

**THE ECONOMICS OF  
SOLID WASTE MANAGEMENT  
IN IRELAND**

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**Alan Barrett and John Lawlor**

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## GENERAL SUMMARY

### *Introduction*

In recent years municipal solid waste management, that is the collection and disposal of refuse discarded by households and commercial enterprises, has become an area of growing concern, both in Ireland and elsewhere. There are a number of reasons for this. First, the amount of solid waste generated has been growing. Therefore, there is simply a greater amount to be dealt with and this is putting a strain on existing disposal facilities. Second, rising environmental concerns have led to the passage of directives and legislation requiring tighter environmental controls on the design and operation of disposal routes for solid waste, such as landfills and incinerators. These controls, while lessening the negative environmental impact of the disposal routes involved, have added significantly to the direct cost of solid waste management. Third, in spite of the improved environmental nature of landfills and incinerators, there is a growing public resistance to the siting of such facilities and hence a decline in the number of readily accessible sites.

Given the nature of the solid waste problem, there is a need to achieve two broad objectives. First, the amount of solid waste generated should be reduced, at least as long as the benefits of such reduction still exceed the costs. Second, given that solid waste will inevitably be generated regardless of reduction efforts, it should be dealt with in an efficient, least-cost, manner where cost is defined to include both the pecuniary costs of solid waste management and the unpriced environmental costs also. Both objectives can be viewed as attempting to achieve desirable environmental outcomes without imposing undue economic burdens.

In this report, we apply the concepts and logic of environmental economics in investigating approaches to the achievement of these two objectives. With regard to waste reduction, we examine ways in which economic incentives can be used to induce people to reduce their waste generation. At the domestic level in Ireland, a large proportion of the population currently faces no additional charges for increased use of solid waste services because such services are financed through flat fees or from general revenue. In such situations, there is no incentive to economise on the use of solid waste services. Even at the commercial level where per-volume user-charges often do apply, the charges rarely reflect the full cost of

disposal. We examine a number of taxing and charging options which can help to overcome this incentive problem.

As some amount of solid waste will always be generated, we must also discuss how best to deal with the amount that is generated. The range of options that exist are often listed according to a ranking based on environmental criteria, with the ranked list, in order of increasing environmental acceptability, as follows: landfill, incineration with energy recovery, recycling and re-use. In choosing an approach to solid waste management a balance must be struck between the costs of the approach and the benefits that the approach provides. As such, we explore the relative costs of the various ways of dealing with solid waste from an Irish perspective, to see if the waste hierarchy makes economic sense in an Irish context. In considering costs, we consider both the pecuniary or internal costs of the approaches and also the unpriced or external costs that are associated with these various options.

#### *Legal and Institutional Structures*

The bodies mainly responsible for waste management in Ireland, either directly or as regulators, have traditionally been the local authorities. Their functions have been set out by the Local Government (Sanitary Services) Acts, 1878 to 1964, and the Public Health Acts Amendment Act, 1907. These gave the local authorities wide powers in relation to collecting and disposing of solid waste, but placed few restrictions on them from an environmental point of view. The legal situation is in the process of changing considerably, however, due to new legislation at both EU and domestic levels.

Considering EU legislation first, a new Directive on packaging waste has recently been adopted, and a Directive on landfill is proposed. The former sets targets for the recovery and recycling of packaging waste, which for Ireland requires the recovery of 25 per cent by weight of packaging waste within 5 years of the adoption of the Directive, and 50 per cent (including 25 per cent recycling) by 31st December 2005. In the absence of incinerators with heat recovery in Ireland, the targets effectively apply to recycling. Within this later target a minimum of 15 per cent by weight of each material must be recycled. The proposed landfill Directive requires high standards of environmental protection for new landfills, especially in the areas of leachate and gas control. Existing landfills will have to be retro-fitted with features to protect the environment.

Turning to the domestic legal situation, the most significant development is the recently published Waste Bill, which will bring major changes to solid waste management in Ireland. Existing domestic legislation will be repealed, a number of EU Directives will be given effect, and the roles of local authorities, the Minister of the Environment and the Environmental Protection Agency (EPA) will be redefined. Most significantly, the EPA will become responsible for the licensing

of all significant waste disposal activities, including landfills, and the Minister will have wide powers of policy direction regarding the more important aspects of waste management.

In addition, in 1994 the government published its *Recycling for Ireland* strategy document (Department of the Environment, 1994a), which reflects the requirements of the EU packaging Directive described above. It sets recovery/recycling targets for each type of packaging waste to be achieved over 5 years. The rates are 25 per cent for most material types, and 55 per cent for glass, giving an overall average rate of 33 per cent. Two notes of caution need to be sounded in relation to these recycling targets:

- (i) In general, targets such as these should only be set after it has been determined that the benefits of achieving these targets exceeds the costs. It is not clear the degree to which this has been done in this case. Indeed, from an economic point of view, it is better to charge the proper price for the use of all resources, and let the market determine the socially optimal level of recycling (and other activities).
- (ii) The EU Directive on packaging waste (and the Irish government's recycling strategy), while requiring action to promote waste reduction and re-use, only set numerical targets on recycling and recovery. This perhaps reflects the fact that reduction and re-use are difficult to quantify. It could, however, give a perverse behavioural incentive to those affected by the Directive. Firms, industries and countries might concentrate on achieving the numerical recycling/recovery targets, to the neglect of re-use and reduction, because their performance could be more easily evaluated against the numerical targets. Given the inter-changeability of reusable and non-reusable (but recyclable) packaging, this could conceivably lead to a move away from re-use, which might have a detrimental environmental effect. Specifically in the case of beverage containers, we have seen that there is much use of reusable packaging in the on-licence trade in Ireland. The on-licence trade is, relatively speaking, more important in Ireland than in other EU countries. Therefore, setting targets at EU level for recycling only may not give due credit to Ireland for the amount of "benign" waste management that is already being achieved here, in the area of beverage packaging.

#### *Overall Magnitudes*

Approximately 9 million tonnes of solid waste arise in Ireland each year, of which over 7 million is industrial waste. The vast majority of this waste ends up in landfill or industrial on-site dumps. Landfills in Ireland are generally small, and many are coming to the end of their useful lives. There is, therefore, likely to be

considerable expenditure on new landfills in the coming years. These new facilities will be much more expensive to build and operate than the old facilities, and there will be considerable rationalisation of numbers in the process. It is expected that the current number of landfills will fall from 100 to 50 over the next 10 years or so.

Comparison with overseas data indicates that Ireland produces relatively low quantities of household waste *per capita*, and that the predominance of the use of landfill here is in common with other countries with a relatively low population density. The other main method of solid waste disposal used internationally is incineration.

#### *Current Levels of Costs and Financing and Incentive Structure*

Local authority costs of solid waste management in 1994 were £65 million, excluding some administration and including inter-authority contributions. These costs relate mainly to domestic and commercial waste; the costs of industrial solid waste management are unknown. Local authority income in this area is much lower, at £18.5 million, indicating that the service is being significantly underpriced, and the Polluter Pays Principle is not being adhered to. The levels of charges vary widely across the country. Forty per cent of households pay no charge for waste disposal services, while charging for commercial waste management services is much more commonplace. Costs of providing the service will increase very substantially in the coming years, as a result of the need to replace and up-grade landfills. We estimate that the capital costs of this could be in excess of £250 million; operating costs will also increase very substantially. This is likely to force local authorities to charge full costs for the service. An alternative may be to privatise the service, and a number of local authorities have already done this, especially for waste collection.

#### *Costs of Solid Waste Services*

In considering costs of solid waste services, two points must be borne in mind. First, regardless of how we decide to deal with this issue, it appears that there will be a need for landfill for the foreseeable future; indeed it will probably remain an essential part of solid waste management for most, if not all, of Ireland. Alternative methods of waste management will not deal with sufficient quantities of waste for them to be able to replace landfill as the main method of waste management in this country. Second, the most important consideration in comparing the costs of various methods of waste disposal is not the average cost of individual disposal routes, but the cost of the overall system. Another way of looking at this is that, given that there will be a number of alternative routes in existence, the avoidable costs (or long-run marginal costs – LRMC) of each are what are important. We need to compare the cost of disposing of a tonne of material by one route with the saving from diverting it from another. If the net

amount represents a saving, then the overall cost to society of solid waste management has fallen, and it makes sense to change the disposal route.

We examine the costs of the various solid waste management options, and try to compare them to determine the optimal mix of waste management routes. Both market costs and external costs are considered. The exercise involves the making of many assumptions, which if they were changed might change the conclusions. In a number of cases, the assumptions made are so significant that the results should be viewed with caution. However, what is presented is based on the best information available. Subject to this proviso, we conclude that:

- (i) Landfill costs are set to increase significantly over the coming years; yet this still appears to be the most cost-effective option for disposal of most of the solid waste arising in Ireland. Economies of scale will apply, and this will mean that considerable savings are achievable from building fewer, larger landfills. However, these savings need to be set against the increased collection costs arising from bringing waste to more distant landfill sites, in the particular circumstances.
- (ii) Incineration is significantly more expensive than landfill in most cases. However, if very high landfill prices prevail in the Dublin area and the number of landfills can be reduced in Dublin as a result of building an incinerator, then incineration might become competitive there.
- (iii) Recycling by the kerbside system, whereby recyclable materials are collected from households and firms, may be a viable option, where a large enough catchment area is available (perhaps 50,000 households) and both landfill costs and recyclables prices are extremely high. This combination of circumstances would probably only prevail in Dublin, although it is difficult to see recyclables prices remaining sufficiently high, in the longer run, to justify kerbside recycling even in Dublin.
- (iv) Recycling by the bottlebank or bring system, whereby individuals bring recyclable materials to a central drop-off point, appears to be quite viable over a significant part of the country.
- (v) Composting, both at the household and centralised level, has the potential to divert sizeable quantities of waste, but the economics of centralised composting and of promoting home composting is not clear.
- (vi) Re-use and reduction at source already occur at the industrial and commercial level, and higher landfill prices will increase the incentive for these. Imposing a re-use system at the domestic level appears not to be cost-effective, and caution should be used if considering such a scheme.

- (vii) Finally, the various options will impact on each other, in terms of the amount and suitability of waste going to each. However, in terms of the quantities that are likely to be involved in the Irish context they are unlikely to affect the economic viability of each other, except in the case of incineration in Dublin, which might be able to replace the building of one landfill if the costs of the latter are very high.

As stated, a number of important assumptions are made in comparing the costs of alternative methods of waste management. Perhaps the most significant assumption centres around the use of energy and the external (environmental) costs thereof (especially in the case of the evaluation of recycling). We make estimates of these external costs, and adjust the costs of the alternative methods accordingly. Ideally, there should be a tax on energy which would "internalise" these external costs. This would mean that any activity that saved energy would automatically become relatively cheaper, and we would not have to worry about giving recycling (or other activities) a subsidy for energy saved. Specifically, the market would give a very precise message to all involved of the environmental benefits and costs of their energy use.

#### *Economic Incentives in Solid Waste*

As there are often no incentives, or only weak incentives, to economise on the use of solid waste services, we explore ways of introducing such incentives. One approach is the imposition of volume- or weight-based user charges that fully reflect both the internal and external costs of waste disposal. The advantage of these charges is that they provide a very direct incentive to households and firms to reduce the amount of waste they put out for collection. Ideally, this would be achieved through waste reduction. It would be a concern that such charges could lead to illegal dumping, but the experience of local authorities who have introduced user charges indicates that this problem can be contained. A review of studies from the US shows that the amount of waste put out for disposal does fall in response to per-use charges.

A range of other approaches exist which could be used to introduce incentives in this area and these are examined. Product or packaging taxes that relate to the waste component of products can be used to internalise the associated costs of disposal. Such taxes then create an incentive to purchase less waste intensive products. Similarly, taxes can be placed on virgin raw materials so as to stimulate a demand for secondary raw materials, thereby diverting materials from the waste stream. Deposit-refund schemes create an incentive to return containers for possible re-use or recycling, again diverting materials from the waste stream. Recycling credits are a way of rewarding recycling and so provide an incentive for groups to undertake recycling projects. Finally, a landfill levy is a surcharge on disposal to landfill that can be used to internalise the unpriced negative



environmental consequences of landfills and that also creates an incentive to divert waste from landfills.

The conclusion emerges from our discussion that per-volume user charges would be the simplest and most effective way of producing the environmentally desirable result of waste reduction. Such charges should reflect the long-run marginal cost of waste disposal and thereby provide an incentive to use solid waste services up to the point that is economically and environmentally efficient. They also satisfy the polluter pays principle which should be an important element of any environmental policy. As mentioned above, some local authorities have already moved in this direction.

As a final note on economic incentives, we explore the idea of providing incentives to communities to accept landfills or other waste facilities through a compensation offer. There are a number of problems with the operation of such a scheme, most importantly that compensation offers could be bid up through the political process to the point that people are not just compensated but actually make windfall gains. It is of interest, however, to think about such a scheme as a constructive approach to overcoming the NIMBY syndrome, i.e., not in my backyard.

In summary, having identified a problem in the incentive structure of solid waste our objective is to suggest how this problem can be rectified in the most economically and environmentally satisfactory way.

## Chapter 1

### *INTRODUCTION*

In this introductory chapter we want to achieve two objectives. First, we want to outline the nature of the solid waste problem that we are investigating and to establish why solid waste should be an area of study from an economic perspective. Second, we want to outline how we approach the topic and how we can make a contribution to a better understanding of the area.

The human activities of production and consumption give rise to solid waste. Given the environmental difficulties associated with dealing with this waste, it is desirable that the amount of waste generated be minimised, at least to the point where the benefits of such reductions exceed the costs involved. While it may be possible to minimise the amount of solid waste which we generate, however, it seems reasonable to say that some amount is an inevitable consequence of human activity. It is also inevitable that this waste will have to be dealt with. The problem then arises of how best to achieve this. As Goddard (1994) points out, until recently this problem was left to engineers with little input from economists. This is changing, however, as the economic dimensions of the issue become more apparent.

Ireland, in common with most developed countries, is generating an increasing quantity of solid waste every year, especially at the household level. The main method chosen here for dealing with solid waste has been, and continues to be, landfill. The reason for this is that landfill in Ireland has been the least expensive way of dealing with solid waste, while at the same time fulfilling perceived requirements regarding public health and environmental standards. This situation is changing, however, for two main reasons. First, new EU guidelines on landfills will soon make landfilling more expensive than it has been up to now. The purpose of these guidelines is to reduce the negative environmental impact of landfills but in so doing they increase the costs of landfilling. Second, although landfill standards are rising, public opposition to landfill siting is on the rise also, as seen most clearly in recent times in the opposition to the proposed new landfill in Kill, Co. Kildare. In addition to rising landfill costs and growing public opposition, the

number of landfills in operation in the country is falling and is set to fall even further in the coming years.

This then is the problem which we face and the reason for this study: a large and growing waste mountain, combined with rising costs of the traditional disposal route and a falling capacity in that route. Broadly speaking what is required is a system of solid waste management that provides methods of dealing with solid waste that balance environmental and economic considerations, while at the same time providing incentives to bring waste generation to an optimal level and to use the most environmentally and economically desirable means of dealing with the solid waste that is generated. Let us now outline how we propose to contribute to the discussion of this problem.

In Chapter 2 we set the scene for our discussion by providing a description of current waste flows in Ireland, including contents and disposal routes. Included in this description are some of the findings of a survey of local authorities regarding their approaches to solid waste management. While local authorities are ultimately responsible for solid waste management, some are now choosing to privatise the collection segment of the system; as can be seen in Table 2.8 of the next chapter, of 90 local authority areas, 13 have contracted out collection. There is also the possibility of the private sector becoming more involved in the disposal segment<sup>1</sup>. Hence, while solid waste is currently predominantly a local authority concern, we are mindful of the growing private sector dimension.

In Chapters 3 and 4 we make a more analytical contribution, along two broad themes. Although waste reduction may be the environmentally preferred option in dealing with solid waste, we postpone most of our remarks on this point to Chapter 4 where it fits in with our discussion on economic incentives in the solid waste area. As such, we start Chapter 3 from the perspective of a given amount of waste which has to be dealt with. We investigate the costs of the various approaches to solid waste management. As already noted, with the costs of landfill rising and its acceptance declining, proposals are being put forward to use alternatives to landfill. Much of the discussion comes from the US and continental Europe and emerging from this discussion is what is known as the waste hierarchy. This is a ranking of the alternative methods of dealing with solid waste based on environmental criteria which puts landfill as the least acceptable method followed by incineration, recycling, re-use and reduction at source. It is this hierarchy which now seems to dominate thinking in this area.

One of the fundamental questions to be addressed in Chapter 3 is how readily this waste hierarchy should be applied to Ireland. Specifically, in Chapter 3 we examine the costs of the various approaches to solid waste, including the

<sup>1</sup> One example is the National Toll Roads' proposal for a landfill site in Mulhuddart, Co. Dublin (Blair, 1995).

environmental costs which are not priced in market transactions, to see if the hierarchy makes economic sense in the Irish context. A sound environmental policy must consider both the costs and benefits of actions. It is possible that, for example, landfill costs in Ireland, both those priced and unpriced in the market, are different from costs elsewhere due to differences in factors such as land valuations, population density and dependence on groundwater for drinking water purposes. Furthermore, population density also alters the economics of an option such as recycling and hence may alter the applicability of the waste hierarchy to Ireland. This examination of the relative costs of alternative approaches to solid waste should enlighten discussion in this area and help in avoiding approaches which, while cost effective elsewhere, may not be so in Ireland. We should point out that environmental considerations will not be absent from our analysis. We merely want to consider costs along with environmental benefits to ensure that environmental protection is not pursued at a cost that may be excessive.

Regardless of the technical approach taken to solid waste management, be it landfill, incineration, recycling, etc., another issue arises which we will address in Chapter 4. This issue is the fact that solid waste services are often unpriced or underpriced. While there are circumstances in which it is optimal to provide a service free of charge, it is generally the case that when a service is underpriced or provided free of charge an excess demand for the service and hence a misallocation of resources will arise. We will argue that solid waste services (SWS) should not be provided free of charge and that introducing economic incentives for people to economise on their use of SWS is an important component of a solid waste strategy. Many people in this country pay nothing for SWS while others pay flat fees. In both cases there is no incentive to economise on the use of the service since additional use of the service does not entail an additional cost to the user. Even those who do face some form of user-charges are rarely paying the full cost. This is particularly true for commercial users of SWS. It is only when faced with the full cost of disposing of their waste that households and firms will have an incentive to minimise the amount of waste going to landfill through reducing the waste they generate or finding alternative routes such as recycling or re-use. In Chapter 4 we examine possible ways of introducing correct incentives into the area of solid waste management. We point out what methods might be used such as user-charges and various taxation schemes. We also analyse the possible effects of the various schemes using economic theory and the empirical work done elsewhere.

Overall, in this report we will attempt to apply two broad economic principles to the solid waste problem, the application of which we feel can lead to a more efficient approach to the problem. First, if an action is to be taken, the benefits of that action should exceed the costs. There is growing evidence that many recycling

programmes that were initiated in the US and Europe were implemented without due recognition of the need to assess the balance between costs and benefits. The consequence of this has been programmes that have imposed heavy economic burdens on communities. Since the benefit of one approach to solid waste management is the diversion of waste from another approach, we need to assess the relative costs of the various approaches if we are to make decisions based on cost/benefit comparisons. Second, the vast majority of resource allocation decisions in a market economy are made in a decentralised way without them being a source of study or concern. This is because a correct pricing structure leads people to use the least costly, most efficient ways to fulfil needs and wants. To use Goddard's (1994) term, because the price mechanism in solid waste has been "short circuited" there is reason to believe that least costly, most efficient ways to deal with waste are not being utilised. By introducing some form of pricing system, it may be possible to bring to the solid waste system some of the advantages of the market system with which we make most of our resource allocation decisions. In addition, by correctly pricing solid waste services people have an incentive to engage in the most environmentally favourable waste strategy, waste reduction.

Finally, in Chapter 5 we offer some conclusions. Our purpose will be to distil what we see as the most important findings and to suggest economically rational ways in which those responsible for the management of solid waste can tackle the waste problem.

## Chapter 2

### *SOLID WASTE FLOWS AND THE CURRENT LEGAL, INSTITUTIONAL AND INCENTIVE STRUCTURES IN IRELAND*

#### *2.1 Introduction*

This chapter sets out the current situation in Ireland *vis-à-vis* the quantities of waste generated, how this is disposed of and the current costs and incentive structure that exist. Brief reference will be made to the waste flows in other European countries. The legal and institutional structure will also be described, as this is experiencing much change at present, with new legislation and regulation coming from both domestic and EU sources.

#### *2.2 Legal and Institutional Structures<sup>2</sup>*

The bodies mainly responsible for waste management in Ireland, either directly or as regulators, have traditionally been the local authorities<sup>3</sup>. The private sector is also involved, especially in the areas of waste collection and industrial waste management. The local authorities' functions have been set out by the Local Government (Sanitary Services) Acts, 1878 to 1964, and the Public Health Acts Amendment Act, 1907. These give local authorities *inter alia* the following powers and obligations:

- (i) They are empowered, but not obliged, to collect household waste. They can charge for this.
- (ii) They *are* required to remove trade/commercial waste if requested to do so by the owner or occupier of the premises, and can charge for so doing.
- (iii) They are empowered to provide "fit buildings or places for the disposal of waste ..... provided no nuisance is created".

There have in the past been no legal restrictions on the disposal of waste by the local authorities within their own areas, while private waste disposal sites require a

<sup>2</sup> Description of the legal framework is largely drawn from Environmental Resources Limited (1993).

<sup>3</sup> Although all the local authorities are independent, in the area of solid waste management the 34 major authorities (i.e., the county councils and the county borough corporations) carry out most of the functions.

licence from the local authority. The legal situation is in the process of changing considerably, however, due to new legislation at both the EU and domestic levels.

Considering EU legislation first, a new Directive<sup>4</sup> on packaging waste has recently been adopted, and a Directive on landfill is proposed. The packaging Directive (EU, 1994) establishes the "waste hierarchy" as the baseline for solid waste management. This ranks in descending order of desirability the various methods of solid waste management, as follows:<sup>5</sup>

reduction at source; re-use; recycling; other forms of recovery (mainly incineration with energy recovery); final disposal (i.e., sanitary landfill and incineration without energy recovery).

The hierarchy seeks to move solid waste management away from the end-of-pipe solutions (such as landfill and incineration) which were the norm in previous decades, towards options that have less environmental impact. Waste management should aim to operate as high up this ranking as possible. However, partly recognising that the hierarchy may not apply always and everywhere, the Directive calls for the completion of life cycle analyses to "justify a clear hierarchy between reusable, recyclable and recoverable packaging". The Directive also sets targets for the recovery and recycling of packaging waste, which for Ireland requires the recovery of 25 per cent by weight of packaging waste within 5 years of the adoption of the Directive, and 50 per cent (including 25 per cent recycling) by 31st December 2005<sup>6</sup>. Within this later target a minimum of 15 per cent by weight of each material must be recycled; there is no such restriction for the 5-year target.

The proposed landfill Directive requires high standards of environmental protection for new landfills, especially in the areas of leachate and gas control. Existing landfills will have to be retro-fitted with features to protect the environment. The proposed Directive also requires that member states "encourage" the full recovery of costs in the setting of prices for landfill sites.

Turning to the domestic legal situation, the most significant development is the recently published Waste Bill (1995), which will bring major changes to solid waste management in Ireland. Existing domestic legislation will be repealed, a number of EU Directives will be given effect, and the roles of local authorities, the Minister of the Environment and EPA will be redefined. Most significantly, the

<sup>4</sup> EU Directives, once adopted, represent a legal obligation on all member states.

<sup>5</sup> This is the definition of the hierarchy as used in EU Legislation. Other versions, widely used, rank re-use, recycling and incineration with energy recovery in descending order of desirability.

<sup>6</sup> In the absence of incinerators with heat recovery in Ireland, the targets effectively apply to recycling.

EPA will become responsible for the licensing of all significant waste disposal activities, including landfills, and the Minister will have wide powers of policy direction regarding the more important aspects of waste management. One of these is the power to impose waste management requirements on producers, retailers and consumers, and the power to exempt these from such requirements if they are participating in "approved (industry-sponsored) waste recovery programmes". This is similar to legislative provisions already enacted in a number of other European countries. With respect to these matters the Bill is mainly an empowering piece of legislation – it gives the Minister powers rather than enacting particular measures itself.

In 1994 the government published its *Recycling for Ireland* strategy document (Department of the Environment, 1994a), which reflects the requirements of the EU packaging Directive described above. It sets recovery/recycling targets for each type of packaging waste to be achieved over 5 years (see Table 2.4). As already stated, in the absence of incineration with heat recovery in Ireland, the targets effectively apply to recycling. The rates are 25 per cent for each material type, and 55 per cent for glass, giving an overall rate of 33 per cent. This rate is higher than the currently proposed Directive 5-year targets, both in overall terms and in terms of each individual material; however they are set with the higher 2005 targets in mind.

The government has also in recent years passed regulations requiring an Environmental Impact Assessment (EIS) for all landfill sites handling more than 25,000 tonnes per annum. Draft guidelines from the EPA impose high standards on the construction and operation of landfills. Much of this is in anticipation of the requirements in the proposed landfill Directive.

With respect to incineration, in terms of regulation, large scale incineration and the incineration of hospital and hazardous wastes are now under the regulatory control of the Environmental Protection Agency (Environmental Protection Agency Act, 1992, first schedule). In terms of policy, the government has recently stated that it would not grant aid the building of a national hazardous waste incinerator, and appears to be encouraging the investigation of alternative methods of dealing with hospital and clinical waste. There is no specific policy in terms of being in favour or against incineration of Municipal Solid Waste (MSW), however the recently published new Alternative Energy Requirement (AER) competition seeks to encourage the development of a 30MW private power station fuelled by biomass. MSW incineration would qualify under this competition.

In summary, the legal situation relating to solid waste management is in a process of change at the moment. The Strategy, the Directives and the Waste Bill (when it is passed) will radically change the nature and costs of waste management in Ireland over the coming years. This represents a clear political commitment to



improved environmental standards and the waste hierarchy (particularly recycling, and the minimisation of the use of landfill)

### *2.3 Overall Magnitudes*

Solid waste in Ireland is classified into 5 main categories – household, commercial, industrial, hazardous, and miscellaneous<sup>7</sup>. Household and commercial together make up Municipal Solid Waste (MSW). The relative quantities under each heading are shown in Table 2.1. While nation-wide data on trends are not available, ESBI/Atkins International (1992) indicates a growth rate for municipal waste of 2-3 per cent per annum in the Dublin region, since the mid-eighties. More recent indications are that waste quantities in Dublin were on a growth path until the early nineties, but may have stabilised since then (ESBI International, 1994). The reasons for this are not clear. Attention is drawn to the uncertainties and disagreements regarding quantities for all the data represented in this chapter. These are highlighted in the notes to each table.

The main disposal routes of this waste are landfill, other on-site disposal, recycling<sup>8</sup> and incineration. The quantities, by type of waste, are shown in Table 2.2. The vast majority at present goes to landfill, or in the case of industrial waste, to disposal on-site. Recycling is a route for some industrial waste and municipal waste, while incineration is used mainly by industry and hospitals to dispose of hazardous waste; there is no incineration of MSW at present in Ireland. Composting is another method of dealing with organic waste, but is currently not used to any significant extent, although Dublin Corporation currently composts its own park green waste. Home composting pilot schemes are in place in a number of local authority areas, and there are plans for centralised composting schemes in Limerick and Dundalk in the near future.

It can be seen that the rates of recycling vary widely across sources of waste. For example, 54 per cent of hazardous waste is recycled, compared with 14.5 per cent of commercial waste and only 1.5 per cent of domestic waste. This pattern is reflected across the European Union (Pearse and Turner, 1992).

<sup>7</sup> This classification is from Department of the Environment Circular 10/78 (ESBI/Atkins International, 1992). Commercial waste comprises waste from offices, shops, restaurants and some small industrial firms; this waste is generally collected as part of the normal waste collection services, either by local authorities or private collectors. Industrial waste is waste from industrial processes, including manufacture and food and animal products processing. Hazardous waste includes hospital waste; miscellaneous waste includes such materials as "cleansing waste" arising from the local authority's street-cleaning services.

<sup>8</sup> Unless otherwise indicated, recycling will denote mechanical recycling (i.e., reprocessing of materials into new products). Incineration with heat recovery is sometimes described as thermal recycling, but this terminology will not in general be used here.

Hazardous waste has increased 36 per cent by weight between 1988 and 1992. However, recycling and incineration thereof has increased by over 50 per cent, and landfilling has fallen by more than 80 per cent over the same period. At present 20 per cent of hazardous waste is exported for disposal (Department of the Environment, 1994b).

Re-use and reduction at source, the two highest ranks in the waste hierarchy, are not included, as they are more in the nature of waste avoidance than waste disposal, and quantitative data on these are difficult to obtain. This highlights a problem with these aspects of waste management, i.e., there is a danger that, because these methods of dealing with solid waste are difficult to quantify, they may gain less attention than other more easily quantifiable methods, especially recycling. If this leads to a de-emphasis of re-use and reduction, the results could be detrimental for the environment. This issue will be returned to later in the chapter.

Table 2.1: *Solid Waste Arising in Ireland, Tonnes Per Annum*

<i>Material</i>	<i>Commercial</i>	<i>Domestic</i>	<i>Industrial</i>	<i>Hazard</i>	<i>Misc.</i>	<i>Totals</i>
Paper	241,579	140,190	90,000			471,769
Glass	51,367	55,958	30,000			137,325
Plastic	36,087	104,763	40,000			180,850
Metals	35,629	32,945	250,000			318,574
Textiles	11,980	71,574	0			83,554
Organic	193,556	379,415	490,000			1,062,971
Mining & quarrying			1,930,000			1,930,000
Effluent sludge			440,000			440,000
Construction & demolition			2,500,000			2,500,000
Others/unknown	196,836	126,820	1,350,000	99,393	105,786	1,878,835
<b>Total</b>	<b>767,034</b>	<b>911,665</b>	<b>7,120,000</b>	<b>99,393</b>	<b>105,786</b>	<b>9,003,878</b>

*Source:* MCOS, 1994; Department of the Environment, Department of the Environment, 1994a, 1994b; Boyle, 1987a, 1987b.

*Notes:*

1. Commercial, domestic and hazardous waste data are for 1994. Industrial and miscellaneous waste data are for 1984.
2. The above indicates that household and commercial waste amount to 1.68 million tonnes. ERL (1992) estimated a figure of 1.97 million tonnes for the same. Both figures come from limited surveys. MCOS (1994) indicate that the quantities of packaging materials arising in the household sector are significantly lower than those estimated by ERL. Kerbside Dublin (1994) estimate that household waste amounts to 759,000 tonnes per annum, similar to the MCOS figure.

