An Economic Approach to Municipal Waste Management Policy in Ireland

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Final Report for Dublin City Council

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TABLE OF CONTENTS

Executive Summary

Ireland is at an important junction in refining and implementing its municipal waste management policies. While significant progress has been made in recent years in encouraging the use of recycling as an alternative to landfill, the State has to meet legally binding targets that will become increasingly challenging from 2010 onwards.

Significant policy decisions are under consideration, both in an effort to meet the targets and in response to political preferences concerning waste management options. In this report we aim to contribute to the policy debate by setting out a roadmap for economically efficient waste policies. We then apply this roadmap to assess specific options.

The objective of public policy using the economic approach is to maximise societal welfare. Competitive markets serve this purpose. However, markets can fail in various ways. In the waste management sector, the main market failures are externalities (costs created by an activity that do not fall on the person or firm carrying out the activity) and market power (the ability of a firm to raise prices above marginal cost in a sustained way). Government intervention may be required to address these externalities.

There are alternative approaches to setting public policy in the waste management sector, in particular through various rules of thumb. In the report, we discuss how several such rules – the waste management hierarchy, the polluter pays principle and the proximity principle – relate to economically efficient policymaking. Each rule of thumb discussed has some degree of consistency with an economically optimal approach, but each also omits important dimensions of the policy choice. Ultimately, such rules of thumb are better used as simple devices for communicating policy priorities to a wide audience than as a firm guide for policy implementation.

We classify the main types of instruments that governments may use to address market power as competition policy and market design. Similarly, there are a range of interventions that may be used to address externalities: taxes and subsidies; assignment of property rights; command and control measures; and provision of information. The instrument chosen in each case should be the one that minimises the net cost to society. Cost-benefit analysis, which in a regulatory context is normally carried out in the context of a Regulatory Impact Analysis, is the appropriate tool for ensuring that significant government interventions are justified and make the greatest possible contribution to societal welfare.

Another approach to assessing the effects of regulation in a specifically environmental context, Strategic Environmental Assessment, omits full consideration of costs. Using it in isolation could lead to unnecessarily costly policies being adopted when less costly policies to achieve the same benefits were available. This form of assessment is thus not sufficient for assessing the relative or absolute effects of options for regulating waste management on societal welfare.

Having set out the options for applying and evaluating government intervention, we consider the characteristics of the municipal waste management sector. The sector exhibits vertical relationships and complementarities across activities; for example, the choice of collection process (three bins vs. two bins) affects the economics of processing waste after collection. This implies that choices about regulation need to take into account costs and benefits across the whole end-to-end system rather than on individual activities in isolation.

There are important economies of scale (particularly in processing and disposal) and economies of scope and density (particularly in collection) in the sector. In addition, some waste management activities, particularly in the processing and disposal/incineration segments, also involve a high proportion of fixed or sunk[1] costs. Some facilities exhibit asset specificity, whereby once a facility has been built it is only economic for it to supply one or a few customers.

Most markets for waste management services are local or regional in scope, as evidenced by high economies of scale, scope and density, together with a relatively high ratio of transport costs to value of waste material. Since there are wide variations in local conditions such as population density, policy measures need to be calibrated to local or regional market circumstances, which may not necessarily coincide with existing administrative boundaries.

Particularly given the need for further substantial infrastructure investment, it is vital to limit policy/regulatory risk and to ensure that such risks, along with demand risk, are borne by the market participants best able to manage them. The high share of fixed capital costs in total costs for many activities in the sector implies that cost of capital will be an important driver of costs for the system as a whole. Cost of capital, in turn, tends to be sensitive to the presence of policy risk: project developers will tend to hold off investing or seek higher returns if there is a possibility that post-investment policy actions will change the economics of the market. This problem, together with that of asset specificity, may be addressed by contracts that minimise risk and allocate it appropriately.

Based on the economics of efficient government intervention and the characteristics of the municipal waste management sector, we set out the following roadmap for development of appropriate policies:

1. There are market failures in waste management, so government intervention is merited and should be directed at addressing them.

2. Since economic markets for waste services and most market failures in the sector are regional or local in their incidence, policymaking should allow for local variations. It seems unlikely that applying the same collection and processing arrangements across all of Europe, or even the whole area of a given country, would be efficient.

3. The main source of market failure in this sector is externalities. Policy should, where possible, use economic instruments to address externalities. In particular, levies and taxes are to be preferred to command and control policies because they allow the price and cost of externalities to be equated and they can readily be adjusted over time as market conditions change, which facilitates higher efficiency. Command and control rules are prone to set implicit prices on externalities that are far out of line with their true cost, particularly when they are applied across a broad range of areas with different cost conditions. Furthermore, command and control mechanisms are likely to inhibit technical change.

