliminate this concern and will control the bias that may arise due to spatially correlated, time-specific unobservables (Frederiksson and Millimet, 2002; Levinson, 2002).

The second estimation procedure adopted is to instrument for the spatial lags. IV estimation is necessary when purely spatial models are examined, and also has the benefit of providing consistent estimates of the parameters even in the presence of spatially correlated error terms (Kelejian and Prucha, 1998; Saavendra, 2000). This is very important because the presence of spatially correlated unobservables could lead one to incorrectly conclude that strategic behaviour is evident. Following Figlio et al. (1999) and Fredriksson and Millimet (2002), among others, the instruments used are some of the attributes included in x_{it} for neighbouring states.

Both of the estimation procedures are run with and without country and time fixed effects, the significance of which are jointly tested using a Wald test as performed by the testparm command in STATA 9.0. The country fixed effects capture time-invariant country-specific attributes. The time fixed effects will control for events that occur in a given period and may impact all countries through a reshaping of attitudes (Fredrkisson and Millimet, 2002).

In examining the nature of the spatial interaction in the data, the estimated regression equation takes the general form:

$$\Delta MWPC_{it} = \alpha_i + \gamma_t + \delta \Sigma \omega_{ijt} \Delta MWPC_{jt} + \beta \Delta X_{it} + \varepsilon_{it}$$

where $\Delta MWPC_{it}$ is the waste variable in country *i* at time *t*; α_i are country fixed effects; γ_t are time fixed effects; ω_{ijt} is the weight assigned to country *j* by country *i* at time *t* ($j \neq i$), where some of the weights may be zero, $\Delta MWPC_{jt}$ is the measure of relevant waste variable in country j; δ is the parameter of interest; X_{it} is a vector of country characteristics (i.e., GDP per capita, population density, and urbanisation); and ε_{it} represents idiosyncratic shocks uncorrelated across time and space. Analogous equations are specified for the proportion disposed at landfill, paper or glass recycled, and finally, the level of the landfill tax.

As explained above, the test for spatial interaction among countries requires the testing of the significance of δ; a non-zero coefficient implies that one country's waste or recycling performance depends on the performance in other countries. The measures of waste management performance are the same as before: (1) MWPC generation, (2) the proportion of MWPC disposed of at landfills, (3) the proportion of paper and cardboard consumed that is recycled, and (4) the proportion of glass consumed that is recycled. The final issue examined is (5) whether the introduction and/or change in landfill taxes in one country is influenced by landfill tax policy in neighbouring countries.

The OLS (i.e. the spatio-temporal model) and IV parameter estimates for these results are presented in tables 5.2 and 5.3 respectively and are discussed below. Each of the regressions is run with and without country and time fixed effects (FE), and all the regressions include per capita GDP, population density, and urbanisation. 'Robust' estimates are also obtained to account for potential heteroskedasticity in the data. For the IV estimation, the instrument set includes population density and urbanisation from neighbouring countries, using the same weighting scheme as the dependent variable[17].

[17] As such, IV estimation for the Ybest weighted variables was not undertaken.

Table 5.2. OLS with and without FE

	Without FE		With FE			Wald test
	Coeff	*t-stat*	*Coeff*	*t-stat*	*robust*	*Prob>F*
Contiguity						
Mwpct-1	0.156	2.37	0.057	0.73	0.91	0.0000
Mwpct-2	0.157	2.39	0.055	0.68	0.76	0.0000
Mwpct-3	0.160	2.43	0.059	0.68	0.70	0.0000
Ldflt-1	0.223	1.64	0.037	0.23	0.41	0.0004
Ldflt-2	0.369	2.64	0.122	0.76	0.66	0.0010
Ldflt-3	0.318	1.38	-0.201	-0.67	-0.40	0.0003
Papert-1	-0.073	-1.01	-0.05	-0.68	-0.83	0.0000
Papert-2	-0.134	-1.76	-0.12	-1.55	-1.68*	0.0000
Papert-3	0.138	1.67	0.134	1.58	1.64	0.0000
Glasst-1	0.107	1.96	0.031	0.52	0.35	0.0446
Glasst-2	0.049	0.89	-0.02	-0.32	-0.26	0.0461
Glasst-3	0.11	1.94	0.047	0.76	0.52	0.0757
Ldtxt-1	0.151	1.88	0.147	1.68	0.98	0.0341
Ldtxt-2	-0.053	-0.63	-0.07	-0.78	-1.47	0.0395
Ldtxt-3	0.043	0.50	0.00	0.00	0.01	0.0562
Distance						
Mwpct-1	0.56	2.54	-0.299	-0.99	-1.13	0.0000
Mwpct-2	0.576	2.61	-0.089	-0.28	-0.30	0.0000
Mwpct-3	0.609	2.74	0.175	0.53	0.55	0.0000
Ldflt-1	-0.026	-0.10	0.579	1.11	1.34	0.0001
Ldflt-2	0.596	2.31	0.817	1.52	1.70*	0.0004
Ldflt-3	0.236	0.69	1.92	2.21	1.33	0.0001
Papert-1	-0.177	-1.16	0.025	0.08	0.07	0.0000
Papert-2	-0.475	-3.01	-0.653	-1.97	-1.95*	0.0000
Papert-3	0.140	0.85	0.351	1.00	0.94	0.0000
Glasst-1	0.594	3.44	0.063	0.21	0.15	0.1420
Glasst-2	0.419	2.31	-0.0774	-0.25	-0.19	0.0936
Glasst-3	0.534	2.79	-0.0148	-0.05	-0.03	0.1515
Ldtxt-1	0.108	1.09	0.33	1.34	0.81	0.0276
Ldtxt-2	-0.095	-0.95	-0.185	-0.72	-1.64	0.0437
Ldtxt-3	0.042	0.40	-0.09	-0.36	-0.75	0.0542
Population						
Mwpct-1	-0.24	-1.42	-10.46	-13.53	-14.13**	0.0000
Mwpct-2	-0.30	-1.79	-8.67	-10.28	-10.13**	0.0000
Mwpct-3	-0.32	-1.87	-6.72	-7.25	-6.43**	0.0000
Ldflt-1	-0.49	-0.57	5.15	1.02	1.05	0.0002
Ldflt-2	-0.11	-0.16	6.13	1.22	1.43	0.0001
Ldflt-3	-0.32	-0.5	13.54	2.59	1.28	0.0000
Papert-1	-0.17	-0.77	-2.05	-1.44	-0.88	0.0000
Papert-2	-0.53	-2.34	3.79	2.63	1.98**	0.0000
Papert-3	-0.34	-1.41	1.77	1.17	0.67	0.0000
Glasst-1	0.45	2.82	-0.58	-0.63	-0.54	0.0831
Glasst-2	0.52	2.97	-0.68	-0.67	-0.83	0.1357

Glasst-3	0.87	4.82	-0.09	-0.09	-0.12	0.6051
Ldtxt-1	0.19	0.77	3.46	1.35	1.64*	0.0247
Ldtxt-2	-0.21	-0.84	2.08	0.79	1.51	0.0414
Ldtxt-3	0.15	0.58	3.09	1.15	1.54	0.0451
Gravity						
Mwpct-1	0.16	1.81	0.04	0.36	0.63	0.0000
Mwpct-2	0.16	1.84	0.06	0.61	0.96	0.0000
Mwpct-3	0.17	1.97	0.12	1.04	1.49	0.0000
Ldflt-1	0.59	2.40	0.41	1.18	1.08	0.0006
Ldflt-2	0.63	2.55	0.05	0.14	0.10	0.0011
Ldflt-3	0.40	1.46	0.63	1.32	1.07	0.0002
Papert-1	-0.164	-1.64	-0.17	-1.32	-1.32	0.0000
Papert-2	-0.33	-3.28	-0.32	-2.37	-2.62*	0.0001
Papert-3	0.066	0.63	0.21	1.46	1.49	0.0000
Glasst-1	0.292	3.14	0.16	1.35	1.07	0.0800
Glasst-2	0.251	2.61	0.11	0.93	0.87	0.1039
Glasst-3	0.288	2.94	0.05	0.44	0.27	0.1515
Ldtxt-1	0.14	2.58	0.18	1.30	0.91	0.0352
Ldtxt-2	-0.072	-0.77	-0.08	-0.56	-1.52	0.0430
Ldtxt-3	-0.039	0.41	-0.05	-0.37	-0.72	0.0541
Simple Average						
Mwpct-1	0.0002	0.30	-0.204	-3.62	-3.22*	0.0000
Mwpct-2	0.000019	0.25	-0.0016	-2.57	-2.58*	0.0000
Mwpct-3	0.0004	0.46	-0.0075	-1.05	-1.02	0.0000
Ldflt-1	-0.0008	-2.46	0.0044	2.00	1.22	0.0051
Ldflt-2	0.00055	1.41	0.0048	2.18	1.27	0.0694
Ldflt-3	-0.0007	-1.08	0.01834	5.81	2.75*	0.0004
Papert-1	-0.0002	-1.33	0.0012	1.19	0.49	0.0000
Papert-2	-0.00039	-2.14	0.0041	3.88	2.48*	0.0000
Papert-3	1.26 x 10-6	0.01	0.0004	0.06	0.03	0.0000
Glasst-1	0.003	1.27	0.0004	0.31	0.19	0.0308
Glasst-2	-0.00001	-0.07	-0.0007	-0.40	-0.36	0.0386
Glasst-3	0.0002	0.75	0.0022	1.67	1.71*	0.0280
Ldtxt-1	0.0003	0.68	0.0072	2.09	2.04*	0.0147
Ldtxt-2	-0.0004	-0.70	0.0058	1.64	2.09*	0.0273
Ldtxt-3	0.0003	0.61	0.0077	2.08	2.17*	0.0258
Ybest						
Mwpct-1	0.00006	6.52	0.00017	7.43	6.69*	0.0000
Mwpct-2	0.00006	7.02	0.0002	9.06	8.06**	0.0000
Mwpct-3	0.00007	7.43	0.0002	10.20	8.77**	0.0000
Ldflt-1	-0.0016	-4.16	0.0050	3.30	2.78**	0.0006
Ldflt-2	-0.0012	-3.24	0.0013	1.38	1.58	0.0016
Ldflt-3	-0.0009	-2.49	0.0009	1.16	1.26	0.0007
Papert-1	-0.0005	-1.41	0.0021	2.58	1.29	0.0000
Papert-2	-0.0004	-0.96	0.0012	1.36	0.66	0.0000
Papert-3	-0.0006	-1.52	-0.0006	-0.66	-0.30	0.0000
Glasst-1	-0.00003	-0.11	0.0028	4.95	4.04**	0.0001

Glasst-2	0.00004	0.15	0.0029	4.66	3.96**	0.0002
Glasst-3	-0.00002	-0.10	0.0031	4.69	4.11**	0.0003
Ldtxt-1	0.0077	1.11	0.1119	5.48	2.47**	0.0000
Ldtxt-2	0.0068	0.92	0.1089	4.96	2.38**	0.0002
Ldtxt-3	0.0086	1.10	0.1109	4.64	2.33**	0.0007

Significant at: **<5% and *10%

Table 5.3. IV with and without FE

	Without FE			**With FE**			**Wald test**
	Coeff	*t-stat*	*robust*	*Coeff*	*t-stat*	*robust*	*Prob>F*
Contiguity							
mwpc	0.574	2.72	3.30	-2.850	-1.69	-1.17	0.0444
ldfl	0.247	0.30	0.33	-0.538	-0.96	-0.87	0.0055
paper	0.017	0.03	0.05	0.216	0.34	0.46	0.0002
glass	0.538	0.56	0.53	0.311	1.04	0.96	0.5744
ldtx	1.062	0.83	2.17	1.266	0.37	0.75	0.9969
Distance							
mwpc	2.108	1.03	1.23	-0.455	-0.27	-0.36	0.0000
ldfl	0.257	0.19	0.21	-2.695	-1.20	-0.97	0.0003
paper	1.05	1.00	0.97	1.02	0.20	0.24	0.0054
glass	1.08	2.44	2.04	1.99	0.82	0.92	0.9830
ldtx	0.873	2.18	2.40	3.47	0.97	1.15	1.0000
Population							
mwpc	0.371	1.73	1.70	-5.39	-1.19	-1.57	0.0000
ldfl	2.604	0.48	0.63	-35.67	-1.15	-0.88	0.0000
paper	1.608	0.71	0.99	-1.92	-0.11	-0.18	0.0000
glass	0.75	4.23	4.41	-3.11	-1.72	-2.85*	0.0814
ldtx	1.89	2.49	1.91	11.48	0.04	0.11	0.9999
Gravity							
mwpc	0.134	0.43	0.49	-1.37	-3.42	-2.20	0.0000
ldfl	0.5763	0.71	0.66	-2.577	-0.98	-0.63	0.0184
paper	0.491	0.52	0.80	1.13	0.68	0.90	0.0077
glass	-0.216	-0.19	-0.18	0.047	0.08	0.10	0.03929
ldtx	1.242	1.58	2.62	1.32	0.87	1.68*	0.9999
Simple Average							
mwpc	0.0004	0.79	0.66	0.0156	2.14	2.43*	0.0000
ldfl	0.0024	2.49	2.55	-0.0336	-3.28	-1.94*	0.0000
paper	0.0002	0.39	0.49	-0.047	-2.06	-1.34	0.0000
glass	-0.00017	-0.21	-0.21	-0.0207	-7.93	-6.99**	0.0000
ldtx	0.0027	2.96	2.46	-0.0625	-14.71	-7.04**	0.0000

Significant at: **<5% and *10%

The overid command in STATA computes versions of Sargan's (1958) and Basmann's (1960) tests of overidentifying restrictions for a regression estimated via instrumental variables in which the number of instruments exceeds the number of regressors: that is, for an overidentified equation. These are tests of the joint null hypothesis that the excluded instruments are valid instruments, i.e., uncorrelated with the error term and correctly excluded from the estimated equation. A rejection casts doubt on the validity of the instruments.

Table 5.4 Sargan and Basman Tests

	Sargan test	**Basman test**
Contiguity		
mwpc	0.671 (0.1962)	1.659 (0.1978)
ldfl	0.758 (0.3410)	0.737 (0.3908)
paper	0.005 (0.9430)	0.005 (0.9433)
glass	0.311 (0.5772)	0.308 (0.5791)
ldtx	0.053 (0.8188)	0.052 (0.8195)
Distance		
mwpc	0.620 (0.4310)	0.615 (0.4331)
ldfl	2.310 (0.1285)	2.265 (0.1324)
paper	0.280 (0.5969)	0.277 (0.5987)
glass	0.146 (0.7021)	0.145 (0.7036)
ldtx	0.034 (0.8527)	0.034 (0.8533)
Population		
mwpc	0.935 (0.3335)	0.927 (0.3355)
ldfl	0.777 (0.3782)	0.755 (0.3849)
paper	1.449 (0.2288)	1.437 (0.2306)
glass	1.392 (0.2381)	1.380 (0.2401)
ldtx	0.006 (0.9380)	0.006 (0.9382)
Gravity		
mwpc	9.892 (0.0017)	9.958 (0.0016)*
ldfl	1.301 (0.2541)	1.268 (0.2601)
paper	0.221 (0.6380)	0.219 (0.6397)
glass	0.325 (0.5686)	0.322 (0.5705)
ldtx	0.001 (0.9731)	0.001 (0.9732)
Simple Average		
mwpc	0.4333 (0.5106)	0.429 (0.5126)
ldfl	0.107 (0.7432)	0.104 (0.7471)
paper	1.002 (0.3169)	0.993 (0.3190)
glass	10.545 (0.0012)	10.613 (0.0011)*
ldtx	1.129 (0.2882)	1.121 (0.2897)

P-values are in parenthesis; (*) indicates instruments are not exogenous

As one can see from the results in Table 5.3, in most cases the OLS-FE and IV-FE is the appropriate model as indicated by the magnitude of the Wald tests. In a small number of cases, the fixed effects are not significant. For these regressions, the Wald test was repeated once the robust command was included in the regression. With the exception of FDWIV for ldtx, these all yielded results of Prob> F = 0.0000 suggesting that the model with fixed effects is the appropriate one. The model without fixed effects is more appropriate for the landfill tax model for all weights used, with the exception of the simple average weighting model.