3. Excluded from the domestic stream is sewage treatment sludge, amounting to 37,686 tonnes of dry solids (TDS) per annum (approximately 850,000 m<sup>3</sup> before drying). At present 42 per cent of sludge generated goes to landfill and 46 per cent is dumped at sea (Cleary, 1991). Dumping at sea will have to be discontinued after 1998, in accordance with the Urban Waste Water Treatment Directive. In addition, the Directive requires an increase in treatment, which will lead to greater quantities of sludge in the coming years. Weston-FTA (1993) estimate that by the year 2005, 112,000 tonnes dry solids (TDS) will be generated, a threefold increase on current quantities, due to the new treatment requirements. The main disposal route for this sludge will probably be landspreading, but alternatives such as incineration and co-composting also exist. It might also possibly end up in landfill.
5. Also excluded from the domestic stream is the waste dealt with by home composting. A recent survey indicated that 29.6 per cent of households in Ireland composted their kitchen and garden waste, and used the compost (Murphy, *et al.*, 1994). The quantities involved are unknown, although 25 per cent of household organic waste is compostable (ERL, 1993), so the quantities are potentially large.
6. Agricultural waste is ignored throughout. Although quantities in this sector are significant, the waste is mainly organic, and most of it is landspread. The issues related to this are more relevant to water pollution than to solid waste management. There is also some plastic waste from agricultural activities, but we do not have data on this.

Table 2.2: *Disposal Routes of Solid Waste in Ireland, Tonnes Per Annum*

<i>Disposal route</i>	<i>Commercial</i>	<i>Domestic</i>	<i>Industrial</i>	<i>Hazard</i>	<i>Misc.</i>	<i>Totals</i>
Landfill	656,122	898,462	972,000	1,950	105,786	2,634,320
On-site dump			2,060,000			2,060,000
Recycle/recovery	110,912	13,203	632,000	54,100		810,215
Incineration				36,936		36,936
Other/unknown			3,456,000	6,407		3,462,407
<b>Totals</b>	<b>767,034</b>	<b>911,665</b>	<b>7,120,000</b>	<b>99,393</b>	<b>105,786</b>	<b>9,003,878</b>

*Source:* Department of the Environment, 1994a, 1994b; Boyle, 1987a, 1987b.

*Notes:*

1. Commercial, domestic and hazardous waste data are from 1994. Industrial and miscellaneous waste data are from 1984.
2. The majority of industrial waste disposed of on-site consists of 1.93 million tonnes of mining and quarrying waste. In some sources disposal on-site is described as landfill.
3. Included in "unknown" are 535,000 tonnes of industrial waste formerly dumped at sea, a practise discontinued in recent years. Indications are that some of this waste is now treated and landspread. Also under this heading is 2,500,00 tonnes of construction and demolition waste. According to the Department of the Environment, the vast majority of this is used either on site or in land reclamation; very little ends up in landfill sites.
4. Twenty per cent of hazardous waste is exported for disposal. A certain amount of commercial and domestic waste is recycled overseas. We are not aware of any other waste category that is exported for disposal or recovery, although it is likely that at least some of the industrial waste that is recycled or recovered is exported for this purpose.
5. It is assumed that all miscellaneous waste is landfilled.

Each of these routes of solid waste management is now examined in more detail, but first the collection of waste is considered. The order of consideration reflects the waste hierarchy ranking, starting at the bottom with landfill as the main means of waste disposal currently used in Ireland.

### 2.3.1 Collection

Waste management falls into two main stages, collection and disposal. Collection is carried out by both local authorities and private companies. Department of the Environment data indicate that 12 local authorities have fully privatised refuse collection. Others have at least partially privatised the collection of commercial MSW. In the Dublin Corporation area, for instance, 46 per cent of MSW is collected by private contractors (ESBI/Atkins International, 1992).

Collection of most waste is quite a standardised activity, from whatever source, notwithstanding that industrial and commercial waste will generally be in larger and more homogenous quantities than domestic. Exceptions are miscellaneous and hazardous wastes, and recyclables. Also, where disposal sites are distant from collection sites, transfer stations are often used to transfer the waste onto vehicles more suitable for long journeys. These are planned for Dublin and other areas in the coming years.

### 2.3.2 Landfill

Table 2.2 shows that landfill and on-site disposal are the main disposal routes for solid waste at present. Most landfills are operated by the larger local authorities, mainly county councils. Some are also privately operated, although these tend to be smaller in size – data from the late eighties indicate that the local authorities take 90 per cent of all waste deposited to landfill. In 1990 there were 94 local authority landfills in operation, broken down in size as follows:

Table 2.3: *The Size of Local Authority Landfill Sites in Ireland in 1990*

<i>Size category (tonnes per annum)</i>	<i>No. of sites</i>
> 100,000	6
25,000 - 100,000	18
5,000 - 25,000	70
Total	94

*Source:* Markey, 1995

*Notes:*

1. Department of the Environment (1994a) quotes the current number of landfills in Ireland – both public and private – at 110.
2. Our survey of 23 large local authorities covered 50 sites, which had an average throughput of 18,000 tonnes per annum.

Irish landfills are characterised by their small size, and by the fact that many are quite old and are coming to the end of their life-spans. There is therefore likely to be considerable expenditure on new landfills in the coming years. In our survey of 23 large local authorities, 16 stated that they expected to experience capacity problems in the near future. In some areas of the country, notably Dublin, there is an acute need for new landfill capacity.

Two factors will emerge. First, new landfills will be built to much higher standards of environmental protection and safety than before, in accordance with public pressure, the proposed EU Directive on landfills and government regulations. Second, because this will increase costs, and economies of scale will apply (Dennison, 1991), the new sites will tend to be fewer and larger. Department of the Environment (1994a) estimates that the number of landfills in the country will reduce to perhaps 50 by the early years of the next decade. In some cases facilities will be shared by more than one county council. Despite these economies of scale, future costs will be significantly greater than at present. This issue will be expanded on in the next chapter.

In this context there is also the problem of increased public opposition to new waste disposal facilities, on environmental grounds, and also as part of the NIMBY (Not In My Back Yard) syndrome. This has doubtless been fuelled by the negative experience of the operation of old landfill sites in the past. The paradox is that the high level of opposition is constraining the establishment of better-run, modern facilities and thereby encouraging the continued operation of the older, objectionable sites, often beyond their designed useful life. The net result is detrimental to the environment.

### 2.3.3 Incineration

As stated, incineration is carried out by industry and hospitals to dispose of hazardous wastes. There is no municipal incineration in Ireland at present, though this option, especially with energy recovery (steam, electricity or industrial heat), is often cited as a potential part of Ireland's waste management strategy. As already stated, the Department of Transport, Energy and Communications has recently publicised a new Alternative Energy Requirement (AER) to generate energy from biomass, the definition of which would include MSW incineration. However, public opposition to incineration is likely to be at least as strong as it is to landfill.

### 2.3.4 Recycling

Recycling is practised to varying degrees in the industrial and commercial sectors, but is much less prevalent in the domestic sector, as Table 2.2 indicates. Glass and paper recycling are the most established sectors, but ferrous metals, aluminium and plastics are also recycled. However, domestic recycling is the area receiving most attention at the moment, especially with respect to packaging

waste. Ireland, in common with its EU partners, has committed itself to increase recovery/recycling of packaging waste over the next 5 years and beyond. Current and targeted recovery/recycling levels are shown in Table 2.4 (the quantities are from the government's Recycling Strategy). As already stated, in the absence of incineration with energy recovery in this country, the targets effectively relate to recycling. Attention is drawn to the wide variation between these "official" figures for current recycling and the quantities stated by industry, referred to in the notes to the table. Further research is required to determine which figures are more accurate.

Table 2.4: *Current and Targeted Quantities of Packaging Waste Recycling in Ireland*

<i>Material</i>	<i>Present situation (1994):-</i>			<i>Target situation (1999):-</i>	
	<i>Tonnes arising</i>	<i>Percentage recycled</i>	<i>Tonnes recycled</i>	<i>Percentage to be recovered/recycled</i>	<i>Tonnes to be recovered/recycled</i>
Paper	138,051	14.0	19,000	25	35,000
Glass	106,302	21.0	22,000	55	58,000
Plastic	114,811	0.1	100	25	29,000
Ferrous metals	31,286	0.3	100	25	8,000
Aluminium	13,544	4.1	550	25	3,000
Totals	403,994	10.3	41,750	33	133,000

*Source:* Department of the Environment, 1994a.

*Notes:*

1. The target quantities to be recycled are calculated on the basis that the quantities of waste arising are unchanged from current levels.
2. Aluminium recycling is estimated at 10 per cent by Fitzpatrick Associates (1993) and ESBI/Atkins International (1992). Soft Drinks and Beer Bottlers Association (SDBBA) (1994) estimates that can recycling reached 15 per cent in 1993. The last two quote rates in Dublin of 21 per cent.
3. Soft Drinks and Beer Bottlers Association (1994) estimates that glass recycling reached 32 per cent in 1993. ESBI/Atkins International (1992) quote recycling rates in Dublin at 35 per cent.
4. Paper recovered for recycling from all sectors amounts to 84,000 tonnes per annum, 22 per cent of the total paper used (Department of the Environment, 1994a).
5. The latest industry estimates for recycling of plastic packaging is 3.5 per cent.
6. Industry sources quote the quantity of metal packaging waste arising as being considerably lower than those cited above.
7. The Recycling Strategy also includes a recycling target of 25 per cent for newsprint and approximately 17 per cent for organic waste.

The targets are broadly in line with those in the EU Directive on Packaging Waste, which requires Ireland to achieve a recovery rate of 25 per cent within 5 years, rising to 50 per cent by the end of the year 2005 (including 25 per cent

recycling). Before going further, it is perhaps appropriate at this point to sound two notes of caution in relation to these targets:

- (i) In general, targets such as these should only be set after it has been determined that the benefits of achieving these targets exceeds the costs. It is not clear the degree to which this has been done in this case. Indeed, from an economic point of view, it is better to charge the proper price for the use of all resources, and let the market determine the socially optimal level of recycling (and other activities). Charging the proper price will be the subject of the next chapter.
- (ii) The Directive (and the Irish government's recycling strategy described later), while requiring action to promote waste reduction and re-use, only set numerical targets on recycling and recovery. This perhaps reflects the fact that reduction and re-use are difficult to quantify. It could, however, give a perverse behavioural incentive to those affected by the Directive. Firms, industries and countries might concentrate on achieving the numerical recycling/recovery targets, to the neglect of re-use and reduction, because their performance could be more easily evaluated against the numerical targets. Given the inter-changeability of reusable and non-re-usable (but recyclable) packaging, this could conceivably lead to a move away from re-use, which might have a detrimental environmental effect.

Returning to the data in Table 2.4, a number of aspects are apparent. First, the current levels of recycling are significantly lower than the target levels, even if one were to accept industry's claimed levels of recycling for glass and aluminium. Therefore, a large effort will be required to meet the targets. Second, the current rate of recycling varies widely across different material types<sup>9</sup> (this variation reflects the economics of recycling each waste type). As one might expect, paper, glass and aluminium are already recycled to some degree, while at the other end of the scale, plastics recycling is minimal. This might call into question the wisdom of setting uniform targets across widely differing materials; targets which better reflect the viability of recycling each material type might be more easily attainable. It is interesting that the EU Directive does allow the quantity recycled of any one material to be a minimum of 15 per cent, perhaps reflecting the difficulties with recycling certain materials.

Third, the quantities of packaging waste arising and the targeted quantity for recycling are modest in comparison with other wastes, as is apparent from Tables 2.1 and 2.2. Even if we achieve the targets, we will be diverting a further 90,000 tonnes of waste away from landfill, out of a current flow of 2,600,000 tonnes per

<sup>9</sup> This reflects the situation in Germany in 1988, prior to the introduction of the DSD system (Klepper and Michaelis, 1993).

annum. This might suggest that we should concentrate our efforts on other waste streams, especially in the industrial area. However, disposing of packaging waste is problematic because of its diversity and the fact that it contributes disproportionately to litter. Therefore, it may be that the attention being given to it is appropriate.

A further point is that while packaging waste arises approximately 50:50 from the domestic and commercial sectors, almost 70 per cent of the recycling is done in the commercial sector. This is as one might expect, since packaging waste from commercial sources should arise in larger, more homogenous quantities, and therefore should be more economical to recycle. This should be kept in mind when trying to determine the most cost-effective methods of increasing packaging waste recycling.

It is worthwhile considering the capacity for increased recycling in Ireland. This is important because recyclables that must be exported will bear an extra transport cost, which may be critical to the viability of recycling, especially when the relevant markets are weak. It appears that between the Republic and Northern Ireland there is adequate processing capacity for recycling paper. All aluminium is exported to Britain for recycling at the moment. Ireland does have very significant excess capacity for recycling glass, and PET plastic (from which most plastic mineral water and soft drinks bottles are made). Irish Glass Bottle Ltd has 50-60,000 tonnes per annum excess capacity for cullet (MCOS, 1994). Wellman International Ltd has a capacity to recycle 55,000 tonnes of PET per annum, compared to Ireland's annual consumption of 7,000-8,000 tonnes (ERL, 1993), but the collected bottles must first go to the Netherlands for initial processing. Ireland, then, is dependant on outside capacity for recycling aluminium and most plastics. This might be of concern for plastics, where the extra transport costs could affect the viability of recycling; it should not be a problem for aluminium, which is a valuable product and has a well established market.

### 2.3.5 Re-use and Reduction at Source

Re-use is well established at the industrial and commercial level, but has become much less prevalent at the domestic level in recent years. It is difficult to quantify the amount of re-use that goes on, but we can get tentative indications for some industries. The on-licence drinks industry is a case in point. Over 80 per cent of the beer sold on-licence in Ireland is conveyed in reusable aluminium kegs, and the majority of bottled drinks sold in the on-licence trade are conveyed in reusable bottles. SDBBA (1993) indicates that there were 247 million refillable glass bottles in circulation in the licensed trade in 1992. Even a modest estimation of the turnover of these bottles and kegs would indicate that this re-use is significantly more important for waste minimisation than achieving the recycling target for glass, set out in Table 2.4. This brings us back to the point made already, that



putting numerical targets on recycling may take attention away from re-use that is already being achieved. A further point is that the on-licence trade is relatively more important in Ireland than in other EU countries. Therefore, setting targets at EU level for recycling only may not give due credit to Ireland for the amount of "benign" waste management that is already being achieved here, in the area of beverage packaging.

A number of changes in consumer behaviour and retail patterns in recent years have meant a reduction in re-use at the household level. The biggest change has probably been the increased domination by supermarkets, combined with the move to weekly shopping patterns. One effect has been on the use of returnable glass bottles<sup>10</sup>. Tetrapak cartons make better use of supermarket shelf space and involve fewer breakages, and hence are favoured by retailers. Supermarkets (and no doubt consumers) also do not want the inconvenience of dealing with returned empty bottles. In addition, consumers are purchasing beverages, milk, soft drinks, etc., in larger quantities, and the larger, lighter and less breakable non-returnable packaging makes this more convenient. More will be said about the economics of this in the next chapter.

Reduction at source is generally agreed to be the most desirable method of waste management. Yet it is also the most difficult to quantify and cost, since the avoided waste is not easy to identify, and most reduction at source occurs in the private sector. Indeed, in many cases the reduction techniques used by companies are part of in-house technology and as such may be commercial secrets. At the domestic level, consumers can reduce the amount of waste they create by changing their purchasing behaviour to avoid high-waste products, or by purchasing more reusable products or packaging.

<sup>10</sup> One particular case in point is the use of glass milk bottles, which now represent approximately 7 per cent of the national market of 538 million litres per annum. Sales in glass bottles are almost exclusively via doorstep delivery in Dublin. Doorstep delivery represents 40 per cent of the national market, but 50 per cent in Dublin; 50 per cent of doorstep sales in Dublin are in glass milk bottles. In terms of waste cartons avoided, this might be diverting 1,300 tonnes per annum from disposal to landfill (each one-pint carton weighs approximately 19 grams). However, as milk bottles are many times heavier than cartons (approximately 530 grams), a high trippage rate would have to be achieved for there to be a net reduction in waste quantities arising (even taking into account a 35 per cent glass recycling rate in Dublin – see Table 2.4). According to industry sources the trippage rate would be quite high, with a bottle lasting 1 – 1.5 years on average, but we could not find an exact trippage rate.

There may be scope for delivering more of the doorstep sales in milk bottles (it may not be very practical to try to convert shops back to selling in milk bottles). If all of the doorstep sales in the country were sold in glass bottles, this might avoid 7,000 tonnes per annum in waste cartons. But the trippage and recycling rates would determine whether there would be a net reduction in waste quantities.

Industry sources claim that much reduction has occurred over the last two decades, in terms of light-weighting, down-sizing and switches in materials. For example, the use of PET instead of glass for packaging beverages has enabled a very significant weight reduction to be achieved, and has allowed larger pack sizes, such as 2 litre and 3 litre packs, to be used, thus bringing about further savings. Similarly, the weight of beverage cans has been reduced by 25 per cent between 1972 and 1990 (Soft Drinks and Beer Bottlers Association, 1993). However, the amount of waste has been increasing steadily at the domestic level in recent years, despite these reductions. This increase probably reflects greater prosperity, a move to more convenience goods, and to one-trip packaging. In general, as we shall see, waste generation seems to increase with wealth.

#### 2.3.6 Overseas

When considering the situation in Ireland, it is useful to compare the flows here with those in other countries. Table 2.5 below shows the level of household waste generated per capita, while Table 2.6 indicates the disposal methods used in various countries. Caution should be used in interpreting these data – widely varying statistics exist depending on what source is used. We have tried to use what appears to be the most up-to-date data available. This indicates that data collection problems are not unique to Ireland; one would expect that as more stringent EU and national legislation comes into operation in the various countries that data quality will also improve.

Subject to these provisos, Table 2.5 indicates a rough correlation between wealth and a higher level of household waste generation, though there are also wide differences between countries where one would expect similar levels of waste generation (e.g. comparing Holland, Belgium and Germany, or Denmark, Sweden and Finland). It is interesting that Ireland has a low level of waste generation, in international terms. However, as stated, conclusions should be drawn with caution, due to the problems with the data.

Data on disposal methods seem to indicate that the amount of landfill used reflects the relative land scarcity in the particular country. For example, Luxembourg uses landfill for only 22 per cent of its waste, whereas Greece uses it for all of its waste. A number of countries are planning very substantial reductions in landfill use in the coming years. For instance, Germany plans to reduce the organic content of waste to landfill to 5 per cent of the total, Norway plans to ban the landfilling of organic waste altogether, while the Netherlands plans to eliminate landfill use for a number of waste streams, including household waste, by the end of the decade. It is interesting that the second main method of waste disposal used internationally is incineration; this and landfill account for at least 80 per cent of waste disposal in all cases.

Table 2.5: *Household Solid Waste per capita in EU Countries Per Annum*

<i>Country</i>	<i>kg per capita</i>
Austria	367
Belgium	358
Denmark	351
Finland	260
France	348
Germany	417
Greece	(note 2)
Ireland	260
Italy	348
Luxembourg	436
Netherlands	484
Portugal	340
Spain	(note 2)
Sweden	(note 2)
UK	347

*Source:* DSD, 1995.

*Notes:*

1. The most quoted source of data on MSW generation per capita (as opposed to household generation) is Eurostat (1994) cited in Moore (1994). However, these data appeared to be relatively out of date, and in a number of cases the quantity of MSW quoted was less than the quantity of household waste generated per the above table. Therefore we have not quoted the MSW data.
2. We could find no household waste data for Sweden, Spain or Greece. DSD (1995) indicates that MSW generation per capita in Sweden is 314kg and in Spain is 323kg, while Eurostat (1994) indicates that the MSW figure in Greece is 300kg. These would suggest that household waste per capita in these countries is very low by international standards.

Table 2.6: *Disposal Methods for Municipal Solid Waste in EU Countries, by Percentage*

Country	Landfill %	Incineration %	Recycling %	Centralised composting %
Belgium	43	54	3	0
Denmark	18	65	18	1
France	48	42	1	9
Germany	46	36	16	2
Greece	100	0	0	0
Ireland	93	0	7	0
Italy	79	18	1	2
Luxembourg	22	75	2	1
Netherlands	45	35	15	5
Portugal	85	0	0	15
Spain	78	6	0	16
UK	84	12	4	0

*Source:* Department of the Environment, 1992, ERL Ltd, 1993; UK figures are from CSERGE *et al.*, 1993 Patel and Higham, 1995 and Royal Commission on Environmental Pollution, 1993; data for Germany, Portugal, the Netherlands and Belgium are from *WARMER Bulletin*, No. 44, February 1995.

#### 2.4 Current Levels of Costs and Financing and Incentive Structure

This section will examine the current level of costs for solid waste management in Ireland. Much of the data come from a survey we carried out of the 33 main local authorities in the country, which asked them about their costs, income, level of service and revenue collecting systems<sup>11</sup>. As stated already, the private sector is also involved in solid waste management in Ireland, but financial data on this sector are generally not available, so we are confined to considering the public sector. Government of Ireland (1994) indicates that total local authority costs of solid waste management in 1994 were £65 million, excluding some administration and including inter-authority contributions. These costs relate mainly to domestic and commercial waste, and as already seen the vast majority of the waste is landfilled. Table 2.7 gives a breakdown of the 1994 costs.

<sup>11</sup> We had 23 responses, including Cork County Council North, West and South as separate responses. The responding areas are home to 62 per cent of the population of the country.

Table 2.7: *Breakdown of Local Authority Waste Management Costs, 1994*

<i>Expenditure Heading</i>	<i>£'000</i>
Operation of landfills	13,156
Provision of landfills	917
Domestic refuse (collection)	29,093
Street cleaning	15,395
Trade and other waste	532
Litter prevention	414
Loan charges	1,618
Miscellaneous (mainly administration)	4,010
<b>Total</b>	<b>65,135</b>

*Source:* Department of the Environment

The above includes capital expenditure as it is spent; depreciation is not included. Income under the same sub-programme amounts to £18.5 million (including inter-authority contributions), the balance of the costs being financed by rates and the Rate Support Grant from central government. It can be seen therefore that the service is significantly under-priced, and the Polluter Pays Principle is not being adhered to<sup>12</sup>. Our survey of and consultations with local authorities indicated that all of them made a loss on their solid waste activities, although two or three were close to break-even and were hoping to achieve this in the coming years. Economic theory would suggest that in such circumstances the service will be over-used by the public, and more waste will be discarded and landfilled than is economically optimal; this is dealt with in more detail in a later chapter.

As stated, the private sector is also involved in solid waste management. We do not have details of the total expenditure in this area, but it is reasonable to assume that the private operators at least recover their costs from their customers<sup>13</sup>. This is one advantage of privatisation from an economic point of view – provided there is competition or the threat thereof, services should be properly priced to reflect the use of resources involved. However, to the degree that private waste collectors are being under-charged for disposal to local authority landfills, it is likely that they in turn are under-charging their customers.

<sup>12</sup> In fact the extent of the under-pricing is greater than the above costs and income suggest, because external costs are not considered. This issue will be addressed in the next chapter.

<sup>13</sup> In the case where private firms are being paid by the local authority to provide the service, there is of course no guarantee that the local authority is passing on the full costs to users of the service.

We now consider the level and types of charging that local authorities use. Turning first to domestic charges, Table 2.8 indicates the situation in 1994. As can be seen, the majority of local authorities charge their domestic customers for waste disposal; however, in most cases charges are fixed, so there is no incentive on householders to reduce the amount of waste they present for collection (this issue will be returned to in a later chapter). Although only a minority have no charging system, these include all the major urban areas except Cork. This means that a very high proportion of households in the country are not paying at all for their refuse services<sup>14</sup>. As for the level of charges, tag charges range from 25p to £2.00 per tag (to be attached to a bag or wheelie bin), annual wheelie bin charges vary from £45 to £65 per bin, while fixed charges range from £16 to £70 per annum.

Table 2.8: *Local Authority Domestic Waste Collection and Disposal Charges*

<i>Charge system</i>	<i>No. of Local Authorities</i>	<i>Percentage of Population</i>
Tag-a-bag or other volume-related	13	18
Fixed charge	41	27
No charge	23	40
Privatised collection	13	15
Total	90*	100

*Source: Irish Independent, 14/9/94*

\* Cork County Council is divided into North, West and South regions for administrative purposes, and they are treated as separate local authorities in this table.

As for collection and disposal of commercial solid waste by local authorities, complete data are not available, but our survey of the major local authorities provides some data, presented in Table 2.9. As can be seen, volume-related charges are more widely used for the commercial sector, with the majority of respondents (weighted by population) applying these charges. Charges vary widely, however, from £30 per annum for one bag per week to £75 per annum for 10 bags per week. Many of the local authorities indicated that they seek to break even on their commercial collection services, at least on a marginal cost basis.

<sup>14</sup> Although it appears that many local authorities are planning to introduce charging in the near future.

Table 2.9: *Local Authority Commercial Waste Collection and Disposal Charges*

<i>Charge system</i>	<i>No. of Local Authorities</i>	<i>Per cent Weighted by Population</i>
Volume-related	10	64
Fixed charge	3	14
No charge	2	3
Privatised collection	8	19
Total respondents to survey	23	100

*Note:*

1. Cork County Council is divided into North, West and South regions for administrative purposes, and they are treated as separate local authorities in this table.
2. Commercial collection is at least partially privatised in all but one of the local authority areas surveyed. The cases of privatised collection in the table are where collection is wholly or mostly privatised.

We now consider direct delivery of waste to local authority landfills, either by members of the public, commercial or industrial concerns, or private waste collectors. The details are in Table 2.10, and as can be seen, charging is more the norm at landfill sites than is the case for collection services. However, indications are that in most cases the charges do not cover the full cost of operating the landfill.

Table 2.10: *Charges for Disposal Direct to Local Authority Landfills*

<i>Charge type</i>	<i>No. of Local Authorities</i>	
	<i>Domestic</i>	<i>Commercial</i>
By weight, volume or by vehicle	16	17
No charges	3	2
No landfills in the area	3	3
Landfills privatised	1	1
Totals	23	23

*Note:* Charges are mainly by vehicle or volume. New landfills will generally be equipped with weighbridges, so in the future weight-related charges may become more the norm.

To summarise briefly the incentive structure that exists at the moment in the public provision of solid waste services:

- (i) the service is far from self-financing, so users are being under-charged, giving an incentive to over-use the service;
- (ii) a large proportion of householders in the state are not being charged at all for solid waste services, and in the majority of areas where charges do

- exist, they are flat rate, which again gives no incentive to economise on use of the service;
- (iii) at the commercial level, charging is more the norm, and charges tend to be more use-related; this may reflect that commercial concerns have more scope to reduce their waste than do households;
  - (iv) charging is the norm for waste delivered directly to landfills, either by vehicle, volume or weight.

The above discussion relates to the current situation. As already indicated, costs are due to increase very considerably in the future, as a result of more stringent rules on the operation of solid waste services, especially landfill. The next chapter will look at this in detail, where we present a model of the costs of modern landfill. This model indicates that the cost of replacing all the current landfills with modern facilities will entail a capital expenditure over the coming years of in excess of £250 million<sup>15</sup>; this will increase annual expenditure on landfill facilities by the local authorities (or private operators as the case may be) to more than £30 million<sup>16</sup>, from the current level of £14 million (see Table 2.7). This will obviously have major repercussions for local authority finances and current levels of charges for the household and commercial sectors. The problem is more acute for solid waste management than for other environmental services provided by the local authorities, because it appears that there will be no capital grants forthcoming either from central government or the EU for the construction of waste management facilities. In this context, it is expected that more local authorities will be forced to bring in charges and to increase their existing charges, in order to finance solid waste management in the future. Alternatively, more of them may privatise their waste management services. The private sector would most definitely pass on cost increases to their consumers, but in the long run it is difficult to see how the local authorities could fail to do so also.

<sup>15</sup> On the basis that the number of landfills will be reduced to 50, the quantity of waste going to those landfills will be 2,600,000 tonnes per annum (as per Table 2.2), and making certain assumption about the size of landfills to be built.

<sup>16</sup> Assuming that landfill construction is financed over 20 years at a real interest rate of 5 per cent, and adding operating costs for the new facilities.



## Chapter 3

### *METHODS OF DEALING WITH SOLID WASTE AND THEIR COSTS*

#### *3.1 Introduction*

In considering the economics of solid waste management in Ireland, a starting point is to determine the costs per tonne of each alternative waste management route. These costs include not only the market costs, but also the external costs and benefits related thereto (that is, the environmental costs and benefits not reflected in the market cost). Having determined these costs, we can hopefully identify which waste disposal routes are most efficient and environmentally appropriate in the Irish context.

This is not a straightforward problem, because costs will vary over scale and between urban and rural regions. Hence the result is likely to be a combination of waste management routes rather than one global solution. Another complication is that data for Ireland are incomplete at best, and in some cases the option being considered does not currently operate in Ireland (e.g. MSW incineration). Therefore data from overseas are used in a number of instances. This is obviously not ideal, as circumstances can differ widely between overseas countries and Ireland (not to mention from region to region within countries). While there are dangers in relying on international data and applying them to Ireland, in some cases we are forced to take this approach. In other situations very strong assumptions have had to be made in order to generate monetary values for cost items. Again, where this has been done the results are less certain than we would have wished, and we have flagged this where appropriate throughout the chapter. That said, the cost estimations are based on the best data we could find, and they give some indications of relative magnitudes for each waste disposal route.

Before we proceed, two important points must be made. First, regardless of how we decide to deal with this issue, it appears that there will be a need for landfill for the foreseeable future; indeed it will probably remain an essential part of solid waste management for most if not all of Ireland. Even if we achieve the recycling target of 33 per cent of packaging waste, this will only divert approximately 100,000 tonnes of waste per annum, 4 per cent of the total currently going to landfill. Incineration also cannot be the full answer, because not

all waste is combustible, an incinerator cannot operate 100 per cent of the time, and there will still remain a residue of  $\frac{1}{3}$  by weight which will have to be disposed of. More importantly, outside Dublin and possibly Cork many alternative options such as kerbside recycling and incineration do not appear to be viable. This is very important in the context of the waste hierarchy, which puts a ranking of desirability in environmental terms on the various methods of waste management, ranging from reduction at source at the top to landfill at the bottom. This hierarchy seeks to move solid waste management away from the end-of-pipe solutions (such as landfill and incineration) which were the norm in previous decades towards options that have less environmental impact. While the hierarchy may be appropriate in large, densely populated countries, its application in a country like Ireland, without any reference to the differing circumstances and costs here, may not be optimal. Part of the aim of this chapter is to provide some illumination on these questions.

Second, there is a factor which must be kept in mind in the comparison of costs of alternative waste management routes. That is, the most important thing is not the average cost of individual disposal routes, but the cost of the overall system. Another way of looking at this is that, given that there will be a number of alternative routes in existence, the avoidable costs (or long-run marginal costs – LRMC<sup>17</sup>) of each are what are important. We need to compare the cost of disposing of a tonne of material by one route with the saving from diverting it from another. If the net amount represents a saving, then the overall cost to society of solid waste management has fallen, and it makes sense to change the disposal route. However, because in many cases the alternatives do not already exist in

<sup>17</sup> It is important to distinguish between short-run marginal cost (SRMC) and long-run marginal cost. SRMC is the familiar concept of the extra cost of providing an extra unit of a good or service. SRMC can be very low, until full capacity of the landfill, factory, etc., is reached. At this point, the cost of an extra unit becomes huge, as a new or bigger facility has to be built. The concept of LRMC gets over this problem, by smoothing out the effect of building a new facility sometime in the future. It charges each unit not only with the extra cost of providing that unit *now*, but also with the fact that producing that unit brings nearer the day when full capacity is reached and a new or bigger facility will have to be built. Bringing forward capital expenditure has a cost because of the time value of money.