4. The few externalities from waste management that have significant effects beyond the regional level, in particular emissions of greenhouse gases such as methane and CO_2, are best addressed using instruments that ensure consistency across sectors and countries rather than sector-specific measures. For example, the

[1] Not recoverable if a firm or facility exits the market.

European Union ("EU") Emissions Trading System ("ETS") controls CO_2 emissions across Europe. Such measures are to be preferred to sector-specific command and control measures.

5. A second source of market failure is the potential for market power. Along with some externalities (such as road congestion) which are difficult to price, it may be proportional to address market power problems by reserving exclusive rights for waste authorities and encouraging them to keep costs low by permitting competition *for* the market, rather than *in* the market. Thus further use of franchising and contracting-out of services should be considered.
6. Regional administrations setting policy for waste collection and processing in their areas should take into account vertical complementarities in waste management processes. This can be done by comparing the wider societal costs and benefits of alternative end-to-end waste management systems, e.g. with different mixtures of source segregation and bulk processing, and selecting the set of options with the highest net benefits.
7. Given the prevalence of asset specificity, long-lived infrastructures and sunk costs in waste processing, all levels of government should aim to avoid creating unnecessary policy and regulatory risk. Regional administrations should consider using contractual arrangements that situate regulatory and demand risks with the parties that can most efficiently bear them and should be free to avoid hold-up problems by credibly committing to contractual conditions in advance.

We also flesh out the policy in two important results: a proposed mechanism to meet the Landfill Directive targets for 2010, 2013 and 2016; and a methodology for setting waste levies for landfill, incineration and mechanical biological treatment ("MBT"). A cap and trade system is advocated for the meeting of Landfill Directive targets, where the cap is based on these targets and the permits to deposit waste in landfill are awarded to existing users of landfill based on 2009 data. Gradually over the period to 2016 an increasing proportion of the landfill permits would be auctioned off instead of being given free to existing users of landfill, so that by 2016 it would reach 100%

In terms of setting levies for incineration, landfill and MBT based on externalities, our approach is to: first, ensure consistency across different sources of externalities such as MBT, cement kilns, power plants and so on; and, second, to consider only *unpriced* externalities, thus avoiding double regulation and double counting. Based on these considerations levies are estimated considering only methane and disamenities and the results are as follows:

Landfill	€44.24 to €54.89 per tonne
Urban Incineration	€4.22 to €5.07 per tonne
Rural Incineration	€0.42 to €0.50 per tonne
MBT	€0.92 to €1.45 per tonne.

Although waste licenses set permissible levels for licensed facilities, via landfill gas concentration limits, methane is such an important contributor to global warming that a differentiated levy should be imposed across all the waste treatment options that are considered here.

To provide background for the assessment of specific policy options, we outline recent developments in municipal waste management policy in Ireland and provide projections of current and likely future waste arisings for Ireland and for the Dublin region.

While considerable progress has been made over recent years in encouraging recycling of municipal waste, meeting the Landfill Directive targets for diverting biodegradable waste from landfill still presents a substantial challenge. The current recession should temporarily halt growth in municipal waste arisings, but we expect growth to resume as the economy recovers, with household formation and rising incomes providing the main impetus. We illustrate the likely effect on diversion of the two municipal waste incinerators that currently have planning permission, and we suggest that substantially more investment and strengthening of collection arrangements are likely to be required if the targets are to be met. Our quantitative analyses suggest that the limits on incineration would be not binding if applied on a national level, but would be binding in the Dublin region (at least) if applied regionally.

In this report, the specific options considered arise from the government's proposed policy direction pursuant to Section 60 of the Waste Management Act, 1996 and the recommendations made in the international review. In particular with respect to the former, we assess the proposals intended to limit the use of incineration of municipal solid waste ("MSW"), with an initial cap of 30% being set, subsequently reduced to 25%.

The proposed Section 60 policy direction, which was released for comment in summer of 2009, marks a radical departure from existing MSW policy with a series of measures that will discourage incineration and favour MBT. We are concerned that this radical departure will considerably increase perceived regulatory risk. Greater perceived risk will raise the capital costs of projects, delaying the building of essential infrastructure and the realisation of environmental benefits. Thus, we think that there is merit in deferring the implementation of the Section 60 policy direction so that it can be considered in the context of the international review as envisaged in the Department of the Environment, Heritage and Local Government's *Strategy Statement* for 2008 to 2010.

The Strategic Environment Assessment ("SEA") methodology is we argue inappropriate, on its own, for evaluating the Section 60 policy direction. A cost benefit analysis, which does take costs into account, is the appropriate methodology. This implementation of the Section 60 policy direction should be deferred until a full Regulatory Impact Analysis ("RIA") is undertaken.