The results from the Sargan tests in Table 5.4 indicate that in nearly all cases, the instruments are exogenous. Fredriksson and Millimet (2002) also obtain results where in some regressions their instruments are exogenous and in others they are not, despite using the same instruments in all their regressions.

The results reveal that the existence of spatial interaction in waste management performance and landfill tax policy is dependent on the type of spatial weight that is adopted. For each variable examined, at least two weights indicate the presence of spatial interaction in the data. The results are summarised in Table 5.5[18].

[18] Recall that the sample average and Ybest weights are not row standardised and are therefore not directly comparable to the other weights. Moreover, the Ybest weight created here presents one approach for the weight construction to examine the role of environmental leadership. Additional weights should ideally be constructed to more rigorously analyse this topic but this lies beyond the scope of this existing study (see Fredriksson and Millimet 2002a for alternative weighting schemes to assess whether a country is affected by the environmental leader).

Table 5.5. Summary of Significant Results

	Significant weights
OLS	
Mwpc	Population, Simple Average, Ybest
Ldfl	Distance, Simple Average, Ybest
Paper	Contiguity, Distance, Population, Gravity, Simple Average
Glass	Simple Average, Ybest
Ldtx	Population, Simple Average, Ybest
IV	
Mwpc	Simple Average
Ldfl	Simple Average
Paper	-
Glass	Population, Simple Average
Ldtx	Gravity, Simple Average

An anomaly in the results occurs in the population-weighted data for MWPC in that the coefficients are much larger. In particular, the coefficients in the results without the fixed effects look "normal" but once the fixed effects are introduced, the coefficients increase dramatically[19].

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[19] The waste data has been checked, and is correct and identical to the data used in all other regressions. The original population weight data was also checked. This is correct given the way in which it has been defined, which is the same as that in F&M (2002). I have experimented by deleting an outlier (Spain, 2000), and the coefficients in FDWIVreg become even larger (on absolute scale) and more significant. (Results with lagged variables would not change because t-1 from the year 2000 would have that data deleted anyway). The nature of this particular weight is such that each element of waste data is multiplied by exactly the same weight (except the appropriate zero's, see table below) hence perhaps small differences in the waste data result in magnified changes using this weight matrix.

Stylised Example of Population Weight Matrix

	AUS	AUT	BEL	DEN
AUS	0	X	Y	Z
AUT	W	0	Y	Z
BEL	W	X	0	Z
DEN	W	X	Y	0

The coefficients on the spatio-temporal weights on LDFL are positive and statistically significant in three of the six weighting indices. They are negative and statistically significant in the IV simple average scheme and albeit negative, statistically insignificant in the other IV models. The coefficients are negative on the temporally lagged simple average weights and positive on the Ybest weights. Note that Fredriksson and Millimet (2002) and Konisky (2005) for example also find positive and negative coefficients depending on the weights that are used.

With regard to paper recycling rates, though the results reveal that the spatially augmented dependent variables are not significant in the instantaneous case where IV is used, the spatio-temporal lags however are statistically significant in five out of six formulations of the weighted variables. The coefficients on the spatio-temporal contiguity, distance and gravity weighted data are negative in the 2-year lag. It is positive in the analogous population and simple average weights, and positive and statistically significant in the first lag of the Ybest weight.

Referring to the performance of glass recycling, this tends to be positive with the spatio-temporally weighted data (with a 3 year lag in the simple average weight, and for all lags with the Ybest weight) whereas in the IV results, the coefficients are negative (i.e., for the population and simple average weights).

With regard to the differences in the weights in relation to the results where they have found to be significant, the following points are in order:

- The results using the Ybest weights need to be conservatively interpreted as this represents just one possible weighting methodology to evaluate whether countries are influenced by the leader in environmental policy. Though the results present some preliminary evidence that the Ybest weight is significant, further analysis is warranted to analyse the existence of this effect with the use of additional weighting approaches.

- It is not possible to directly compare the results from the simple average weights and the distance, contiguity, population and gravity weights because the simple average weight has not been row-standardized.

- The OLS results from the regression on paper/cardboard recycling provide the most concrete evidence for the existence of spatial interaction, as is revealed by the number of weights with statistically significant coefficients. The results using weights that incorporate an element of distance, imply that an increase in the neighbours recycling rate will lead to a decrease in one's own recycling rate (for paper/cardboard). The results using the population weight indicate that an increase in a neighbours recycling rate where population levels are high will lead to an increase in one's own recycling rate.

The results provide some support to the evidence by Konisky (2005) who examines state enforcement of hazardous waste pollution control regulation, namely the Resources Conservation and Recovery Act (RCRA) across the U.S. Using data on the annual number of sampling inspections taken by state governments divided by the number of regulated facilities under RCRA, he finds strong evidence of strategic interaction. More specifically, using contiguity, inverse distance and other weights, he finds that a 10% increase in competitors enforcement effort leads to about a 6% to 16% increase in one's own enforcement effort.

The results for the case of the landfill tax are elaborated in more detail as this is arguably the more interesting variable to examine. To summarise briefly, the landfill tax change is modelled as a function of state economic variables (change in income per capita) and state demographic variables (change in POPD and URB), as well as changes in landfill taxes in neighbouring countries. This is quite similar to a model by Besley and Case (1995) for tax-setting behaviour across U.S. states. Note that the nominal landfill tax, as opposed to the real landfill tax (adjusted for inflation) is the relevant variable of interest because this is the level of the tax as perceived by the public[20]. There may be strategic interaction in the imposition and level of landfill taxes as voters and politicians tend to be sensitive to events outside their boundaries. Introducing a tax may be easier for a government who can refer to similar taxes in comparable regions (Heyndels and Vuchanen, 1998). In the OLS model, the results are statistically significant when the population, simple average and Ybest weights are used; in the IV model, the results are statistically significant when the gravity and simple average weights are used. Recall that the gravity weight is a function of both population and distance from country *i*, thus the results from these weights (which are comparable as they are both row-standardised) lend support to the conjecture of spatial interaction in waste policy, suggesting that a government reacts to landfill taxes introduced in larger and geographically more proximate countries. Specifically, a 10% increase in another countries landfill taxes will lead to a 16.4% to 16.8% percent increase in one's own country's landfill tax. The simple average weight (which is not row-standardised) is significant in both models. Furthermore, these weights are positive in four of the cases, and negative in the IV simple average weight model. A positive parameter coefficient indicates that as neighbouring countries increase their landfill disposal taxes, country *i* will increase its' own landfill tax in response. The positive and significant coefficient on the Ybest weight provides some indication that countries may perhaps be responding to the environmental leader when selecting their landfill tax policy, and is therefore in support of the so-called California effect. However, further analysis would be necessary to assess the robustness of these results, including the use of different types of weights and models to examine this issue in more detail.

In comparing these results with the IV results of Levinson (2002) on U.S. hazardous waste taxes, he finds that the instrumented variable of other states hazardous waste taxes is insignificant when they are weighted by an inverse distance square and by tons of waste exported. However when these weights are combined with a post-1992

[20] Case (1993), Besley and Case (1995), Hernandez-Murillo (2003), Brueckner and Saavendra (2001) and Levinson (2002) use nominal tax rates in their analysis as well.

dummy[21], the coefficients on the parameters (0.52 and 0.57 respectively) become statistically significant.

5.5 Conclusions and Policy Implications

The theoretical literature on interjurisdictional environmental regulatory competition is now well established. This paper adds to the empirical literature on interactive environmental policy behaviour by examining waste management performance and policy-making. Using spatially weighted values of the dependent variables, this chapter has investigated the degree of national policy interdependence in waste management performance and landfill tax-setting across OECD countries. The results reveal that some form of spatial interactions are present in the data, and that these are dependent on the type of spatial weight that is adopted. This has important implications for practitioners in the field of interjurisdictional policymaking in that the selection of weighting methodology might lead one to conclude that there is no strategic behaviour when an alternative weighting structure may have led to the opposite conclusion, and vice versa. The importance of selecting the most appropriate weights, and that indeed several weights should be tested for, should not be underestimated.

In addition, given the restrictive assumptions under which local environmental authority will lead to efficient regulations, it is unlikely that the waste policies selected are efficient. If it is the case that tax competition leads to inefficiently low taxes on pollution and reduces welfare, then the policy implications are that co-ordination of environmental taxes or standards among a group of countries may improve welfare under certain circumstances. The results presented here provide some evidence to suggest that waste management and landfill taxes in one country do impact the decisions of neighbouring countries. This is consistent with the literature

[21] In 1992, the Supreme Court ruled the practice of states explicitly imposing higher taxes on disposal of waste by out-of-state entities than they imposed on local waste generators unconstitutional. Levinson (2002) argues that since 1992, the tax asymmetry has taken on more subtle forms.

on strategic environmental policy-making, appealing to capital competition and transboundary pollution spillovers as the motivating factors.

There is also some support to indicate that countries respond to the environmental leader, as is indicated by the results using the Ybest weight. However, these results are preliminary and further analysis is suggested to examine if this finding holds under different model specifications.

A further suggestion for future research is to examine whether countries' regulatory waste management expenditures are influenced by the magnitude of expenditures made in neighbouring countries, as well as to devise alternative instruments for waste management policy. Given the large fraction of waste management expenditures in the total environment budget, waste is a particularly interesting topic to examine in the realm of strategic environmental policymaking.

Appendix 5.1. The Haversine Formula

Presuming a spherical Earth with radius R, and that the locations of the two points in spherical co-ordinates (longitude and latitude) are lon1, lat1 and lon2, lat2, then the Haversine formula is given by:

Δlon = lon2 - lon1

Δlat = lat2 - lat1

a = (sin(Δlat/2))^2 + cos(lat1) * cos(lat2) * (sin(Δlon/2))^2

c = 2 * atan2(sqrt(a), sqrt(1-a))

d = R * c

and will give mathematically and computationally exact results. The intermediate result c is the great circle distance in radians. The great circle distance d will be in the same units as R[22].

The historical definition of a "nautical mile" is "one minute of arc of a great circle of the earth." Since the earth is not a perfect sphere, that definition is ambiguous. However, the internationally accepted (SI) value for the length of a nautical mile is

[22] Most computers require the arguments of trigonometric functions to be expressed in radians. To convert lon1, lat1 and lon2, lat2 from degrees, minutes, and seconds to radians, these must first be converted to decimal degrees. To convert decimal degrees to radians, the number of degrees is multiplied by pi/180 = 0.017453293 radians/degree.

Inverse trigonometric functions return results expressed in radians. To express c in decimal degrees, multiply the number of radians by 180/pi = 57.295780 degrees/radian. (The number of RADIANS must be multiplied by R to get d.)

1.852 km (or 1.151 miles). Thus, the implied "official" circumference is 360 degrees times 60 minutes/degree times 1.852 km/minute = 40003.2 km. The implied radius is the circumference divided by 2π: R = 6367 km = 3956 mi (Source: Math Forum[23]).

[23] http://mathforum.org/library/drmath/view/51879.html

Appendix 5.2. An Example of a Weight Matrix: The Inverse Distance Matrix

lat	long	lat rad	long rad	Country	Australia	Austria	Belgium	Canada	Czech Republic	Denmark	Finland	...etc	Turkey	United Kingdom	United States	SUM
-27	-133	2.042035	-2.32129	Australia	0	0.030037	0.028774	0.029031	0.030388	0.029779	0.031958		0.034521	0.028229	0.028347	1
47.33	-13.33	0.744732	-0.23265	Austria	0.002472	0	0.045411	0.00551	0.113524	0.035855	0.01769		0.017965	0.027196	0.004288	1
50.83	-4	0.683645	-0.06981	Belgium	0.002073	0.03976	0	0.005427	0.037705	0.044532	0.015952		0.011333	0.057807	0.004143	1
60	95	0.523599	1.658063	Canada	0.013579	0.031318	0.035234	0	0.03198	0.036471	0.037611		0.024933	0.038846	0.082402	1
49.75	-15.5	0.702495	-0.27053	Czech Rep	0.002423	0.110025	0.041736	0.005453	0	0.043783	0.020217		0.017679	0.026755	0.004182	1
56	-10	0.593412	-0.17453	Denmark	0.002953	0.043208	0.061291	0.007732	0.05444	0	0.034209		0.016237	0.05379	0.005698	1
64	-26	0.453786	-0.45379	Finland	0.005452	0.036679	0.037776	0.013719	0.04325	0.058858	0		0.025912	0.038239	0.009659	1
46	-2	0.767945	-0.03491	France	0.002956	0.051433	0.080928	0.007437	0.041432	0.03626	0.018118		0.016139	0.048257	0.005869	1
51	-9	0.680678	-0.15708	Germany	0.002119	0.06026	0.088461	0.005233	0.064512	0.055453	0.017695		0.012823	0.038105	0.003988	1
39	-22	0.890118	-0.38397	Greece	0.005269	0.061675	0.037224	0.009422	0.055057	0.034316	0.025651		0.063815	0.029129	0.007596	1
47	-20	0.750492	-0.34907	Hungary	0.002837	0.077851	0.031699	0.005829	0.087152	0.032407	0.020434		0.026136	0.022744	0.004531	1
65	18	0.436332	0.314159	Iceland	0.006107	0.034562	0.046316	0.025007	0.03634	0.051772	0.045281		0.021132	0.061926	0.016318	1
53	8	0.645772	0.139626	Ireland	0.00365	0.035003	0.066902	0.011458	0.034466	0.047544	0.025109		0.015923	0.139091	0.008589	1
42.83	-12.83	0.823272	-0.22393	Italy	0.003405	0.097735	0.044073	0.007168	0.06161	0.033182	0.01963		0.025751	0.029751	0.005719	1
36	-138	0.942478	-2.40855	Japan	0.034221	0.026019	0.025671	0.028982	0.026909	0.027735	0.03241		0.028153	0.025857	0.023964	1
37	-127.5	0.925024	-2.22529	Korea Rep	0.032449	0.026974	0.026322	0.027096	0.027939	0.028549	0.033663		0.030017	0.026325	0.022049	1
49.75	-6.17	0.702495	-0.10769	Luxembourg	0.001993	0.050106	0.151675	0.005006	0.044243	0.040014	0.015023		0.011567	0.040455	0.003848	1
23	102	1.169371	1.780236	Mexico	0.0191	0.028032	0.030443	0.066715	0.02809	0.030196	0.029609		0.02347	0.032271	0.159632	1
52.5	-5.75	0.654498	-0.10036	Netherlands	0.002158	0.040562	0.144653	0.005657	0.042988	0.067161	0.018535		0.0119	0.059149	0.004268	1
-41	-174	2.286381	-3.03687	NZ	0.125899	0.027772	0.027248	0.036593	0.028144	0.028302	0.030416		0.030757	0.027453	0.040444	1
62	-10	0.488692	-0.17453	Norway	0.00363	0.031648	0.040213	0.010343	0.037088	0.078018	0.062318		0.016969	0.045955	0.007293	1
52	-20	0.663225	-0.34907	Poland	0.003105	0.061018	0.038672	0.006866	0.107059	0.054618	0.031268		0.023286	0.029078	0.005195	1
39.5	8	0.881391	0.139626	Portugal	0.004418	0.037165	0.045545	0.011641	0.032952	0.031592	0.020354		0.019475	0.042654	0.00969	1
48.67	-19.5	0.721345	-0.34034	Slovakia	0.002578	0.074353	0.031553	0.005458	0.114119	0.034569	0.020512		0.021899	0.022434	0.004206	1
40	4	0.872665	0.069813	Spain	0.004064	0.039946	0.047462	0.01021	0.034355	0.03132	0.019462		0.019359	0.04111	0.008423	1
62	-15	0.488692	-0.2618	Sweden	0.003711	0.031979	0.037073	0.010042	0.038366	0.07205	0.087528		0.018054	0.039232	0.007111	1
47	-8	0.750492	-0.13963	Switzerland	0.002433	0.088879	0.06963	0.005756	0.05686	0.035593	0.016442		0.015272	0.034251	0.004499	1
39	-35	0.890118	-0.61087	Turkey	0.007953	0.050298	0.036239	0.012281	0.051071	0.037722	0.034989		0	0.030757	0.009799	1