In practical terms there are a number of ways of calculating LRMC. One method, known as Marginal Capacity Cost, is defined as the difference in capital costs between the most likely size of facility and the next biggest size, divided by the difference in quantities of waste dealt with in each (each term expressed in present value terms). Marginal operating costs are then added to this (in many cases average operating costs are used if they are not materially different from the marginal costs). Scott and Lawlor (1994) contains a more detailed discussion of LRMC, and the alternative methods of calculating it.

Ireland, and would have to be developed from scratch, the marginal cost is often effectively the average cost. More specifically, given that currently the vast majority of solid waste goes to landfill, we will generally be comparing the LRMC of landfill with the average costs of the alternatives, to determine which is cheaper.

In the rest of this chapter each route will be considered in turn. There will be a general discussion, followed by an exploration of the average costs, and where appropriate the LRMC. Each will then be compared with the cost of landfill. Finally there will be a summary and conclusions.

### 3.2 Collection

The first element of cost to be examined is general MSW collection. This is a vital component of the costs of landfill, and also of incineration<sup>18</sup>. Our survey of local authorities revealed an average collection cost of £38 per tonne, with a range from £22 to £65. This is considerably higher than the cost of landfill, as we shall see. It is interesting that collection is the highest cost element of solid waste management, although it generally attracts less attention than final disposal. There are likely to be changes both in the nature and cost of MSW collection in the future, due to increased efficiency and rationalisation of the number of landfills in the coming years.

Considering efficiency first, there is reason to believe that for some of the larger local authorities collection costs are higher than they should be, due to inefficiencies and out-dated work practices. Dennison (1994) details some of these:

- (a) "task and finish" – this is the system whereby each collection crew has a particular route to collect, and once this is finished they are entitled to go home. In many cases the route takes 3.5 to 5.5 hours to complete, much shorter than the official working day.
- (b) bin type – traditionally a collection crew consists of a driver and 4 collectors. However, where wheelie bins are used this can be reduced to a driver and 2 collectors. In addition, more houses can be serviced in the same time-span, thus further increasing productivity. The use of wheelie bins is very apparent where collection has been privatised.
- (c) "driver" status – drivers do not in general assist in collection, and thus they may be under-employed. While there are practical difficulties with using drivers to collect in urban areas with many short stops, the abolition of driver status so that all operatives can drive and collect may improve productivity.
- (d) changes in collection times – changes to avoid rush-hour may increase productivity.

<sup>18</sup> Collection is also a consideration with recycling and re-use, but these will be dealt with in the discussions of those disposal routes.

It is difficult to estimate what effect greater efficiency might have on collection costs, but one way is to consider what costs are where the service has been privatised. Provided there is competition, the privatised costs should indicate the cost of providing the service efficiently. A number of local authorities around Ireland have privatised their collection services, but there is as yet no study of how this has affected costs. Dennison (1994) discusses studies in other countries where collection has been widely privatised. These report that in the USA private collection costs roughly 20 per cent less than public collection. In Canada, private collection is approximately 30 per cent cheaper, except where public and private collectors compete with each other (in that case there is no difference in cost). In the UK, costs fell by 22 per cent in areas where services had been contracted out; in many cases the contractor was an arms-length subsidiary of the local authority.

The second factor that will effect collection costs is the rationalisation of landfills. As stated in the previous chapter, it is expected that the number of landfills will halve over the next decade or so. This will result in longer journeys to landfill sites and the increased use of transfer stations, which will tend to increase collection costs.

We can make a rough estimation of what collection costs might be in the future. Extra efficiency might reduce costs by say 20 per cent, bringing average costs down from £38 to £30 per tonne. As for the rationalisation of landfills and the use of transfer stations, ESBI/Atkins International (1992) estimated for Dublin Corporation that this might cost an extra £12 per tonne. Reference to financial data in the Waste Management Plans of some other counties indicates that this level might not be untypical of the rest of the country. Adding this to the £30 per tonne gives a possible future cost of £42 per tonne for collection. We use this cost for collection in all circumstances, regardless of size of waste disposal facility. This should not be inappropriate, unless large regional landfills are built, remote from the area of collection. Higher collection costs would probably then be incurred.

Considering the LRMC of collection, it appears that collection costs are largely fixed, since the truck and its crew must pass and pick up at each household or business, regardless of the quantity of waste left out for collection (assuming that some quantity will be left out). The work practises referred to already would contribute to this, but research from the UK, where collection has been much more open to competition, indicates that costs there also remain largely fixed. Touche Ross (1991) estimate that in the UK short-run marginal costs of collection are 10 per cent of average costs in urban areas and 20 per cent in rural areas (distance travelled is more important in the latter). They estimate that in the longer run, approximately 27 per cent of costs are variable. If we take this latter figure and

apply it to our average collection cost of £42 per tonne, it gives a LRMC of collection of £11 per tonne.

### 3.3 Landfill

We have seen that landfill is the major disposal route for waste in Ireland. If we include on-site industrial dumps, it accounts for over 70 per cent of waste disposal. For the domestic sector this rises to 98 per cent. Landfill is widely regarded as the least desirable method of disposal, with international legislation and policies aimed at encouraging the alternatives. However, landfill is also the easiest and most straightforward method of disposal, notwithstanding that recent and upcoming legislation will make it significantly more technically demanding and expensive.

It should be noted, however, that even if one accepts that in general landfill is the least desirable disposal route, there may be circumstances where it represents the most environmentally sound method. This will be the case for the residue of waste which cannot be usefully eliminated by the alternative methods. In another sense, where a particular region has access to plentiful, properly constructed and operated landfill space, is remote from recycling markets and incinerators, and produces relatively little waste, landfill may be the least-cost and environmentally best disposal method. This may well apply to large areas of Ireland, when new modern landfills are in place throughout the country. In this context, optimising the use of landfill (in conjunction with the other waste management options) will be the most important task of solid waste management in Ireland for the foreseeable future.

#### 3.3.1 Landfill Costs – Internal and External

Dennison (1995) develops a model which estimates a range of costs for different size landfills, summarised in Table 3.1. Average and Long-Run Marginal Cost (LRMC)<sup>19</sup> are given. This shows that costs are affected by economies of scale, with considerable levelling off at the higher landfill sizes. This is more apparent from a graph (see Figure 3.1). The considerable reductions in average costs from the smaller landfill sizes (25,000 and 50,000 tonnes per annum) to the larger sizes indicate that considerable savings can be made by building fewer, larger landfills, although this saving would have to be balanced against higher collection costs. That said, there will be a considerable degree of variability in the cost levels for particular sites, so the cost for an individual site may not fit well into the pattern in the graph. This appears as if it may be the case for landfill sites in Dublin, where significantly higher costs are being predicted, despite the fact that it seems that the landfills to be built in Dublin will probably be designed to deal with 300,000 to 400,000 tonnes per annum – considerably bigger than those in our

<sup>19</sup> LRMC is calculated by combining the Marginal Capacity Cost with the marginal operating costs, as described in the first footnote of this chapter.

model. The remoteness of prospective sites from the sources of waste, as well as very stringent planning requirements, may be factors in pushing up costs in Dublin.

Table 3.1: *Landfill Costs per Tonne, by Size of Landfill*

<i>Annual Tonnage</i>	<i>Average Cost per Tonne</i> £	<i>Long-Run Marginal Cost</i> <i>per Tonne</i> £
25,000	25	7
50,000	16	6
100,000	11	5
150,000	9	5
200,000	8	4
250,000	7	

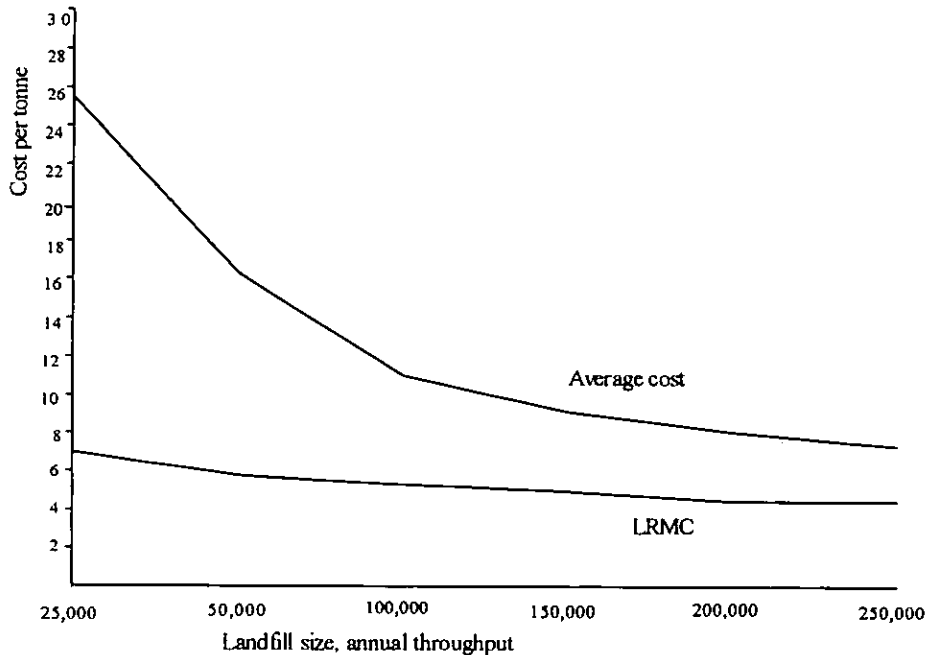
*Source:* Dennison, 1995

*Note:*

1. The above costs do not include the revenues and costs of landfill gas exploitation. Dennison estimates that this might be viable for sites with an annual throughput of 100,000 tonnes or more, and would reduce costs by approximately 30p per tonne.
2. Landfill costs in Dublin appear as if they may be higher than those indicated here, despite the fact that such landfills would probably also deal with higher quantities of waste (perhaps 300,000 to 400,000 tonnes per annum). Predictions of costs as high as £25 per tonne have been made. These numbers are very speculative, however, and are made more so by the fact that both proposed new landfill sites in Dublin are going through planning or legal procedures at the time of writing.

In addition to the internal (or market) costs considered above, there also exist external (or environmental) costs which must be included to calculate the full cost of disposal to landfill. No data exist on these costs for Ireland, but CSERGE *et al.* (1993) have carried out a study for the UK, based on the external costs that might apply to a new, state-of-the-art landfill, and we have used their findings. They divided external costs into (1) a disamenity cost, or fixed externality and (2) a variable externality. The disamenity cost is the cost imposed by the existence of the landfill, regardless of the quantity of waste going into it. This is mainly manifested by reductions in property valuations in the vicinity of the site. The variable externality is due to the pollution from the site, and is directly related to the quantity of waste going into the site, e.g. air pollution, noise pollution, traffic congestion, etc.

Figure 3.1: Average Cost and LRMC per Tonne of Landfill



CSERGE found that no evaluations of disamenity cost had been carried out for the UK, but they review a number of studies of this kind from the USA. The most applicable study, Roberts *et al.* (1991), put the disamenity cost at £139<sup>20</sup> per household per annum, in terms of willingness to pay to avoid being located within

<sup>20</sup> US\$227 @ US\$1.6381 = £1. Strictly speaking, this cost may include more than the pure disamenity, since residents may be factoring in some of the use-related costs to their willingness to pay calculations. There is some evidence from actual negotiated compensation agreements in the USA that compensation increases with the size of the facility (Nieves *et al.*, 1992). This suggests that disamenity costs are not completely fixed; however we will work on the basis that they are, for simplicity's sake.

Another US study by Nelson *et al.* (1992) found that house prices rose by 6.2 per cent per mile from the landfill, which is higher than the Roberts estimate, but the effect decayed after 2 - 2.5 miles, so overall the two estimates are broadly comparable.

Kiel and McClain (1995) illustrate that house prices in areas remote from the facility may increase, as demand for such houses increases. Therefore, the disamenity cost to society as a whole may not be as large as the above calculation would suggest. They also find that the effect on house prices changes over time, with the effect maximised when the facility comes on line, and falling after a number of years of operation. Neither of these effects is considered here.

4 miles of the landfill site. They warned that this cost level might not be transferable to the UK, and this proviso would apply also with respect to Ireland<sup>21</sup>. However, we will use this value in the absence of something better, at least to demonstrate how these costs might affect the cost of landfilling.

CSERGE estimated that the variable external costs related to landfill with energy recovery (from methane gas) were £1-2 per tonne, and without energy recovery were £3-4 per tonne<sup>22</sup> – the difference being due to the benefit of displacing the burning of fossil fuels (see Appendix I<sup>23</sup>).

The next step is to apply these costs to the waste that goes into landfill. The variable external cost is a straightforward per tonne figure, but the fixed externality for a given landfill site would depend on the throughput of waste and the housing density in the area. If we were to take a hypothetical landfill site in Dublin city, with a throughput of 200,000 tonnes per annum, and 45,000 households within 4 miles in any direction of the landfill (this is the approximate household density in the Dublin area), then the disamenity cost might be

$$139 \times 45,000 / 200,000 = \text{£}31 \text{ per tonne.}$$

The same site built in a sparsely populated area, with say 1,000 houses within 4 miles of the site, would have a disamenity cost of

$$139 \times 1,000 / 200,000 = \text{£}0.70 \text{ per tonne.}$$

Given the decentralised structure of waste management in Ireland, such large sites will be the exception, however. Most sites will be smaller – perhaps 50,000 tonnes per annum<sup>24</sup> – and are also likely to be built in less populated areas. If we

<sup>21</sup> The biggest concern in using these numbers for Ireland is probably the difference in market prices of houses in the two countries. We contacted a number of estate agents and auctioneers to ask their opinion of the level of disamenity costs that might apply in Ireland. While they agreed that there was definitely a cost in terms of reductions in property valuations, they stated that there were too many variables to come up with a "rule of thumb" on the issue. One point that was made was that the uncertainty aroused by proposals to build landfills had the effect of putting property sales and developments "on hold", until the uncertainties were resolved. Once it was known whether and where a landfill was to be built, the property market in the area would reactivate again.

<sup>22</sup> The CSERGE calculation is based on the externalities that would arise in the case of a modern landfill, so the externalities from older landfills might be considerably greater. However, since all landfills will be required to be upgraded to modern standards over the coming years, we will not investigate this issue.

<sup>23</sup> There are a number of uncertainties about the CSERGE findings and these are set out in the notes to the appendix. Some of these suggest that external costs are understated, while one point would suggest that they are overstated, so the net effect is not clear. Because of this we have not adjusted the figures.

<sup>24</sup> Taking the tonnage currently going to landfill of 2.6 million tonnes per annum (see Table 2.2), and dividing by the projected number of landfills in Ireland in the future (50) gives an average tonnage of 52,000 tonnes per annum.



take this size of site, with 1,000 households within 4 miles of the site, the disamenity cost would be –

$$139 \times 1,000/50,000 = \text{£}3 \text{ per tonne}$$

The above calculations demonstrate the sensitivity of this approach to the housing density around the site, as well as to the method of estimating the disamenity cost. In particular, an environmentally valuable site with few houses would carry a low external cost, which is perhaps not valid<sup>25</sup>. However, if only environmentally unremarkable sites are used then this approach is acceptable.

Assuming that only sparsely populated and environmentally unremarkable areas would be considered for siting landfills in the future, one could take the estimate of £3 per tonne disamenity cost, and add the variable external cost, to get a total external cost of £4-7 per tonne, depending on whether energy recovery will be included. It appears that for the majority of Irish landfills energy recovery may not be viable due to their small size<sup>26</sup>, so for landfill sites in Ireland the external costs might be nearer to £7 per tonne. For convenience we will use this rate for all landfill sites, regardless of size. Larger sites may be closer to more densely populated areas, which would increase the disamenity cost, but would have a higher tonnage throughput and might include energy recovery, which would both reduce variable external costs per tonne; so using a constant rate for all sizes is perhaps not inappropriate.

We can then calculate a hypothetical average cost per tonne of collection and disposal to modern landfill, as set out in Table 3.2. Attention is drawn to the note on the possible costs of landfill in Dublin.

Table 3.2 gives estimated average costs of landfill in Ireland, having made a number of important assumptions, mainly in relation to the externalities. A different set of assumptions would result in different figures. Estimation of external costs in the Irish context would result in a more robust estimation; however this would require a very detailed survey to be carried out, and that is beyond the scope of this study. We will therefore proceed with the above numbers, while keeping in mind the limitations thereof.

<sup>25</sup> In the language of environmental economics, only use value is being considered by this method; non-use values such as option and existence values are ignored. Option values relate to the benefit society obtains from having the option to use a particular (environmental) asset. Existence values relate to the benefit from knowing that a particular asset exists, rather than from using it *per se*.

<sup>26</sup> Dennison (1995) indicates that at least 1 million tonnes of waste must be in place before gas extraction is viable.

Table 3.2: *Average Costs of Disposal to Landfill*

<i>Landfill Size (tonnes per annum)</i>	<i>Categories of Cost per Tonne</i>			
	<i>Collection</i>	<i>Disposal</i>	<i>External</i>	<i>Total</i>
	£	£	£	£
25,000	42	25	7	74
50,000	42	16	7	65
100,000	42	11	7	60
150,000	42	9	7	58
200,000	42	8	7	57
250,000	42	7	7	56

*Note:* As stated already, there are indications that average costs of landfill disposal in Dublin may be as high as £25 per tonne. This would give an overall average cost of £74 per tonne, including collection and external costs.

The next step is to estimate the LRMC of landfill. This will tell us the resources saved by diverting waste from landfill. If many of the costs of landfill are fixed, we may not save many resources by (say) recycling or re-use. In such a case, these activities are more difficult to justify. We have already estimated the LRMC of the landfill site itself (see Table 3.1), and of collection. This leaves the external cost.

Our estimate of the external costs of £7 per tonne was made up of a variable cost of £4 and a fixed (disamenity) cost of £3. The latter arises due to the existence of the landfill, regardless of the quantity of waste going through it. Therefore, the only way that it could be included in a LRMC cost is if one argued that there would be *more* (rather than larger) landfill sites if the quantity of waste were to increase<sup>27</sup>. Given the rationalisation in site numbers and the move towards larger sites, this is perhaps unlikely. This allows one to argue that only the variable external costs should be included in LRMC<sup>28</sup>. We can, therefore, calculate a LRMC for our various sizes of landfill as follows:

<sup>27</sup> We are also assuming that landfills will have to be built, whatever the waste management strategy will be.

<sup>28</sup> Not having to consider the fixed external cost in the LRMC is also convenient, in that the estimation of this is the least satisfactory element of our calculation of landfill costs.

Table 3.3: *Long-Run Marginal Costs of Disposal to Landfill*

Landfill Size (tonnes per annum)	Categories of Cost per Tonne			
	Collection	Disposal	External	Total
	£	£	£	£
25,000	11	7	4	22
50,000	11	6	4	21
100,000	11	5	4	20
150,000	11	5	4	20
200,000	11	4	4	19
250,000	11	4	4	19

*Note:*

1. The LRMC of the 250,000 tonne landfill site is assumed to be consistent with that of the 200,000 tonne site.
2. We have already mentioned that landfill costs in Dublin may be significantly higher than those in our model. Taking a case where the average cost of landfill in Dublin might be as high as £74 per tonne (including collection and external costs, as indicated in Table 3.2), the LRMC of such a landfill might be as high as £25 per tonne (including collection and external costs). Once again we reiterate that these numbers are very speculative.

LRMC appears to be very constant over the various landfill sizes. The totals are considerably less than the average costs, and indicate the net resource saving from diverting a tonne of waste from landfill to some other disposal option.

### 3.3.2 The Sustainability of Landfill?

Before leaving the question of the cost of landfill, there is one other aspect which we have not discussed, and that relates to environmental sustainability. In other words, can we continue to landfill our solid waste without imposing significant environmental burdens on future generations? There are a number of characteristics of landfill, as opposed to alternative waste management routes, which raise questions of sustainability, including:

- (i) Sites take up considerable areas of land, and have limited life-spans, at the end of which a new site must be found. Given the increasingly stringent criteria for site selection, it is conceivable that we will eventually run out of suitable sites.
- (ii) Landfills will produce gas and leachate for long periods of time after their useful lives. Notwithstanding the potential to harness the gas as fuel, these pollutants will impose burdens on future generations.

The future costs of dealing with these problems may be significant, and one might argue that current disposal to landfill should carry some "sustainability

levy" to reflect this cost. However, estimation of the appropriate cost would be extremely difficult, and there are a number of arguments against imposing a levy for sustainability reasons.

First, though we may run out of suitable sites in years to come, there are alternatives – reduction, re-use, recycling, incineration, etc.<sup>29</sup> – and as suitable sites become scarcer their costs will increase<sup>30</sup>, thus making these other options more viable. If we actually run out of sites, the cost of landfill will become infinite, and alternative methods will deal with all of the solid waste stream. Over time, also, these alternative technologies should improve and become more cost effective, and so may take over from landfill before this becomes very scarce. In addition, old landfills are converted back to some other use at the end of their lives, usually agriculture or amenity, so it is not the case that the land is useless after being used for landfill.

Second, new approaches to landfill management are attempting to tackle the problem of long-term pollution<sup>31</sup>. These methods are increasingly being used on new landfill sites (their costs are included in the landfill costs represented in this study).

In summary, then, although there are issues of sustainability relating to landfill, the existence of alternative waste disposal methods, and new landfill management strategies, appear to deal with these issues. We conclude that a levy or tax to deal with the question of sustainability *per se* would not be appropriate. The question of a landfill levy to deal with the external costs as described earlier in this section will be returned to in a later chapter.

### 3.4 Incineration and Refuse Derived Fuel (RDF)

As with our consideration of landfill, we will have a general discussion of incineration, followed by an estimation of its average costs and LRMC. We will then compare it with the costs of landfill. RDF will also be considered.

<sup>29</sup> Further alternatives may be developed in the future.

<sup>30</sup> Assuming the owner of the land which has been chosen as a landfill site is fully compensated to reflect its value for landfill.

<sup>31</sup> For example, the "dry tomb" approach attempts to minimise the amount of moisture going into the landfill, to reduce the quantities of gas and leachate produced. Other approaches seek to minimise the amounts of biodegradable material going into the landfill, again with the aim of reducing gas and leachate emissions. This approach is being adopted in a number of European countries, including Germany, where new regulations requires that biodegradable materials are to be reduced to 5 per cent of disposals to landfill. The "flushing reactor" method, on the other hand, increases the levels of moisture going into the site, in order to speed up the production of gas and leachate, and stabilise the landfill over a shorter period of time. This assists in the control of the pollutants, and also make gas exploitation more viable.

Incineration of municipal solid waste has the advantages of reducing the weight and volume of waste significantly, and of generating energy in the form of steam or heat. The steam can be used to generate electricity, or either steam or heat can be used for district heating or industrial processes. More recently plants producing a combination of the two have been developed. These are known as combined heat and power (CHP) plants, and they increase the energy-efficiency of the process very considerably. Incineration has been used in Scandinavian countries since before the Second World War, and is also widely used in countries where lack of land makes landfill problematic. As such, its applicability to Ireland may be limited, except in Dublin, where landfill capacity may become scarce in the coming years. Incineration is also seen as a method of dealing with the organic portion of MSW, where the dumping of such waste to landfill is raising concerns relating to methane production<sup>32</sup>. In this context incineration becomes more attractive, as the ash from incineration is relatively biologically inactive<sup>33</sup>, and the main gas released by incineration is CO<sub>2</sub>, which is less damaging as a Greenhouse gas than methane<sup>34</sup>. Another consideration is that incineration with energy recovery is a valid method of recovery under the EU packaging Directive and the Irish government's Recycling Strategy, and in its absence very high levels of recycling would have to be achieved, which might be extremely expensive. However, this is a question of meeting legally set targets rather than of economics, and we will not consider it further here.

Incineration is a highly technical and capital intensive operation, to which significant economies of scale apply. It seems that at present levels of technology and energy prices a minimum throughput of 200,000-250,000 tonnes per annum are required for financial and technical viability (Royal Commission on Environmental Pollution, 1993). This would also appear to rule out incineration in many areas of Ireland<sup>35</sup>. However, it may be that in future years the technology may be viable and environmentally acceptable on smaller scales. As against this, more stringent pollution control requirements applying to incinerators are making

<sup>32</sup> Even with energy recovery from landfill gases, it appears that the maximum collection efficiency is 40 per cent (Royal Commission on Environmental Pollution, 1993). When this collected methane is burned the carbon content is converted to carbon dioxide.

<sup>33</sup> Although Nealon (1995) indicates that some methane continues to be generated in landfills that take only incinerator ash.

<sup>34</sup> The Greenhouse effect of methane is 7.5 times stronger than the effect of the same weight of carbon in the form of carbon dioxide (Royal Commission on Environmental Pollution, 1993).

<sup>35</sup> Although one option may be to construct regional incinerators which would be supplied by a number of counties.

the process more expensive, and increasing the size requirement for economic viability.

Apart from the above factors, a number of technical issues are relevant. First, incineration is affected by the existence of alternative waste disposal routes, because of their effect on feedstock, i.e., the material to be incinerated. For instance, increased recycling of paper and plastics will reduce the potential feedstock for incineration. On the other hand, increased recycling of non-combustibles prior to incineration will make the latter more viable, by reducing the toxicity of the remaining ash. Second, not all solid waste is combustible and incinerators will not be in service 100 per cent of the time, so alternative waste disposal routes will also have to be in place. Finally, incineration of waste, even when the feedstock is 100 per cent combustible, will only reduce the weight of the waste by 67 to 75 per cent (Moore, 1994), and the volume by 90 per cent (Grehan and Dodd, 1994). Ferrous metals can be recovered from this ash, but the balance is generally landfilled (although in some countries it is used as infill for road construction). This residue is regarded as a hazardous waste, due to the concentration of heavy metals therein. Hence incineration will not on its own deal with the entire solid waste problem: it must operate in conjunction with one or more alternative disposal routes.

A variation on straightforward incineration is the production of refuse derived fuel (RDF). In this process non-combustibles in the MSW are removed and the combustible element, often compressed into briquettes or pellets, is used as a fuel. In contrast to incineration, only 33 per cent of the MSW is suitable for conversion to RDF; a further 3.4 per cent of the materials can be recovered (mainly metals), and the balance of over 60 per cent must be otherwise dealt with. There are a number of technical and quality problems with RDF, which mean that it cannot be burnt in every furnace type; usually it is used in cement kilns and steel smelters. Another possibility is to burn the RDF to generate electricity, rather than sell the product as fuel. At present there are no users of RDF in Ireland, so markets would need to be established before RDF production could be considered.

#### 3.4.1 Costs of Incineration

Estimating the costs of incineration for Ireland is difficult, because no MSW incinerators have been built here, and also because the technology and environmental regulations are changing rapidly. The best estimates we could get are UK figures from Patel and Higham (1995), who have calculated costs for two sizes of incinerators – 400,000 and 200,000 tonnes per annum – incorporating electricity generation (at 450kWh per tonne of MSW). These plants have a generating capacity of 24MW and 12MW respectively, and a lifespan of 20 years. We have taken their calculation of the minimum per tonnage charge ("gate fee") required to break even, and adjusted them to comply more closely with Irish

conditions<sup>36</sup>. The results are summarised in Table 3.4 (adjustments to Patel and Higham's calculation are stated in the notes to the table).

Table 3.4: *Capital Costs and Required Gate Fees for Incineration with Electricity Generation*

Item	Plant Size (tonnes per annum)	
	200,000	400,000
Capital cost (£)	47,300,000	80,900,000
Required gate fee (£ per tonne) where electricity price is 2.5p/kWh	32	25
Required gate fee (£ per tonne) where electricity price is 4p/kWh	25	18

Source: Patel and Higham, 1995.

Notes:

1. Site acquisition is excluded from capital costs, but this is not a significant cost.
2. The current UK electricity pool price is 2.5p/kWh, whereas the UK Non-Fossil Fuel Option (NFFO) price is 4p/kWh – the same as the price used for the ESB's first Alternative Energy Requirement competition (AER1).
3. Patel and Higham assume a disposal cost of £10 per tonne for the residue (35 per cent by weight). We have used £25 per tonne as being more realistic.
4. A discount rate of 10 per cent is used by Patel and Higham. The rate generally used to evaluate public projects in Ireland is 5 per cent (Department of Finance, 1984 and 1994)<sup>37</sup>, and we have used the latter.

It can be seen that the economics of incineration is sensitive to the price paid for the electricity generated. Determining the appropriate price to pay is therefore important, and it is interesting to look at practise in Ireland in this area. The ESB has recently completed its first Alternative Energy Requirement (AER1) competition to source 75 MW of capacity from alternative fuel sources (of this total, 15MW were reserved for generation from biomass and waste). For this, the ESB paid an "environmental credit", which increases the price paid to such producers. The rate varied by time of delivery, from 6.2p at peak times to 2.4p at

<sup>36</sup> Patel and Higham's costs are based on compliance with current British environmental standards. It appears that if standards which are applied in other EU countries, e.g., the Netherlands, were used, then costs would be higher (Glas and Grehan, 1995).

<sup>37</sup> If private capital were used to finance the incinerator, a return in excess of 10 per cent would probably be required. However, we assume throughout this study that financing is provided at public sector rates.

off-peak times, with an average of 4p/kWh<sup>38</sup>. A new AER scheme has recently been put in place, aimed specifically at securing a single plant of up to 30MW capacity to produce energy from biomass (which would include MSW incineration). The price payable for the electricity is subject to competition, with a cap of 4p/kWh, but it is expected that the price will not exceed 3.5p/kWh.

What should be paid for the electricity from an incinerator? There are two relevant factors – the value of the electricity, and the environmental benefit from substituting away from burning fossil fuels to generate electricity. The latter is an externality (i.e., it is not usually priced in market transactions), and we will deal with this later. As for the former, one approach is to pay a price equal to the cost to the ESB of generating the same electricity itself, i.e., the long-run marginal cost (LRMC) of electricity generation in the ESB. Exact figures for this are not available, but it is generally considered to lie between 2p and 3p per kWh. Mays (1995) has estimated that the cost of generating electricity from a modern combined cycle gas turbine (CCGT) plant in Europe is 2.5p per kWh. It might not be unreasonable to use this figure, on the basis that the next generating station that the ESB might build would probably be of this type.

The high capital cost associated with incineration also has other implications, in that it makes such projects very sensitive to changes in circumstances, such as changes in regulations, and in waste flows. Both these eventualities can render facilities obsolete or surplus to requirement, while still leaving large capital charges to be paid. These risks can be mitigated by building plants to higher standards than currently apply, and by entering into long-term contracts with local authorities and industry for the supply of waste. But they certainly make

<sup>38</sup> AER1 was dominated by wind farm projects, with the majority of the extra capacity coming from this source and the other energy sources not taking up their allocated capacities. A number of points are worthy of note:

- (i) An advantage of MSW incineration as an alternative source of electricity is that it is more dependable than wind power, where the ESB would have to keep significant reserves in position, in order to deal with any unexpected drop in supply. This might suggest that the price paid for electricity from an incinerator should be higher than that from a source such as wind power.
- (ii) The differential rates structure in AER1 reflect the ESB's peak demand times. However, this may not reflect the supply characteristics of municipal waste incineration, which would be much more constant. This may have consequences for the economic viability of such incinerators, in that higher incineration and/or storage capacity might be required in order to avail of the peak tariffs. One possible way around this might be for the local authority to use the electricity itself in some of its other facilities, thus making it less dependent on the ESB's pricing arrangement. The practicality of this would depend on the electricity demand pattern in the facility in question, however.



incineration less attractive than less capital-intensive facilities, particularly landfills. These risks may make it appropriate to use a higher rate of discount for an incinerator than for a project subject to fewer uncertainties such as a landfill.