54	2	0.628319	0.034907	UK	0.003029	0.035458	0.086081	0.008911	0.035994	0.058197	0.024046		0.014323	0	0.006637	1
38	97	0.907571	1.692969	USA	0.015287	0.0281	0.031007	0.095008	0.028279	0.030986	0.030529		0.022937	0.033359	0	1

CHAPTER 6

A Choice Experiment to Evaluate Household Preferences for Kerbside Recycling in London

6.1 Introduction

Recent developments in national and European Union (EU) waste management policy has prompted considerable interest into alternative waste management programs that would divert a portion of the municipal solid waste (MSW) stream from landfills. This is particularly relevant in certain European countries that have recently become signatories to the EC Landfill Directive (99/31/EEC)[1] and are far from attaining their targets. A prime example of this is the United Kingdom (UK), that currently has one of the poorest records in Europe with regard to the proportion of MSW that is sent to landfills (Eurostat, 2003). This is in the order of 80%, though it is expected to decrease in the future as a result of government policy, including the implementation of the landfill tax and the requirement that 25% of MSW is recycled[2].

The EC Landfill Directive sets targets to reduce the landfilling of biodegradable municipal waste to 75% of 1995 levels by 2006, 50% by 2009, and 35% by 2016, though for the UK and Greece, these deadlines have been extended[3]. Biodegradable waste is defined as waste that is capable of undergoing anaerobic or aerobic decomposition, such as food and garden waste, and paper and paperboard (Article 2). Failure to meet the targets of the Directive would mean that the UK could face a non-compliance fine of up to £500,000 per day after the first target date in 2010. Furthermore, the government has reserved the right to pass on any European fine imposed on the UK for missing the Landfill Directive targets onto the local authorities or devolved administrations responsible for the UK missing its targets. This could mean that failing councils would be responsible for their share of fines reaching £180 million a year until the Directive's demands are met[4].

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[1] Official Journal L 182, 16/07/1999 P. 0001 – 0019.

[2] Government national recycling targets for England are: 17% recycling or composting by 2003-4; 25% recycling or composting by 2005; 30% recycling or composting by 2010 and 33% recycling or composting by 2015 (Waste Strategy, 2000).

[3] The Directive allows member states which landfilled over 80% of their municipal waste in 1995 to postpone the targets by up to four years. The Government intends to use this four year derogation, making the target dates for the UK 2010, 2013 and 2020 respectively.

In this chapter, a stated preference choice experiment (CE) method is employed to estimate household's valuation, in terms of willingness-to-pay (WTP) for kerbside waste-separation and collection services in London. The purpose of the study is to examine the determinants of household recycling behaviour and to estimate the recycling service attributes that are valued most highly by the public. Recycling service attributes valued in this study include the kerbside recycling of a number of 'dry' materials (i.e., paper, glass, aluminium, plastic, and textiles), the composting of food and garden waste, as well as the frequency of kerbside recycling collection. Facing budget constraints and strict recycling targets, this information could help local authorities to prioritise the recycling services and facilities they offer to their residents.

The contribution of this study to the literature is threefold. Firstly, several studies have employed stated preference methods (e.g., the contingent valuation method, contingent ranking method) to estimate the economic value of recycling (see, e.g., Jakus et al. 1996; Lake et al. 1996; Tiller et al. 1997; Huhtala, 1999; Kinnaman, 2000; Caplan et al. 2002; Aadland and Caplan, 2003, which are reviewed in section 6.2). There is to date only one existing CE study on recycling for Macao, China (Jin et al. 2006)[5]. The CE presented in this paper is the first such study applied to estimation of the WTP for the kerbside collection of dry materials, compost and textiles. Secondly, to this date there is only one study that examines recycling behaviour in London (in the borough of Kensington and Chelsea) (Robinson and Read, 2005). Consequently, there is an urgent need for more information on recycling costs and benefits in London so as to develop efficient and effective recycling services. Finally, studies on composting are limited to Sterner and Bartelings (1999) who study *inter alia* the determinants of composting in a small Swedish municipality, and Kipperberg (2003) who examines composting of yard and food waste in Seattle. Since around 40 percent of household waste could be composted, this is an extremely important part of

[4] www.letsrecycle.com

[5] Another more general application of CE to waste management does exist, namely that by Garrod and Willis (1998) who examine lost amenity due to landfill waste disposal.

the waste stream, which should be studied in greater detail[6]. Moreover, the collection of compost is a relatively new feature of waste services provided in London and was recently introduced in the borough Richmond-upon-Thames in November 2005. This study therefore represents a timely and interesting opportunity to estimate the economic value of composting to households, an issue which has not generally been examined previously.

The chapter is organised as follows: Section 6.2 discusses the motivation for this research and reviews existing studies that have examined WTP for recycling. The theory underlying the choice experiment method is described in section 6.3, along with some of its previous applications. Section 6.4 discusses the design and administration of the CE survey, and the results are presented and analysed in section 6.5. Section 6.6 discusses policy implications and finally, section 6.7 concludes.

6.2 Previous Literature

Large-scale waste disposal experiments in which kerbside waste and recyclables are weighed can be extremely expensive and time-consuming and require that policy evaluation is *ex-post*[7]. Instead, stated preference techniques are able to evaluate hypothetical changes in policy and to determine which policies are valued most highly. Several studies have taken this route to estimate WTP for recycling and find that households value recycling. These include Jakus et al. (1996), Lake et al. (1996), Tiller et al. (1997), Huhtala (1999), Kinnaman (2000), Caplan et al. (2002), and Aadland and Caplan (2005). In an earlier study, Lake et al. (1996) conduct a contingent valuation study using 285 households in the village of Hethersett, South Norfolk, U.K. and estimate a mean household WTP of £35.60 to continue a green bag kerbside recycling scheme. Other studies using contingent valuation such as Jakus et

[6] When organic waste is deposited at a landfill, biodegradation results in the generation and release of methane and carbon dioxide into the atmosphere, contributing to the global problem of climate change. Estimates suggest that 6% of all methane emissions from the atmosphere occur from landfill sites (Beede and Bloom, 1995). In the UK, landfill gas methane emissions contributed around 25 percent of total methane emissions in 2001, and about 2 percent of UK total greenhouse gas emissions (Source: http://www.environment-agency.gov.uk/yourenv/eff/resources_waste/213982/207743/?lang=_e).

al. (1996) and Kinnaman (2000) estimate households are willing to pay an average of \$5.78 and \$7.17 per month, respectively, for kerbside recycling in the U.S. Tiller et al. (1997) estimate rural households would pay an average of \$4.00 per month for drop-off recycling facilities in a rural/suburban area of Tennessee. Caplan et al. (2002) uses contingent ranking analysis to examine residents support for kerbside services that would enable separation of green waste and recyclable material from other sold waste. Using a sample of 350 individuals in Ogden, Utah, they find residents are WTP approximately 3.7-4.6 cents per gallon of waste diverted. Aadland and Caplan (2003) use data on more than 1,000 households in Utah to value either their actual kerbside recycling program or a hypothetical program if one is currently not provided by their community. They find that WTP for kerbside recycling is approximately \$7 per month and that young, well-educated women who are members of environmental organisations, who recycle out of an ethical responsibility, who are not frequent drop-off users and who reside in large households are willing to pay the most for these programs. Only very recently has there been an application of the choice experiment approach to estimate WTP for recycling, namely that by Jin et al. (2006) who examine preferences for kerbside recycling collection, noise reduction, and frequency of collection using a sample of 244 individuals in Macao, China. Studies that are specific to composting are more limited and include Sterner and Bartelings (1999) who study the determinants of disposal, recycling and composting in a small Swedish municipality, and Kipperberg (2003) who examines composting of yard and food waste in Seattle. Another study by Daneshvary et al. (1998) looks specifically at kerbside textile recycling (Table 6.1 presents a summary of these findings).

[7] Several important studies of this kind were reviewed in Chapter 2.

Table 6.1 Existing WTP Estimates for Recycling and Composting

Study and Location	WTP bid levels and estimates
Jakus et al. 1996 *Tennessee, USA*	Implied WTP to recycle paper and glass is \$5.78 per household per month.
Lake et al. 1996 *South Norfolk, U.K.*	Use dichotomous choice CV approach to estimate WTP per annum for recycling. Mean WTP for kerbside recycling is £35.69 per annum (£2.97 per month)
Tiller et al. 1997 *Tennessee, USA*	Estimate household WTP for drop-off recycling in a rural/suburban area of Tennessee. Using contingent valuation, the most conservative mean household WTP is near \$4.00 (£2.22) per household per month.
Daneshvary et al. 1998 *Southern Nevada*	Use univariate analyses and binary logit regression to determine resident's support for kerbside textile recycling policy from 817 mail surveys.
Sterner and Bartelings, 1999 *Tvaaker, Sweden*	Elicited WTP to have someone else sort their waste ~ 420 SEK per year. Alternative measure of WTP was via time spent ~ 2500 SEK per year
Aadland and Caplan, 1999 *Ogden, Utah*	Mean WTP for kerbside recycling estimated at \$2.05 per household per month.
Huhtala, 1999 *Helsinki, Finland*	WTP for recycling of FIM 110 (£12.84) per month per household.
Kinnaman, 2000	WTP estimate of \$7.17 per month for kerbside recycling
Caplan, et al. 2002 *Ogden, Utah, USA*	WTP of 3.7-4.6 cents per gallon of waste diverted that enables separation of green waste and recyclable material from other solid waste.
Kipperberg, 2003 *Seattle, USA*	Average WTP for composting is \$49 (£27.16) per household for yardwaste and \$12 (£6.65) per household for foodwaste.
Aadland and Caplan, 2005 *40 western U.S. cities*	Generate random values between \$2-10 for WTP. Overall mean WTP of \$5.35 per month (£2.97 per month)
Jin et al. 2006 *Macao, China*	WTP for waste segregation and recycling at source is \$0.80 WTP for noise reduction in waste collection and treatment \$0.77 WTP for increase in frequency of collection (2x per day) is \$0.10. All estimates are per person per month.

The only London specific recycling study examines recycling behaviour in the borough of Kensington and Chelsea by Robinson and Read (2005). This is a revealed preference study that addresses household participation in recycling, types of services used, frequency of recycling, the materials recycled and the problems encountered. The study does not report on any information that was collected with regard to the socio-economic characteristics of the households. As such, this study presented here is the first study to specifically investigate the household determinants in recycling preferences in the London area, and it is the first that employs the choice experiment approach to estimate WTP for the kerbside collection of dry materials, compost and textiles.

All three methods mentioned above i.e., contingent valuation (CV), contingent ranking (CR) and choice experiments (CE), fall under the category of a stated preference elicitation technique. Stated preference methods assess the value of non-market goods by using individuals' stated behaviour in a hypothetical setting. The CV method is able to elicit individuals' preferences, in monetary terms, for changes in the quantity or quality of a non-market environmental resource. Valuation is dependent or 'contingent' upon a hypothetical situation or scenario whereby a sample of the population is interviewed and individuals are asked to state their maximum WTP (or minimum willingness to accept [WTA] compensation) for an increase (decrease) in the level of environmental quantity or quality.

In CR, individuals are asked to rank a discrete set of hypothetical alternatives from most to least preferred. Each alternative varies by price and a variety of other choice attributes. The CR method can offer several advantages over contingent valuation (Caplan et al. 2002). For example, Smith and Desvouges (1986) note that "although rankings of contingent market outcomes convey less information than total values obtained by contingent valuation, individuals may be more capable of ordering these hypothetical combinations than revealing directly their WTP for any specific change in these amenities".

However, there are also disadvantages with the use of CR. Firstly, respondents are not asked to make a choice (as they are in a real setting), but rather to rank the alternatives. Though this may provide the analyst with information on *preferences*, this is not choice. Secondly, individual respondents are assumed to use the response scale in a cognitively similar fashion (Hensher et al. 2005, p.90).

In contrast, in a CE, individuals are given a hypothetical setting and asked to choose their preferred alternative among several alternatives in a choice set. The CE is a multi-attribute stated preference elicitation technique because each alternative is described by a number of attributes or characteristics. A monetary value is included as one of the attributes, along with other attributes of importance, when describing the profile of the alternative presented. Thus, when individuals make their choice, they implicitly make trade-offs between the levels of the attributes in the different alternatives presented in a choice set (Alpizar et al. 2003). Furthermore, the CE method avoids many of the problems associated with the CV method such as information bias, design bias (starting point bias and vehicle bias), hypothetical bias, yea-saying bias, strategic bias (free-riding), substitute sites and embedding effects (see Boxall et al. 1996; Bateman et al., 2003; Hanley et al. 1998).

The choice experiment method was initially developed by Louviere and Hensher (1982) and Louviere and Woodworth (1983) in the marketing economics and transportation literature. More recently it has been applied in the field of environmental economics for valuation of non-marketed environmental goods. Earliest applications are those by Adamowicz et al. (1994) on recreation and Boxall et al. (1996) on recreational moose hunting. More recent applications include *inter alia* Layton and Brown (2000) on climate change, Rolfe et al. (2000) on forests, Carlsson et al. (2003) on wetlands, and Birol et al. (2006) on home gardens. Choice experiments are becoming ever more frequently applied to the valuation of non-market goods. This method gives the value of a certain good by separately evaluating the preferences of individuals for the relevant attributes that characterize that good, and in doing so it also provides a large amount of information that can be used in determining the preferred design of the good. The next section outlines the theory behind this preference elicitation technique in more detail.

6.3 Choice Experiment Method: Theory and Models

The CE method has its theoretical grounding in Lancaster's attribute theory of consumer choice (Lancaster, 1966) and an econometric basis in random utility models (Luce, 1959; McFadden, 1974). Lancaster proposed that consumers derive utility not from the goods themselves but from the attributes they provide. Consider a household's or individual's choice and assume that utility depends on choices made from a set C, i.e. a choice set, which includes all possible recycling alternatives. The individual is assumed to have a utility function of the form:

$$U_{ij} = V(Z_{ij}, S_i) + e(Z_{ij}, S_i) \qquad (1)$$

where for an individual i, a given level of utility will be associated with any alternative recycling scheme j. Utility derived from any of the recycling scheme alternatives depends on the attributes of the recycling scheme Z_j and the social and economic characteristics of the individual S_i, since different individuals are likely to receive different levels of utility from these attributes.

The random utility theory (RUT) is the theoretical basis for integrating behaviour with economic valuation in the CE method. According to RUT, the utility of a choice is comprised of a deterministic component (V) and an error component (e), which is independent of the deterministic part and follows a predetermined distribution. This error component implies that predictions cannot be made with certainty. Choices made between alternatives will be a function of the probability that the utility associated with a particular option j is higher than those for other alternatives[8]. Assuming that the relationship between utility and attributes is linear in the parameters and variables function, and that the error terms are identically and independently distributed with a Weibull distribution, the probability of any particular alternative i being chosen can be expressed in terms of a logistic distribution. Equation (1) can be estimated with a conditional logit (CL) model (McFadden 1974; Greene 1997, pp. 913-914; Maddala 1999, pp. 42), which takes the general form:

[8] $Prob_{ij} = Prob\,[(V_{ij} + e_{ij}) \geq (V_{jh} + e_{ih})\ \forall\ h \in C, j \neq h]$ In words, the probability of an individual choosing alternative j is equal to the probability that the utility of alternative j is greater than (or equal to) the utility associated with alternative h after evaluating each and every alternative in the choice set of h = 1, ...i...H alternatives.

$$P_{ij} = \frac{\exp(V(Z_{ij}, S_i))}{\sum_{h \in C} \exp(V(Z_{ih}, S_i))} \qquad (2)$$

Equation (2) states that the probability of an individual choosing alternative j out of the set of h alternatives is equal to the ratio of the (exponential of the) observed utility index for alternative j to the sum of the exponentials of the observed utility indices for all J alternatives, including the j alternative (Hensher et al. 2005, p. 86). The conditional indirect utility function generally estimated is given by:

$$V_{ij} = \beta + \beta_1 Z_1 + \beta_2 Z_2 + \ldots + \beta_n Z_n + \delta_1 S_1 + \delta_2 S_2 + \ldots + \delta_l S_m \qquad (3)$$

where β is the alternative specific constant (ASC) which captures the effects on utility of any attributes not included in choice specific attributes. The number of recycling scheme attributes considered is n and the number of socio-economic and attitudinal characteristics of the respondent employed to explain the choice of the recycling scheme is m. The vectors of coefficients β_1 to β_n and δ_1 to δ_l are attached to the vector of attributes (Z) and to vector of interaction terms (S) that influence utility, respectively. Since social, economic and attitudinal characteristics are constant across choice occasions for any given respondent, these only enter as interaction terms with the recycling scheme attributes.

The assumptions about the distributions of error terms implicit in the use of the CL model impose a particular condition known as the independence of irrelevant alternatives (IIA) property. This property states that the probability of a particular alternative being chosen is independent of other alternatives. Whether the IIA property holds can be tested by dropping an alternative from the choice set and comparing parameter vectors for significant differences. If the IIA property is violated then CL results will be biased and hence a discrete choice model that does not require the IIA property, such as random parameter logit (RPL) model, should be used. Inclusion of socio-economic and attitudinal characteristics is also beneficial in avoiding IIA violations, since these are relevant to preferences of the respondents and can increase the deterministic component of utility while decreasing the error one (Rolfe et al. 2000; Bateman et al. 2003).

Though the use of socio-economic and attitudinal characteristics help to detect conditional, observed heterogeneity, these methods do not detect for unobserved heterogeneity. It has been demonstrated that heterogeneity can be present in significant residual form even when conditional heterogeneity is accounted for (Garrod et al., 2002). Unobserved heterogeneity in preferences across respondents can be accounted for in the RPL model. The random utility function in the RPL model is given by:

$$U_{ij} = V(Z_j(\beta + \eta_i), S_i) + e(Z_j, S_i) \tag{4}$$

Similarly to the CL model, utility is decomposed into a deterministic component (V) and an error component stochastic term (e). Indirect utility is assumed to be a function of the choice attributes (Z_j) with parameters β, which due to preference heterogeneity may vary across respondents by a random component η_i, and of the social, economic and attitudinal characteristics (S_i) if included in the model. By specifying the distribution of the error terms e and η, the probability of choosing j in each of the choice sets can be derived (Train, 1998). By accounting for unobserved heterogeneity, equation (2) now becomes:

$$P_{ij} = \frac{\exp(V(Z_j(\beta + \eta_i), S_i))}{\sum_{h \in C} \exp(V(Z_h(\beta + \eta_i), S_i))} \tag{5}$$

Since this model is not restricted by the IIA assumption, the stochastic part of utility may be correlated among alternatives and across the sequence of choices via the common influence of η_i. Treating preference parameters as random variables requires estimation by simulated maximum likelihood. Procedurally, the maximum likelihood algorithm searches for a solution by simulating m draws from distributions with given means and standard deviations. Probabilities are calculated by integrating the joint simulated distribution.

Recent applications of the RPL model have shown that this model is superior to the CL model in terms of overall fit and welfare estimates (Breffle and Morey, 2000; Layton and Brown, 2000; Carlsson et al., 2003; Lusk et al., 2003; Morey and Rossmann, 2003). It should also be noted however that even if unobserved heterogeneity can be accounted for in the RPL model, the model fails to explain the sources of heterogeneity (Boxall and Adamowicz, 1999). One solution to detecting the sources of heterogeneity while accounting for unobserved heterogeneity is by including respondent characteristics in the utility function as interaction terms. This enables the RPL model to pick up preference variation in terms of both unconditional taste heterogeneity (random heterogeneity) and individual characteristics (conditional heterogeneity), and hence improve model fit (e.g., Revelt and Train, 1998; Morey and Rossmann, 2003).

The CE method is consistent with utility maximisation and demand theory (Bateman et al., 2003). When parameter estimates are obtained, welfare measures can be estimated using the following formula:

$$WTP = \frac{\ln \sum_k \exp(V_k^1) - \ln \sum_k \exp(V_k^0)}{\alpha} \tag{6}$$

where WTP is the welfare measure, α is the marginal utility of income (generally represented by the coefficient of the monetary attribute in the CE), and V_k^0 and V_k^1

represent indirect utility functions before and after the change under consideration. For the linear utility index the marginal value of change in a single attribute can be represented as a ratio of coefficients, reducing equation (6) to:

$$WTP = -1\left(\frac{\beta_{attribute}}{\beta_{monetary\ variable}}\right) \quad (7)$$

This part-worth (or implicit price) formula represents the marginal rate of substitution between income and the attribute in question, i.e., the marginal WTP for a change in any of the attributes. Compensating surplus welfare measures can be obtained for different recycling services scenarios associated with multiple changes in attributes, i.e., equation (7) simplifies to

$$Compensating\ surplus = -(V^0 - V^1)/\beta_{monetary\ variable} \quad (8)$$

6.4 Survey Design and Administration

6.4.1 Design of Choice Sets

A choice experiment is a highly 'structured method of data generation' (Hanley et al., 1998), relying on carefully designed tasks or "experiments" to reveal the factors that influence choice. Experimental design theory is used to construct profiles for the environmental good in terms of its attributes and levels of these attributes. Profiles are assembled in choice sets, which are in turn presented to the respondents, who are asked to state their preferences in each choice occasion.

The first step in choice experiment design is, therefore, to define the recycling service in terms of its attributes and levels these attributes take. Prior to the development of the CE questionnaire, a focus group was conducted in November 2005 to obtain background information and perceptions of recycling from residents in London. A pilot survey was then carried out in December using the contingent valuation (CV) method and an open-ended elicitation format, to obtain bid estimates of WTP for the existence of a kerbside recycling scheme. This also served to test the language used in the survey and to ensure that respondents were able to understand the concepts and the manner in which they were described (Dillman, 2000). A total of 30 pilot surveys were collected where WTP ranged from £0 to £20 per month. The mean monthly WTP for recycling scheme services obtained from the CV surveys was £9.53.

The selection of recycling attributes for the final CE survey was conducted as a result of an extensive literature review, the focus group, and the CV pilot study. The five attributes selected, along with their respective levels, are reported in Table 6.2.

Table 6.2. Recycling Attributes and their Levels

Attributes	Definition	Levels
Number of materials collected	Paper and glass, aluminium, plastic	2, 3, 4
Compost Collection	Food and garden waste	Yes, No
Textile Collection	Clothing and textiles	Yes, No
Frequency of collection per month	Number of times per month recycling vehicles pick-up	2, 4, 8
Cost per month (£)	Increase in monthly bills per household	1, 2, 5, 10, 20

A large number of unique recycling service descriptions can be constructed from this number of attributes and levels[9]. Statistical design methods (see Louviere et al., 2000) were used to structure the presentation of the levels of the five attributes in choice sets. More specifically, an orthogonalisation procedure was employed to recover only the main effects, consisting of 24 pair wise comparisons of recycling service profiles. These were randomly blocked to three different versions with eight choice sets[10]. Each respondent was presented with eight choice sets, each containing two recycling scheme profiles and an option to "opt out" by selecting neither, in which case the respondents were told that there would be no kerbside recycling at all. Such an "opt out" option can be considered as a status quo or baseline alternative,

[9] The number of recycling services scenarios that can be generated from 5 attributes, 2 with 2 levels, 2 with 3 levels and one with 5 levels is $3^2*2^2*5=160$.

whose inclusion in the choice set is instrumental to achieving welfare measures that are consistent with demand theory (Bennett and Blamey, 2001; Bateman *et al.*, 2003; Kontoleon, 2003). Figure 6.1 provides an example of a choice set.

Figure 6.1 Example of a Choice Set

Choice Experiment 1.1

Which of the following schemes do you favour? Option A and option B would entail a cost to your household. Alternatively, you might favour neither scheme: Monthly bills would not rise, but all rubbish left for collection would be deposited at landfills or incinerated.

	Choice A	**Choice B**	**Choice C**
Materials Collected	Paper, glass and aluminium	Paper and glass	Neither Option A nor B: I do not wish to participate in kerbside recycling
Collection of Compost	No	No	
Collection of Textiles	Yes	Yes	
Frequency of Collection	Fortnightly	Weekly	
Cost per Month	£5	£2	

6.4.2 Selected Boroughs and Sampling

The CE survey was implemented in January and February 2006. A stratified sampling approach was adopted for the survey. Randomly selected individuals were surveyed in primarily three areas of London, namely the boroughs of Camden, Kensington and Chelsea, and Richmond-upon-Thames. Though surveys were conducted in other parts of London, due to time and budget constraints it was necessary to focus in certain areas. The boroughs were chosen so as to represent a variety of commercial and residential areas, the types of homes that predominated, and the recycling and composting services offered. This is explained in more detail below:

[10] The optimal number of choice sets presented to each respondent varies depending on the difficulty of the choice tasks, and the conditions under which the survey is conducted, where 4 to 16 choice sets are generally considered to be efficient (Louviere et al. 2000).

Camden, District of Bloomsbury: Located in central London, this borough covers an area of 22 km^2 (2,180ha). Camden is a highly commercial area and the respondents surveyed are therefore likely to reflect a greater diversity in terms of socio-economic characteristics and the areas in which they reside.

Kensington and Chelsea, District of Bayswater: The Royal Borough of Kensington and Chelsea is one of the most densely populated areas of the United Kingdom. It has 164,000 residents and 83,000 households in an area of slightly under 12 km^2. It is cosmopolitan, with marked ethnic diversity and a wide range of housing types (Robinson and Read, 2005).

Richmond-upon-Thames, District of Barnes: Located in Southwest London (part of Outer London), Richmond-upon-Thames covers an area of 57 km^2 and is not entirely urbanised. The kerbside collection of food waste for compost was recently introduced in this borough. The housing composition leans more towards detached, semi-detached (terraced) residences, rather than flats and block mansions (i.e. single-family vs multi-family dwelling) which may affect recycling rates.

Table 6.3. Background Information on Selected Boroughs

	Camden	Kensington and Chelsea	Richmond-upon-Thames	Westminster
Households with kerbside collection	53,869	63,358	60000	52,000
Recycling and composting rate (TCR)	19.1%	18.08%	23.8%	15.3%
Collection per month	Weekly	Twice weekly	Weekly	Weekly
'Dry' Materials	Empty aerosols Glass bottles/jars Light cardboard Paper Mixed cans Textiles Shoes	Empty aerosols Paper Cardboard Glass Cans/tins Plastic bottles	Paper Glass bottles/jars Mixed cans Yellow pages Aluminium foil Textiles Shoes Yellow pages	Empty aerosols Paper Cardboard Glass bottles/jars Tins/cans Plastic bottles
Foodwaste	No (home	No (home	Yes	No (home

collection	composting only)	composting only)		composting only)

Source: www.capitalwastefacts.com

Face-to face interviews were conducted on the street, in parks, and other public areas[11]. The CE survey was administered to be representative of the population in terms of gender and age, and only individuals aged 18 or over were surveyed.

The final data set consists of 188 useable surveys[12]. With respect to sample size, Hensher et al. (2004) discuss sampling for choice data and explain that in practice, the somewhat arbitrary number of 50 decisions per alternative has been suggested as an experimental lower limit which provides adequate variation in the variables of interest for which robust models may be fitted. The sample size used in this analysis therefore lies above the lower limit (as there are three alternatives in the recycling CE).

Given that there were 3 versions of choice sets, each with 8 choice sets, this constitutes a total of 1504 observations for the analysis (i.e., 188 * 8). The number of surveys collected for each of the London boroughs is reported in Table 6.4 with a map of the London boroughs depicted in Figure 6.2 below[13].

Table 6.4. Number of surveys from each London borough

Borough	**Sample size**
Barking and Dagenham	1
Barnet	3
Brent	6
Bromley	1
Camden	12
Croyden	1
Ealing	10
Enfield	1
Greenwich	2
Hackney	3

[11] Though door to door surveys may be more appropriate, this was not possible to due time and budget constraints as well as safety considerations for the surveyors. Face-to-face interviews are preferred over mail surveys and telephone interviews (Arrow et al. 1993, NOAA Panel Guidelines).