As well as these costs, there are also external costs that are not captured in the market prices of incineration. As described above in the discussion of landfill, they can be divided into two categories – the fixed externality, reflecting the disamenity of the site, and variable externalities, directly related to the amount of waste processed.

Concerning the disamenity cost, as with landfill, no studies have been carried out to quantify it, either in Ireland or the UK. Kiel and McClain (1995) estimate for a town of 20,000 inhabitants in the US that house prices rose by £4,033 per mile<sup>39</sup> distance from the incinerator site, with the effect decaying at roughly 3.5 miles. Using such figures in an Irish context is less than satisfactory. The ideal approach would be to carry out a detailed survey of the situation in Ireland; however, as with landfill, this is beyond the scope of this study<sup>40</sup>. We are forced to use the US estimations, which at least give us some indication of what the costs might be.

Using the same approach as with landfill, the US estimation translates to a cost of £61 per tonne in a densely populated area and £9 per tonne in a sparsely populated area, for an incinerator burning 200,000 tonnes per annum. If we consider a plant with an annual throughput of 400,000 tonnes, these values are halved, to £30 per tonne and £4 per tonne respectively.

As for the variable externality, CSERGE *et al.* (1993) calculate that there is a net external *benefit* of £4 per tonne of waste incinerated, with energy recovery, due to the benefit of reduced pollution from other energy sources (see Appendix I). Again, applying these figures to an Irish context is less than ideal, but they give a benchmark to work with.

Combining the fixed and variable external costs gives us the total externality. Whether this is a net benefit or cost depends on the throughput of waste and density of population in the vicinity. Using the example of a 200,000 tonne per annum plant it will range from £5 (9 - 4) to £57 (61 - 4) per tonne external cost. For the larger incinerator size, the costs range from £0 (4 - 4) to £26 (30 - 4). To maintain consistency of comparison with landfill, if we assume that the incinerator is built at a sparsely-populated site, then the appropriate externality is £5 per tonne for the smaller facility and £0 for the larger.

<sup>39</sup> US\$6,607 @ US\$1.6381 = £1.

<sup>40</sup> Such a survey would be even more problematic than for landfill, as no MSW incinerators currently exist in Ireland. A Contingent Valuation Method (CVM) survey could be used, which would ask people how much they would be willing to pay to avoid having a hypothetical MSW incinerator built near them (or how much they would accept to have an incinerator built near them).

We can now estimate the average costs for a hypothetical incinerator, burning 200,000 or 400,000 tonnes per annum. Collection costs are assumed to be the same as for landfilling; there is no reason to believe that they would be any different<sup>41</sup>.

Table 3.5: *Average Costs per Tonne of Incineration with Electricity Generation*

	<i>Plant Size (tonnes per annum)</i>	
	<i>200,000</i>	<i>400,000</i>
	<i>£</i>	<i>£</i>
Waste collection	42	42
Incineration and disposal of residue	32	25
External costs	5	0
<b>Total cost per tonne</b>	<b>79</b>	<b>67</b>

It is interesting that the overall cost per tonne does not vary a lot by size of incinerator plant. This is because collection costs still dominate the costs of processing and disposal. Attention is drawn once again to the assumptions underlying these calculations, and the sensitivity of costs to changes in these assumptions.

Looking briefly at the costs of RDF, Moore (1994) estimates a cost of £12 per tonne of RDF produced, at a facility processing 50,000 tonnes of MSW per annum, assuming the RDF is used to generate electricity, and that electricity is sold at a price of 6.5p/kWh. Viability is highly dependant on the electricity price achieved. If we use 2.5p/kWh (the price we used for electricity from MSW incineration), the cost per tonne goes up to £47 per tonne. Adding £5 per tonne external costs, as with incineration<sup>42</sup>, and £42 per tonne for collection, gives an overall average cost of £94 per tonne. On this basis the costs of disposal of solid waste by converting it to RDF would not appear to be competitive; however a larger facility than the one costed by Moore (1994) might be more viable.

Returning to incineration, as with landfill, we need to consider not only the average costs but also the LRMC of incineration. This is a slightly unusual case, because there is no incinerator in place at present in Ireland. In such a circumstance the appropriate cost to take for the facility itself is the average cost

<sup>41</sup> As with landfill, we have used the same collection cost, regardless of size of incinerator. This is correct if we are considering incinerators to be built in Dublin; however, it might be inappropriate for a regional incinerator, where collection costs would be higher.

<sup>42</sup> Strictly speaking we should recalculate the external cost for RDF, but we use the same figure as incineration for illustration purposes.

as calculated above. For collection, it is appropriate to take the LRMC as calculated for landfill, i.e., £11 per tonne. As for the external costs, the variable costs should definitely be included, but the disamenity costs are less clear. The latter do not change with the size of the facility, but they depend on whether the incinerator is built or not. Since this is the stage of decision-making at the moment, these costs are avoidable, and therefore should be included in LRMC. The LRMC of incineration, therefore, can be stated as follows:

Table 3.6: *Long-Run Marginal Costs per Tonne of Incineration with Electricity Generation*

	<i>Plant Size (tonnes per annum)</i>	
	<i>200,000</i>	<i>400,000</i>
	<i>£</i>	<i>£</i>
Waste collection	11	11
Incineration and disposal	32	25
External costs	5	0
<b>Total cost per tonne</b>	<b>48</b>	<b>36</b>

Comparing these costs briefly with the LRMC of landfill (Table 3.3), it can be seen that incineration is considerably more expensive. The case of Dublin is different from the rest of the country, not only because it appears that landfill may be very costly, but also because incineration might replace one out of two or three landfills that would be built in Dublin<sup>43</sup>. In other words, for that one "marginal" landfill in Dublin, we are dropping the assumption that it will be built regardless of what other waste disposal alternatives are used. We assume that the landfill in question would be taking 300,000 to 400,000 tonnes per annum, about the same scale as the incinerators considered here. In this circumstance we should compare the LRMC of incineration as in Table 3.6 with the LRMC of landfill calculated in the same way, i.e., LRMC of collection (£11) + total external costs (£7) + average disposal costs (£25) = £43 per tonne. This appears to show, subject to all the assumptions and uncertainties, that incineration might be viable in the Dublin area, if it could replace a landfill.

### 3.5 Recycling

Recycling of MSW is made up of three stages of activity – collection, segregation and reprocessing. In addition, there are two main approaches to recycling – "bring" and "collect". With the "bring" system, waste generators bring the recyclables to a central point such as a bottlebank – they effectively carry out part of the collection and segregation stages. With the "collect" system, the recycler collects from the waste generators' premises, and the collected items are

<sup>43</sup> We assume that more than one landfill would be built to dispose of Dublin's waste.

brought to a "Materials Recovery Facility" (MRF) for sorting and possibly reprocessing<sup>44</sup>.

The collect system is further divided into two sub-categories:

- (i) unsegregated – where all waste, both recyclable and non-recyclable, is collected from the waste generator, and is brought to a "dirty" MRF for separation. Much of the waste collected will not be recyclable, and a proportion of the recyclables collected will be unusable due to contamination. The advantage of this system is that because it avoids the need for separate collection, it does not depend on household participation for its success<sup>45</sup>.
- (ii) pre-segregated – where the waste generator segregates the recyclables from the non-recyclables, and may even separate out the different types of recyclables. Here the waste is brought to a "clean" MRF for further segregation.

The two approaches to recycling – bring and collect – will be looked at in the rest of this section, and the costs of these operations in Ireland and overseas will be examined.

#### 3.5.1 "Collect" Systems and Their Costs

The only example of a collect system at the domestic level in Ireland is the Kerbside Dublin pilot project<sup>46</sup>. Set up in 1991, this is an (initially) three year pilot project which provides 25,000 out of a total of 290,000 households in Dublin with a weekly kerbside collection system (a small amount of waste is obtained from mobile bring and drop-off facilities). Householders put out a green box with their recyclables in it, the same day as the normal local authority collection. Table 3.7 shows the various materials collected and their relative importance in terms of weight. The participation rate among households in the catchment area is 76 per cent, and 23 per cent by weight of their domestic refuse is collected for recycling (Madden, 1995). The remaining refuse continues to be collected by the local authority.

Collect systems are invariably expensive to operate. Kerbside Dublin requires an operating subsidy of over £500,000 per annum<sup>47</sup>, with funding for this coming

<sup>44</sup> In the case of much industrial recycling the entire process may be carried out by the waste producers themselves.

<sup>45</sup> The system is mainly used in the USA, where MSW includes less kitchen waste than in Europe, due to the use of domestic waste disposal units (Moore, 1994). We are not aware of any example of this approach outside North America.

<sup>46</sup> There are a number of collect systems in the commercial sector, notably for glass recycling.

<sup>47</sup> FÁS labour is used for sorting, so in effect the subsidy is greater. Employment was 23 full-time and 30-40 part-time FÁS workers in 1993 (Fitzpatrick Associates, 1993). Discussions with Kerbside Dublin indicate that extra labour costs might amount to

from the Dublin local authorities, the Department of the Environment, and industry<sup>48</sup>. The project is also connected with the European Recovery and Recycling Association (ERRA), an industry-sponsored body, which is involved in a number of similar schemes in other countries.

Recyclables prices and the mix of materials collected are major variables in the cost of "collect" recycling. The main materials are paper and board, which account for 54 per cent by weight; according to Madden (1995) the price received for this material is critical to the costs of the Kerbside Dublin operation. The percentages and prices of collected materials are given in Table 3.7. Price volatility is a major problem for all recycling activities – prices have fallen significantly during the early 1990s, and have recovered dramatically in recent months. The average price of recyclables in September 1993 was £17 per tonne; by January 1995 it had recovered to £35 per tonne and by June 1995 it had increased to £60 per tonne. The German DSD system, which increased the supply of recyclables in the European market during the early 1990s, is often blamed for the fall in prices. The subsequent installation in Germany of greater recycling capacity, the general economic recovery and high demand in the paper industry are credited with causing the recent turn-around.

Using data from Madden (1994 and 1995) and IBEC (1995) we can estimate costs for the current Kerbside Dublin operation size, and for its increase to a catchment area of 50,000 urban households<sup>49</sup> (see Table 3.8). January 1995 prices are used, and tonnage is taken to be 4,500 tonnes per annum for the existing size of operation, increasing to 9,000 tonnes per annum for the larger size. The net cost is £124 per tonne for the current operations size, and £78 per tonne for the larger size<sup>50</sup>.

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£100,000 per annum if FÁS schemes were not used.

<sup>48</sup> Notably the newspaper industry is not contributing, although it generates a large amount of the waste collected by Kerbside Dublin.

<sup>49</sup> Madden (1995) estimates that this is the optimum scale of operation, i.e., there are considerable economies of scale to be earned by expanding from the current size. He indicates that Dublin is the only city in Ireland where this catchment area of 50,000 households could be achieved within a reasonable geographical area. In Cork, the maximum would be approximately 40,000 households.

<sup>50</sup> Costs for maintaining the existing Kerbside Dublin operation would be somewhat lower, because the existing operation has had considerable investment already in plant and equipment, which is to a degree at least a sunk cost. If we take that the annual depreciation charge of £75,000 represents this investment, its exclusion from the costs calculations would reduce costs at the 25,000 household level of operations by £17 per tonne, and would reduce costs at the larger operation level by £8 per tonne.

Table 3.7: *Proportions of Materials Collected by Kerbside Dublin, and Prices Paid for Recyclables, in 1993 and 1995.*

Material	Percentage of	Price per Tonne	Price per Tonne	Price per Tonne
	Total by Weight	Sept. 1993	Jan. 1995	June 1995
	%	£	£	£
Paper/beverage cartons	60.9	0	25	60
Glass	16.1	20	20	20
Steel cans	4.4	20	12	40
Aluminium cans	1.2	400	660	650
Clear PET	3.1	120	140	200
Green PET	1.2	40	50	100
Clear PE	1.5			130
Coloured PE	0.8			75
PVC	0.3			90
Miscellaneous	10.5			
Average		17	35	60

Source: Fitzpatrick Associates, 1993; Madden, 1995.

Note: Total quantity collected in 1993 (estimate) was 4,470 tonnes. Miscellaneous includes contaminated, unsolicited or otherwise unusable materials which are disposed to landfill.

These rates are dependant on sales revenues; if we were to use the very high prices being achieved in June 1995<sup>31</sup>, the net costs would fall to £99 and £53 per tonne; if prices returned to their September 1993 levels, net costs would increase to £142 and £96 per tonne.

The analysis so far ignores externalities, which might be significant. In order to estimate the true cost of recycling we should adjust the cost for the external benefits and costs associated therewith. Strictly speaking, most of the externalities associated with recycling refer to avoided external costs of the alternatives. For example, the external costs avoided by diverting waste from landfill, or by producing aluminium from recycled rather than virgin material.

<sup>31</sup> Madden (1995) and industry sources indicate that the price of paper, which is the most important material collected, is expected to remain at its June 1995 level for the next 1 to 2 years. Whether it would continue at such a high level in the longer run must be open to question, however.

Table 3.8: *Estimated Average Cost of Kerbside Dublin's Operation at Present Level (25,000 Households), and with Coverage Doubled to 50,000 Households, Using January 1995 Prices*

	<i>Annual Costs for Current Operations</i> £	<i>Cost per Annum for Doubled Coverage</i> £
Existing running costs	540,000	540,000
Add adjustment for true cost of labour	100,000	100,000
Extra costs to increase operation size		300,000
Fixed costs – depreciation	75,000	75,000
Total costs	715,000	1,015,000
<i>Gross cost per tonne</i>	<i>159</i>	<i>113</i>
Less: total sales income (@ £35 per tonne)	157,500	315,000
Net funding requirement	557,500	700,000
<i>Net cost per tonne</i>	<i>124</i>	<i>78</i>
<i>Cost per household</i>	<i>12.7</i>	<i>19.7</i>
<i>Cost per participating household (76 per cent)</i>	<i>16.7</i>	<i>25.9</i>

Source: Kerbside Dublin, 1994; Madden, 1995.

*Notes:*

1. Tonnage collected currently includes approximately 5 per cent dropped off at the MRF. Strictly speaking, therefore, the net cost per tonne for recyclables collected from the kerbside is slightly understated. However, one could argue that the publicity generated by the kerbside collection encourages the drop-off quantities.
2. The participation rate is defined as the proportion of households that put out recyclables for collection at least once per month. The set-out rate (the average proportion of households who put out waste) is somewhat lower, at 60 per cent.

In a first best or ideal world, these other activities should carry these extra costs, and there would then be no need to adjust the cost of recycling for them. This should certainly be the case with landfill – it should carry a levy equal to the external costs of landfill, which we have already estimated. With other costs, it may not be technically or politically possible to charge the activity in question with its external costs, and in these circumstances one could justify a subsidy for recycling as a second best solution. The items involved would include:

- (i) External costs arising from the initial exploitation of raw materials, e.g., air and water pollution from mining activities (in theory these should be charged as a tax on the activity in question).
- (ii) External costs arising from transport of the raw materials to the processing location (in theory these should be levied by a fuel tax).

- (iii) External costs relating to the production process (again, in theory pollution taxes should internalise these costs).

Recycling itself also generates external costs, which should be levied on the activity in question. These include external costs related to collection, sorting, transport and reprocessing of the material (a fuel tax would internalise much of the collection and transport externalities).

The estimation of these values is a difficult task, the results of which would be controversial. One thing that is clear is that considerable amounts of energy are saved in the production process from recycling (Stanners and Bourdeau, 1995; Ogilvie, 1992). One approach would be to take the energy saved in these processes and try to estimate external costs relating to this energy. We could then treat this as an external benefit of recycling. However, estimating the external costs from energy usage is a very complex and inexact process. Variables include the fuel type, location of energy usage and generation (in the case of electricity), and the technology used. Large uncertainties lie around the costs of global warming, among other things. As a result, any estimation needs to be used with a large degree of caution. We have made an estimate of the costs involved (see Table 3.9), but because of the uncertainties and variables it should be treated as largely illustrative.

Table 3.9 takes data from Ogilvie (1992) for the energy used in production from virgin and secondary (recycled) materials. These are Swiss data, and therefore, once again, are not ideal for use in an Irish context; however, they give us a benchmark to work with. As for putting an external cost on this energy, a rate of 0.43p per kWh or £49 per tonne of oil equivalent (TOE) is estimated (see Appendix III). The data are used to estimate the savings from using secondary materials, and to weight the saving for the materials collected by Kerbside Dublin. The weighted external benefit of recycling would be £31 per tonne. If Kerbside Dublin were to receive a subsidy to account for the external benefits of recycling, this amount of £31 (say £30 for convenience) per tonne might be appropriate. Alternatively they might receive a subsidy for each type of material collected, based on the individual external cost difference calculated for each material above.

We are now in a position to estimate the average cost of the Kerbside Dublin operation, including some amount for the external benefits of recycling (see Table 3.10)<sup>52</sup>. For the present level of operations and the January 1995 prices, the net cost is £94 per tonne (124 - 30); for an increased size of operation of 50,000 households, the net cost is £48 per tonne (78 - 30)<sup>53</sup>. If we used the exceptionally

<sup>52</sup> Keeping in mind that if the external costs of energy consumption were properly reflected in an energy tax, there would be no need to subsidise recycling as described above, as the market value of any energy-saving activity should increase automatically.

<sup>53</sup> For Kerbside Dublin's existing operations, deduct £17 per tonne for the small operations level and £8 per tonne for the larger operations level, as before.



high prices being achieved in June 1995, costs would be roughly £25 per tonne lower. However it is questionable whether such prices would be sustainable in the longer run. A fall back to pre-1995 prices would add approximately £18 per tonne to the net costs. January 1995 prices are perhaps a reasonable average price that might be expected in the longer run, although it is very difficult to predict this with any confidence.

Table 3.9: *Estimation of External Benefits of Recycling, in Terms of Energy Saved*

<i>Material</i>	<i>Aluminium</i>	<i>Glass</i>	<i>Paper</i>	<i>Tinplate</i>	<i>Plastics</i>
Energy used to produce from virgin material (kWh per tonne)	47,594	2,187	10,786	9,257	
Energy used to produce from recycled material (kWh per tonne)	4,337	1,640	5,226	5,560	
Difference	43,257	547	5,560	3,697	34,420
External cost of difference per tonne (@ 0.43p/kWh)	£186	£2	£24	£16	£148
Weighting by reference to quantities collected by Kerbside Dublin	0.02	0.18	0.68	0.05	0.08
<b>Weighted average external benefit per tonne of recycling by Kerbside Dublin</b>					<b>£31</b>

*Source:* Ogilvie, 1992.

*Note:* We found one study (British Glass, 1989) of glass recycling, which examines the entire recycling process from the point of view of energy saving only. It concluded that the overall net energy saving was 5,550 MJ per tonne of cullet used. This would give an external cost difference of £7 per tonne. If that were incorporated into the above table it would increase the overall external costs difference to £32 per tonne.

Table 3.10: *Costs of Kerbside Recycling, Net of Sales Revenue and External Benefits*

<i>Pricing Scenarios</i>	<i>Net Cost per Tonne per Number of Households Served</i>	
	25,000	50,000
£17 per tonne (pre-1994)	112	66
£35 per tonne (January 1995)	94	48
£60 per tonne (June 1995)	69	23

We can compare these costs with the LRMC of landfill, on the basis that kerbside recycling would only be viable in Dublin (we would emphasise again that there are very large uncertainties in the estimations carried out in this section, so the conclusions should be viewed with caution). Reference to Table 3.3 shows that kerbside recycling is considerably more expensive under most circumstances. Kerbside can be competitive where recyclables prices are exceptionally high (as

they were in June 1995), and the larger kerbside operation size is used. However, it is difficult to see how prices would remain at such high levels in the longer run – high prices should stimulate increased supply, causing the price to fall over time.

### 3.5.2 The Cost of "Collect" Systems Overseas

As stated, Kerbside Dublin is the only collect system operating in Ireland at the domestic level; it is useful to consider costs in other countries, especially where different systems are used. Kerbside Dublin is an example of what is usually described as the "blue box" approach – where householders put their waste into an open box and the collectors do some separation at the kerbside, using compartmentalised trucks. There are a number of alternatives, such as the "green bin" (a separate wheelie bin for recyclables, with no sorting at the kerbside) and the "green bag" (no separation, but using a semi-transparent plastic sack so that the collectors can see what is being collected). Collection can also be combined with local authority collection, in a compartmentalised truck. Experience overseas indicates that costs vary considerably between the different methods. There are indications that the Kerbside Dublin system – "blue box" with some separation at the kerbside and a separate collection from the local authority – is one of the more expensive options. Some studies have been made of costs in the USA and the UK. US data indicate average costs for "collect" recycling at £89 to £119 per tonne recycled (National Solid Waste Management Association, 1992, 1993); UK data are summarised in Table 3.11.

Table 3.11: *Costs of "Collect" Recycling in the UK, per Tonne Recycled*

<i>Source</i>	<i>Blue Box</i> £	<i>Green Bin</i> £	<i>Green Bag</i> £
Coopers & Lybrand (1993)	83 - 172	54 - 69	54 - 74
Warren Springs Laboratory (1993a, b)	64 - 128	-	73

As can be seen, there is a wide range of costs, but it does seem that the "blue box" system is more expensive than the alternatives. However, we do not have sufficiently detailed information to come to a conclusion that the alternatives would be cheaper in the Irish context. In particular the costing systems and the recyclables prices used are not known (although it is likely that the latter would be lower than current levels). If future kerbside pilot schemes are put in place in Ireland, it would be valuable to use the other receptacle systems, in order to see if they would be more cost effective.

One point that is worthy of consideration is whether there should be combined collection of recyclables and non-recyclables in a compartmentalised truck, or separate collection for each stream. Intuitively one would expect the former to be more cost effective. However, indications from the UK are that combined

collection adds significantly to the cost of the normal waste collection system, and that it may not always be the most efficient option (De Búrca, 1995). The main reasons for this are:

- (a) collecting two streams instead of one slows down both;
- (b) recyclables will represent a significantly smaller quantity, so a separate collection of these could cover a larger area in one run;
- (c) the combined collection would have to use compartmentalised trucks; if one compartment filled before the other the truck would have to return to the depot to unload, even though it was only partly full; seasonal variability in waste quantities makes it difficult to fix compartment sizes;
- (d) if the recycling and MSW facilities are in different locations, costs will be greater.

It is not possible to state that Kerbside Dublin could reduce its costs by adopting combined collection with the local authority. Indeed, Fitzpatrick Associates (1993) in its economic evaluation of Kerbside Dublin indicated that combined collection would not be cost effective. As with the receptacle systems, if there was to be another scheme put in place in the future it could be designed with combined collection, to examine the relative cost-effectiveness of this approach.

### 3.5.3 "Bring" Systems and Their Costs

Bring systems, whereby waste producers bring their recyclables to a central point for subsequent collection by the recycler, are quite well established in Ireland. The main system is the glass bottlebank system organised by Rehab, but there are also systems in place for recycling aluminium cans, and to a lesser extent textiles and more recently paper. Given the high price of aluminium (see Table 3.7) the economics of aluminium recycling is less in doubt<sup>54</sup>, and so most of the remaining discussion will concentrate on glass.

The major cost differential with bring as opposed to collect systems is that the very considerable cost of door-to-door collection and segregation is avoided. As a result, most of these systems operate with less explicit subsidisation than is required for a kerbside system, though they do enjoy some subsidisation. For instance, in the case of glass, both the Department of the Environment and Irish Glass Bottle have subsidised the capital cost of constructing cullet<sup>55</sup> plants in Dublin and Cork, and in many cases local authorities assist in the provision and preparation of bottlebank sites, and also may pay a lump sum subsidy to Rehab. In addition Rehab gets grants from a number of industry associations.

Although door-to-door collection and segregation by a central agency is avoided with the bring system, these costs still arise, though in this case they are

<sup>54</sup> Coopers & Lybrand (1993) indicates that aluminium recycling in the UK usually generates a financial surplus.

<sup>55</sup> Broken glass of a uniform colour in a form suitable for recycling.

borne by the householder. Measuring these costs is a difficult exercise, since householders may or may not be making specific trips to the bottlebank. There is also the "feel good" factor, whereby householders get utility from participating in recycling. Since recycling is purely voluntary in Ireland, one might assume that this "feel good" factor out-weighs the time costs associated with bringing bottles to the bottlebanks, for those who do participate. In addition, there are costs involved in central collection, segregation, quality control, contaminant removal, and most significantly, transport to processing facilities.

In terms of the market for recycled glass, the price paid for cullet delivered to Irish Glass Bottle is £40.50 per tonne at the moment, and the price for unprocessed glass delivered to a cullet plant is £20 per tonne. Unlike other materials these prices have remained quite constant over time<sup>56</sup>.

As for estimating the cost of bring recycling in Ireland, we do not have exact data on this question. One point is that there would be a difference between costs in the large urban areas, where a high turnover enables the recovery of the capital costs of bottlebanks and transport costs are not excessive, and costs in other areas. In smaller population centres the bottlebanks generate less material and transport costs become very considerable. One rural authority we talked to estimated that they subsidised Rehab to the tune of £30 per tonne of glass recovered in their county; this excludes any subsidies Rehab receive on a national level<sup>57</sup>. In addition there are a small number of private firms that operate bring recycling systems around the country as a profit-making business, though they receive some capital grant aid, and they also benefit from the subsidisation of the cullet plants. Coopers & Lybrand (1993) indicates that recycling using the bring system (excluding aluminium) in the UK costs £16 to £37 per tonne. In the absence of better data we have taken a hypothetical cost of £30 per tonne for bring recycling of glass in Ireland, though this is subject to correction.

There are also externalities related to recycling using the bring system. These are more or less the same as for recycling using the collect system, already discussed, except that the externalities relating to initial collection will be different. The problems with estimating these values are the same, but again as a minimum we can use the estimates of energy savings as an external benefit. We saw from Table 3.9 that the external benefits per tonne were £186 for aluminium, £148 per tonne for plastics, £7 for glass (taking the entire recycling process into account), £24 for paper and £16 for tinplate. If a subsidy were to be paid for recycling under

<sup>56</sup> Although cullet prices in the UK have been falling considerably in recent times (De Búrca, 1995).

<sup>57</sup> However, Rehab has a specific mandate with respect to social employment, and it is difficult to separate the costs relating to this function from costs related to recycling *per se*.

the bring system, these values might be appropriate<sup>58</sup>; however we again emphasise that these numbers are mainly illustrative, due to the large uncertainties involved in their estimation.

Comparing bring system costs with those of landfill is difficult, since the above discussion is incomplete. However, the quite high external benefits calculated indicate that recycling by the bring system might be justifiable even where costs of operation are high. Taking glass, if say the costs of operating bottlebanks amount to £30 per tonne, then deducting external benefits of £7 would give a net cost of £23 per tonne. This is comparable to the LRMC of landfills. In the large urban areas, where the costs of operating bottlebanks should be lower, bring recycling of glass should be more competitive. Considering aluminium, this activity is already profitable, given the high prices for the material. The external benefits from the high energy savings would justify a considerable subvention of aluminium recycling, which should stimulate higher recycling levels. Paper recycling has been given a major stimulus by very high prices since mid-1995 (£60 per tonne) and this has led private enterprise to establish a network of paperbanks in recent months, mainly around Dublin. Presumably this activity is profitable in its own right; a subvention reflecting the external benefits from energy savings would give it a further boost. Finally, we repeat that if the external costs of energy consumption were properly reflected in an energy tax, there would be no need to subsidise recycling as described above, because the market value of any energy-saving activity should increase automatically.

<sup>58</sup> There is an interesting issue with respect to these "recycling subsidies". In the case of aluminium and tinplate, the collected materials are not recycled in Ireland, but are exported to the UK. The energy saving from reprocessing, therefore, accrues to the UK economy (this would be reflected in the recyclables price), as does much – if not most – of the relevant external savings. One might argue that Ireland should not pay for external benefits which will accrue to another country. However, the materials in question have originally been consumed in Ireland, so other countries have suffered external costs from the original production process, the fruits of which have been enjoyed in Ireland. On that basis it is appropriate for Irish society to pay a recycling subsidy.

Of course, should the countries that manufacture the aluminium and tinplate introduce appropriate energy taxes, they will have internalised the external costs in question. It is probable that this tax would be passed onto consumers in Ireland, and, therefore, we will have paid the full cost of what we consume. In addition, the price of recyclables should increase to reflect the energy tax avoidable by its use. In that circumstance there would be no need for Ireland to subsidise the recycling of these materials, at least with respect to the external costs of energy use.

### 3.6 Composting<sup>59</sup>

A further possibility for waste diversion is the composting of organic waste. This process reduces the volume and weight of the waste by up to 50 per cent, and in theory produces a usable product, i.e., soil conditioner. Over 570,000 tonnes of organic waste are landfilled per annum in Ireland, and this element of waste causes most of the environmental problems with landfill – i.e., odours, gas emissions, rodent infestation, settling, etc. Its production is also highly seasonal, and its removal from the waste flow would help to even out peaks in MSW quantities. In theory approximately 50 per cent of municipal organic waste could be available for composting (ERL, 1993). There are two possible approaches – home composting and centralised composting. Economic pricing of household waste collection would encourage the former, and a number of local authorities around the country are involved in home composting pilot projects, to encourage its more widespread use. The option of centralised composting is problematic in that the technology, economics and markets for it are unproved (ERL, 1993); specifically:

- (i) There needs to be careful removal of contaminants prior to processing, as the presence of such contaminants, notably metals, could severely affect the usability of the finished product, especially for crop production (Department of the Environment, 1992). These contaminants must then be disposed of separately.
- (ii) Immature compost can exhibit toxic characteristics, which can inhibit plant growth (Moore, 1994).
- (iii) The abundant availability of topsoil and the existence of an established high quality soil conditioner in moss peat may be the biggest problem for the viability of composting in Ireland.
- (iv) The composting process can take 4 to 6 months, requiring extensive storage facilities (Moore, 1994).
- (v) Maturing compost will produce leachate if exposed to rain, and odour problems are also significant (Veiga-Pestana, 1995).
- (vi) A further point is that centralised composting might displace home composting, to no net benefit. However, there are no baseline data on the amount of home composting that already occurs, and no indications of how much might occur if higher refuse disposal charges were in place. It is, therefore, difficult to estimate how significant this potential problem might be.