[12] The response rate was about 70%.

[13] There are a total of 33 boroughs in London. The sample consisted of randomly selected respondents residing in 28 boroughs.

Hammersmith and Fulham	11
Haringey	7
Havering	2
Hillingdon	1
Hounslow	4
Islington	3
Kingston-upon-Thames	3
Kensington and Chelsea	25
Lambeth	7
Redbridge	1
Richmond-upon-Thames	24
Southwark	4
Sutton	2
Tower Hamlets	5
Waltham Forest	1
Wandsworth	8
Westminster	22
Missing data	18
TOTAL	188

Figure 6.2. Map of London Boroughs

6.4.3 Data Preparation and Coding

The data were then coded according to the levels of the attributes. Attributes with two levels (i.e., collection of compost and collection of textiles) entered the utility function as binary variables that were effects coded. For the collection of compost, yes level was coded as 1 and no level was coded as –1. Similarly, for the collection of textiles, if the service was available (yes), the level was coded 1 whereas no collection of textiles (no) was –1. The levels for the number of materials collected and the frequency of collection per month were entered in cardinal-linear form and consequently took the levels of 2, 3, 4 and 2, 4, 8 respectively. Similarly, the payment attribute was coded as 1, 2, 5, 10, and 20. The attributes for the 'neither kerbside recycling scenario' option were coded with zero values for each of the attributes.

In addition to collecting data on the socio-economic characteristics of the individual, several motivational and attitudinal questions were also asked, including their preferences for the use of economic incentive methods to reduce waste generation and encourage recycling.

The survey included nine specific questions probing household's motivations for and against recycling (see Appendix 6.2). The questions were selected based on previous literature and adapted from other recycling surveys including Aadland (2003) and Halvorsen and Kipperberg (2003). The questions were phrased as: "*I recycle partly because…*" and "*I find it difficult to recycle partly because…*" with options to choose between strongly agree, partly agree, partly disagree, strongly disagree, and don't know. To facilitate analysis, these were coded as binary variables where agree entered as 1 and disagree as 0.

The attitudes of the respondents for environmental issues were elicited through a series of questions that are now widely used to measure pro-environmental orientation (Dunlap et al. 2000). The New Ecological Paradigm Scale (revised from the 1978 New Environmental Paradigm Scale) originally consists of a set of 15 questions which are designed to tap into five hypothesised facets of an ecological worldview. These are the reality of limits to growth, antianthropocentrism, the fragility of nature's balance, rejection of exemptionalism, and the possibility of an ecocrisis. Given the nature of this particular survey, that many other questions needed to be addressed, and that it was unrealistic for the survey duration time to last for more than 20 minutes, it was necessary to extract only a few of the full NEP Scale questions to include in the survey. Four of the 15 questions were selected, such that each of the five facets mentioned above were addressed, and so that two of the questions were worded so that agreement indicated a pro-ecological view, and two were worded so that disagreement indicates a pro-ecological view. The questions are the first four questions in the "Attitudes" section of the Recycling Survey in Appendix 2. These were measured on a 1 to 5 scale, with 1 reflecting a strong anti-ecological view, 3 reflecting "Don't know" and thus uncertainty with regard to the question, and 5 reflecting a strong pro-ecological view. An NEP score was then also created in which the total was added across the four questions.

In addition to this measure of pro-environmental orientation, actual household behaviour was assessed via questions eliciting each respondents purchase of organic produce, donations to environmental organisations, the purchase of environmental publications, fair-trade products and shopping at environmentally friendly shops. These were measured on a Likert-scale ranging from zero (never) to 4 (always). An environmental consciousness index (ECI), ranging from 0 to 20, was calculated using the Likert scores. Respondents were also asked whether they are a member of an environmental group, along with a series of questions on household characteristics, including age of the respondent, age of the oldest person in the household, highest education level attained in the household, occupation, type of home, the number of people in the household, the borough of residence, car ownership and household income. A selection of the descriptive statistics on the recycling behaviour, attitudes and socio-economic characteristics of the households are reported in Table 6.5.

Table 6.5. Descriptive Statistics of Respondents, N=188

Social and Economic Characteristics	
	Mean (s.d.)
Age of the respondent	36.80 (13.20)
Household size	2.71 (1.43)
Income (£/hh)	69,684 (52,853)
	Percent
Gender (female=1, 0 otherwise)	56%
Education (university degree and above=1, 0 otherwise)	77%
Occupation (full-time=1, 0 otherwise)	78%
Type of home (house=1, 0 otherwise)	44%
Dependent children (yes=1, 0 otherwise	26%
Tenure (own house=1, 0 otherwise)	52%
Car (yes=1, 0 otherwise)	57%
Recycling Behaviour and Services	
	Percent
Household recycles (yes=1, no=0)	81.9%
Household composts (yes=1, no=0)	23%
Kerbside recycling in the borough(yes=1, no=0)	0.845
Kerbside composting in the borough (yes=1, no=0)	0.276
Household used drop-off site (yes=1, no=0)	0.889
	Mean (s.d.)
% of paper recycled	50.67 (37.8)
% of glass recycled	53.61(39.46)
% of can recycled	38.44 (40.01)
% of plastic recycled	27.00 (34.73)
% of textiles recycled	37.17 (37.71)
% of food recycled	11.97 (28.64)
% of garden waste recycled	14.69 (30.85)
Minutes per week spent on recycling	17.9 (24.14)
Minutes walk to drop-off site	13.19 (18.59)
Use drop-off site (yes=1, no=0)	0.47 (0.50)
Motivations for Recycling	
	Percent
To contribute to environment (yes=1, no=0)	99%
To be a responsible person (yes=1, no=0)	96%
It is a pleasant activity (yes=1, no=0)	46%
Neighbours recycle (yes=1, no=0)	31%
It is required by the local authority (yes=1, no=0)	48%
Difficulties in Recycling	
	Percent
Lack of storage space (yes=1, no=0)	55%
Inconvenient /poor service (yes=1, no=0)	37%
Lack of information (yes=1, no=0)	35%
Lack of time (yes=1, no=0)	19%

On average, the households' mean income and percentage of those with university degrees are higher than the London population means for these variables (www.ons.gov). This may be explained by the fact that many of the respondents are from Kensington and Chelsea and Westminster, two of the relatively wealthier boroughs of London. Based on data from the most recently available 2001 UK Census for London, females represent 51.6% of the population; households with one or more cars represent 62.5% of the London population; average London household size is 2.35; and owner-occupied housing represents 56.5% of the London population (www.statistics.gov.uk/census2001). In comparison with the social and economic characteristics of the sample collected for the CE, the sample has a slightly larger proportion of women (56%); a smaller proportion of owner-occupied housing (52%); a smaller percentage of households owning one or more cars (57%) and larger average household size (2.71). Overall however, the values are comparable and the sample seems representative in this regard.

In the sample, 81.9% are recyclers of one form or another, whereas 23.0% compost food and/or garden waste. The average household spends on average about 18 minutes per week separating, sorting and preparing their materials for recycling/composting. 88.9% of the sample were aware of a drop-off site nearby, the average walking distance to which is about 13.2 minutes, and 47.0% of the sample had at one point or another used a drop-off site.

With regard to motives for recycling, 99% said that they wanted to contribute to a better environment, 96% want to think of themselves as a responsible person, 46% say that recycling is a pleasant activity in itself, 31% feel they should recycle because their neighbours recycle and 49% perceive it as a requirement by local authorities.

55% of the respondents said that they found it difficult to recycle because they do not have enough space in their households to store their recyclables, 37% said that it was not convenient for them to recycle because recycling services are poor. 35% felt that they had not been provided with adequate information regarding recycling and 19% said they did not have time to recycle.

Respondents were also encouraged to freely comment and express themselves on aspects of recycling that were important to them. Some respondents stated that they did not want twice a week collection because it is too frequent, unnecessary, and some even said caused adverse effects to the environment due to vehicle emissions and congestion. On the other hand, one respondent stated that if recycling collection was only offered fortnightly, she would prefer neither recycling option (i.e. option C) and would use the drop-off facility.

6.5 Results

6.5.1 Conditional Logit Models

The CE was designed with the assumption that the observable utility function would follow a strictly additive form. The model was specified so that the probability of selecting a particular recycling services scenario was a function of the attributes only and did not include an alternative specific constant (ASC). This is because the three alternatives were unlabeled[14]. Using the 1504 choices elicited from the 188 respondents, the highest value of the log-likelihood function was found for the specification with all attributes in linear form. The results of the CL estimates for the sample are reported in the first column of Table 6.6.

[14] It is unlabeled in the sense that "Option A", "Option B" and "Neither Option" do not convey meaning to the respondent on what the alternatives represent in reality (e.g. a brand, or car vs. bus, etc) and do not provide any useful information to suggest that there are unobserved influences that are systematically different for alternatives A and B. In this case the use of ASC makes no behavioral sense (p. 371). The ASC is a parameter for a particular alternative that is used to represent the role of unobserved sources of utility. One of the main benefits of using unlabeled experiments is that they do not require the identification and use of all alternatives within the universal set of alternatives. Further benefit: The IID assumption imposes the restriction that the alternatives used in the modeling process be uncorrelated. This assumption is less likely to be met under labeled experiments than under unlabeled experiments (p. 113). The correct way to proceed is to exclude constant terms for all (unlabeled) alternatives i.e, we constrain the average unobserved effect for all (unlabeled) alternatives to be zero. (Hensher, et al. 2005).

The overall fit of the model, as measured by McFadden's ρ^2 indicates a good fit[15], and the coefficients are statistically significant and intuitively correct. All of the recycling services attributes are significant factors in the choice of recycling services, and *ceteris paribus*, any single attribute increases the probability that a recycling scenario is selected. In other words, respondents' value kerbside recycling services scenarios that result in a greater number of materials recycled, the availability of compost and textile collection, and a greater frequency in collection. The sign of the payment coefficient indicates that the effect on utility of choosing a choice set with a higher payment level is negative.

Table 6.6. Conditional Logit (CL) Model and CL Model with Interactions

	CL Model		**CL Model with Interactions**	
Attributes and Interactions	Coefficient	Standard Error	Coefficient	Standard Error
Materials	0.3376***	0.0335	0.3751***	0.0455
Compost	0.8411*	0.0507	0.0990*	0.0684
Textiles	0.1117**	0.0504	0.0422	0.0683
Frequency of collection	0.3511*	0.01954	0.0021	0.0259
Payment	-0.1357***	0.0079	-0.3407***	0.0573
Pay*ECI	-	-	0.00933***	0.0024
Pay*Education	-	-	0.10995***	0.0357
Pay*Walk	-	-	0.001316***	0.0005
Pay*TCR	-	-	-0.00070	0.0021
Pay*Income	-	-	$0.2x10^{-6}$*	$0.1x10^{-6}$
Pay*Sex	-	-	0.00726	0.0171
Pseudo R^2	0.14113		0.13235	
Log likelihood	-1419.28		-785.6741	
Sample size	1504		1504	

Significance at *** 1%; ** 5%; * 10%

[15] The ρ^2 value in multinomial logit models is similar to the R^2 in conventional analysis except that significance occurs at lower levels. Hensher et al. (2005, p. 338) comment that values of ρ^2 between 0.2 and 0.4 are considered to be extremely good fits.

Pseudo R^2 is computed as 1- (unrestricted log-likelihood/ restricted log-likelihood) and is an alternative measure of goodness-of-fit for probabilistic choice models (McFadden, 1974; Garrod and Willis, 1998).

As explained in section 6.3, the assumptions about the distribution of error terms implicit in the use of the CL model impose a particular condition known as the IIA property. The IIA assumption states that the ratio of two probabilities of any two alternatives should be preserved despite the presence or absence of any other alternative within the set of alternative included in the model (i.e. Pi/Pj will remain unaffected by the presence or absence of any alternative within the set of alternatives modeled) (Hensher et al. 2005, p. 519). To test whether the CL model is appropriate, the Hausman and McFadden test (1984) is used. The IIA test involves constructing a likelihood ratio test around the different versions of the model where the choice alternatives are excluded. If the IIA holds then the model estimated on all choices should be the same as that estimated for a sub-set of alternatives. The results are shown in Table 6.7 below, indicating that the IIA property cannot be rejected at the 5% level. Therefore the CL model is appropriate for estimation of this data.

Table 6.7. Test of Independence of Irrelevant Alternatives

Alternative Dropped	$\chi^2(5)$	Probability
Option 1	11.9300	0.03576
Option 2	53.1380	0.0000
Neither Option	29.0844	0.0000

The basic conditional logit model assumes homogenous preferences across respondents (i.e. that tastes do not vary). As mentioned in section 6.3, it is the random parameter logit model that is able to accommodate the presence of unobservable preference heterogeneity in the sampled population. Nevertheless, it is possible to account for and identify observed conditional preference heterogeneity in the CL framework. This is undertaken via the interaction of individual-specific characteristics with the attributes of the choices[16]. This approach allows the β's to vary across individuals in a systematic way as a function of individual characteristics.

The analyst can thus assess the distributional impacts of a particular policy change. Due to multicollinearity problems however, it is not possible to include all the interactions between the socio-economic and attitudinal characteristics of the respondents and the recycling attributes (see Breffle and Morey, 2000)[17].

Various combinations of demographic and attributes were used[18]. The last two columns in Table 6.6 present the results from a specification that includes interaction terms between the payment attribute and socio-economic and attitudinal characteristics. Using the Swait-Louviere log-likelihood test it can be seen that the model with the interaction effects outperforms the simple model[19]. The results indicate that households who have university degrees, higher income and ECI levels, as well as those who have to walk longer distances to the recycling drop-off points are willing to pay more for kerbside recycling services. Moreover, similar to results found in Robinson and Read (2005), there is no statistically significant difference between the WTP for recycling services between women and men. Finally, the variable TCR, which reflects borough-level total current rate of recycling (i.e. recycling and composting) and is a proxy for borough level performance indicators, is also not significant[20].

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[16] Morey et al. (2002), Rolfe et al. (2000), and Scarpa et al. (2003) provide some recent examples of this approach.

[17] Appendix 6.4 reports the correlation matrix for the data in this sample.

[18] Other combinations of interaction effects provided little improvement to overall fit and explanatory power to the model. The specification presented in Table 6.5 is convenient since it can easily be compared and contrasted with the results of the RPL model below.

[19] $[-2 \times (LL_1 - LL_2)] = -2\ (1419.28 - 785.67) = 1267.22 = \chi^2$ where the critical $\chi^2(6) = 12.59$ at $\alpha =$ 0.05

This consist of the test statistic $-2(LL_1 - LL_2)$ where LL_1 and LL_2 refer to the log-likelihood statistics for the model with and without and . The test statistic is asymptotically follows a χ^2 distribution with degrees of freedom equal to the difference in the numbers of parameters in estimated in the two models.

[20] Data for TCR is obtained from www.capitalwastefacts.com. This variable is included so as to account for potential locational preference heterogeneity in the data.