Lack of markets appears to be a major problem internationally, where centralised composting is undertaken (Veiga-Pestana, 1995). ERL (1993) points

<sup>59</sup> For the purposes of this study the term composting will be used to describe both aerobic and anaerobic biodegradation of solid waste.

out that every centralised compost plant built in the UK over the last twenty years has failed. In Denmark compost from centralised plants is generally landfilled due to a lack of markets; however in the Netherlands compost does appear to be saleable (Glas and Grehan, 1995). Moore (1994) points out that currently in Ireland 50,000 tonnes of used mushroom compost is landfilled annually, although it could be used as a soil conditioner. As against this, mature compost is largely inert, so its landfilling is less problematic than other materials (indeed, it could serve a useful purpose as covering material). Also, the proper pricing of landfill would encourage producers of compost to find alternative means of disposal, although it might make composting less economically viable if some of it still had to be landfilled.

Department of the Environment (1994a) states "it would be reasonable to aim at diverting some 100,000 tonnes of organic waste from landfill within the period of this strategy", but does not address further the feasibility of this. This quantity represents 17 per cent of the total organic waste arising. If we could divert this amount, the effect would be similar to meeting the packaging waste recycling target, and given the biodegradable nature of organic waste, this might be more beneficial.

One possibility is to concentrate on central composting of green garden waste, which is less problematic than other organic fractions of MSW, as there is a lower level of contamination - Dublin Corporation already has a composting facility for its own park (and householders') green waste. Another possibility is the co-composting of MSW with sewage sludge, increasing quantities of which will be produced in the coming years. However, this technology is at the developmental stage, and the economics of it are largely unknown. There are two centralised composting schemes at the planning/proposal stages in Ireland at the moment, one in Limerick and the other in Dundalk (the latter to be operated by the local authority, Rehab and a private company). In addition, there are a number of home composting pilot schemes in place around the country. It will be of interest to see how these schemes operate, how much waste they will divert and how much they will cost.

### 3.6.1 Costs of Composting

There is no experience of the cost of centralised composting in Ireland, as this has not yet been utilised here. Estimates of costs for one of the proposed pilot projects in Ireland, processing 15,000 tonnes per annum of organic waste into 7,500 tonnes of compost, indicate capital and running costs of approximately of £25-30 per tonne (net of projected sales at £10 per tonne, and including only the LRMC of collection). In this scheme collection of organic waste and other waste will occur on alternate weeks; a second bin for each household to hold the organic waste is included in the costs.

Overseas experience indicates that the most basic centralised composting system will cost a minimum of £30 per tonne. However, this is highly dependant on the feedstock, the technology, the availability of markets and the quality specifications for the final product (Veiga-Pestana, 1995). Coopers & Lybrand (1993) indicates that centralised composting with kerbside collection in the UK costs £88 to £98 per tonne (UK schemes using a bring system have costs as little as £5 to £15 per tonne – presumably this excludes householders' costs in bringing the organic waste to the composting site). If the costs estimated for the Irish pilot project are achieved, they will represent a significant reduction on the cost levels overseas. They may also be competitive with landfill in Dublin, if costs there prove to be very high. However, the actual experience with the two Irish pilot projects would have to be evaluated before strong conclusions could be drawn.

### 3.7 Re-use

In general, re-use is considered superior to recycling of waste, as there is no reprocessing involved. However, there are extra costs with re-use, which make the comparison less clear-cut than might first appear. These can be listed as follows:

- (i) Reusable packaging needs to be stronger, requiring more materials and weighing more than one-trip packaging. This means that more resources are used to produce and transport the product.
- (ii) Reusable packaging must be stored, washed and transported back to the filler for re-use, all of which requires extra resources.
- (iii) To ensure return, a deposit system must be in place, entailing administration, storage, and security costs.

These costs are not trivial, and may in certain cases make re-use less environmentally sound than other options.

Re-use is well established for industrial and commercial packaging, and one can assume that in these cases it must be financially superior to alternative one-trip packaging. With respect to beverage packaging, over 80 per cent of the beer sold on-licence in Ireland is conveyed in reusable aluminium kegs, and the majority of bottled drinks sold in the on-licence trade is also conveyed in reusable bottles. Industry organisations are in favour of extending the return system in the on-licence trade (i.e., pubs, hotels and restaurants) to those bottled sales which are in one-trip bottles. This might divert perhaps 17,000 tonnes from landfill and recycling per annum (Soft Drinks and Beer Bottlers Association, 1993)<sup>60</sup>.

<sup>60</sup> These organisations would like to see the refill system in the on-licence trade made mandatory, in order to ensure that all sectors of the trade participate, and the cost is shared. There are dangers in this, however, in that re-use may be enforced where it is not economically the most efficient option, and mandatory participation in the re-use system may act as a barrier to entry to the trade.



Beverage packaging in the supermarket/off-licence sector is almost exclusively one-trip, however, and the main reason appears to be the extra convenience for the retailer and the consumer. Industry organisations are against returning to a deposit-return/refill system for this sector, citing the extra cost, inconvenience, low trippage rates<sup>61</sup> and studies which report that refill systems at this level are environmentally inferior to recycling<sup>62</sup>.

There has been much study comparing the environmental and economic impacts of reusable and one-trip packaging. These studies use methodologies which are variously described as "life-cycle assessments (LCA)", "cradle-to-grave analyses", "eco-balancing", etc. As the names suggest, they attempt to evaluate all environmental impacts of a product life-cycle, including the internal and external costs and benefits of all the inputs into the provision of the good in question. However, the methodologies are far from standardised<sup>63</sup>, and the results are ambiguous as to which method is more environmentally benign. Perchard (1995) reviews some European literature in this area, and concludes that where distribution distances are low (in continental terms) and trippage rates are high, refillable containers are environmentally superior. However, as distances increase and trippage rates fall, the more lightweight non-refillable containers are better. In studies in areas of the UK where reusable packaging is still used, trippage rates were found to be between 17 and 23 for on-premises sale, and at 3 for off-licence sales. If these trippage rates reflect the situation that might apply in Ireland, then one would need to be careful about re-introducing deposit/refill systems here. One would need to be confident that consumers could be persuaded to increase their return rates. Publicity and education might achieve this; the wider use of volume-related refuse charges would give an economic incentive to return containers.

In a similar vein, Pöll and Schneider (1993) carry out an *ex ante* evaluation of the Austrian refillable packaging ordinance which requires between 60 and 95 per

<sup>61</sup> The average number of times a reusable container is actually used.

<sup>62</sup> Although the Restricted Practices Commission (1987) in comparing soft drinks prices in the Republic of Ireland (where PET bottles were used) with those in Northern Ireland (where returnable glass bottles were used), in 1987, comments "the (Northern Ireland) market continues to take the returnable bottle in retail outlets and where this container is used, there is a significant difference in the net selling prices to the consumer ...the cost of the PET containers is 17p to 18p compared with 2p for glass returnable bottle based on 10 trips". Since 1987, PET prices have fallen to approximately 10p per bottle, and the use of returnable glass bottles in Northern Ireland has become much less prevalent. Research in other parts of the UK indicate that the trippage rate of 10 may not be realistic.

<sup>63</sup> Although a number of organisations have developed methodologies (Anonymous, 1995).

cent of drinks and beverages to be supplied in returnable containers by the end of 1996. Looking at the case of fruit juice, they conclude that the ordinance –

- (i) will have a detrimental effect on the environment, as meeting the terms of the ordinance will require using more resources than were used in the previous situation<sup>64</sup>;
- (ii) will have a detrimental effect on the economy, by increasing costs in the industry, and by possibly encouraging a cartel to enforce compliance with the ordinance;
- (iii) will reduce consumer welfare by causing an increase in prices and by forcing a move away from a preferred packaging type<sup>65</sup>; aggregate waste quantities may be reduced, but only because of a reduction in consumption of the product.
- (iv) may not be effective as it will encourage consumers to shop in neighbouring countries ("shopping tourism"), thus importing packaging waste.

These arguments counsel against the use of imposed targets for achieving high levels of re-use (or for that matter recycling), especially when those levels are significantly higher than those already being achieved by the market. The "shopping tourism" argument is most relevant in the case of a small country bordered by many other states, as is the case with Austria; however, it might apply to some degree to Ireland. If cost-increasing measures are imposed in the Republic and not in the UK, this might encourage cross-border shopping, which would damage the economy in the Republic and succeed only in importing packaging waste from the UK. Thus, even those measures that are justified in terms of internalising external costs might not be effective. However, in reality the UK has higher recycling/recovery targets to meet under the EU packaging Directive than

<sup>64</sup> Pöll and Schneider calculate what they call an "eco-price" for one litre of fruit juice from a particular company, supplied in returnable and non-returnable containers. This is the actual market price adjusted for taxes and environmental externalities, to generate a price which reflects the actual resources used in providing the product. They find that while the market price of the product in returnable packaging is 51 per cent more expensive than the same product in non-returnable packaging, the eco-price of the former is still 47 per cent more expensive than the eco-price of the latter. This implies that the "returnable" product uses more resources and hence is less environmentally friendly than the non-returnable variety. Since markets for juice in both types of container are competitive, the market price is assumed to be a fair reflection of the costs of resources used. It is interesting that taking into account externalities does not significantly alter the relative prices.

<sup>65</sup> Pöll and Schneider emphasise the importance of consumer preferences when determining the merits of various packaging types.

Ireland. It is, therefore, unlikely that Ireland would put in place significantly more stringent requirements in this regard than the UK.

It is often argued that considerable economies of scale apply to re-use, i.e., that once a system is set up, the marginal cost of increasing the level of re-use would be small. This is undoubtedly true, but there are two reasons why it would not necessarily apply in the Irish situation:

- (a) For most beverages sold in supermarkets, there is no existing re-use system. Supermarkets would have to establish their return infrastructure from scratch. Industry would also have considerable capital expenditure, since the beverages sold in supermarkets and off-licences are in different size containers from those sold in the on-licence trade, and in the case of milk most suppliers do not have an existing returnable milk bottle system.
- (b) The small size and dispersed nature of the Irish market would make economies of scale difficult to achieve outside Dublin.

Apart from this, there are considerations that may favour the use of non-refillable containers. First, because the use of non-refillable containers encourages centralised production, economies of scale can be secured. Second, a manufacturer using non-refillable containers may be better able to change and improve its packaging, whereas one who uses refillable containers will have capital tied up in the system and may be less able to change.

### 3.7.1 Costs of Re-use

We can reasonably assume that where industry is currently voluntarily operating a reusable packaging system, this is financially superior to the alternatives<sup>66</sup>. The above discussion indicates that there is a question mark over whether a reusable packaging system at the domestic level (effectively the only material involved would be glass) would be environmentally or economically cost effective, but what might it cost?

It is difficult to identify costs of a deposit-return system for the domestic sector, since such systems do not widely exist in Ireland<sup>67</sup>. We are therefore again forced to consider overseas data. Studies in the USA would suggest costs of perhaps £350 per tonne<sup>68</sup> processed through such a system. Using this in the Irish

<sup>66</sup> This assumption might not be valid if government regulations required industry to participate in re-use schemes, but there are no such cases that we are aware of at the moment in Ireland.

<sup>67</sup> Except for a small section of the liquid milk market.

<sup>68</sup> Alter (1993) gives an estimated cost for a nation-wide deposit-return system at US\$421 per tonne of material returned in 1988 prices, net of recycling revenues. Converting this to 1995 IR£ gives £356 per tonne. Naughton *et al.* (1990) estimates a cost of US\$322 per tonne for the California container deposit legislation, in terms of loss of consumer surplus. In 1995 IR£ this is £230 per tonne. The average of these two is £290 per tonne. Alter indicates that 80 per cent of the tonnage is glass, with most of the

context has its dangers, not least because of differences in population densities and sizes, and energy costs (these might suggest that the system would cost more in Ireland). However, it gives us some benchmark to work with.

As against this, there are a number of savings from re-use:

- (i) The materials used to produce from virgin materials are saved; these costs might be £30 per tonne<sup>69</sup>;
- (ii) The energy used to convert the raw materials or cullet is also saved; however a large amount of energy, water and materials is used to process the returned bottles, and this might cancel out any energy saving<sup>70</sup>;
- (iii) The costs of disposing of or recycling this glass is avoided; we have seen that the LPMC of landfill is £19-22 per tonne, so this is the saving if the material is diverted from landfill. Diverting it from recycling may save a different sum.
- (iv) External savings will also accrue from reductions in material and energy use, and also from reductions in littering, but since re-use itself also incurs external costs it is not clear that the net externality would be positive.

If we were to take a gross cost of £350 per tonne, less £30 for savings in virgin materials and £20 for avoided disposal, this would give a net cost of £300 per tonne. This is very approximate, given that we are using US data, and the uncertainties stated above. Notwithstanding, it appears that this is a very high cost option, and care should be taken to fully consider the costs before such a system is implemented in Ireland.

### 3.8 Reduction at Source

Reduction at source is generally considered to be the most benign method of dealing with waste, as the waste is not created in the first place. Its extent is difficult to determine, both because we are measuring the absence rather than the presence of waste, and also because the methods of waste reduction are often industrial processes and are, therefore, not in the public domain. One major factor

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rest being aluminium. If we "add back" the net revenues from recycling of, say, £60 per tonne to the above we get a gross cost of £350 per tonne.

Alter points out that in the USA the imposition of deposit systems failed to encourage the market to move back to refillable containers, and that most collected containers are recycled rather than re-used. Since industry has the option to re-use these glass containers, but chooses instead to recycle them, one could assume that costs for a re-use system would be at least as great as the costs calculated here.

<sup>69</sup> The current price for glass cullet at Irish Glass is £40.50 per tonne. This would reflect the costs of virgin materials and also energy saving in using cullet instead of raw materials.

<sup>70</sup> The literature is divided as to which option uses more energy, detergents, etc. (c.f. Pöll and Schneider, 1993).

that determines the level of reduction at source is the cost of the alternatives, i.e., waste generation and disposal; where these alternatives are more expensive than waste reduction, then industry has an incentive to engage in the latter. This also applies at the commercial and domestic levels (at the domestic level the main method of source reduction is to avoid buying disposable products, products with excessive packaging or products which otherwise generate a lot of waste).

From the economist's point of view, then, the best method of encouraging waste reduction is to properly price the alternatives; the main alternative being disposal to landfill. As already seen, this is significantly under-priced at present. Proper pricing of landfill would certainly lead to more reduction at source; however, without data on the marginal cost of reduction at source it is impossible to predict how much would occur. We will return to this issue in the chapter on pricing and incentives.

In some other countries a more directive and interventionist approach has been taken. A case in point is the Netherlands, where a "covenant" was agreed between the government and industry in 1991, to reduce the level of packaging dumped in the environment to nil by the year 2000. The waste hierarchy is applied, and waste reduction plays a pre-eminent role. The covenant, which is a legal contract, has very detailed targets and deadlines for –

- reductions in the amount of waste put onto the market,
- reductions in the use of environmentally damaging materials ("qualitative reduction"),
- a move from one-trip to reusable packaging (if research shows this to be superior),
- recycling and finally incineration.

Industry is obliged to draw up annual implementation plans to meet the targets in the covenant, and the government is obliged to draw up appropriate regulations. It is interesting, however, that cost and economics are rarely mentioned in the covenant. The aim is to eliminate the dumping of packaging waste, while the cost of doing so is given less consideration.

One needs to be cautious of imposing reduction at source (or other waste management methods), by use of regulations. This may impose costs on the various sectors of the economy out of proportion with the benefits from waste reduction. We have not come across an economic assessment of the Dutch system, but it would be worthwhile determining whether the benefits of this approach do indeed exceed the costs.

While sounding a note of caution with respect to introducing regulations in Ireland to impose source reduction, the widespread use of this approach overseas will impact on Ireland, even in the absence of domestic legislation. This is because of the openness and smallness of our economy. If multinational firms must reduce

the packaging levels on their products destined for the major European countries, they are unlikely to maintain a different packaging style for a small market such as Ireland. So we may benefit from source reduction carried out elsewhere. On the other hand, Irish exporters will be forced to engage in source reduction in order to gain or maintain access to other markets. This is likely to impact more on smaller, indigenous firms than on multinationals operating out of Ireland, which would have the resources and economies of scale to pay for the source reduction. Another factor is that packaging reduction, if it also entails weight reduction, should help firms in peripheral regions, as transport costs should be reduced. However, if longer distances to markets also mean that the scope for reducing packaging is limited, peripheral regions might be at a disadvantage. In summary, it is difficult to know whether source reduction in other countries would have a positive or negative impact on the Irish economy.

#### 3.8.1 Costs of Reduction at Source

By its nature, reduction at source is difficult to cost. However, one can assume that, where it is already happening in Ireland, it must have a financial benefit or at least a zero cost (this may not be the case in other countries, where regulations may be imposing levels of reduction beyond what would be financially justified).

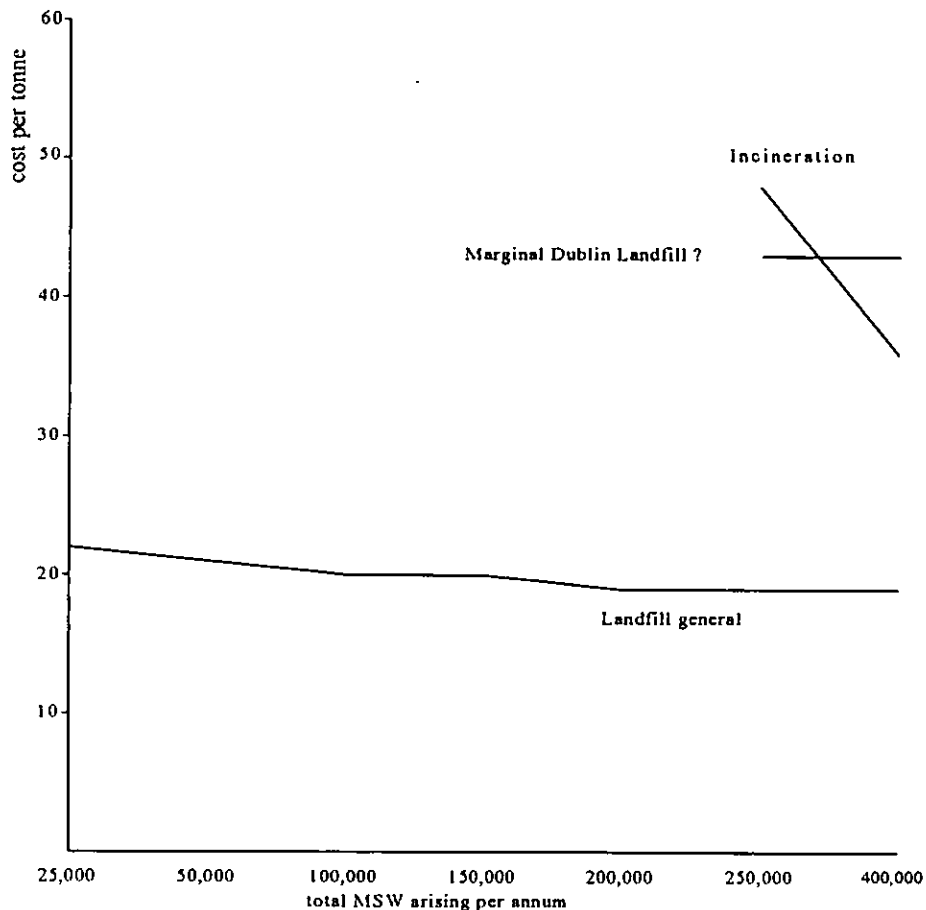
External benefits may be significant, but again it would be very difficult to estimate these. Many of the external benefits relate to the avoidance of external costs related to alternative methods of waste management. As stated already, the proper pricing of waste disposal, energy use and industrial processes that have an environmental impact would ensure the optimal level of waste reduction.

### 3.9 *The Optimal Combination of Waste Disposal Routes*

Having examined the costs of the various options for solid waste management, we can now try to draw some conclusions about the optimal combination of disposal methods. We will do this by comparing the costs of each. As already stated, from an economic point of view, the best way to achieve the optimal use of resources is to charge the proper price (reflecting internal and external costs) for them, and let the market determine the socially optimal way of using them. In this context, we should charge the proper price for the use of all waste management options, and the market will determine the best combination of them to use. A point to note is that this approach obviates the need to set specific targets for the use of various waste management options, as has been done for recycling/recovery in Ireland, and for other options in a number of other countries.

#### 3.9.1 Comparing the Costs

An appropriate starting point is to consider the question of incineration versus landfill. We have estimated the comparable costs, in Tables 3.3 and 3.6, and they can be represented in a graph (see Figure 3.2). This shows that comparing landfill

Figure 3.2: *LRMC of Landfill and Incineration, per Tonne Processed*

LRMC costs in our model with incineration costs, landfill is considerably cheaper. We have also included a possible LRMC for marginal landfill in Dublin, as described in Section 3.4.1<sup>71</sup>. This shows that if incineration could reduce the number of landfills to be built in the Dublin area, then it might be competitive. However, as already stated, there are large uncertainties in the estimation of all these costs, and this needs to be kept in mind before drawing strong conclusions from the foregoing.

The next options we consider are recycling and composting. Recycling can be by collect (kerbside) or bring systems. Regarding kerbside, as with incineration, it appears that a kerbside operation would only be viable in Dublin (and possibly in

<sup>71</sup> Marginal in the sense that an incinerator could replace one of the landfills that might be built to take Dublin's waste.

Cork), so we are effectively comparing these costs with those of landfill disposal in Dublin. Taking an operational size of 50,000 households, and including external benefits, costs are £48 per tonne; for a size of 25,000 households the costs are £94 per tonne (maintaining the existing Kerbside Dublin operation would cost somewhat less). Costs are very dependant on market prices for recyclables, however. Kerbside, at an operation level of 50,000 households, can be competitive with landfill in Dublin if (1) landfill is very expensive and (2) recyclables prices are very high (as they are at the time of writing), but it is difficult to see how these prices would be maintained in the longer run.

Turning to the bring system, the main materials involved are glass and aluminium. With regard to glass, it is not known how much it costs to run the nation-wide Rehab system. Taking the level of subsidisation from local authorities outside Dublin – £30 per tonne – and deducting the estimated external benefit from recycling glass – £7 per tonne – gives a cost of £23 per tonne, but this is largely speculative, and varies widely from site to site within one area. If this were roughly accurate, however, it can be seen that it is comparable with the LRMC of landfill. In larger cities, both landfill and glass bring systems should be cheaper. In Dublin, where it appears as if landfill might be very expensive and the cost of operating bottlebanks low, glass bring systems should have a clear advantage.

As for aluminium, the economics of such recycling should be even stronger than with glass, because of the high market prices achievable. In addition there is a high energy saving, which we estimate might generate an external benefit of £180 per tonne. One could justify quite a high subsidisation of aluminium on this basis, in order to capture these external benefits. This would give a further boost to aluminium recycling. However, there are many substantial assumptions made in the estimation of the external benefits, and care needs to be taken in using the above figure.

Other materials, including paper and textiles, are being recycled using the bring system – the system of paperbanks is being expanded very significantly during 1995, in response to very high prices. Plastics are not recycled on any substantial scale (notwithstanding PET recycling by Wellman International), but given the large energy saving, which we estimate might generate an external benefit of £150 per tonne, there is an argument for a high subsidy on this activity (again, given the assumptions used in estimating this figure, care should be taken in using it). Obtaining plastics in usable quantities and suitably segregated is problematic, however. While it should be viable at an industrial and commercial scale, viability for the domestic sector is not clear, even given the high subsidy the above external benefits would suggest.

As a final point on recycling, we have discussed the external benefits of saving energy through recycling. Ideally, energy use should carry a tax to cover the



external costs related to its use. If such a tax were implemented, there would be no need to subsidise recycling as described above, because the market value of any energy-saving activity should increase automatically.

As for composting, there are no up-and-running municipal waste composting schemes in Ireland at present, so no data exist on the costs. A number of pilot schemes are in place or planned, and these should yield very useful information on costs, marketability, technical difficulties, etc. Estimations from one centralised composting pilot project suggest costs of £25-30 per tonne, which might be competitive with higher landfill costs in Dublin. However, overseas experience suggests that cost might be considerably higher than this. It is not possible to comment further, until actual experience is evaluated. Home composting may be able to achieve the same results as centralised composting, at lower cost, but again actual experience will have to be evaluated.

Re-use is used widely in industry and in the on-licence drink trade, and one can assume that it costs less than other options in these circumstances (regulations and legislation is enforcing re-use in certain cases overseas, but we are not aware of this in Ireland as of yet). It is extremely difficult to cost re-use however. Future increases in landfill prices will give an incentive for more re-use – volume-related charges for rubbish collection should encourage consumers to buy products with less or reusable packaging. As for introducing a re-use scheme for beverage containers in the domestic sector, indications are that this option would be very expensive – data from the USA indicate that net costs could be in the order of £300 per tonne processed through the system. This is much more expensive than any of the alternatives, and caution should be exercised if such an option is considered.

Much of what has been said about re-use also applies to reduction at source. This alternative is almost impossible to cost. The proper pricing of alternative waste disposal methods should ensure the optimal level of reduction at source. In countries such as the Netherlands it is being imposed by law; care needs to be taken if this approach is used, to make sure that the level of reduction imposed is justified by reference to the costs and benefits thereof.

### 3.9.2 Interaction of the Alternative Options

Another consideration is that the level of activity in one waste management system will affect both the economic and technical viability of the others. For example, as mentioned already, increased recycling affects incineration by removing combustibles, which will make incineration less viable, and by removing non-combustibles, which will make it more viable and will reduce incineration pollution. Increasing landfill costs will make both recycling and incineration more attractive, but will also increase the costs of disposing of the residues from these

streams. Finally, more reduction at source will reduce the feedstock of all options, thus tending to make them less viable (given the economics of scale that apply).

These cost variabilities are extremely difficult to enumerate; one would require a model of the behavioural interactions between all the different waste disposal routes, and this would then have to be related to the economics of scale for each route, to determine the effects on costs. Our data are inadequate to support such a model. However, one can make a number of comments on this issue, based on the information already available.

Landfill, as the main disposal route, will also be the largest aggregate cost. The effect of increased use of alternative disposal routes on the aggregate and per tonne cost of landfill are of concern. We have seen already that savings can be made by building fewer landfills, and by building large as opposed to small facilities. However, as Table 3.3 indicates, the LRMC of landfill does not vary enormously with scale, and, therefore, the effect of a small change in quantities landfilled on cost per tonne will not be great. In this context the following points are relevant:

- (i) Increased recycling or re-use of packaging is unlikely to have a significant effect on the quantity of waste going to landfill, and, therefore, is unlikely to affect the economics of scale in landfill.
- (ii) The factors that might have the potential to affect the costs of landfill are incineration, and possibly large scale composting:
  - (a) Incineration would probably only be viable in Dublin, where the waste deposited to landfill in 1990 was approximately 800,000 tonnes (ESBI/Atkins International, 1991). If an incinerator were built to take as much of this waste as possible, then assuming 75 per cent of the waste is combustible, the incinerator runs 90 per cent of the time and 10 per cent by volume remains as ash, this will leave the equivalent of over 300,000 tonnes to be landfilled. This would obviously have a very significant impact on the number of landfills required in Dublin. Savings would be achieved by reducing the number of landfills required, rather than on the cost per tonne landfilled. It is likely that an incinerator, if it were to be built at all, would be somewhat smaller than that described above, to avoid over-dependence on this route<sup>72</sup>. However, an incinerator built to take (say) 300,000 to 400,000 tonnes per annum would be able to replace a similarly-sized landfill in Dublin, and, as we have seen, this could be cost effective.

<sup>72</sup> This could cause serious problems in the event of even short-term closure, due to technical or regulatory difficulties.

- (b) Large-scale composting might remove a maximum of 40,000 tonnes from the waste stream in Dublin annually, or 140,000 tonnes from the country's waste stream, even if all the compost was marketable. These quantities would not have a significant impact on the requirements for landfills.

Similar points can be made in relation to the effect of alternative waste management routes on the viability of incineration. If an incinerator were to be built in Dublin to process 400,000 tonnes per annum, it would have to be operated at this level of throughput in order to recoup its capital cost. Any reduction in throughput would have a serious impact on the cost per tonne. The point is often made that this would discourage reduction, recycling, composting, etc. However, given that such an incinerator would process at most 50 per cent of Dublin's waste currently being landfilled, it is difficult to see how these other activities would significantly impact on it. Given a waste throughput in Dublin's landfills of 800,000 tonnes per annum, and assuming that 75 per cent of this is combustible, the feedstock for incineration in Dublin would be 600,000 tonnes per annum. As stated above, composting might remove 40,000 tonnes per annum from Dublin's waste stream, and this fraction has a high moisture content, so it is not ideal for incineration. Reduction, re-use and recycling are potentially significant as they can remove paper and plastics from the stream – perhaps 200,000 tonnes of these arise in Dublin per annum, although some of this is already being diverted, and should be removed from the equation. But they are also beneficial for incineration, in that they remove glass and metals, of which perhaps 100,000 tonnes arise in Dublin per annum (again, some of this is already being diverted). In practise the amount of diversion that might be achieved for these materials would be far less than the total arising. For example, extending kerbside recycling to the whole of the Dublin area is estimated to divert 35,000 tonnes of combustibles per annum. Therefore, while it is theoretically possible that if extremely high diversion rates were achieved that they might impact on the feedstock for a large incinerator in Dublin, in practise this is highly unlikely.

Looking beyond landfill and incineration, the major factor affecting the viability of recycling, re-use, etc., is the pricing of landfill. If this is properly priced (including externalities), these other options should be utilised to an optimal degree.

To summarise, while the alternative waste disposal options do interact, it does not appear that their interactions will seriously affect the economic and technical viability of each, at least in the scales of operations envisaged here. The one concern is that disposal to landfill be properly priced.

### 3.10 Conclusions

This chapter has examined the costs of the various solid waste management options, and tried to compare them to determine the optimal mix of waste management routes. Both market costs and external costs were considered. The exercise has involved the making of many assumptions, which if they were changed might change the conclusions. In a number of cases, the assumptions made were so significant that the results should be viewed with caution. However, what has been presented is based on the best information available. Subject to this proviso, we can conclude that:

- (i) Landfill costs are set to increase significantly over the coming years; yet it still appears to be the most cost-effective option for disposal of most of the solid waste arising in Ireland. Economies of scale will apply, and this will mean that considerable savings are achievable from building fewer, larger landfills. However, this saving needs to be set against the increased collection costs in the particular circumstances.
- (ii) Incineration is significantly more expensive than landfill in most cases. However, if very high landfill prices prevail in the Dublin area and the number of landfills can be reduced in Dublin as a result of building an incinerator, then incineration might become competitive there.
- (iii) Recycling by the kerbside or collect system may be a viable option, where a large enough catchment area is available (perhaps 50,000 households) and both landfill costs and recyclables prices are extremely high. This combination of circumstances would probably only prevail in Dublin, although it is difficult to see recyclables prices remaining sufficiently high in the longer run.
- (iv) Recycling by the bottlebank or bring system appears to be quite viable over a significant part of the country.
- (v) Composting, both at the household and centralised level, has the potential to divert sizeable quantities of waste, but the economics of centralised composting and of promoting home composting is not clear.
- (vi) Re-use and reduction at source already occur at the industrial and commercial level, and higher landfill prices will increase the incentive for these. Imposing a re-use system at the domestic level appears not to be cost-effective, and caution should be used if considering such a scheme.
- (vii) Finally, the various options will impact on each other, in terms of the amount and suitability of waste going to each. However, in terms of the quantities that are likely to be involved in the Irish context they are unlikely to affect the economic viability of each other, except in the case of incineration in Dublin, which might be able to replace the building of one landfill if the costs of the latter are very high.