6.5.2 *Random Parameter Logit Models*

Recent applications of the RPL model have shown that this model is superior to the CL model in terms of overall fit and welfare estimates (Breffle and Morey, 2000; Layton and Brown, 2000; Carlsson et al. 2003; Kontoleon, 2003; Lusk et al. 2003; Morey and Rossmann, 2003). The RPL model is estimated using LIMDEP 8.0 NLOGIT 3.0. All the parameters except the payment attribute were specified to be normally distributed (Train, 1998; Revelt and Train, 1998; Morey and Rossmann, 2003; Carlsson et al. 2003), and distribution simulations were based on 500 draws. The results of the RPL estimations are reported in the first column of Table 6.8. RPL model estimates of the sample result in significant derived standard deviations for all four attributes indicating that the data supports choice specific unconditional unobserved heterogeneity in preferences among the respondents. The log likelihood ratio test rejects the null hypothesis that the regression parameters of CL and RPL are equal at 0.5% significance level[21]. Hence improvement in the model fit can be achieved with the use of the RPL model. On the basis of this test it can be concluded that the RPL model is appropriate for analysis of the data set presented in this paper

Table 6.8. Random Parameter Logit (RPL) Model and RPL Model with Interactions

	RPL Model		**RPL with Interactions**	
Attributes and Interactions	Coefficient (s.e.)	Coeff. Std. (s.e.)	Coefficient (s.e.)	Coeff. Std. (s.e.)
Materials	0.8306*** (0.2313)	0.9357*** (0.2918)	0.3751*** (0.0638)	0.1109 (0.2576)
Compost	0.2854** (0.1331)	1.0455*** (0.4355)	0.0989* (0.0706)	0.0238 (1.2506)
Textiles	0.2255** (0.1154)	0.785** (0.5190)	0.0422 (0.0712)	0.0301 (1.1790)
Frequency of collection	0.1043** (0.0511)	0.457*** (0.1896)	0.0021 (0.0275)	0.0485 (0.1263)
Payment	-0.294*** (0.0787)	-	-0.341*** (0.0617)	-
Pay*ECI	-	-	0.0093*** (0.0025)	-
Pay*Education	-	-	0.1099*** (0.03115	-
Pay*Walk	-	-	0.001316*** (0.00056)	-

[21] $-2(1419.28 - 1398.66) = 41.24 > \text{critical } \chi^2(4) = 9.49$

Pay*TCR	-	-	-0.0007 (0.00249)	-
Pay*Income	-	-	0.00000023* (0.0000001)	-
Pay*Sex	-	-	0.00726 (0.0185)	-
R2	0.15352		0.14985	
Log likelihood	-1398.657		-784.5487	
Sample size	1504		1504	

Significance at *** 1%; ** 5%; * 10%

Even if unobserved heterogeneity can be accounted for in the RPL model, the model fails to explain the *sources* of heterogeneity (Boxall and Adamowicz, 1999). One solution to detecting the sources heterogeneity while accounting for unobserved heterogeneity is by including interactions of respondent-specific social, economic and attitudinal characteristics with choice specific attributes and/or with ASC in the utility function. This enables the RPL model to pick up preference variation in terms of both unconditional taste heterogeneity (random heterogeneity) and individual characteristics (conditional/systematic heterogeneity), and hence improve model fit (e.g., Revelt and Train, 1998; Kontoleon, 2003; Morey and Rossmann, 2003). The caveat of multicollinearity mentioned above carry over. Moreover, the selection of a particular multivariate distributional function describing the random parameters may be hard to justify (Bateman et al. 2003; Hensher et al. 2005).

The indirect utility function is extended to include these interactions and the RPL model with interactions was estimated using LIMDEP 8.0 NLOGIT 3.0. The results are reported in the last two columns of Table 6.10. This model has a better/higher overall fit compared to the RPL model, with a ρ^2 of 0.1498. The Swait-Louviere log likelihood ratio test rejects the null hypothesis that the regression parameters for the RPL model and the RPL model with interactions are equal at 0.5% significance level, implying that improvement in the model fit is achieved with the inclusion of social, economic and attitudinal characteristics in the RPL model[22]. In contrast to the RPL model estimated above, the RPL model with interactions does not result in significant

[22] $-2(1398.66-784.55) = 1228.22 > \text{critical } \chi^2$

derived standard deviations for the four attributes (number of dry materials, compost, textiles, and frequency of collection) indicating that data does not support choice specific unconditional unobserved heterogeneity for these attributes.

6.5.3 Willingness to Pay Estimates

As explained in section 6.3, the parameter estimates obtained from the different models can be used to estimate welfare measures. Table 6.9 reports the implicit prices, or marginal willingness to pay (WTP) values for each of the recycling attributes with the respective 95% confidence intervals. These were calculated using equation 7 and the WALD procedure in LIMDEP 8.0 NLOGIT 3.0. The results from the CL model indicate that these are all positive implying that households have a positive WTP for increases in the quantity of each attribute. The results suggest households are WTP on average £2.4 to £2.8 per month for each additional increase in the number of materials collected (paper and glass, aluminium, plastic) and the value households attach to composting services are in the range of £0.62 to £0.97 per month. The results indicate that on average respondents in London do not attach significant values to the collection of textile materials, and neither do they significantly value the frequency of kerbside collection.

Table 6.9. Marginal WTP for recycling services (£/households/month) and 95% C.I.

Attribute	CL Model	CL Model with Interactions	RPL Model	RPL Model with Interactions
Materials	2.488 (2.238-2.738)	2.496 (2.184 – 2.808)	2.8230 (2.5902 – 3.0558)	2.495 (2.158 - 2.832)
Compost	0.619 (0.240-0.998)	0.658 (0.194 – 1.122)	0.9700 (0.6209 – 1.319)	0.658 (0.189 - 1.127)
Textile	0.823 (0.447-1.199)	--	0.7664 (0.4381 – 1.0947)	--
Frequency	0.259 (0.118-0.400)	--	0.3544 (0.2249 – 0.4839)	--

-- indicates that the Wald procedure resulted in insignificant WTP estimates for this attribute. See Henscher, 2005.

6.6 Discussion and Policy Implications

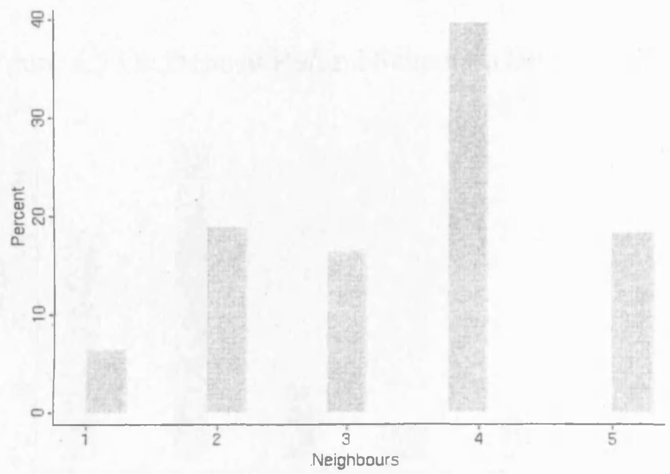
The results reported from the recycling CE study are comparable to those found by other studies using the CV method, thus providing strong evidence of convergent validity. For example, Lake *et al.* (1996) found that residents in South Norfolk, UK; are WTP £2.97 monthly for kerbside recycling services. Similarly, Kinnaman (2000) found that respondents are WTP \$7.17 (£4.13) for kerbside recycling services, whereas Aadland and Caplan (2003) found lower values of \$2.05 (£1.18) per household per month for kerbside recycling services in Utah, USA. For specific recycling services, Jakus *et al.* (1996) found that households in Tennessee, USA are WTP \$5.78 (£3.33) for recycling of paper and glass, and for composting services Kipperberg (2003) estimated that households in Seattle, USA are WTP \$1 (£0.55) per month for composting of households food waste and up to \$4.08 (£2.26) per month for recycling of garden waste.

Overall the results suggest positive WTP for different kerbside recycling services attributes, and in particular for the number of materials collected and composting. Ultimately, the benefits of providing recycling services can be compared with the costs of recycling in a comprehensive cost-benefit analysis (CBA). Based upon previous estimates on the cost of recycling in the UK[23], the results from this analysis suggest that the estimated household willingness to pay for recycling is greater than the costs.

[23] The total cost of recycling includes cost of collecting, recycling, and providing households with recycling containers, as well as the cost of specially designed vehicles for collecting recyclables and the cost of any sorting facilities (for materials not sorted at the kerbside). Savings also need to be taken into account e.g. reduced costs of waste going to landfill or incineration, money raised by selling recyclable materials to reprocessors (e.g. steel industry), and the need for fewer refuse collection vehicles for collecting reduced amount of rubbish to landfill/incineration, as well as the avoidance of non-compliance penalties. Ecotec (2000) has estimated that the gross costs of providing household recycling service for dry recyclables (e.g. newspapers, cans, plastics, textiles) in the U.K. is £7.5-£20 (average of £11.5) per household per year. The average net cost (accounting for revenue from sales, reduced disposal costs etc) is £9. Composting costs depend on type of composting plant, the collection system and the avoided disposal costs. The average net cost of providing a kerbside collection service for compostable materials is £8 per household per year. The average net cost of providing doorstep recycling and composting service is £17 per household per year (Source: Friends of the Earth website. *Fact sheet: Recycling, Can local authorities afford it?).*

Additional issues of interest that can be investigated using the recycling survey data is the motivational factors that neighbours play in affecting household recycling behaviour. To some degree, this issue was the focus of Chapter 5 on spatial interaction effects, albeit at the macroeconomic level. As mentioned earlier, Gamba and Oskamp (1994) find that households are indeed motivated by the recycling behaviour of their neighbours. Preliminary statistics suggests that this may not be the case in the sampled population collected here. More specifically, with regard to the motivational question: "*I recycle partly because my neighbours recycle; I feel I should too*", 6.5% of the sample strongly agreed, 18.9% partly agreed, 16.6% partly disagreed, 39.6% strongly disagreed, and 18.3% did not know.

Figure 6.3 On Neighbours as a Motivating Factor to Recycle



A further aim of the survey is to tentatively investigate which waste policy instruments to encourage recycling the public would find more favourable. Respondents were asked whether households should be charged for the collection of their unsorted waste (regular rubbish) if containers are provided for recyclable waste and whether there should be a deposit refund scheme. Strongly agree was converted to 1, partly agree to 2, partly disagree to 3, strongly disagree to 4, and don't know to 5. The frequency of responses are depicted in Figures 6.3 and 6.4 below, suggesting that there is a general consensus in favour of economic instruments to encourage recycling, and that the preference is towards the deposit-refund scheme.

Figure 6.4 On Pay-As-You-Throw Programs (PAYT)

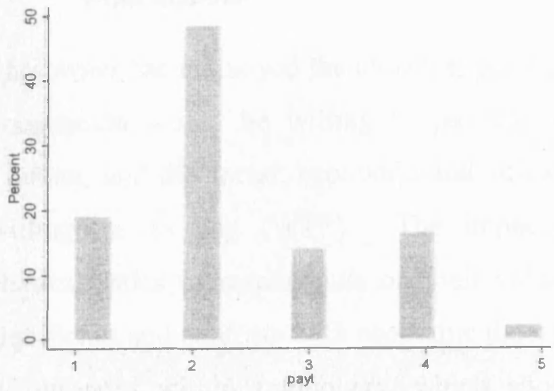
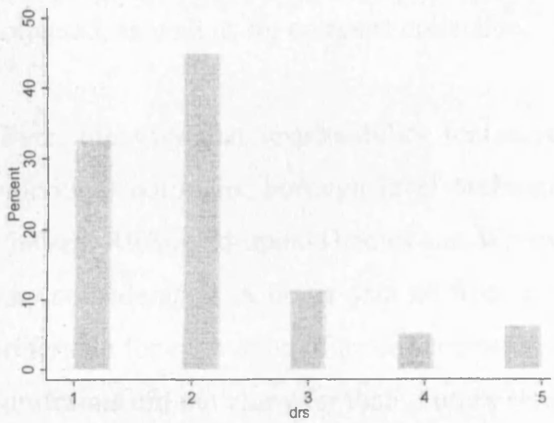


Figure 6.5 On Deposit Refund Schemes (DRS)



Finally, given the way that responsibility for recycling across London is assigned to the individual boroughs, ideally it would have been interesting to collect data from a much larger number of households and to estimate marginal WTP for recycling services at the borough level. This would enable boroughs to target their respective residents more specifically, based on their socio-economic and attitudinal characteristics. As an example, WTP estimates have been calculated here for the three boroughs for which the sample sizes were the largest, namely Kensington and Chelsea (N=25), Richmond-upon-Thames (N=24) and Westminster (N=22). The results on WTP per attribute for each borough are shown in Appendix 6.5.

6.7 Conclusions

This paper has employed the choice experiment method to estimate what, if anything, households would be willing to pay for specific kerbside recycling services in London, and the social, economic and attitudinal characteristics that determine their willingness to pay (WTP). The impacts of social, economic and attitudinal characteristics of respondents on their valuation of recycling service attributes are significant and conform with economic theory. Considerable preference heterogeneity is observed within Londoners, which should be taken into consideration when designing provision of kerbside recycling services. The results indicate that on average households are WTP the most for an increase in the number of dry materials collected, as well as for compost collection.

Given the way that responsibility for recycling across London is assigned to the individual boroughs, borough level preferences were estimated for Kensington and Chelsea, Richmond-upon-Thames and Westminster. These, however, did not seem to vary considerably. A larger data set from a wider array of boroughs would have been preferable for estimation of more accurate borough level preferences, however budget constraints did not allow for that. Future research with a larger data set is prompted.

As for the economic and policy instruments that might be employed to create incentives for recycling, the results of the survey reveal that the public seems to find the introduction of such instruments acceptable, and there is a greater preference for the introduction of deposit refund schemes rather than pay-as-you-throw, or unit-pricing, programs. Further research on economic and policy instruments to create incentives for recycling is also required. Appropriate economic incentives and efficiently designed recycling services can help London meet its recycling targets in the most effective and least-cost manner.

Appendix 6.1. Introduction Sheet to the Choice Experiment

B. CHOICE EXPERIMENT

Introduction Sheet

In this section we would like to find out what aspects of kerbside recycling services are important to households. Kerbside recycling refers to the doorstep collection of material that you have sorted from the rubbish you normally generate. We have identified 5 recycling characteristics and present these with different levels. Recycling characteristics and their levels include:

A. ***Paper/Glass/Aluminum/Plastic collection.*** This refers to the kerbside collection of these materials. This service would enable 2, 3, or 4 of these materials to be collected from your doorstep. If the service is not available, all materials will be sent to landfill or incinerated.

B. ***Compost collection.*** This refers to the possibility of recycling your biodegradable food waste and/or your green garden waste by leaving it in a separate container on your kerbside for collection. This recycling collection service is either available or it is not. If it is not available, all compostable waste will be sent to landfill or incinerated.

C. ***Textiles collection.*** This refers to the kerbside collection of textiles (i.e. clothing and fabric) for recycling. This service is either available or it is not. If it is not available, all textile waste will be sent to landfill or incinerated.

D. ***Frequency of collection per month.*** The kerbside collection can occur 2, 4, or 8 times per month. These levels apply to the collection of all types of recyclable materials mentioned above.