This chapter has considered the costs to society of the various options for solid waste management. These costs suggest the most viable options that are available. The next question is, having determined which are the most viable waste management options, how should we go about actualising them? From the economist's point of view, proper pricing of the various options is the most appropriate way to do this. The following chapter will consider proper pricing for waste services and the incentive effects thereof.

## Chapter 4

### *ECONOMIC INCENTIVES IN SOLID WASTE*

#### *4.1 The Incentive Problem in Solid Waste Management*

All households and firms are, to a greater or lesser extent, generators of solid waste. They are, therefore, also demanders of solid waste services (SWS). In a market economy, those who demand goods or services are required to pay for them. In this way, there is an incentive to economise on the use of the good or service. Very often, however, demanders of solid waste services are not required to pay for their use of SWS on the basis of the amount of the service they use. Hence, there is no incentive to economise on the use of SWS through reductions in the amount of waste generated. In this section, we want to address this issue of the lack of incentives to use SWS optimally and how it can be corrected. In so doing, we will consider various charging and taxing options, with a view to identifying the optimal way of introducing incentives. As part of this discussion on incentives, we will also consider the use of incentives in the form of compensation to those who suffer through the development of a landfill.

Before discussing the issue of incentives, we will consider briefly why it is that SWS have not been directly charged for and why it is that this is changing. Previously, the collection and disposal of solid waste were considered to be largely issues of public health which were to be solved by engineers. The objective of public authorities was to remove solid waste from residential and working areas in the least costly, most efficient manner. When land for landfill siting was plentiful and concerns regarding the environmental impact of poorly designed landfills were limited or only partly understood, the disposal of waste was cheap and so the economic dimension of the process was of limited concern. A number of factors are now changing, however, and are having the effect of bringing the economic dimension of solid waste management to the fore. One such change is the growth of the volume of waste in modern industrial/consumer societies. This growth in waste volume has occurred as land available for disposal purposes has declined. This amounts to a rise in demand for landfill space and a decline in supply, thus creating a price rise. In addition, there is the growing public resistance to landfill siting. This has contributed to the decline in supply just discussed, but it has had

an additional effect. The design of landfills has been improved in an effort to reduce negative environmental impacts which in turn should help alleviate public resistance. These improvements have made landfills more expensive to construct and operate.

With rising costs of disposal and increasing difficulties in finding acceptable landfill sites, the authorities responsible for waste disposal must find ways to address these twin difficulties. What is required is a way of recouping the costs of solid waste management while at the same time providing people with an incentive to economise on their use of SWS and so taxing or charging structures that can achieve these ends are gaining attention. We will now proceed to discuss such taxing and charging structures, although we should point out that some authorities, both in Ireland and elsewhere, are already addressing the difficulties by charging directly for SWS. As reported in Chapter 2, Table 2.8, 13 local authorities have volume-related charges for domestic collection and disposal, although as the table shows, two-thirds of the population covered by the survey face a fixed-charge or no charge and hence no economising incentive.

Where direct charging for SWS is not in force a problem in the incentive structure of solid waste management arises. This problem can be conceptualised in the simple diagram shown in Figure 4.1<sup>23</sup>. This diagram shows the demand curve for SWS, which in an Irish context we can take to mean the collection and landfilling of waste. This is a market demand curve and is made up of the demand for SWS of households and firms. For now, we assume that the demand curve is downward sloping, meaning that more of the service is demanded as the price falls. We will address the validity of this assumption below. For simplicity, the private marginal cost (PMC) and social marginal cost (SMC) of providing SWS are assumed to be constant, the difference between the two being the costs arising from unpriced externalities associated with landfills such as groundwater contamination.

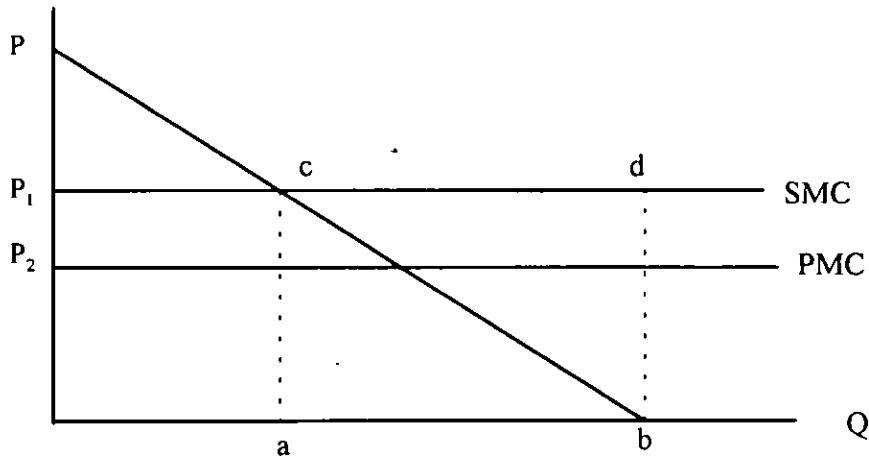
Under standard assumptions, economic theory tells us that the optimal output of SWS is at quantity (a). Such a quantity will be demanded if the price of SWS is  $P_1$ . At this point the social marginal cost is equal to the marginal benefit as measured by the price the marginal user is willing to pay to discard their last unit of waste. At an output level above (a), social marginal cost is above marginal benefit and so welfare is not maximised. Similarly, at output levels below (a) marginal benefit is above social marginal cost so welfare could be increased by increasing the provision of the service.

In the case of goods and services that do not give rise to negative externalities, the optimal output can be arrived at by ensuring that price equals private marginal cost, such as price  $P_2$  in the figure. In the presence of negative externalities, such a

<sup>23</sup> The discussion here is largely based on Jenkins (1993), Chapter 1.

price leaves output above the optimum. Hence, price must be set so that the externalities are included thereby producing price  $P_1$  and optimal output (a).

Figure 4.1: *The Market for Solid Waste Services*



In many instances, both in this country and others, users of SWS do not face additional charges for additional units of SWS demanded and so the price they face is zero. This is true where no charge is imposed and SWS are financed through tax revenue, and also where a flat fee is imposed. In these cases the zero price leads to a demand for service represented by quantity (b) in the figure. At this level of output social marginal cost is above marginal benefit and so the output level is sub-optimal. In the case of local authorities who have privatised collection a problem can also arise. If those collectors are not being charged the full marginal cost of the service, the demand for the service will be greater than the optimum.

#### 4.1.1 Pricing in Practice

On a practical point, the issue arises as to how prices can be imposed on SWS. A number of options exist and it is useful to mention some of these options here. One approach is to require households and firms to buy specific bags from the waste collection agency, whereby only those bags will be accepted at collection. Similarly, tags can be sold by the waste collection agency and only bags with these tags collected. Another possible system is to require the household or firm to subscribe annually, say, to a set volume of collection, say a 32 gallon bin, and to charge higher amounts for higher volumes. As noted above, a number of local authorities have already adopted these types of volume related charges.



All these approaches are volume based and so give rise to a phenomenon known as the "Seattle stomp". Since volume and not weight is charged for, there is an incentive to compress waste and thereby to dispose of the same amount of waste, albeit with a lower volume. It is possible, of course, to overcome this problem by installing a weight based system. Such a system does exist whereby wheelie-bins are weighed when placed on a truck and the household is billed accordingly.

We can set out a number of desirable features of a pricing system. These features are as follows:

1. The charge should be use-related – the more waste people dispose, the more they should pay. In this way an incentive exists to economise on the use of solid waste services.
2. There should be no cross-subsidisation between customers. Again, this indicates that the use-related system is preferable. With a flat fee, small users of the service subsidise large users.
3. Cost recovery must be an important element of the charging system, to ensure that users are paying the full cost of the service. This requirement is in line with the Polluter Pays Principle.
4. Ideally, types of waste which are more difficult or expensive to dispose of should attract a higher charge.
5. The system should be simple, so that the incentives are clear to customers, and it is not too complicated to administer. A complicated system may be so expensive to administer as to outweigh any benefits generated.
6. Proper monitoring and penalising of illegal dumping will be important because of the incentive created by charges for solid waste services.
7. Finally, it would be desirable that the use-related fee be based on the long-run marginal cost of providing the service. In disposing a tonne of waste to a landfill, the time at which a new landfill will be required is brought forward. This element of the cost of landfilling should be reflected in the charging system to ensure again that users face the full cost of the use of the service.

It would be difficult for one system to satisfy all these criteria. Indeed, some of them might be mutually exclusive; for instance, a system that charges differently for different material types may be too complicated to administer in a cost effective way. Trade-offs between the requirements will be necessary but all considerations should be kept in mind to ensure that an optimal system is arrived at.

#### 4.1.2 A Sample Charging System for Solid Waste Collection and Disposal to Landfill<sup>74</sup>

We will attempt to design a charging system based on LRMC (long-run marginal cost), which will preserve economic efficiency while at the same time ensuring cost recovery for the local authority. As we saw in Figure 3.1, LRMC is falling with scale, with the curve flattening out at larger sizes. Economic efficiency requires that the consumer of the service is charged the marginal cost. However, if this is done in the context of falling marginal costs, the provider of the service will not recover all costs (see diagram). This can be rectified by having a two-part charge – a use-related charge based on LRMC, and a fixed charge designed to recover the balance of costs<sup>75</sup>.

If we take a typical landfill site with an annual throughput of 50,000 tonnes, we saw from Table 3.3 that the LRMC for such a site (including collection and external costs) was £21 per tonne, whereas from Table 3.2 the average cost was £65 per tonne. Therefore a charge based on the LRMC would leave £44 per tonne to be recovered by a fixed charge.

If we assume that the average household produces 1 tonne of solid waste per annum, then the preceding numbers can be related to a single household, producing 19kg of waste per week. On a weekly basis this translates into a 40p use-related charge and a 85p fixed charge, a total of £1.25 per week. The charge could be collected from households using a tag-a-bag or wheelie-bin approach and we shall consider each in turn:

(a) *Tag-a-Bag System:*

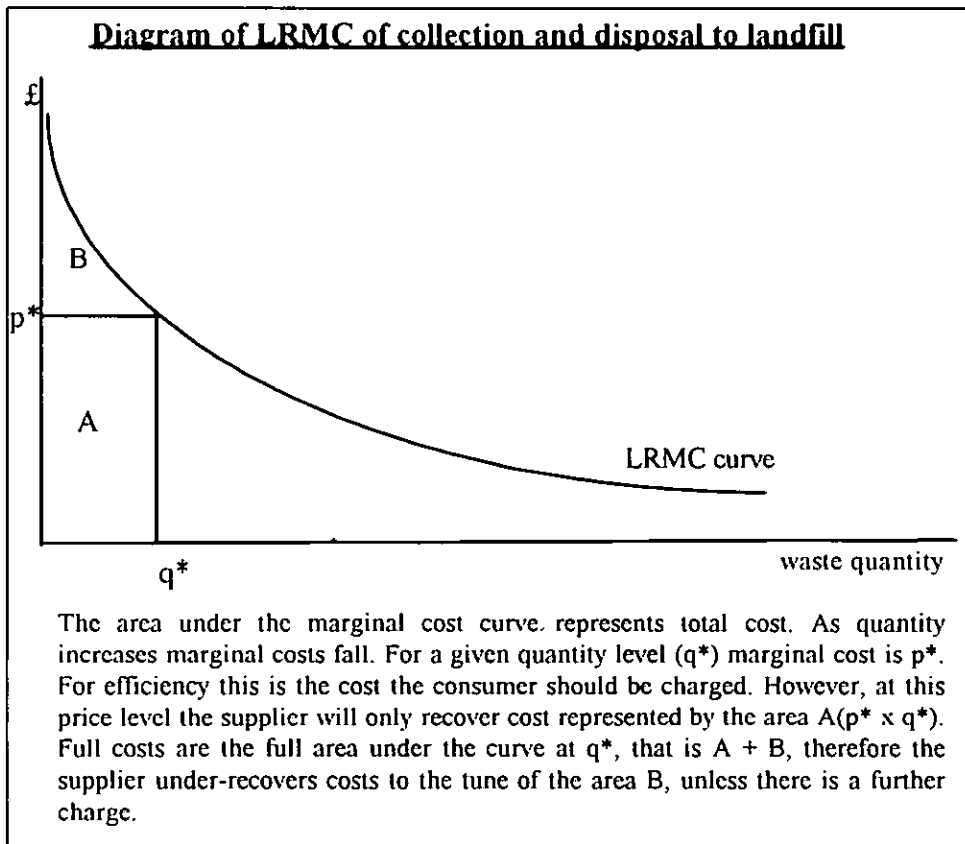
Taking the LRMC charge first, this can be charged at 40p for a tag or bag containing 19kg (it may be convenient to round this up to of 20kg). The balance required for cost recovery could be charged by an annual fixed charge of £44.

(b) *Wheelie-Bin:*

For the LRMC charge, a wheelie-bin capable of containing (say) 20 kg could be charged at 40p per pick-up. Smaller or larger bins would be charged accordingly. A convenient way of charging this might be to use a tag system as in (a) above, where the tag would have to be attached to the bin for it to be picked up. Where technology permits in the future, each bin could be weighed on pick-up, and charged at 2p ( $40 \div 20$ ) per kg collected. The balancing charge could be charged as in (a) above. Where bins are provided by the local authority, the cost of the bins would be added to the fixed charge.

<sup>74</sup> The approach here follows Scott and Lawlor (1994).

<sup>75</sup> These fixed charges do not affect efficiency because they do not affect people's behaviour. Fixed costs can alternatively be recovered by some other non-distortionary method such as a fixed property tax.

Figure 4.2: *LRMC of Collection and Disposal to Landfill*

The above suggested system is very much a *pro forma* approach. Each local authority would need to re-calculate the costs in terms of its own circumstances. A similar system could be devised for dealing with commercial waste.

We can also suggest a system for charging for waste delivered directly to a landfill site. From Tables 3.2 and 3.3 we can see that the relevant costs per tonne break down as follows, using the site taking 50,000 tonne per annum as an example, and including external costs<sup>76</sup>.

Anyone entering the landfill site to deposit waste would pay a weight-based charge and a fixed charge. The weight-based charges should be £13 per tonne. As for the fixed costs, these could be set per type of vehicle, by reference to the usual

<sup>76</sup> These costs include external costs relating to collection, although a collection service is not being considered here. However, the agents delivering waste to the landfill site would presumably incur such external costs when collecting the waste, and, therefore, it is not inappropriate to charge them for these externalities.

weight of waste transported therein. For example, if we assume that a waste freighter carries on average 8 tonnes, then a standard fixed fee for a freighter to enter the site could be set at  $\text{£}13 \times 8 = \text{£}104$ . Other standard fees could be calculated for other vehicle types and skips.

The above charges relate to the costs of modern landfill. New landfill sites will increasingly replace the old landfills in the coming years, as a result of the latter coming to the end of their useful lives, and of recent and proposed domestic and EU legislation. In the interim, however, the old landfills will continue to be used, and one might ask what should be charged for the use of these. One argument is to charge by reference to the actual costs of running these landfills. However, from an economic viewpoint, they should be charged by reference to their LRMC, just as new landfills should be. That said, calculating the LRMC for old landfills is not simply a matter of comparing the capital cost of the existing site with the cost of the next biggest site. This is because LRMC is concerned with the cost of the next phase of capital investment, and this will be a modern landfill, whose costs will be radically different from those of old sites. Therefore, the LRMC of these old landfills is equal to the LRMC of the new landfills that will replace them. This is effectively the cost to society of filling up the old landfill<sup>77</sup>. This is the minimum that should be charged – if actual running costs including externalities are greater than this, then the difference should be recovered by a fixed charge, of the type described above.

As a final point, the above calculations include the estimated external costs<sup>78</sup>. The mechanics of levying these costs needs consideration. There are two types of costs involved – the variable costs and the disamenity (fixed) costs. The way usually recommended for dealing with these costs is for a central agency to charge the landfill operators for them, and for these operators to pass the charges on to the consumers of the service. The central agency should levy the variable charge by reference to the tonnage going into the landfill, and the disamenity charge as a fixed charge per landfill, regardless of level of activity.

<sup>77</sup> Charging in this way will give an incentive to minimise the use of the old landfill, which means that it might remain in operation for longer. One might argue that this would be detrimental to the environment by postponing the building of modern landfill sites with their superior environmental protection. If it were felt that the net effect of keeping the old landfill open longer (by reducing the amount of waste being disposed of) was negative, the solution might be to close the old landfill before it was full.

<sup>78</sup> One cost that is not included is the extra cost of policing illegal dumping, if this is encouraged by the introduction of use-related charging. This issue will be dealt with later in the chapter, but if there is a significant extra cost it should be added to the charges levied on consumers of the service, if it cannot be recovered from those responsible for illegal dumping.

The case of the disamenity costs is interesting, as these do not vary with the amount of waste going through the facility. There is an argument for not levying this charge, as it would simply be passed on as a fixed charge, and would not have an effect on the amount of waste generated<sup>79</sup>. There are two possible counter-arguments to this:

- (i) If society prefers to have fewer large landfill sites instead of a larger number of small sites, then the disamenity charge should be levied on site operators. This will give them an incentive to minimise the number of sites.
- (ii) If the landfill operators have to compensate the local residents for the disamenity suffered by them, this will become an internalised fixed cost of the landfill, and will be passed onto the users of the landfill in the same way as any other cost. Needless to say the central agency would not then include disamenity costs in its levy.

#### 4.2 *The Elasticity of Demand of Solid Waste Services*<sup>80</sup>

We now return to the assumption that the demand curve for SWS is downward sloping. It could be the case that this assumption is invalid and that the demand curve is actually vertical, whereby the same amount of solid waste is discarded regardless of the price of disposal. If this is so then the above discussion is invalid also. If the same quantity of SWS will be demanded regardless of price, the issue of optimal versus sub-optimal levels of output does not arise. It is important for us, therefore, to establish the probable slope of the demand curve for SWS.

For the demand curve to slope downward it is necessary for users of SWS to be able to (a) reduce the amount of waste they generate and/or (b) to reduce the amount of waste they discard for collection and landfilling. Waste generation can be reduced, for example, by buying low packaging intensive goods or re-using supermarket bags. Waste discarded for collection can be reduced by recycling, composting or in the undesirable way of littering.

In theory then, it is certainly possible for the demand curve to slope downward. We now want to review some empirical examinations of this question. Most of the relevant studies attempt to measure the price elasticity of demand for SWS. Findings of negative elasticities would indicate that the demand curve is indeed downward sloping, i.e., as the price goes up the quantity of the service

<sup>79</sup> This comes back to the central assumption that the landfill is unavoidable. If one could argue that there is an option to build or not build the landfill, then these disamenity costs also become variable.

<sup>80</sup> By price elasticity of demand economists mean the percentage change in quantity demanded relative to the percentage change in price. Essentially what is being measured is the responsiveness of quantity demanded to price.

demand goes down. If the demand curve is vertical the price elasticity would be zero, i.e., no change in quantity demanded when the price changes.

Jenkins (1993) contains a review of the literature on this point. The earliest study she refers to in which the direct effect of user-fees on demand for SWS is considered is that of Wertz (1976). Using a very crude approach in which there were only two data points Wertz estimates an arc elasticity of -0.15. A later study by Efaw and Lanen (1979) found no significant response of demand to a user-fee but like Wertz their empirical work is crude. Two other studies are referred to by Jenkins although the approach in these is somewhat indirect. As a proxy for user-fees the authors, McFarland *et al.* (1972) and Skumatz (1990b), use waste utilities' revenues per ton. The problem with this approach is that average revenue tells us nothing about fee structure so it is not clear that a response to user fees is being measured. This problem is particularly serious in the work of McFarland *et al.*, so their elasticity estimate of -0.46 should be viewed cautiously. For Skumatz the problem is less serious since it is known that, for part of her sample, user-fees were in operation. Her elasticity measure of -0.14 is therefore more believable.

In her own empirical work, Jenkins collected data from nine communities in the US, five of which had user-fees and four of which did not. Along with user-fee data, she collected data on such things as average household income, household size, etc. She estimates an elasticity of -0.12 at sample means and concludes that a switch from no user-fee to a \$1 charge per 32-gallon container would lead the average household to reduce their waste discarded by 15 per cent. Jenkins goes on to measure welfare gains from user fees by calculating the area of the triangle (bcd) of Figure 4.1 under different assumptions on the social marginal cost of waste disposal.<sup>81</sup> For example, for the city of Seattle she calculates that a move from a zero user fee to a fee of \$1.71 per 32-gallon container would produce a welfare gain of \$5 million.

While Jenkins' empirical work is superior to the earlier studies mentioned it still suffers from the problem that it is an indirect look at the effect of user-fees since it does not consider how a single community reacts to a price change. A recent study that adopts this direct approach is that of Fullerton and Kinnaman

<sup>81</sup> The area of the triangle is taken to measure welfare losses and gains because it is the difference between marginal social cost and marginal benefit. We take the demand curve to represent marginal benefit because it shows how much the marginal consumer is willing to pay for an each additional unit of the service. Starting at point (c), an extra unit of the service costs more than the benefit it provides as seen in the fact that the marginal consumer is willing to pay an amount below the SMC for that unit of the service. For each unit beyond point (c), SMC exceeds marginal benefit and by summing the differences we get a measure of the net social cost of providing a level of service (b) as opposed to (a).

(1994). They gathered data from households in Charlottesville, Virginia before and after the introduction of a per-unit user-fee for solid waste services. These data included the weight and volume of solid waste discarded, the weight recycled and the socio-economic characteristics of the households. Using the weight measure of discarded solid waste, they estimate an own price elasticity of  $-0.075$ ; using the volume measure, they estimate an own price elasticity of  $-0.227$ . As the timeframe of the study was only a few months, it could well be that this is a short-run elasticity. With more time to adjust, the elasticity measure could be higher. The fact that the volume elasticity is higher is as expected since the charge is based on volume as opposed to weight. The authors also estimated a cross-price elasticity for recycling of  $0.074$ <sup>82</sup>.

Fullerton and Kinnaman's data indicate that reduction in per capita garbage discarded was 1.53 pounds per week. They go on to establish what method was used most by households to reduce waste discarded. The percentage who report using each method as their main responses to the charges are shown below along with the observed reduction in their waste. It should be noted that each household was allowed to report more than one response.

Table 4.1: *Household Response to SWS User-fees (from Fullerton and Kinnaman)*

<i>Household indicating:</i>	<i>%</i>	<i>Pounds</i>
Did not reduce	25.3	-0.05
Recycled more	65.3	-1.76
Composted more	30.6	-2.25
Demanded less package	17.3	1.26
Other	10.7	-5.1
All households	100	-1.53

Some interesting points emerge from this table. For most people, 65.3 per cent, additional recycling was the dominant response to the user-fee. The free kerbside recycling programme which was in operation facilitated this. As such programmes are almost non-existent in Ireland, such a result may not hold here. The group which reported that they demanded less packaging were not successful in reducing the weight of their discarded waste, although they did reduce the volume of this waste by 0.12 pounds. The biggest reduction in weight came from those who reported "other" as their dominant response to the user-fee. The authors believe this may represent illegal dumping of waste so this is a worrying result.

<sup>82</sup> By cross price elasticity we mean the effect of a change in the price of service A on the quantity demanded of service B. In this case what is being measured is the responsiveness of the demand for recycling when the price of SWS rises.

The social costs of this illegal dumping depend on precisely what is being done with the waste; for example, is it burnt in a crowded urban setting or placed in bins belonging to others?

In summary, the studies that have considered the slope of the demand curve for SWS in general find negative elasticities. All the studies are from the US, however, so care should be taken in extrapolating the results to Ireland. To the extent that the results are applicable, it would appear that charging for SWS could reduce the amount of waste going to landfill. It would be a very useful exercise to replicate in an Irish setting some of the studies described to assess the impact that charging would have.

#### *4.3 Getting Relative Prices Right*

Fullerton and Kinnaman's examination of the alternatives used in response to the user-fee for disposal leads us to the interaction between the various alternatives available to the household. Morris and Holthausen (1994) and Kennedy and Laplante (1994) have looked at this issue and their work, along with the Fullerton and Kinnaman results, raises certain considerations which we should be aware of.

When faced with user-fees for waste disposal, households and firms have alternatives available to them. In general, the greater the number of substitutes available for a good or service, the greater will be the own-price elasticity of demand of the good in question. The availability of free kerbside recycling to the households in the Fullerton and Kinnaman data set most likely increased the price elasticity of demand of these households for SWS. In Ireland, as free kerbside recycling is so rarely available, it could well be that measured elasticities would be lower. If this is the case, then the potential welfare gain from user-fees would be a good deal lower also.

It could then be argued that in order to raise the own-price elasticity of demand for SWS and thereby generate a greater potential reduction in waste disposed through user-fees, free kerbside recycling should be provided to greater numbers of households in Ireland. Such a policy could be economically inefficient however. If disposal is charged for and recycling is free, there is a strong incentive to recycle as opposed to dispose. The only cost to the household would be the inconvenience of separation and if this inconvenience is valued less than the cost of disposal, recycling will occur. The potential problem that arises is similar to that shown under the partial equilibrium analysis of SWS when a zero price is charged for collection and disposal. An excess supply of recycled materials can arise. This could be manifest through a glut of recycled material leading to falls in prices, thereby rendering the recycling effort uneconomic. Alternatively, the glut could be such that materials collected as recyclables could be landfilled anyway, in the absence of buyers.



The important point is that the relative prices of disposal and recycling should reflect the relative marginal costs of the two activities in order to achieve economic efficiency. Such reasoning encompasses a situation in which landfilling is sufficiently expensive and recycling sufficiently cheap that user-fees for disposal with free kerbside may be an efficient pricing strategy.

If both disposal and recycling services are being charged for or if a cost is faced such as the time taken to bring bottles to a bottle bank, an incentive exists for households to look beyond these methods of disposal. Referring again to Fullerton and Kinnaman, we can see that included in these alternatives are waste reduction (under which heading we can include composting) and littering. If littering is considered costless by the individual there will exist a strong incentive to litter. This could arise if (a) no penalties exist for littering or (b) there is no enforcement. In order to ensure that littering does not arise it is necessary that the penalties be sufficiently high and the probability of prosecution be high enough that on average the cost of littering will exceed the costs of the more acceptable alternatives.

This concern that volume-related charges could lead to increased littering can be explored to a limited degree by considering the experience of local authorities who have implemented per-unit user-fees. In our survey we asked if a litter problem had arisen and if it had been solved. While eight local authorities in our survey said a litter problem had arisen following the introduction of per-use charges, six of the eight said the problem was solved. We spoke at greater length to two local authorities about this littering issue and a number of points emerged. It was the feeling of those we spoke with that the main cause of littering was not user-fees but a lack of environmental awareness. The increasing environmental awareness of recent years had reduced the social acceptability of littering. This increased resentment of those who dump illegally has made the prosecution of offenders that much easier. It was even said that a warning from the Gardai was often enough to curtail an individual's illegal dumping.

With proper relative prices attached to disposal, recycling and littering, there exists an incentive to reduce waste up to the point where the marginal benefit of doing so equals the marginal cost. (The marginal benefit of waste reduction comes from the avoided charges; the marginal costs comes from the inconvenience generated by the shift in consumption to less waste intensive goods.) If the charges are correct an efficient use of the various approaches to solid waste management will result.

#### *4.4 The Politics of Charges*

Before proceeding to consider other approaches to incorporating incentives in this area, the political issue of resistance to charges should be addressed. Charges in other areas such as water have been greeted with resistance and in some cases

refusal to pay. This adds to the costs of collection and makes politicians wary of proposing charges. It appears to us that part of the resistance to charges comes from a sense that the imposition of charges amount to double charging. People feel that their taxes have been used to finance these services so that direct charges amounts to double charging.

One way to avoid this argument against charging is to link the imposition of charges to tax cuts, thereby showing that what is occurring is a shift in financing method and not an effective tax increase. The recently introduced tax allowance for people who pay charges acts in this manner. At a time of rising costs for waste disposal, it would be necessary for revenue raised through charges to be greater than that given back in tax cuts to ensure adequate financing of SWS. Our reason for believing that charges may have greater acceptability if imposed in combination with tax cuts stems from results of a survey on environmental attitudes carried out by the ESRI (Murphy, Scott and Whelan, 1994). When asked about methods of payment for improved environmental services, including waste disposal, 3 per cent said they favoured increased taxes while 44 per cent said they preferred charges for the amount used (53 per cent favoured fixed charges).

#### *4.5 Other Economic Instruments*

The discussion of incentives has focused so far on charging households and firms at the point of disposal. Let us now consider other approaches to creating incentives in the area of solid waste where intended effects are to lead to waste reduction or other ways of diverting waste from landfill. We will look at five such approaches: (i) product taxes; (ii) raw material taxes; (iii) deposit-refund schemes; (iv) recycling credits and (v) landfill levies. We will also look briefly at an example of the use of incentives, the German "Green Dot" system. Much of the discussion here is taken from the Environmental Resources Limited Report of 1992.

##### *4.5.1 Product Taxes*

Product or packaging taxes are taxes on goods which relate to their waste component. The waste component is made up of the packaging associated with the product and also the waste arising when the product is no longer in use. Such a tax can be used to achieve a number of objectives, not all of which are mutually exclusive. First, the tax can be used to internalise the external costs of the disposal of the product and its packaging. Second, the tax can be used to alter behaviour, such as enticing consumers to substitute less waste intensive products into their consumption bundles. Third, the tax can be used to raise revenue to fund waste disposal or other waste services, or government activities in general.

From an economist's viewpoint, the first objective, that of internalising external costs, is the most readily defensible. The tax can be imposed on either the consumer, providing an incentive to alter consumption patterns, or the producer,

providing an incentive to alter production patterns. Either way, the production of relatively waste-intensive products should decline thereby reducing waste.

The way in which a product tax can ultimately affect the waste stream, and the group which bears the incidence, depends on how the consumers and providers of the products involved react to price changes. In order to explore the effects of such a tax, we will examine a number of scenarios.

*Scenario 1:* (Figure 4.3) In the first scenario, consumers of the product reduce their demand for the product in response to increases in price, giving the downward sloping demand curve (D). Producers, on the other hand, increase their supply of the product in response to price increases, giving the upward sloping supply curve (S). In the absence of any tax, this market for SWS would operate like any other market and reach equilibrium where the price of the product would be  $P_1$  and the quantity traded would be  $Q_1$ .

Let us now consider how the imposition of a product tax on the producers alters the outcome. As the producers must pay the tax they will want a higher price for the product at each level of output. This is represented in the diagram by the upward shift in the supply curve from S to S'. The size of the upward shift will be equal to the size of the tax.

In effect, there is a new supply curve and so a new equilibrium emerges. Demand and supply now intersect at a price of  $P_2+t$  and  $Q_2$  so these are the new price and quantity that prevail. When faced with the tax, producers attempt to pass the price increase onto consumers. But in response to higher prices, consumers reduce their demand. In the end, consumers pay a price of  $P_2+t$  but producers only receive  $P_2$ . The difference,  $t$ , is the tax which goes to the government.

There are two main points to be learned from this scenario. First, the effect of the tax is to reduce to quantity of the product traded and so the inflow to the waste stream will be reduced accordingly. Second, although the tax is imposed on the producer, consumers bear part of the burden of the tax in the form of increased prices. This is a familiar point about taxes in general that has particular relevance in the environmental context. Application of the Polluter Pays Principle may dictate that taxes be imposed on producers but as this scenario shows, the producer may not bear the full cost. This point will emerge more fully in Scenarios 2 and 3.

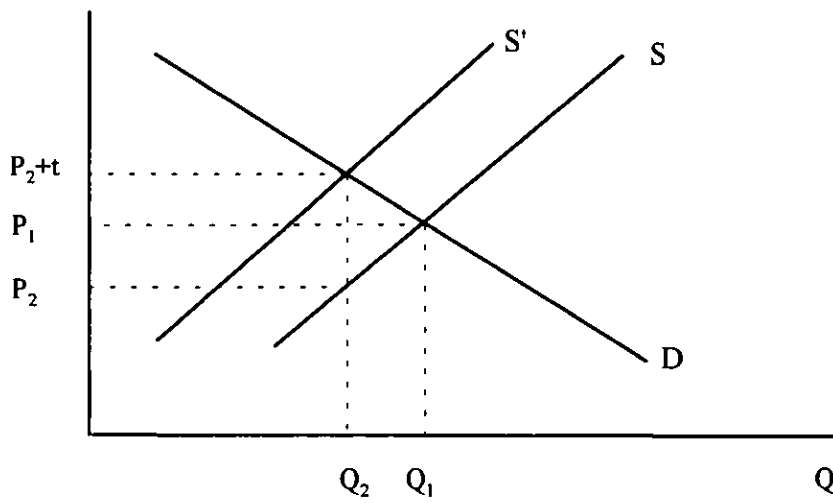
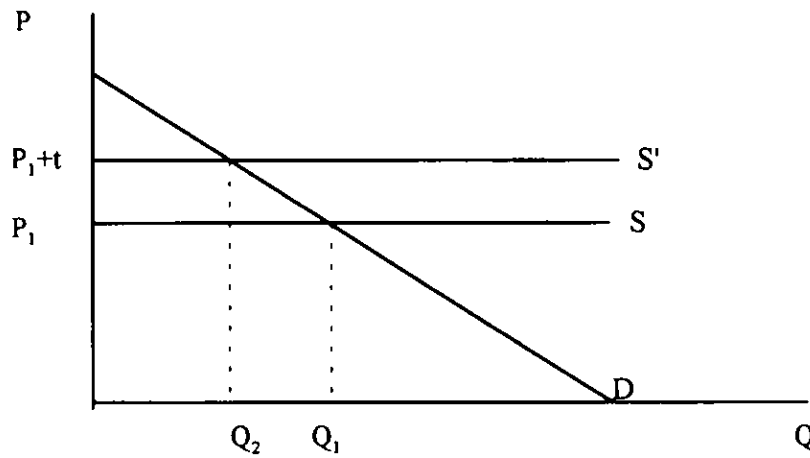
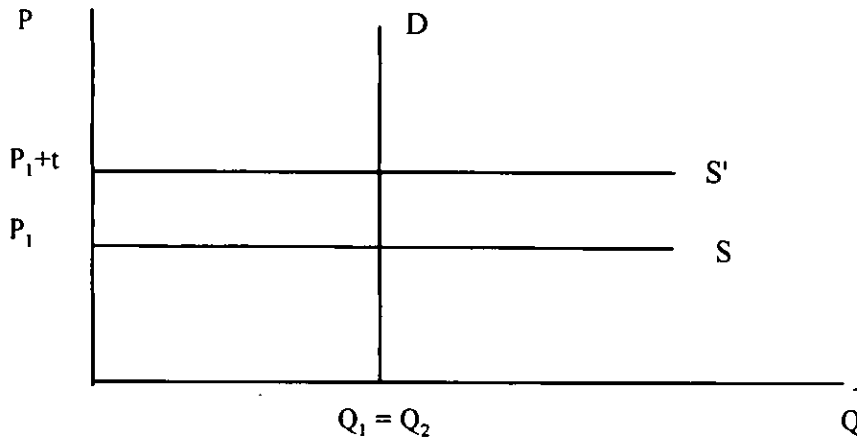
Figure 4.3: *Scenario 1 on Product Taxes*Figure 4.4: *Scenario 2 on Product Taxes*

Figure 4.5: Scenario 3 on Product Taxes



*Scenario 2:* (Figure 4.4) In this scenario demand, as before, is downward sloping. Producers act differently in this case, however. For a price of  $P_1$  (or better) producers simply supply the amount that is demanded, giving a horizontal supply curve. Such a situation might arise if a producer is producing for a global market and the market depicted in the diagram is the Irish sub-set of that global market. The producer's decisions on levels and forms of production are determined in the global context and demand in Ireland is simply met at the going price, again giving the horizontal supply curve. In the absence of the tax, the market price of the product will be  $P_1$  and the quantity traded  $Q_1$ .

As in Scenario 2, the imposition of the tax on the producer has the effect of shifting the supply curve upward from  $S$  to  $S'$  because the producer will want a higher price for each level of output. Again, the shift is equal to the amount of the tax. The new intersection of supply and demand is at a price  $P_1+t$  and the quantity traded is  $Q_2$ .

As before, the reduction in the quantity traded of the product will reduce the inflow into the waste stream. Note, however, that in this case, all of the burden of the tax falls on the consumers. The rise in price is exactly equal to the tax,  $t$ , and so the producers have passed all the tax onto the consumers.

*Scenario 3:* (Figure 4.5) In this case, producers act as they did in Scenario 2 but consumers act differently. Whereas in the first two scenarios consumers adjusted the quantity they demanded of the product in response to price changes, in this scenario quantity demanded does not vary with price. This may seem like an extreme case but it can be regarded as the polar case in which people have a

sufficiently strong need for a particular product that they will purchase a given quantity regardless of the price.

Once again, the initial price and quantity are  $P_1$  and  $Q_1$  respectively. Also, the initial effect of the tax is to shift the supply up from  $S$  to  $S'$ . The price of the product rises to  $P_1+t$  so the consumers are once again absorbing the full burden of the tax. In this case, however, there is no reduction in the quantity of the product traded because of the nature of consumer demand. As a result, there will be no reduction in the waste stream.

The purpose in presenting these scenarios has been to show that the effect of the product tax depends on the reactions of producers and consumers. In designing a product tax structure, the points made must be kept in mind. In general, the agents who are least responsive to price changes (for example, the consumers in Scenario 3) will ultimately bear the incidence of the tax, regardless of where it is imposed. And if either demand or supply is very unresponsive to price changes, the tax will have little effect on the waste stream because there will be little change in the quantity traded of the good.

Product taxes can also be used in an effort to encourage industry to set up recycling schemes or deposit refund schemes. For example, Norway taxes non-returnable beverage containers, thereby providing an incentive to operate a deposit-refund scheme. In order for the tax to be successful in achieving these sorts of objectives the taxes must be sufficiently large to make the alternative worthwhile. In using product taxes for this reason, it should first have been established that the encouraging of re-use or recycling is an optimal objective. If the costs of re-use or recycling, both internal and external, are greater than the corresponding costs of landfilling, a product tax that creates a strong incentive towards recycling may reduce welfare.

Another possible objective of a product tax would be the raising of revenue for the establishment of recycling schemes or some other purpose. As with the previous objective, questions should be answered before imposing such a tax for such a purpose. In purely economic terms, revenue should be raised in the most cost effective manner, where cost is defined broadly so as to include costs arising from administration and from distortions to the economy. It could be, however, that the environmental linkage between the tax and its purpose make it politically more acceptable. As stated before, it should also be established before a recycling scheme is set up that it is the optimal thing to do.

The design and coverage of a product tax depends on the trade-off made between administrative ease and effectiveness. For example, in order to correctly internalise external costs, a large number of tax rates must be applied reflecting the contributions of different products to the waste stream. Clearly though, a higher number of rates will create greater administrative difficulties.

#### 4.5.2 Raw Material Taxes

Raw material taxes come in two forms. First, virgin raw materials can be taxed while secondary raw materials remain untaxed, thus creating an incentive for the use of secondary raw materials. The tax on virgin raw materials can be set so as to internalise the eventual disposal cost. The logic then in leaving secondary materials untaxed is that their waste component has already been charged for. The second type of raw material tax is where all raw materials are taxed but the rates differ according to estimates of the rate of recycling. Thus, a raw material that is never recycled will face a higher tax rate than one which is recycled a number of times. In both cases, the incentive to use recycled materials is intended to lead to the diversion of waste from the waste stream.

As we saw under product taxes, although raw material taxes can be used to internalise costs of disposal, their actual effect on the waste stream may not be that significant. In order for producers to switch from using virgin raw materials to secondary materials, the tax must be set at a level that makes the secondary material relatively cheaper. If the tax is set such that virgin materials remain cheaper, producers will continue to use virgin materials. As production will have become more costly, output may fall and in this way the flow into the waste stream will be reduced. However, this generally is not the intended effect of a raw material tax.

The notion of the tax making production more costly brings us to another point. A tax on raw materials, levied only in Ireland, would put domestically produced goods at a competitive disadvantage and so this problem would have to be addressed. Regarding goods sold in Ireland, those imported would have to be taxed on a product basis so as to restore the competitive balance. This tax could be computed on the basis of the recycled component of the good in question but by moving in this direction the information requirements of the tax would become large. Domestically produced goods sold abroad would require a refund of the tax in order to restore a competitive balance.

Another issue which arises in the context of foreign trade is the possibility that recycled materials could be imported into this country in an effort by firms to avoid a virgin material tax. Should this recycled material make its way into the waste stream following one use, the effect on the Irish waste stream is the same as if virgin materials were used since no real diversion has occurred from the domestic perspective. It is true that waste is diverted from the country which exports the recycled material but it would seem unfair that Irish consumers and producers would bear the cost of waste diversion for another country. For this reason the imposition of such a tax may make more sense at a transnational level.

As was the case under product taxes, even though a raw material tax might be levied on producers this does not mean that the full cost of the tax is necessarily

borne by producers. The tax will raise the costs of producers and so an effort will be made by them to pass this increase onto consumers or workers. Again, their ability to do this depends on the relative responsiveness of producers and consumers but the question arises that if consumers ultimately pay the tax, would it be better to levy it directly in the form of disposal charges, for example?

#### 4.5.3 Deposit Refund Schemes

Deposit-refund schemes operate through an additional charge being placed on an item when it is purchased, and this charge being subsequently refunded when the item or its container are returned. Such schemes have been widely used in the US and Europe, although typically for a limited range of products such as beverage containers. This type of scheme has been extended to car hulks in Greece and Norway and vehicle batteries in the US (OECD 1994).

These schemes can be used to internalise the costs of disposal. By setting the charge equal to the marginal social cost of disposal, a consumer who purchases an item and discards as opposed to returning it incurs the cost of disposal. A consumer who returns the item and thus keeps it from the waste stream avoids the disposal cost. Typically, however, the objective of these schemes has not been internalising costs but rather the objectives have been to generate high rates of return of containers with a view to re-use and recycling and to reduce litter. In order to achieve these objectives it has been necessary to set deposits at levels greater than those that would internalise disposal costs.

One case study of a deposit refund scheme applied to beverage containers (Porter, 1983) found that the scheme was successful in generating high rates of return (95%) and in reducing the amount of litter due to beverage containers (85%). The more difficult issue in evaluating the scheme, however, is putting money values on the costs and benefits. In order to put a value on the reduced litter it would be necessary to know how much people would be willing to pay in order to reduce litter by the amount observed. To our knowledge, no such authoritative estimate has ever been made. On the cost side, one of the most difficult items to estimate is the value consumers place on the inconvenience of having to store containers and return them. On the basis of the costs of deposit-refund schemes presented in Section 3.4, which omitted the valuation of reduced litter and increased consumer inconvenience, all we can say is that the valuation of reduced litter would have to be high and consumer inconvenience would have to be low for deposit-refund schemes to be efficient. Without valuations, however, nothing positive can be said.

In discussing product taxes some points were made concerning the small nature of the Irish market for some products relative to the global market and this issue is relevant in the context of deposit-refund schemes also. The expectation underlying a deposit-refund scheme might be that producers might be enticed to



switch from one-trip packaging to reusable packaging. However, if a producer is producing for a large market, most of which does not have the scheme, it may not be profitable to switch to reusable packaging. In this case, although the producer is forced to take back the containers these containers may not be re-used and may be disposed. In this case, no diversion from landfill has been achieved but additional cost has been incurred.

#### 4.5.4 Recycling Credits

Recycling credits are another method of encouraging the diversion of waste from landfill and towards recycling. Under such a system, local authorities or other bodies responsible for waste management would make a payment to recyclers equal to the amount the authority saved by not having to collect and dispose of the material in question. While there are no such schemes in Ireland, the grants that local authorities pay to groups such as Rehab or can recyclers are in the nature of recycling credits. Overseas, the UK introduced legislation in 1990 to formalise a system of recycling credits to be paid by Waste Disposal and Collection Authorities, but we have found no evaluation of this system so far. Recycling credits have also been used in Canada in the past, but it appears they were not successful in encouraging large-scale recycling, and were subsequently abandoned (Touche Ross, 1991).

As pointed out in an earlier discussion, it is important that recycling credits be set in such a way that a situation is generated in which the relative prices on landfilling and recycling reflect their social cost. It is important to avoid setting the credit too high, otherwise an over-supply of recycled materials can be generated. For example, if landfill is being correctly charged for, recyclers are implicitly rewarded by not facing the disposal charge. If a recycling credit is imposed on a situation where landfill charging exists, recyclers are getting a double reward, i.e., the avoided landfill charge plus the recycling credit. Such a strong incentive to recycle may create an over-supply of recycled material.

#### 4.5.5 Landfill Levy

A landfill levy is a tax on disposal of waste to landfill. The tax can be levied on either operators of landfills or disposers to landfill. This approach to introducing economic incentives has been adopted by Britain and Denmark so we will give it a more lengthy consideration here than the other instruments. Much of the discussion is taken from Coopers & Lybrand (1993).

Like the other taxes, a landfill levy can be used to achieve a number of objectives. Again like the other taxes, it can be used to internalise the external costs of disposal to landfill. Second, it can be used to encourage the diversion of waste from landfill. Third, it can be used to raise revenue for purposes of running the solid waste management system or for other purposes entirely.

The levy can take a number of forms, each with advantages and disadvantages. First, it can take the form of a per weight or volume charge, constant across all forms of solid waste. This would be a simple approach to the levy and as it would be constant across the country it may be viewed as fair and hence more acceptable than other, more variable, approaches. A second approach, that of a constant *ad valorem* charge, would vary regionally to the degree that landfill charges differ across the country and so may be seen as unfair. However, to the degree that differences in landfill prices reflect higher environmental damage from some landfills, it may be appropriate that the levy faced at some landfills be higher than others. A third approach to the levy would be a variable levy based on either weight, volume or cost of disposal charged, with the levy depending on the content of the waste being disposed of. As some waste imposes higher environmental costs than others, the idea would be that this should be reflected in the levy. The administration of such a variable approach and the information required could make it impractical so either of the other two approaches would be preferred.

Let us now consider the possible effects of a landfill levy. To an extent, the effects depend on where the levy is imposed, i.e., operators of landfills or disposers of waste to landfill and on the ultimate incidence of the levy, i.e., if it is pushed back or forward by agents in the solid waste arena.

Consider first the case in which the levy is imposed on operators of landfills. Very often in the Irish context operators of landfills and disposers to landfill will be the same entity, the local authority, but for the purposes of the present discussion we will assume them to be separate. When faced with the levy the operator in the short run can absorb the levy and pay it from existing revenues or attempt to pass on the cost of the levy in the form of higher disposal charges to landfill users. In the longer run, the operator can shift away from the use of landfill and attempt to avoid the levy by providing other means of dealing with the waste such as incineration. The actual outcome will depend on the ease with which each alternative can be pursued. Assuming that not all the levy can be passed on, however, the existence of the levy would encourage the landfill operator to consider alternatives to landfill.

If the levy is imposed on the users of the landfill, alternative actions are open to them also. Like the operators, the users could absorb the charges but are likely to attempt to alter their behaviour in response to the levy. Again like the operators, they could seek alternatives to landfill and so avoid the levy. At present in Ireland, however, the extent of alternatives to landfill is limited so this approach would be a more long-term effect. A more practical alternative for landfill users in the short run would be to pass the cost of the levy onto the consumers and producers from whom they collect the waste. If charging is done on a per-unit basis, the effect of

the levy would then be the same as the effect of per-unit charges and an incentive would be created to generate less waste or to find disposal routes other than collection for landfilling. As has already been discussed, however, with so few being charged on a per-unit basis in Ireland at present, this particular positive effect of the landfill levy would only be generated to a limited extent. With much financing of solid waste services done through general taxation or flat fees, the levy would be passed on in the form of higher taxes or higher flat fees, neither of which results in an incentive to economise on the use of solid waste services.

Depending on where the effect of the landfill levy is felt, incentives may be generated to divert waste to methods of dealing with waste which are not subject to the levy, and to reduce the level of waste generated. But if the hope is that the landfill levy will create incentives at the level of households and firms, it may be simpler to factor in the external costs of landfill into per-unit user charges.

#### 4.5.6 The German "Green Dot" System

In the context of this discussion of incentives in the area of solid waste, a reference to the German "Green Dot" system is appropriate. This system arose out of a German packaging law from 1991. The law gave firms the option of being individually responsible for taking back their packaging waste or participating in a nation-wide collection and recycling system. Since the latter was the only realistic option, a company (DSD) was formed to co-ordinate the nation-wide system. This company imposed packaging levies ("licence fees") on manufacturers, designed to reflect the cost of collecting these products. Other firms were set up to organise the recycling of the collected materials and this was financed either by the packaging manufacturers or by the companies that used the packaging.

The system has certainly increased the amount of packaging waste collected for recycling and there is evidence that the quantity of packaging waste has fallen somewhat. However, the cost has been enormous, bringing the system to the verge of collapse on a number of occasions. The design of the system was flawed in that it failed to consider the processing capacity of the materials collected, and the markets for the final product. The result was that vast quantities of recyclables were collected, some ending up in landfills due to a lack of markets, while the world recyclables market was flooded with material from Germany. This caused a glut and severely affected the viability of recycling in other countries. This problem has abated somewhat recently, due to increased recycling capacity in Germany (and other countries), and general economic recovery leading to increased demand for recyclables. (See Appendix II for a more detailed description of the "Green Dot" system.)

#### 4.5.7 Economic Instruments and the Small and Open Nature of the Irish Economy

Although some reference has already been made to the influence the small and open nature of the Irish economy might have on the effects of the economic instruments discussed, it is worthwhile emphasising some points in this regard.

One of the objectives of instruments such as product taxes, raw material taxes and deposit-refund schemes is to alter product and packaging design in such a way that flows into the waste stream are reduced. There are two broad avenues along which this objective can be met: (a) the tax or regulation can be placed on producers, thereby influencing production and design directly; (b) the tax can be placed on consumers whereby they alter their demand patterns; producers then respond to these demand shifts by producing less waste intensive products.

Difficulties can arise along either route for a small and open economy such as Ireland if it is trying to influence decisions on production and design. Many of the goods sold in Ireland are produced for a global market and Ireland is just a small segment of that market. As decisions regarding production and design are made with a view to the global market, changing circumstances in Ireland may have little impact on decisions. As such, even if producers suddenly face a tax in Ireland related to the packaging component of their product, it may not be worthwhile for them to change the packaging just for the Irish market. Similarly, a shift in the demand patterns of Irish consumers may have little impact on decisions regarding the waste component of a product or its packaging. For these reasons, taxes which have the objective to reduce the waste associated with certain products may have a limited effect. In circumstances such as this, it may be desirable to enact policy at, for example, an EU level. By changing conditions in the global market, it is more likely that objectives regarding product and packaging design be achieved.

Co-ordinated policy on solid waste may be desirable for two other reasons. First, regulations on packaging or taxes which render some forms of packaging uneconomic can be used by countries to create artificial barriers to trade. If domestic producers are already using a particular form of packaging or have access to the form of packaging at cheaper prices than producers elsewhere, the required use of the packaging gives domestic producers a competitive advantage. In this way, a policy that appears to have an environmental objective is really a non-tariff trade barrier. Second, the adoption by a large country such as Germany of waste-related production requirements can have a large impact on production elsewhere because a large proportion of output will be sold in the German market and so their requirements are binding. Similarly, as seen in the experience of the German "Green Dot" system, changed circumstances in the German waste-related market leads to changes in waste-related markets elsewhere (in that case, the German policy gave rise to a flooding of the market for recyclables and a collapse in price). By co-ordinating policy, it is more likely that policies appropriate to a

range of countries can be found rather than the policies of large countries spilling over into the production processes of other countries.

A final point regarding the small and open nature of the Irish economy relates to the issue of cross-border trade. Price differentials on products between Northern Ireland and the Republic of Ireland create an incentive for people to purchase in the cheaper jurisdiction. Taxes or deposit-refund requirements applied in the Republic but not in Northern Ireland may increase the retail price of certain products in the Republic and thus generate cross-border shopping. This has negative impacts on border retailers and reduces the effectiveness of the policies to either reduce waste or to raise revenue.

#### *4.6 Economic Incentives as a Carrot Rather than a Stick*

To complete this discussion of economic incentives in the area of solid waste management, let us address one other issue. In the case of either a landfill or an incinerator, costs are imposed on those who live close to the waste disposal facility. If the benefits to society from such facilities outweigh the costs to society, including the costs specific to the locality, then society at large will be better off from the construction of the facility. Those who suffer, however, are likely to protest and may be able to prevent a socially worthwhile project. What is more, given that the loss per household to the losers will be much greater than the gain per household to the winners, the losers have a much greater incentive to mobilise. The losers will also be geographically concentrated and so organising to resist will be easier. In sum, a project that will benefit society at large can be delayed and even stopped by a minority.

The Pareto principle in economics suggests a way of reconciling the interests of the many and the few. A Pareto improvement is said to occur if some are made better off without anyone being made worse off. A Pareto improvement is possible in principle in the case of, e.g., the landfill construction. As long as the net social benefits are positive, it should be possible to compensate those in the vicinity of the landfill in such a way that they are at least as well-off after the landfill is built as they were before. By collecting tax from the many and giving it to the few in the form of compensation all are made better off and nobody is worse off. Apart from the justness of the proposal, such a compensation scheme has the advantage that it could be designed to provide an incentive to those close to the landfill site to accept the development. The current system of legal wrangling poses additional costs on both sides and is inefficient in that resources are diverted to the legal case which could be used with greater benefit elsewhere.

The question arises, however, as to what the correct level of compensation should be. While it would be desirable that the compensation be sufficient to make the landfill siting acceptable, care would have to be taken not to make the

compensation too generous because of the deadweight loss generated<sup>83</sup>. In order to truly compensate and to avoid windfall gains, it would be necessary to have an estimate in monetary terms of the cost to households of being close to the landfill. Studies have been done to provide just such estimates. One group of studies, using the hedonic pricing approach, estimates the effect of the landfill on the price of houses. The level of compensation could therefore be equal to the estimated fall in house price. The other group of studies, using the contingent valuation method, estimates what people would be willing to pay to keep the landfill away or what they would be willing to accept as compensation. Again estimates from such studies could be used in determining levels of compensation. As an alternative, local groups could be allowed to compete for a landfill/compensation package if a number of sites satisfying environmental requirements were identified. This competition may be able to ensure that compensation requests are reasonable in that an unreasonable request from one group would be underbid by a more reasonable request from another group. In the absence of such competition there would be an incentive for communities to overstate the compensation they require, thus creating the deadweight loss.

There are clearly a number of additional problems with this approach which would need to be addressed before it would be taken. First, it would have to be determined who should be compensated, for example, those within a mile of the landfill, two miles, etc. A political difficulty would emerge over the setting of the boundary with the possibility of compensation being extended to those who are not hurt. Second, the funds required for the compensation might be significant and would require either tax increases or expenditure reductions elsewhere. Both alternatives would clearly give rise to the possibility of the landfill resistance being replaced with another form of resistance.

This compensatory approach to landfill siting has been in operation in Wisconsin since 1981 (Nieves *et al.*, 1992). In that year a law was enacted that mandated negotiations between landfill developers and host communities. Since 1982 all landfill developments and expansions have had the negotiation/arbitration mandate applied and, according to Nieves *et al.*, Wisconsin has been unusually successful in siting waste disposal facilities. The forms of compensation have included: payments to host community governments; direct payments and property value protection to property owners; free disposal fees for host communities.

#### 4.7 *The Role of Government*

We have already made reference to possible roles for the government in various parts of our discussion but we will end this chapter with a general note on this issue. Economic theory tells us that the government should intervene in a

<sup>83</sup> By deadweight loss we mean a payment above and beyond that which would be necessary to bring about acceptance of the landfill siting.

market economy when there are circumstances that lead the private market to produce sub-optimal outcomes. Intervention can take the form of taxing, subsidising or regulating, depending on whether the government wants to promote or constrain certain activities.

Much of our discussion has focused on the desirability of providing an incentive to reduce waste. This can be done at the level of local government through user charges or at the level of central government through a landfill levy, product taxes etc., all of which have been discussed above. Let us now consider briefly if the government should also intervene through subsidising.

Subsidising should be done when it is believed that the private market is providing too little of a good or service. This can happen if there are benefits associated with a service or product that are not captured by the market price, i.e., positive externalities. It is often argued that recycling is one such activity and hence it should be subsidised by the government. From what we have learned in Chapter 3 regarding the costs of recycling, we would make the following comments regarding the subsidising of recycling. For much of the country, the costs of collect systems of recycling are such that they may well exceed any benefits, including the environmental benefits of diversion from landfill. Hence, government subsidising would be uneconomic and inadvisable. In the case of bring systems, however, government subsidising may be justifiable. Indeed, many local authorities are already doing just that and so this form of intervention is already in existence. The important point as always is that the benefit of any subsidy should exceed the cost and the cost should be measured in terms of the alternative uses of the funds. Any money spent on recycling programmes is money diverted from some other programme and so it must be established that the recycling programme is providing the greatest net benefit.

Another possible area of government spending is in the area of information provision. An advertising campaign which informs people of the benefits of waste reduction and suggests ways in which waste can be reduced may provide better value for money than any recycling programme. Such a campaign, especially if combined with user-charging, would increase awareness of the waste problem facing this country and hopefully lead to action on the part of individuals to help deal with the problem.

## Chapter 5

### *CONCLUSIONS*

At the outset of this report we described the nature of the problem which we sought to address. In essence, the problem is one of a waste mountain that has grown in recent years combined with increasing costs of the traditional route, landfill, plus a growing public resistance to landfill siting. We have approached the problem following two broad issues: (i) the relative costs of the various approaches to dealing with the solid waste that is generated and (ii) the lack of economic incentives to reduce the demand for solid waste services through waste reduction or diversion from landfill. From all that has been presented, we now want to distil our principal conclusions.

We will present our conclusions regarding costs first. These are as follows:

1. Landfill costs are set to increase significantly over the coming years; yet it still appears to be the most cost-effective option for disposal of most of the solid waste arising in Ireland. Economies of scale will apply, and this will mean that considerable savings are achievable from building fewer, larger landfills. However this saving needs to be set against the increased collection costs in the particular circumstances.
2. Incineration is significantly more expensive than landfill in most cases. However, if very high landfill prices prevail in the Dublin area and the number of landfills can be reduced in Dublin as a result of building an incinerator, then incineration might become competitive there.
3. Recycling by the kerbside or collect system may be a viable option, where a large enough catchment area is available (perhaps 50,000 households) and both landfill costs and recycling prices are extremely high. This combination of circumstances would probably only prevail in Dublin, although it is difficult to see recyclable prices remaining sufficiently high in the longer run to justify kerbside even in Dublin.
4. Recycling by the bottlebank or bring system appears to be quite viable over a significant part of the country.



5. Composting, both at the household and centralised level, has the potential to divert sizeable quantities of waste, but the economics of centralised composting and of promoting home composting is not clear.
6. Re-use and reduction at source already occur at the industrial and commercial level, and higher landfill prices will increase the incentive for these. Imposing a reuse system at the domestic level appears not to be cost-effective, and caution should be used when considering such a scheme.
7. Finally, the various options will impact on each other, in terms of the amount and suitability of waste going to each. However, in terms of the quantities that are likely to be involved in the Irish context, they are unlikely to affect the economic viability of each other, except in the case of incineration in Dublin, which might be able to replace the building of one landfill if the costs of the latter are very high.

Two other points can be made, relating to the above issues. First, a number of questions relating to the comparison of costs of alternative methods of waste management centre around the use of energy and the external (environmental) costs thereof. This is especially the case in the evaluation of recycling. We have put estimates on these external costs, and adjusted the costs of the alternative methods accordingly; however this is less than perfect, and a number of assumptions have had to be made in the process. Ideally, there should be a tax on energy which would "internalise" these external costs<sup>84</sup>. This would mean that any activity that saved energy would automatically become relatively cheaper, and we would not have to worry about giving recycling (or other activities) a subsidy for energy saved. Specifically, the market would give a very precise message to all involved of the environmental benefits and costs of their energy use.

Second, much emphasis had been placed of late on the achievement of recycling targets for packaging waste. Two notes of caution can be made in relation to this:

- (i) In general, targets such as these should only be set after it has been determined that the benefits of achieving these targets exceeds the costs. It is not clear the degree to which this has been done in this case. Indeed, from an economic point of view, it is better to charge the proper price for the use of all resources, and let the market determine the socially optimal level of recycling (and other activities). This has been the approach taken in this study.
- (ii) The EU Directive on packaging waste (and the Irish government's recycling strategy), while requiring action to promote waste reduction and re-use, only set numerical targets on recycling and recovery. This perhaps reflects the fact that reduction and reuse are difficult to quantify. It could,

<sup>84</sup> Although accurately calculating this tax would be difficult.

however, give a perverse behavioural incentive to those affected by the Directive. Firms, industries and countries might concentrate on achieving the numerical recycling/recovery targets, to the neglect of re-use and reduction, because their performance could be more easily evaluated against the numerical targets. Given the inter-changeability of reusable and non-reusable (but recyclable) packaging, this could conceivably lead to a move away from re-use, which might have a detrimental environmental effect. Specifically in the case of beverage containers, we have seen that there is much use of reusable packaging in the on-licence trade in Ireland. The on-licence trade is, relatively speaking, more important in Ireland than in other EU countries. Therefore, setting targets at EU level for recycling only may not give due credit to Ireland for the amount of "benign" waste management that is already being achieved here, in the area of beverage packaging.