E. ***Cost per month.*** This is the amount in say, monthly fees (e.g. via council tax), that you would be required to pay each month to support the continued existence of kerbside recycling services above. The five levels of payment presented are £1, £2, £5, £10 and £20.

We have generated various recycling scenarios and present these as pairs in a series of cards. We would like you to indicate out of each pair, which recycling scenario you would prefer. Please imagine you are being offered various kerbside recycling options. Each of the following 8 "choice experiments" will present you with three different recycling services scenarios: Option A, Option B, and Option C. Please compare each scenario in the following cards and tell me which one you prefer in each case. In making your choices, please keep in mind your monthly household income and expenditures, and your relevant preferences and budget constraints. Please imagine your household is ACTUALLY paying this amount to obtain the recycling services.

Appendix 6.2: RECYCLING SURVEY

University College London (UCL) is conducting a study on recycling preferences and attitudes in London. As part of this study, we are carrying out a survey in which we would like you to take part. You have been randomly selected and your participation is voluntary. All questions are hypothetical and the data obtained is strictly confidential. The survey should not take longer than 10 minutes. The completion of the exercise can help policy-makers determine the key recycling factors of importance to households, and to prioritise recycling efforts. Please keep in mind that there are no right or wrong answers.

A. RECYCLING BEHAVIOUR AND SERVICES

1. Does your household recycle? Yes ☐ No ☐

2. If yes what/how much (percentage)?

Material	0%	1-25%	26-50%	51-75%	>75%
Paper	☐	☐	☐	☐	☐
Glass	☐	☐	☐	☐	☐
Cans/tins	☐	☐	☐	☐	☐
Plastic	☐	☐	☐	☐	☐
Textiles	☐	☐	☐	☐	☐
Other______	☐	☐	☐	☐	☐

3. Does your household compost (food/garden waste)? Yes ☐ No ☐

4. If yes what/how much (percentage)?

Type	0%	1-25%	26-50%	51-75%	>75%
Food waste	☐	☐	☐	☐	☐
Garden waste	☐	☐	☐	☐	☐

5. If your household does recycle, how much time per week (in minutes) does your household spend separating, sorting and preparing materials for recycling? ____

6. Do you have kerbside (i.e. doorstep) recycling in your borough?
Yes ☐ No ☐ Don't know ☐

7. Do you have kerbside composting in your borough?
Yes ☐ No ☐ Don't know ☐

8. Have you been provided with storage containers for recyclables? Yes ☐ No ☐

9. Do you purchase special recycling bags to put your recyclables in? Yes ☐ No ☐
If yes, what is your average monthly expenditure on bags? £_____

10. Do you have a compost bin? Yes ☐ No ☐
If yes, price paid for bin? £_____ Don't know ☐

11. Do you have a garden? Yes ☐ No ☐

If yes, do you compost in your own garden? Yes ☐ No ☐

12. Is there a recycling drop-off site nearby for paper/glass/compost/etc?

Yes ☐ No ☐ Don't know ☐

If yes, the proximity (minute walk): _____

13. Additional information

	Always	Often	Sometimes	Rarely	Never
Do you use the recycling drop-off site?	☐	☐	☐	☐	☐
Do you recycle when you are not at home (e.g. at office or throwing cans into public recycling receptacles)?	☐	☐	☐	☐	☐

B. CHOICE EXPERIMENT

[PRESENT INTRODUCTION SHEET AND CHOICE SETS]

14. Answer sheet

Choice Set	Option A	Option B	Option C
1	☐	☐	☐
2	☐	☐	☐
3	☐	☐	☐
4	☐	☐	☐
5	☐	☐	☐
6	☐	☐	☐
7	☐	☐	☐
8	☐	☐	☐

15. If you selected Option C in all 8 choice sets, please explain your reasons below.

	Agree	Disagree	Don't know
The government is responsible for this service	☐	☐	☐
Recycling is not an issue that concerns me	☐	☐	☐
I don't believe my payment would be used correctly	☐	☐	☐
I cannot afford to pay for kerbside recycling services	☐	☐	☐
Other (please explain):			

C. MOTIVES AND ATTITUDES

16. Motives and Attitudes for recycling waste

MOTIVES **I recycle partly because…**	**Strongly Agree**	**Partly Agree**	**Partly Disagree**	**Strongly Disagree**	**Don't Know**
I want to contribute to a better environment	☐	☐	☐	☐	☐
I want to think of myself as a responsible person	☐	☐	☐	☐	☐
It is a pleasant activity in itself	☐	☐	☐	☐	☐
My neighbours recycle; I feel I should too.	☐	☐	☐	☐	☐
I perceive it as a requirement by local authorities	☐	☐	☐	☐	☐
Other (please specify)					
I find it difficult to recycle partly because…	**Strongly Agree**	**Partly agree**	**Partly disagree**	**Strongly Disagree**	**Don't know**
I do not have enough space in my household to store recyclables	☐	☐	☐	☐	☐
It is not convenient for me to recycle –current recycling services are poor	☐	☐	☐	☐	☐
I have not been provided with adequate information regarding recycling	☐	☐	☐	☐	☐
I do not have time to recycle	☐	☐	☐	☐	☐
Other (please specify)					

ATTITUDES	**Strongly Agree**	**Partly agree**	**Partly disagree**	**Strongly Disagree**	**Don't know**
Humans are severely abusing the environment	☐	☐	☐	☐	☐
Humans have the right to modify the natural environment to suit their needs	☐	☐	☐	☐	☐
Earth is like a spaceship with limited room and resources	☐	☐	☐	☐	☐
Balance of nature is strong enough to cope with impacts of modern industrial nations	☐	☐	☐	☐	☐
Often recycling waste causes more harm to the environment than throwing it away	☐	☐	☐	☐	☐
The U.K. landfills more and recycles less than most other European countries	☐	☐	☐	☐	☐
Reducing the amount of rubbish generation is very important	☐	☐	☐	☐	☐

17. How much (if at all) do you think a household in your community is paying per month for recycling/composting services?

a) Recycling: £0 £_____ b) Composting:£0 £ _____

18. In your opinion, to encourage recycling should households be charged for collection of unsorted waste (regular rubbish) if containers are provided for recyclable waste?

Strongly agree	Agree	Oppose	Strongly oppose	Don't know
☐	☐	☐	☐	☐

19. In your opinion, to encourage recycling, should there be deposit refund schemes whereby you pay a deposit on e.g. a beverage container but receive a refund if you return it for re-use?

Strongly agree	Agree	Oppose	Strongly oppose	Don't know
☐	☐	☐	☐	☐

20. How often does your household do the following?

	Always	Often	Sometimes	Rarely	Never
Buy organic produce	☐	☐	☐	☐	☐
Give charitable donations to environmental organisations e.g. WWF, RSPB	☐	☐	☐	☐	☐
Buy fair-trade products e.g. fair-trade coffee or fair-trade chocolate	☐	☐	☐	☐	☐
Buy environmentally based journals such as The Ecologist or National Geographic	☐	☐	☐	☐	☐
Shop at environmentally sustainable/friendly shops e.g. Body Shop, OXFAM, charity shops	☐	☐	☐	☐	☐

21. Does anyone in your household belong to an environmental organisation/group?

Yes ☐ No ☐ If yes which one? _______

D. HOUSEHOLD CHARACTERISTICS

22. Age: ______ Age of oldest person in household: ______

23. Gender: Female ☐ Male ☐

24. Highest level of education anyone in your household has completed

a) Upper secondary school (up to 18 years) ☐
b) Professional Qualification ☐
c) University Degree ☐
d) Postgraduate/Doctorate Degree ☐
e) Other (please specify) ______

25. Occupation of adult in your household with the highest income

a) Full-time job ☐
b) Part-time job ☐
c) Unemployed ☐
d) Pensioner ☐
e) Student ☐
f) Other (please specify) ______

26. Ethnicity

a) Asian ☐
b) Black ☐
c) Caucasian ☐
d) Hispanic ☐
e) Other (please specify) ______

27. What type of a home do you reside in?

a) Detached ☐
b) Semi-detached ☐
c) Attached ☐
d) Flats/block mansions ☐
e) Other (please specify) ______

28. Number of people in your household: ______

29. Number of dependent children in your household: _____

30. Tenure status

a) Home owner ☐
b) Renter ☐
c) Other (please specify) ______

31. What city/borough do you live in (or first 3 letters of postcode)? ______

32. Does your household own a car? Yes ☐ No ☐

33. In which one of the following categories of income brackets does your annual <u>household</u> income lie (before tax)?

a) £0 - £10,000 ☐
b) £10,000 - £25,000 ☐
c) £25,000 - £50,000 ☐
d) £50,000 - £75,000 ☐
e) £75,000 - £100,000 ☐
f) £100,000- £150,000 ☐
g) £150,000 and above ☐

Did you find the survey difficult to understand? Yes ☐ No ☐
Do you have any comments regarding the survey that you would like to make?

Thank you very much for your time and effort in completing this survey!

Appendix 6.3. Description of the 24 choice sets of the choice experiment

		Recycling scenario A					Recycling scenario B				
V	CS	Material	Compost	Textiles	Freq	£	Material	Compost	Textiles	Freq	£
1	1	3	NO	YES	2	5	2	NO	YES	4	2
1	2	4	YES	YES	8	20	3	YES	YES	4	1
1	3	3	NO	NO	8	20	2	YES	NO	4	2
1	4	2	YES	NO	8	2	2	NO	YES	4	1
1	5	3	YES	NO	4	20	4	NO	YES	2	5
1	6	4	NO	YES	2	20	2	NO	NO	2	1
1	7	3	NO	YES	2	5	2	YES	YES	4	2
1	8	4	YES	YES	8	20	3	YES	YES	4	1
2	1	3	NO	NO	8	20	2	YES	NO	4	5
2	2	2	YES	NO	8	2	2	NO	YES	4	1
2	3	3	YES	NO	4	20	4	NO	YES	2	5
2	4	4	YES	YES	2	2	2	YES	NO	2	1
2	5	3	NO	YES	2	5	2	NO	YES	4	1
2	6	2	YES	YES	4	2	3	YES	YES	4	1
2	7	4	YES	YES	8	20	3	YES	NO	4	20
2	8	3	NO	NO	8	20	2	NO	NO	2	1
3	1	2	YES	NO	4	5	4	NO	YES	2	5
3	2	2	YES	NO	8	2	4	NO	YES	2	20
3	3	4	NO	NO	2	2	4	NO	NO	8	2
3	4	2	YES	NO	2	10	3	NO	NO	8	10
3	5	4	YES	YES	4	1	2	YES	NO	8	10
3	6	4	YES	NO	8	2	2	YES	YES	8	2
3	7	4	YES	NO	8	2	2	NO	YES	2	1
3	8	4	YES	NO	8	2	2	YES	NO	8	1

Appendix 6.4. Correlation Matrix for the Data

```
cor ager  ageh sex edu occup home hhsize depchd nodc tenure car inc emem eci nep
(obs=166)
```

	ager	ageh	sex	edu	occup	home	hhsize	depchd	nodc	tenure	car	inc	emem	eci	nep
ager	1.0000														
ageh	0.7137	1.0000													
sex	0.1320	0.2009	1.0000												
edu	-0.1911	-0.1640	-0.0408	1.0000											
occup	-0.2882	-0.2304	-0.1308	0.0383	1.0000										
home	0.1793	0.2959	0.1432	0.0408	0.0408	1.0000									
hhsize	-0.2544	-0.0544	0.0062	0.0496	0.1699	0.3309	1.0000								
depchd	0.1096	0.0416	0.0550	0.0550	-0.0137	0.1677	0.3673	1.0000							
nodc	0.1194	0.0591	0.0176	0.0948	0.0209	0.1772	0.4715	0.8842	1.0000						
tenure	0.4748	0.4444	0.0669	0.0306	-0.0890	0.3454	-0.0735	0.2148	0.1856	1.0000					
car	0.0505	0.1523	-0.1595	0.0980	0.1582	0.3060	0.2647	0.2857	0.2848	0.3207	1.0000				
inc	0.0038	-0.0356	-0.0052	0.2536	0.3287	0.2555	0.3235	0.2333	0.2652	0.1656	0.2717	1.0000			
emem	0.0211	0.0804	0.0670	0.1942	-0.0417	0.1880	-0.0607	0.0607	0.0186	0.1471	0.1786	-0.0533	1.0000		
eci	0.1275	0.1017	0.1527	0.2102	-0.1383	0.0458	-0.0486	0.1298	0.1059	0.1581	0.1880	0.1021	0.4155	1.0000	
nep	-0.0132	-0.0778	0.0613	0.0846	-0.0558	-0.1372	-0.0293	-0.0111	-0.0219	0.0503	-0.0324	0.0341	0.0647	0.1359	1.0000

Appendix 6.5. Borough Level WTP Estimates

Kensington and Chelsea

```
--> WALD
    ; Fn1 = Bmat/(Bpay+bpe*9.2+bped*0.76+bpw*10.9+bpt*18.1+bpinc*84170+bps*0.56)
    ; Fn2 = Bcomp/(Bpay+bpe*9.2+bped*0.76+bpw*10.9+bpt*18.1+bpinc*84170+bps*0...
    ; Fn3 = Btex/(Bpay+bpe*9.2+bped*0.76+bpw*10.9+bpt*18.1+bpinc*84170+bps*0.56)
    ; Fn4 =Bfreq/(Bpay+bpe*9.2+bped*0.76+bpw*10.9+bpt*18.1+bpinc*84170+bps*0....

+-----------------------------------------------+
| WALD procedure. Estimates and standard errors |
| for nonlinear functions and joint test of     |
| nonlinear restrictions.                       |
| Wald Statistic             =      79.86371    |
| Prob. from Chi-squared[ 4] =        .00000    |
+-----------------------------------------------+
+---------+--------------+----------------+--------+---------+
|Variable | Coefficient  | Standard Error |b/St.Er.|P[|Z|>z] |
+---------+--------------+----------------+--------+---------+
 Fncn(1)     -2.572367406       .38868745   -6.618   .0000
 Fncn(2)     -.6786879768       .48556506   -1.398   .1622
 Fncn(3)     -.2897311594       .48981346    -.592   .5542
 Fncn(4)  -.1413809207E-01      .18810186    -.075   .9401
 (Note: E+nn or E-nn means multiply by 10 to + or -nn power.)
```

Richmond-upon-Thames

```
--> WALD
    ; Fn1 = Bmat/(Bpay+bpe*9.6+bped*0.78+bpw*11.2+bpt*23.8+bpinc*77610+bps*0.67)
    ; Fn2 = Bcomp/(Bpay+bpe*9.6+bped*0.78+bpw*11.2+bpt*23.8+bpinc*77610+bps*0...
    ; Fn3 = Btex/(Bpay+bpe*9.6+bped*0.78+bpw*11.2+bpt*23.8+bpinc*77610+bps*0.67)
    ; Fn4 =Bfreq/(Bpay+bpe*9.6+bped*0.78+bpw*11.2+bpt*23.8+bpinc*77610+bps*0....

+-----------------------------------------------+
| WALD procedure. Estimates and standard errors |
| for nonlinear functions and joint test of     |
| nonlinear restrictions.                       |
| Wald Statistic             =      43.52835    |
| Prob. from Chi-squared[ 4] =        .00000    |
+-----------------------------------------------+
+---------+--------------+----------------+--------+---------+
|Variable | Coefficient  | Standard Error |b/St.Er.|P[|Z|>z] |
+---------+--------------+----------------+--------+---------+
 Fncn(1)     -2.600218049       .47832511   -5.436   .0000
 Fncn(2)     -.6860360315       .49819750   -1.377   .1685
 Fncn(3)     -.2928680359       .49599929    -.590   .5549
 Fncn(4)  -.1429116310E-01      .19010159    -.075   .9401
 (Note: E+nn or E-nn means multiply by 10 to + or -nn power.)
```