On the issue of incentives our conclusions are as follows:

1. For many users of solid waste services in Ireland there is little or no incentive to economise on their use of the service. Many domestic users in particular face flat fees or no fees, while the fees faced by commercial users are often well below the true cost of the service. This creates an excess demand for the service.
2. There are a number of ways to introduce correct incentives into the system that can lead to diversion of waste from landfill and waste reduction. Those we have discussed are: user charges, product taxes, raw material taxes, deposit-refund schemes, recycling credits and a landfill levy. A strong incentive to discourage littering is also an important element of an overall strategy; the probability of getting caught and the penalties must be sufficiently high to make this an unattractive disposal route.
3. It appears that the simplest and most direct way of introducing correct incentives is through user charges, an approach that has already been adopted in certain parts of the country. Such an approach avoids many of the information requirements of some of the taxing schemes. It also avoids some of the distortionary effects of the taxing schemes, such as effects on trade. Perhaps the greatest advantage of user charges is that they can be set and administered by those responsible for solid waste management, the local authorities.
4. User charges, if they are to generate a more efficient approach to solid waste, must be per-unit charges and should properly reflect the long-run marginal cost of disposal. Only in this way do households and firms face the full cost of their use of the service.

5. If disposal to landfill is charged for, households and firms should also face correct relative prices for the other methods of disposing of their solid waste. For example, if collection for recycling is more expensive, even allowing for environmental benefits, people should pay more for it. Otherwise, a strong incentive to recycle exists even though recycling may not be optimal from a national perspective.
6. Economic incentives can also be used in solid waste management in an effort to overcome the NIMBY (not in my back yard) syndrome. By offering compensation to those negatively affected by the siting of landfills, it should be easier and hence less costly to find new landfill sites. It also seems just that the minority, who suffer through a landfill siting that benefits the majority, should be compensated for their loss. This compensation could take the form of direct payments to residents and property owners or payments to local governments. Offers of such compensation would have to be made so as to ensure that no major gains would be made by those involved but that the compensation merely restored pre-landfill wealth levels.

A general theme that has emerged throughout our analysis is the paucity of data in this area and the resulting difficulties in drawing strong conclusions on many of the issues we have raised. One way in which this paucity could be alleviated would be through pilot projects in carefully selected areas. By analysing waste disposal habits in certain communities before and after the introduction of, for example, user charges in a manner similar to the study of Fullerton and Kinnaman, it would be possible to get a greater insight into the effects of innovations in solid waste management.

Whatever route is chosen in the coming years regarding solid waste management, one thing is clear: with changing public attitudes towards the environment, backed up by domestic legislation and the EU Directive on landfills, major changes will take place in Ireland in the near future in the area of solid waste management. The hope of this report is that the changes made will be economically and environmentally rational, in the sense of giving due consideration to the costs and benefits of alternative approaches and providing the correct economic incentives for their use.

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## Appendix I

### ESTIMATION OF VARIABLE EXTERNAL COSTS

Table A.1.1: *Estimation of Variable External Costs from Disposal of Solid Waste to Landfill and Incineration in the UK*

<i>Externality type</i>	<i>Landfill</i>				<i>Incineration</i>	
	<i>stg£ per tonne</i>				<i>stg£ per tonne</i>	
	<i>Urban, No Energy Recovery</i>	<i>Urban, With Energy Recovery</i>	<i>Rural, No Energy Recovery</i>	<i>Rural, With Energy Recovery</i>	<i>Urban, With Energy Recovery</i>	<i>Regional, With Energy Recovery</i>
Global pollution – CO <sub>2</sub>	0.32	0.46	0.32	0.46	2.55	2.55
Global pollution – CH <sub>4</sub>	2.36	1.36	2.36	1.36	0	0
Air pollution	0	0	0	0	2.01	1.14
Transport pollution	0.1	0.1	0.46	0.46	0.26	0.42
Transport accidents	0.23	0.23	0.55	0.55	0.2	0.33
Leachate	0.45	0	0.45	0	0	0
<i>less</i>						
Pollution displacement	0	1.12	0	1.12	9.40	9.40
<b>Total</b>	<b>3.45</b>	<b>1.03</b>	<b>4.14</b>	<b>1.72</b>	<b>-4.38</b>	<b>-4.96</b>

*Source:* CSERGE *et al.* (1993)

*Notes:*

1. The above calculations exclude disamenity costs, and in the case of incinerators they exclude the “air toxics” – heavy metals, dioxins and other organic compounds. The damage from these is unvalued, “with the balance of probability being that such a value would be close to zero or zero”. This is a controversial assertion; there is much concern and debate about the damage caused by dioxins (see, for example, Greenpeace, 1994).
2. Pollution displacement benefits are calculated by reference to existing power stations. Wallis and Watson (1994) recalculate the value by reference to new generation coal and gas-fired power stations and estimate that the pollution displacement becomes negative (i.e., becomes a cost as opposed to a benefit). Our estimations in the body of the text, using the CSERGE numbers above, indicate incineration is more expensive than landfill. If the Wallis and Watson values were used, incineration would appear even more expensive.
3. Global pollution from CO<sub>2</sub> appears to be overstated in the above, since it takes into account all CO<sub>2</sub> emitted from landfills and incinerators. This is not correct, since all of the carbon in the biodegradable element of solid waste was already removed from the atmosphere when the materials (food, wood, etc.) were grown. These materials are almost all grown specifically for the purpose for which they were used, and are generally replaced by a new crop. Products made from fossil fuels (mainly plastics) are the only significant source of CO<sub>2</sub> from landfills and incinerators that contribute to global warming in net terms.

## Appendix II

### *THE GERMAN PACKAGING ORDINANCE*

#### *1. Legal Background and History*

The German Packaging Ordinance, which spawned the Grüne Punkt<sup>85</sup>/DSD system, was passed in 1991. This laid down the most comprehensive legislation of its kind ever to be implemented, the stated objectives of which were that:

- (1) packaging be manufactured from environmentally compatible materials, which could be reused or recycled, and that packaging be reduced, and
- (2) packaging waste should be avoided by reducing it to the minimum necessary, and by refilling it, or where this is not feasible by reusing or recycling it.

Under the legislation, those who "bring into circulation" packaging or packaged goods are obliged to take back all the packaging that they use on their products. The only way to avoid this individual obligation is to become part of a nation-wide recycling system. As such, the law establishes "producer responsibility" for dealing with packaging throughout its life-cycle.

The law introduced quotas for the percentage of various packaging types which must be recycled, which will increase over the coming years. In addition, there are quotas on the use of refillable beverage containers, which must be achieved if a mandatory deposit-and-return system is not to be imposed (see Table A2.2)<sup>86</sup>. Although this is a Federal law, its operation is in the hands of the individual Federal States (Länder), so exact conditions may differ somewhat from state to state.

<sup>85</sup> The Grüne Punkt (Green Dot) is the symbol used on all packaging which is part of the recycling system described here.

<sup>86</sup> As an historical note, when the government was developing the legislation it also considered a packaging tax approach. The tax system would have directed the revenue raised to the local authorities, who would have then been responsible for both recycling and disposal. However, in consultation with industry, the government decided that the packaging tax would be "too economically disruptive", and therefore chose the current system. The reasons for industry's opposition to the packaging tax is not clear, since the DSD system that was developed is in essence quite similar; as we shall see. However, it may be that they felt they could do a more efficient job than the local authorities in organising recycling, or that they preferred to keep the revenues raised and the recycling infrastructure in private sector hands. Whatever the reason, it appears that the private sector, as in other instances, preferred a more regulation-based approach over a tax-based system.

## 2. *How the System Works*

As stated, the law created individual liability for packaging, unless firms participated in a nation-wide recycling system. Naturally, firms decided to opt for the latter. *Duales System Deutschland GmbH (DSD)* was set up in September 1990 in response to the ordinance (operations commenced in 1991). It is a non-profit making company, whose function is to set up and operate a national packaging recycling system for Germany. Its shareholders include packaging manufacturers, manufacturers of packaged goods (called "fillers", because they fill the packaging), wholesalers and retailers, and recyclers. The system works in two stages:

### (1) *Collecting and Sorting* –

This operates side-by-side with the local authority rubbish collection system (hence "dual"). Consumers keep their packaging waste separate, bringing paper and glass to local paper- and bottlebanks, and putting the rest into a sack or bin for kerbside collection<sup>87</sup>. Regional waste disposal firms, either private or public sector, collect and then sort the waste. Any non-recyclables collected are returned to the local authorities for disposal.

### 2) *Recycling* –

The sorted recyclables then become the responsibility of the "guarantors". These are companies in the packaging manufacturing industries which guarantee that the various collected materials will be recycled<sup>88</sup>. The guarantors either recycle the materials themselves, or organise for it to be done by others. The recycled product is the property of the companies that carry out the recycling, and they are free to do with it as they wish.

In addition to these arrangements, retail participants in the DSD system undertake – "after some time for adjustment – to accept for sale only products labelled with the 'green dot'" (Klepper and Michaelis, 1993).

The system started operation in 1991, and since then has gradually spread over the entire country. By the end of 1992, 72 million of the population had been

<sup>87</sup> There is no sanction if the consumer puts the wrong items into the bag. This contrasts with the requirement in Austria; however, enforcement of this appears to have caused problems in that country. This is the extent of consumers' involvement in the system.

<sup>88</sup> At the outset of the system it was realised that sufficient recycling capacity did not exist for the quantities of materials involved, so new recycling companies were established by the various packaging manufacturing industries. These companies became the "global guarantors" for each industry (glass, plastics, steel, etc.). The global guarantor for paper was set up by DSD itself. They now dominate the recycling industry in Germany, leading to complaints of monopolisation. This is understandable, since all consumer packaging waste is in effect being processed by these companies.

covered, out of a total of 80 million. In effect 1993 was the first full year of its operation.

### 3. *How the System is Funded*

The two stages are also funded separately. Collecting and sorting are funded as follows: the fillers who join the system (they do not have to become shareholders of DSD) must conclude a contract with and pay a "licence fee" to DSD, which is actually a charge for the amount of packaging they use each year. In the case of imported goods, the importing agent is obliged to take back the packaging, so any foreign company wishing to sell consumer products on the German market must also become part of the system. The fees are the only income of DSD, and cover collection and sorting elements of the process (they also cover part of the cost of recycling plastics). The fees are divided into a weight-related element (see Table A2.1) and a volume- or area-related element, ranging from DM0.10 to DM1.20, depending on size. Over time, the fees have been increased considerably and have become more differentiated to reflect the costs of recycling the various materials. Prior to the fee regime, the fee was purely weight-based, and was approximately equal to the weight-related element in the current fees.

Table A.2.1: *DSD Licence Fees - Weight-related Element*

	<i>DEM per Kg of Packaging Used</i>
Plastics	2.95
Beverage cartons	1.69
Other composites	2.1
Aluminium	1.5
Tinplate	0.56
Paper	0.4
Glass	0.15
Natural materials	0.2

*Source:* DSD

*Note:* These rates apply from October 1994.

As well as paying the DSD licence fees, fillers must also have an Acceptance and Recycling Guarantee, from the above-mentioned guarantors, to ensure the recycling of their packaging. The funding of recycling is quite complex, and depends on materials involved. Materials fall into three categories (Klepper and Michaelis, 1993):

- (i) Those with an established market, whose price makes recycling viable without subvention; here the collected material feeds into the normal recycling structure that existed prior to the establishment of the present system. An example is glass.
- (ii) Those materials which require further subvention to make recycling viable. This subvention is paid for by the *packaging manufacturers*, who set up the *guarantor companies*.
- (iii) Plastics; here the *fillers* must negotiate a contract with the guarantor to pay for recycling the product. The reprocessing fees for plastics ranges from 10 pfennigs per unit of volume below 0.05 litres to DM5.00 per unit of volume over 30 litres. In addition, it appears that some of the DSD fees collected on plastics are used to fund plastics recycling.

In the first two cases "global guarantees" are given by the guarantors to recycle all packaging materials of the type in question. There is no individual contract between the filler and the guarantor, and the filler only has to pay its licence fees to DSD. As stated, with plastics the filler must negotiate a contract with the guarantor.

Once the filler has paid its licence fee to DSD for collection and sorting, and recycling is covered by a global or individual guarantee, it can use the Grüner Punkt (Green Dot) on its packaging.

#### 4. *The Effects of the System on Packaging Levels and Recycling*

Little research has been done on the effect of the system. DSD statistics show that they exceeded their recycling quotas in 1994, as seen in Table A2.2. Significantly higher targets will come into play from July 1995, although only *composites and aluminium*<sup>89</sup> appear as if they may have problems achieving these.

To determine the effect of the system we must know the level of recycling that existed before the DSD system was installed. This is problematic, since prior to DSD there was only separate collection of paper and bottles. In particular there were no statistics for plastics (according to Ministry of the Environment personnel). Klepper and Michaelis (1993) give estimates of the level of recycling of primary and secondary packaging waste in West Germany in 1988. These data are less than ideal, because they do not differentiate between domestic and

<sup>89</sup> Aluminium has been a problematic material for DSD – it was the only material that did not reach the 1993 quota in that year, achieving only 6.8 per cent recycling. This may appear strange, given the material's high value. From discussions with DSD personnel, the main reason for this is that the major use for aluminium in packaging in Germany is as the top of steel beverage cans, and separation is technically difficult in this form.



commercial waste<sup>90</sup>. However, they give some indication of the autonomous level of recycling that did occur.

Table A.2.2: *Recycling Quotas and Percentages Achieved*

	<i>Quota from 1/1/93</i>	<i>Quota from 1/7/95</i>	<i>Achieved in 1994</i>
Glass	42	72	71
Paper and Cardboard	18	64	71
Tinplate	26	72	56
Aluminium	18	72	32
Plastics	9	64	52
Composites	6	64	39
Refillable packaging	72		77.2

*Source:* Ministry of the Environment, Bonn

*Note:* The rate achieved for refillable packaging is for 1993.

Table A.2.3: *Primary and Secondary Packaging Recycling in West Germany in 1988*

<i>Material</i>	<i>Percentage Recycling</i>
Glass	38
Tinplate	39
Aluminium	3
Paper/cardboard	6
Plastic	2

*Source:* Klepper and Michaelis, 1993.

As can be seen, the level of recycling varied widely, and seemed to mirror the value of the recyclables and the technical difficulty of recycling (aluminium's use in combination with steel made separation and recycling difficult). Tinplate is a material which has traditionally been recycled in Germany, which explains the high level of recycling of this material. This is what one would expect in an unregulated system. Comparing these recycling rates with Table A2.2, it can be seen that major increases in recycling have been achieved across all materials. The biggest increases have occurred in those materials which previously were largely

<sup>90</sup> In addition, Klepper and Michaelis (1993) indicate that the full economic costs of waste disposal are not being charged to customers at the moment, because external costs and "scarcity rents" (to reflect the scarcity of landfill) are not being charged for. Presumably they were not being charged in 1988 either, so the figures understate the amount of recycling which might occur if full costs were charged.

unrecycled, and the average recycling levels are now much more even across materials.

Data are also available for the total amount of packaging used each year. The aggregate quantities used in Germany has fallen since the introduction of the system in 1991 – see Table A2.4.

Table A.2.4: *Aggregate Amounts of Packaging Materials Used in Germany*

<i>Year</i>	<i>Packaging Used millions of tonnes</i>	<i>Percentage Change on Previous Year</i>
1991	12.79	
1992	12.28	- 3.9%
1993	11.77	- 4.2%

Source: DSD.

The aggregate of packaging fell by over 1 million tonnes over the time period covered. This fall is more or less evenly spread over the various types of packaging, with the exception of composite/beverage cartons, the use of which remained constant. DSD estimates that the reduction is the equivalent of 15kg per head of population; current per capita consumption is 145kg, indicating a reduction of approximately 10 per cent. It is interesting that the reduction in 1992 over 1991 was considerable, even though the system was only in its early stages, and was not applied nation-wide. Was this because firms were preparing well in advance for the new regime, or because the level of packaging was being reduced anyway? It would be interesting to know the level of packaging used in prior years, to see if the new system had any net effect on the trend. Apart from the aggregate quantities reported above, DSD (1992b) quotes data from a survey that indicate that between 1990 and 1992 there was a move from plastics to glass and paper, and within plastics from PVC to the more easily recyclable PE and PP.<sup>91</sup>

While the collection side of the system has developed very quickly, inadequate capacity for processing recyclables in Germany has been a problem – especially for paper and plastics. In brief, the ordinance set very high targets to be achieved in a short period of time. It was possible to set up the collection infrastructure within the specified timeframe, but not the reprocessing capacity. As a result, a

<sup>91</sup> This survey also showed that the Packaging Ordinance was the most quoted reason for changing packaging types (56 per cent of firms). Forty-nine per cent quoted general environmental awareness, and 42 per cent quoted requirements imposed by the retail trade (DSD, 1992b).

significant proportion of the collected recyclables have had to be exported<sup>92</sup>. This has caused the much publicised problems of reduced prices for these materials on the European market, which has damaged the viability of recycling activities in the countries concerned. Other countries have complained that these exported recyclables are subsidised, as the collection and sorting is paid for by the producers (via licence fees), so the guarantors get these materials collected and sorted for nothing. This, and the quantities of materials involved, give the German guarantors an obvious advantage over other European recyclables dealers.

#### 5. *Economic Analysis of the System*

In economic terms, the system imposes a two-part recycling tax on manufacturing industry. There is an explicit collection and sorting tax on the fillers, which goes to one company (DSD), who sub-contracts the sorting and collecting of packaging waste. This tax is based on the material, weight and volume of the packaging. DSD is in a sense a revenue-collecting bureaucracy. However, its financial accounts are published each year and its activities are subject to much scrutiny, so there is a strong incentive on it to minimise its costs. There is also an implicit recycling tax on packaging manufacturers who pay for the recycling of all materials except plastics. With plastics there is an explicit recycling tax on the *fillers*, based on the weight of the material. These recycling taxes are paid to recycling guarantors who are increasingly monopolising the recycling industry. These issues will be expanded on further in this section.

We could find no study which carried out a comprehensive cost-benefit analysis of the system. Hence, the following analysis is largely qualitative. However, by highlighting the incentive effects and some of the efficiency issues it does point out some of the factors which affect the net costs and benefits of the system. First, we consider the financial costs and benefits of operating the system.

##### (a) Finances, Costs and Benefits

The financial cost of running DSD GmbH was DM3.4 billion in 1994 (2.1 billion in 1993, the first full year of its implementation<sup>93</sup>). This is equal to the total

<sup>92</sup> Much of the plastic was exported (in the early stages at least) unsorted to South East Asia, especially China. There they are sorted and re-processed into low quality plastic products. *The Economist* (1993) points out that this is having a detrimental effect on Third World "scavengers", by reducing the price of recyclables in these countries.

<sup>93</sup> DSD suffered a financial crisis in Autumn 1993, as a number of fillers failed to pay their full licence fees, while continuing to use the Grüne Punkt. Effectively they were understating the amounts of packaging they used. Because of the complexity of the waste packaging stream, and the fact that packaging on products for export and transport and industrial packaging are not covered by DSD, it was very difficult to determine which companies were under-paying their licence fees.

licence fees paid by fillers for the collection and sorting of packaging waste – the costs of actual recycling are unknown. This represents DM40 (£17) per head per annum, or DM0.029 per unit of packaging (on the basis that approximately 110 billion units of packaging are used each year in Germany, [DSD, 1992b]). In 1992 DSD estimated that the sorting and collecting systems will require a capital investment of DM7 billion by 1995<sup>94</sup> (DSD, 1992a). Advertising and publicity campaigns are also a large element of the system's costs.

Plastics recycling appears to have been a major contributor to the high cost of the system. The material is especially problematic, because of the many varieties thereof, the lack of a "traditional" plastics recycling industry, and the cheapness of the primary raw material – oil. In effect plastics recycling is not economical – in 1993 primary material cost DM0.80 - 1.60 per kg, while secondary material cost DM3.00 per kg. As a result, in 1993 the recycling of plastics had to be subsidised by DM600-800/tonne (£255-340/tonne). According to Ministry of Environment personnel, this is the key financial problem in the whole system<sup>95</sup>.

Other costs and benefits do arise from the system, that are not included in the purely financial calculations. These externalities include consumer and producer inconvenience, although consumers may not regard it as a cost – they may get satisfaction from a sense of doing something positive for the environment. A benefit not included above is the reduction in municipal waste which has been achieved – this in turn reduces the amount of landfill and incineration which must be undertaken. As stated, a comprehensive study of the cost and benefits does not yet appear to have been carried out.

#### (b) Incentives

As can be seen, the system very much enshrines Producer Responsibility – the cost of the system falls on the packaging producers and fillers. Retailers must simply stock "green dot" products, and consumers are asked to separate out their packaging waste for collection or delivery to bottlebanks and paperbanks, though

<sup>94</sup> It also estimated that the system would have operating costs of DM2 billion per annum, which appears to have been an under-estimate.

<sup>95</sup> New legal measures are proposed to deal with this problem. The new law distinguishes between two approaches to recycling plastics:

- (a) granulation, for use in new plastic products;
- (b) hydration, to convert it back to oil.

The latter is considered less desirable, as it uses more energy, and reduces the material to a lower chemical level. However, it is cheaper, requiring a subsidy of only DM250 per tonne. The law will require plastics to be recycled 30 per cent each (as a minimum) by granulation and hydration, with the balance dealt with by incineration with energy recovery.

there is no sanction if they fail to do so. The incentive effects on the various players in the system can be analysed as follows<sup>96</sup>:

- (1) The requirement on retailers to take back and sort packaging (in default of a "DSD-type" system) puts a large incentive on them to promote the DSD among their suppliers. They can do this by selling only those products which carry the "green dot", thus forcing their suppliers (domestic and foreign) to join the system. Because the retail sector in Germany is dominated by a small number of companies, it is easier for them to do this.
- (2) Fillers must pay licence fees to DSD for collection of recyclables, and directly or indirectly must pay for their recycling. Hence there is a strong incentive on this group to use less packaging, and to move to those forms of packaging which are easier to reprocess.
- (3) Packaging manufacturers have an incentive to develop more recyclable materials, as the imposition of recycling costs is initially on them (with the exception of plastics). However, to the degree that they can pass the costs on to their customers, this incentive is blunted.
- (4) Recyclers/Guarantors have little incentive to minimise their costs and to develop new ways of using recycled waste. This is because collected packaging waste must be recycled, and packaging producers and fillers must pay for it<sup>97</sup>. Also the method of global guarantees has favoured a small number of large companies, squeezing out "traditional" recyclers who existed before the "green dot" system was set up. Thus the system is encouraging monopolisation within each material type, with possibly higher costs, as well as reducing the incentive to innovate. Klepper and Michaelis (1993) conclude that only competition *between* packaging materials (as opposed to within material types) is being encouraged. They point out that innovation in the packaging manufacturing industry was one of the main hoped-for results of the system.
- (5) The incentive effect on consumers is less direct than on the other players. At one level, products with less packaging or more recyclable packaging should enjoy a competitive advantage because they should cost less, but the effect of this may be very small if the proportion of overall costs represented by packaging is small. Apart from this, there is no direct incentive on consumers to reduce their level of packaging waste. They do not pay directly for the

<sup>96</sup> This is drawn mainly from Klepper and Michaelis (1993).

<sup>97</sup> To the degree that the recycling companies are owned by the packaging manufacturers, there is an incentive to control costs, as the latter pay for the recycling; but as already stated, if the recycling costs can be passed on this effect is lost. In the case of plastics, where the fillers pay for the recycling, even this incentive effect is not present.

disposal of this waste, as it is taken away for free<sup>98</sup>. This can give an incentive to separate packaging waste from other waste, but only if general waste charges are weight-based (and the incentive will only be strong enough if full costs are charged). In fact the level of consumer participation in the scheme is very high, and Klepper and Michaelis surmise that this is more due to "green consciousness" than to any economic effect. However, this does not necessarily imply that consumers are taking action to reduce the amount of packaging waste they generate.

(c) Efficiency

A number of features of the system affect its economic efficiency, some of which were alluded to in the previous section on incentives:

1. The Packaging Ordinance imposes high recycling and re-use targets on industry, with no apparent analysis of whether these levels are efficient or not. The critique of similar legislation in Austria by Pöll and Schneider (1993), referred to in Chapter 3, points to some of the problems with this approach.
2. A major problem is that the system applies a levy on packaging, equal to the cost of recycling it. This is problematic, from a number of points of view:
  - (i) It is economically inefficient – what should be levied is the cost of disposing of the material to landfill, including all the relevant external costs. This would encourage recycling up to the point where it uses more resources than (and is therefore less environmentally sound than) landfilling. By levying a charge equal to the cost of recycling, it funds recycling regardless of whether it is the most efficient option.
  - (ii) There is no incentive on the recycler to minimise costs, as its costs are guaranteed by the system. This would incline the system to be more expensive than it need be.
3. The costs of complying with the Packaging Ordinance or joining the DSD system may act as a barrier to trade, and a barrier to entry to the packaging industry. In addition, the organisation of DSD appears to be leading to a monopolisation of the recycling industry<sup>99</sup>. Both these processes will damage the efficiency of the economy.

<sup>98</sup> Glass and paper must be brought to bottle and paper banks, while other packaging is collected from the kerbside. This might conceivably give an incentive to the householder to use plastics instead of paper and glass.

<sup>99</sup> The Environmental Council in Germany estimates the "four or five companies are likely to share some 40 per cent of the domestic waste market in coming years". Gerelli (1994) reports that the German anti-trust authorities have initiated measures to prevent DSD from moving into the areas of commercial and industrial packaging, as they fear the monopolisation of the recycling industry. A proposed change in the law will require the guarantors to use public tendering for the recycling contracts. This may address

4. Finally, the under-pricing of waste disposal (through not charging externalities) and of energy distort the situation. These are outside the control of the system, but not of the policy-maker.

As can be seen, there are a number of features of the system that would lead one to suspect that it is not economically efficient. A comprehensive economic analysis of the system is needed to answer the question of whether the benefits outweigh the costs.

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some of the monopolisation problems with the system.

A counterpoint to the monopoly argument is that there will continue to be competition between the various materials. However, if plastics suffer a major cost disadvantage *vis-à-vis* other materials, the producers of these other materials may be able to increase their prices somewhat in tandem. This is especially possible if there are few substitutes available (e.g., in the case of beverage containers).

## Appendix III

### *EXTERNAL COSTS OF ENERGY USE*

A number of attempts have been made to estimate the external costs of energy use, mainly with respect to electricity generation. Most studies are incomplete, in so far as they evaluate only some of the costs involved; examples are CEPN *et al.* (1994) and CSERGE (1992). These two studies consider the situation in specific countries; the CEPN study deals with power stations in Germany and the UK, while the CSERGE study considers the situation in a UK context. There are dangers in transferring results from one country to another; Newbery (1990) and McCoy (1991) in two studies of the costs of acid rain, demonstrate the wide differences between different countries in terms of the damage done by emissions. Most studies differentiate between the local external costs, from SO<sub>2</sub>, NOX, etc., and the global costs of global warming, ozone depletion, etc., caused by factors such as CO<sub>2</sub>. However, since there are no studies which try to estimate the external costs of energy use in Ireland, we are forced to use overseas values. Given that much of the pollution from electricity generation in Ireland falls in the Irish sea and the North sea, one would expect that external costs in Ireland would be somewhat lower than in other countries. However, this may be compensated for by the fact that many costs are not evaluated in the studies considered.

Lawlor (1995) attempts to estimate rudimentary external costs from the CEPN study, and comes up with an average cost of 1.3p per kWh of electricity generated<sup>100</sup>. A similar exercise on the CSERGE study would come up with a higher value, but as the CEPN study is more recent, and there is reason to believe that the costs in Ireland would be lower than in other countries, we have gone with the CEPN estimates. Making the following important assumptions: (i) that electricity is generated at an average 33 per cent efficiency; (ii) that the external costs from using these fuels is the same whether burned in a power station or in manufacturing industry; this translates into an external cost of 0.43p per kWh at end use, or £49 per tonne of oil equivalent (TOE). Short of specifically calculating

<sup>100</sup> Energy quantities are often measured in kWhs, as a means of comparison between different fuel types. However, when using this measure and comparing electricity with other fuel types one must differentiate between kWhs at generation and at point of use. This distinction is needed because a significant amount of energy is lost in generating electricity from other fuels. Hence, a tonne of oil converted to electricity and used in a manufacturing process will do less work than the same tonne of oil used to fuel the manufacturing process directly. The former case is described as "at generation" whereas the latter is described as "at point of use". A tonne of oil will generate 3,773kWh of electricity, whereas the actual energy in the oil is equivalent to 11,632kWh (Scott, 1992).



a set of external costs for Ireland, which is beyond the scope of this study and would in any case be incomplete, this appears to be the best estimate achievable. However, we emphasise that due to the many assumptions made the value is mainly illustrative.

## GLOSSARY OF TERMS

<i>Aerobic decomposition/digestion</i>	Biodegradation of organic materials in the presence of oxygen. By-products of this are carbon dioxide and water. Composting is a form of this.
<i>Anaerobic decomposition/digestion</i>	Biodegradation of organic materials in the absence of oxygen. By-products of this are methane and carbon dioxide.
<i>Composites</i>	Liquid containers made from a combination of materials – paper, cardboard, plastic and aluminium. For example, beverage cartons are usually 70 per cent paper, 25 per cent plastic and 5 per cent aluminium. Their main advantages are that they are air- and light-proof, and are extremely light in comparison with alternative packaging.
<i>Cullet</i>	Broken glass of uniform colour, in a form suitable for recycling.
<i>Dioxins</i>	Highly toxic chlorine-based chemicals formed as a by-product of many industrial and other processes involving chlorine, including incineration.
<i>External costs/benefits</i>	Those costs and benefits which accrue to the production or use of a good or service, but which are not reflected in the market price thereof (e.g., the costs of pollution).
<i>Externalities</i>	See external costs/benefits.
<i>Ground water</i>	Water from underground sources – wells, aquifers, water tables, etc.
<i>Landfill (sanitary)</i>	"A method of disposing of refuse on land without creating nuisance or hazards to public health or safety, by utilizing the principles of engineering to confine the refuse to the smallest areas, to reduce it to the smallest practical volume, and to cover it with layers of earth at the conclusion of each day's operation or at such more frequent intervals as may be necessary." (American Society of Civil Engineers, 1959, cited in Rouhani and Kangari, 1990).
<i>MRF (Materials Recovery Facility)</i>	A facility or depot where collected MSW or recyclables are brought for sorting, baling, etc., prior to further processing (for example before transfer to a recycling plant).
<i>Mechanical recycling</i>	See recycling.

<i>MSW (Municipal Solid Waste)</i>	Household and commercial solid waste.
<i>PET (Polyethylene Terephthalate)</i>	The plastic mainly used in large mineral water and soft drinks bottles.
<i>Primary packaging</i>	Packaging used by private and commercial consumers to transport and protect the produce until the time of consumption.
<i>Primary (raw) materials</i>	Materials used in a manufacturing process for the first time.
<i>Recycling</i>	Reprocessing in a production process of waste materials for the original purpose or for other purposes, excluding energy recovery.
<i>Secondary (raw) materials</i>	Materials which have previously been used in a manufacturing process, and have been prepared for another use in such a process.
<i>Secondary packaging</i>	Packaging used in addition to primary packaging to protect the product against theft or to apply advertising, and which can be removed without reducing the possibility to transport and protect the product.
<i>Thermal recycling</i>	Incineration with energy recovery.
<i>Transport packaging</i>	Packaging used exclusively to protect the product from the producer to the sales outlet.
<i>Trippage</i>	The number of times a reusable packaging is actually used.
<i>Virgin (raw) materials</i>	See primary raw materials.

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