Westminster

```
--> WALD
    ; Fn1 = Bmat/(Bpay+bpe*9.2+bped*0.59+bpw*9.7+bpt*15.3+bpinc*72730+bps*0.64)
    ; Fn2 = Bcomp/(Bpay+bpe*9.2+bped*0.59+bpw*9.7+bpt*15.3+bpinc*72730+bps*0.64)
    ; Fn3 = Btex/(Bpay+bpe*9.2+bped*0.59+bpw*9.7+bpt*15.3+bpinc*72730+bps*0.64)
    ; Fn4 =Bfreq/(Bpay+bpe*9.2+bped*0.59+bpw*9.7+bpt*15.3+bpinc*72730+bps*0.64)$

+-----------------------------------------------+
| WALD procedure. Estimates and standard errors |
| for nonlinear functions and joint test of     |
| nonlinear restrictions.                       |
| Wald Statistic             =      92.24706    |
| Prob. from Chi-squared[ 4] =        .00000    |
+-----------------------------------------------+
+---------+--------------+----------------+--------+---------+
|Variable | Coefficient  | Standard Error |b/St.Er.|P[|Z|>z] |
+---------+--------------+----------------+--------+---------+
```

```
Fncn(1)      -2.256412223        .32310953   -6.983   .0000
Fncn(2)      -.5953270294        .42382955   -1.405   .1601
Fncn(3)      -.2541444616        .42965081    -.592   .5542
Fncn(4)   -.1240155806E-01       .16504050    -.075   .9401
(Note: E+nn or E-nn means multiply by 10 to + or -nn power.)
```

CHAPTER 7

CONCLUSIONS

7.1 Introduction

The thesis has focused on municipal solid waste management and policy in developed countries. More specifically, it has explored the underlying factors that influence the generation, disposal and recycling of municipal solid waste. Understanding the causes and driving forces that significantly impact waste trends is a critical first step in designing efficient and sustainable waste management policies that can positively impact the economic, environmental and social outcomes. The analysis has been undertaken at the OECD level using macroeconomic country data, and at the household level using original survey data from London, UK. This chapter aims to bring the main conclusions together and discusses the policy implications for sustainable MSW management. Section 7.2 summarises the key findings of the thesis and highlights the contributions to the waste management literature. The waste management policy implications are discussed in section 7.3 and finally, section 7.4 suggests areas in the field that could benefit from further research.

7.2 Main Findings and Contributions to the Literature

The thesis has employed three methods, namely panel data econometrics techniques, spatial econometrics and stated preference methods, to investigate the determinants of municipal solid waste generation, disposal and recycling. This was undertaken using cross-sectional time-series macroeconomic level data from the 30 member countries of the OECD, and household level choice experiment survey data, using a sample of 188 households in London area.

The major results of this thesis are as follows:

(i) Despite recent findings by the OECD (2002) that there has been a relative decoupling, and in some case an absolute decoupling, between per capita municipal waste generation and income in certain countries, an empirical re-investigation of the hypothesised (inverted U-shaped) environmental Kuznets curve in Chapter 3 suggests that waste generation levels continue to increase monotonically with income. Further analysis reveals that in addition to income, the degree of urbanisation is another significant determinant of MSW generation rates across OECD countries. Policy variables, as measured by the waste legislation and policy index, also tend to exert some influence on waste generation, though it is important to note that these results need to be interpreted with care given the aggregate nature of the waste legislation and policy index.

(ii) The main determinants of the percentage of MSW generated disposed of at landfill are income, urbanisation and population density. The results suggest that landfill taxes are also significant, and can be effective policy instruments for diverting the waste stream away from landfill disposal.

(iii) Paper/cardboard and glass recycling rates (as a percentage of apparent consumption) are positively affected by income, urbanisation and population density. The R^2 value on the preferred model for paper/cardboard recycling rates is however low, indicating that a large proportion of the variation in the variable remains unexplained. In contrast, this issue does not arise in the analysis of glass recycling rates.

(iv) In an analysis to identify and analyse the potential existence of spatial interaction in waste management and policy-making, the results reveal that countries do indeed seem to be influenced by decisions made in other countries. For example, landfill taxes in one country tend to be influenced by landfill taxes in other countries based on their size and geographical proximity. The results using the Ybest weight to assess for the so-called 'California effect' need to be interpreted more cautiously as only one methodology to test for this particular effect was used and additional weights would lead to more robust conclusions.

(v) At the household level, London residents attach the highest stated preference values, in terms of willingness to pay, to an increase in the number of dry materials collected and the availability of kerbside compost collection, as revealed in the recycling attributes choice experiment survey. The WTP estimates are about £2.5 per household per month for an increase in the amount of dry materials collected, and £0.65 per household per month for kerbside compost collection. Despite the fact that the sample size is relatively small, the sample lies within the lower limit that is considered acceptable, and the sample statistics are fairly representative of the London population. Moreover, the results indicate that London households seem to look favourably upon the introduction of economic policy instruments to encourage and stimulate recycling levels, with a slight preference towards deposit refund schemes over pay-as-you-throw programs.

With regard to the contributions of this thesis to the literature on sustainable waste management and policy, these are as follows:

(i) Only three previous studies have investigated the existence of an EKC for MSW generation and these were undertaken in the 1990s. The analysis in chapter 3 provides an update of the EKC analysis using data from 1980 to 2000 and adopts a panel data approach to examine whether MSW generation continues to increase monotonically.

(ii) Moreover, chapter 3 adds to the scant literature on macroeconomic studies on the determinants of per capita municipal solid waste generation. As previously mentioned, only Beede and Bloome (1995) examine the determinants of total MSW generation, whereas Johnstone and Labonne (1994) look at household waste generation. The results obtained here conform with previous studies but provide additional insight into the determinants of waste generation rates.

(iii) Using a similar dataset and methodology, chapter 4 examines the determinants of landfill disposal of waste and of recycling rates for paper/cardboard and glass. The analysis seeks to contribute to the literature on MSW landfill disposal and recycling by examining a number of potentially significant factors using macroeconomic panel data. The chapter analyses the relative importance of economic growth and population density, as well as demographic and policy characteristics in OECD countries.

(iv) The analysis of the determinants of landfill disposal rates is, to my knowledge, the first of its kind. The analysis of paper/cardboard recycling rates builds on and extends a previous study by Berglund et al. (2002) by using panel data instead of cross-sectional data, and by including two public policy variables that may have a significant impact on paper/cardboard recycling. Moreover, the approach is extended to analyse the determinants of glass recycling rates which have also not previously been examined.

(v) Chapter 4 presents a first attempt to incorporate the influence of national public waste policies on MSW generation, disposal, and recycling rates. The lack of such policy variables in previous studies is generally recognised (e.g. Berglund et al. 2002, Johnstone et al. 2004), and this chapter seeks to address this issue more explicitly.

(vi) The majority of the spatial interaction studies have focused primarily on the U.S., and no previous study examines the possible existence of strategic interaction or behaviour of environmental policy in an OECD country context. Furthermore, no previous study has examined this issue in the context of municipal solid waste management. Given the magnitude of the waste problem and the large fraction of total environmental expenditures on this resource, this is an important environmental issue that merits further consideration. The analysis in Chapter 5 extends the spatial econometrics literature to the examination of spatial interaction in the imposition of landfill disposal taxes, as well as waste management performance more generally at the OECD level.

(vi) Estimation of household willingness to pay for various kerbside recycling attributes via the stated preference choice experiment method is the first application of this evaluation technique to recycling in a developed country[1]. Moreover, WTP estimates are derived for the kerbside collection of dry materials and textiles, as well as compost. Studies on the latter are scant, and this is the first study that examines recycling and composting preferences in London, UK.

7.3 Policy Implications for MSW Management and Policy

The policy implications of the EKC analysis in chapter 3 suggest that there continues to be an upward trend in MSW generation levels and that significant policy intervention will need to be undertaken to mitigate and reverse these trends. More stringent and aggressive waste management policy will be necessary to decouple the trend between economic growth and waste generation levels.

Further analysis into the determinants of MSW waste generation reveal that income is not the most significant factor but rather that urbanisation seems to play the dominant role in waste generation. The results from the analysis of landfill disposal and recycling rates for paper/cardboard and glass are more encouraging from a policy perspective. Higher incomes are associated with a general movement along the waste hierarchy to the more preferred methods (i.e. from landfill disposal to recycling). Moreover, the results provide some evidence to suggest that landfill taxes have a significant impact on diverting waste away from landfill disposal, and inducing higher rates of paper/cardboard and glass recycling. This implies that governments wishing to divert waste higher up on the waste hierarchy are likely to do so successfully via the introduction of landfill taxes.

The results from the spatial interaction analysis suggest that landfill tax level choices in one country are affected by decisions made in other countries. Whether this is due to policy convergence in general or due to strategic government behaviour is less clear, but the results suggest that there are cascading effects in waste policy-making such that an

[1] The only other available study that uses a CE to examine recycling preferences is by Jin et al. (2006) who conduct their study using a sample of 241 respondents in Macao, China.

increase in the landfill tax level in one country will lead to increases in other countries (especially so from large populated countries and those in close proximity).

With regard to the policy implications from the household choice experiment survey conducted in London, the results suggest that local authorities may wish to focus on increasing the number of dry materials offered for kerbside recycling collection, and moreover, that the kerbside collection of compost is another service that the London public value highly. Given that biodegradable municipal waste is an important contributor to methane gases causing climate change, and the recent landfill diversion targets for biodegradable waste in the EC Landfill Directive, more funds should be allocated to the issue of compost collection. These services are provided in many areas in Germany and northern Italy, and further lessons can be learned from the borough of Richmond-upon-Thames in London, which has recently introduced a kerbside compost collection service for food waste. Textile collection and increases in the frequency of recyclables collection do not seem to be particularly important features of a kerbside collection service.

7.4 Directions for Future Research

The need for further analysis on the determinants and underlying causes for the increases in waste generation levels as well as the methods in which waste is disposed of is evident, given the massive expenditures spent on this resource and the environmental externalities that prevail in the waste sector. A key requirement for this, is the systematic collection of comparable and high quality data in this area. Waste data is not generally perceived to be as reliable and accurate as other available environmental data and this hampers the ability to conduct data analysis. Further efforts should be devoted to collecting these data at an international level to allow for larger cross-country studies, as well as the imports and exports of waste across countries, comparable data on incineration levels, as well as on the recycling data on materials other than paper/cardboard and glass. This would for example benefit from the type of analysis undertaken in chapters 3 and 4 of the thesis. Of particular relevance to the analysis undertaken in chapter 4 would be the collection of comparable cross-country and time-series data on average landfill prices over time. These could then be combined with the panel data on landfill taxes to more accurately assess the price-elasticity of demand for landfill disposal, and the effects this may have on recycling rates. In addition, better quality national data is required for existing national waste policies that have been implemented across countries to more sustainably address the waste management issue.

With regard to the analysis conducted in Chapter 5, this may benefit from further research where total waste management expenditures are examined. Additional alternative proxies for waste policy stringency could also be useful.

Finally, the results presented in Chapter 6 are to some degree contextual in that the social and economic characteristics of households in London may not be representative of the UK as a whole. Benefit transfer (BT) methods, which are used to estimate values for one context by adapting an estimate of benefits from some other context, might nevertheless prove useful, and further studies in similar and different areas can be conducted to examine how accurate BT methods might be (Bateman et al. 2003). Moreover, information obtained on the benefits of recycling can be aggregated over the relevant population and weighted against the total costs of providing the different recycling attributes. In this way, the results can be used to conduct a comprehensive cost-benefit analysis for a socially efficient design for recycling services.

The data set obtained from the household survey is fairly rich and further research and analysis can be undertaken to examine different aspects of recycling preferences and behaviour. For example, intrinsic motivations for recycling can be further explored to determine whether these are an important factor in recycling. This can be undertaken via discrete choice models i.e. binary choice models such as the probit model, or ordered probit and logit models for responses that include more than two outcomes (Greene, 1997). One could also compare urban vs. rural recycling preferences (see e.g. Hanley, Wright and Adamowicz, 1998) in the UK to examine whether there are different waste policy implications for these two demographics, and thus help decision-makers tailor their waste collection services in the most effective and efficient way. Given a larger dataset, latent class model (LCM) analysis may also prove to be useful, by accounting for preference heterogeneity. This was not possible with a sample size of 188.

Given that the recycling survey also collected revealed preference data (as opposed to only stated preference data), further analysis could be undertaken to compare these results with the ones obtained here. One caveat with the revealed preference data is that the data collected here is self-reported as opposed to actually measured, a factor that clearly lies beyond the budget and time scope of this particular research. In the final analysis, it would perhaps be possible to combine the revealed and stated preference methods for estimating the benefits of recycling (Adamowicz et al. 1994).

An additional issue that may benefit from further consideration is the time spent on recycling. Time costs reflect another consideration in the costs of recycling. For example, Bruvoll, Halvorsen, Nyborg (2002) examine households' recycling efforts in Norway, with a focus on the time a household spends on sorting waste. Reasons identified for sorting and recycling waste include a perception that it was mandatory, and for moral motives. On average, respondents are WTP U.S. $20 per year to have a company take over sorting of the waste, if this were possible. Sterner and Bartelings (1999) study the determinants of waste disposal, recycling and composting using data from nearly 600 households in Tvaaker, Sweden where a weight-based billing system for household waste had recently been introduced, along with the establishment of recycling centres. They find that the amount of time and effort invested in recycling exceeded the returns from lower waste management bills (see also e.g., Bruvoll, 2002; Halvorsen, 2004; Berglund, 2005). In the sample used for the CE survey, the respondents indicated that the mean time spent per household on sorting, separating and preparing their materials for recycling/composting is 18 minutes per week. This is equivalent to 15.6 hours per year [(18x52)/60]. Average gross weekly household income for London was £711 in 1999-

2002[2]. A back-of-the-envelope calculation indicates that households are WTP £5.33 per month based on their incomes[3].

More generally, an agenda for further action organised around improving the usability of waste information and improving co-ordination and sharing of good practices would be a useful next step in international collaborative efforts directed towards addressing the waste management issue in a manner that is consistent with sustainable development.

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[2] www.statistics.gov.uk

[3] £711 / 40 hour work week = £17.775 per hour. Divide by 60 (minutes) = £0.29625. Multiply by 18 (minutes) = £5.33.

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[1] Questionnaire by Aadland et al. for their recycling study: www.uwyo.edu/aadland/research/recycle/datareport.pdf

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