The corollary of the 30% cap on incineration is a 70% target for recycling of MSW waste. This latter target is reached by taking current recycling rates for EU Member States, by type of MSW waste, and assuming that Ireland can match the best performers. This is a flawed methodology. Member States vary across a whole range of dimensions such as productivity by sector, labour costs, urban/rural split, household type (apartment compared to a house), market size, policy interventions,[2] and so on. Differences in recycling rates are thus fully consistent with this general picture and no doubt reflect conditions particular to each Member State. Thus the idea that Ireland and presumably each Member State can achieve the highest recycling rate of the highest Member State without taking into account these differences is extremely unlikely to be successful or cost effective. Thus, we think that the Department of the Environment, Heritage and

[2] For example, in Germany and the Netherlands, paper is collected separately and thus more valuable.

Local Government should carefully specify how and by what policy and other mechanisms Ireland will reach any recycling target that it ultimately adopts.

Government intends to disadvantage incineration by banning local authorities entering into contracts that direct waste to incineration and that contain “take or pay” contracts while landfill levies are to be structured in such a way as to not advantage incineration. Since the local authority may be best placed to assume certain risks through a take or pay contract, removing this option is likely to be economically inefficient and needlessly raise costs. Levies should be set to reflect the externalities – environmental and other damage – which may or may not result in landfill paying a higher levy than incineration.

Turning to the incineration cap, we note that it is a command and control measure as opposed to an economic instrument such as a levy. Command and control mechanisms are likely to impose needless costs on the economy, particularly where, as in the present case of incineration, there is no underlying rationale for the 30% target selected. Much better in our view would be to ensure that the private cost of using each waste processing option will be set at a level reflecting the externalities it causes. Of course, setting such prices requires detailed information as to the costs and benefits.

The economic impact of any set of targets obviously depends on the degree to which they are attained. The EU Landfill Directive targets are binding, in the sense that that they are set by the EU and that there are fines for non-compliance. In contrast, while the 70% recycling target is not likely to be attained, the caps on incineration may well be effective in restraining incineration capacity. Hence, as a result more MSW than planned may end up in landfill, making the attainment of the Landfill Directive targets much harder and more costly to meet.

Competitiveness is a broad concept concerned with the ability to compete successfully while at the same time raising living standards. Government has an important role to play in ensuring an appropriate framework within which investment and other decisions vital to economic development can be made that goes well beyond the night watchman functions of providing law and order and securing property rights. Consistent, credible and predictable policies are likely to create a framework that is conducive to economic development and, as such, contribute positively towards enhancing Ireland’s reputation as a place to do business. Our findings suggest that the Section 60 policy direction on MSW policy and corollary policies will not contribute positively towards that reputation and will thus harm economic development and competitiveness.

This conclusion is not changed by the release of the international review on 19 November 2009, despite being supported by 65 annexes and a RIA on the structure of the waste levy. The purpose of the international review is to provide a roadmap for waste management policy in Ireland. A fresh beginning so to speak, a blueprint for change. The Minister for the Environment, Heritage and Local Government anticipates that he will bring “… a new policy statement to Government with a view to its publication in the New Year.” Despite some positives the international review does not provide a roadmap for the way forward. We analyse the reasons for these failures and comment on the lessons drawn by the Minister for the Environment, Heritage and Local Government on the international review for future policy.

The international review provides a number of important pointers concerning the way forward. It pushes a number of the right buttons in designing a waste management policy for Ireland. The international review quite correctly acknowledges the need to consider the economic implications of what is being proposed and that policy should be cost-effective and efficiently delivered. The international review also champions the principle that waste management policy should internalise any externalities. This principle is of paramount importance in the environmental area. Here there are a number of important *unpriced* externalities that need to be *priced* and hence internalised.

Apart from affirming these important general principles or considerations in framing waste policy, the international review makes twenty-five recommendations aimed at providing clarity to policy so that implementation can move forward swiftly. Many of these recommendations are to be welcomed. Basing residual waste levies on externalities is a useful application of the general principle set out above. The refundable compliance bonds to ensure compliance with construction and demolition ("C&D") targets is a novel approach that merits consideration, assuming C&D targets are necessary.

The recommendations concerning the introduction of competition *for* the market rather than competition *in* the market for household waste collection provide the basis for the realisation of substantial increases in efficiencies in collection, with householders benefiting through lower collection prices. The recognition that in applying producer responsibility, producers are to be responsible for full financial responsibility for delivering the services required to meet their obligations is likely to make this a much more effective policy instrument.

For a roadmap to be successful, however, it must provide unambiguous instructions, which can be easily understood. In the context of the international review this means that the recommendations must be clear, have a carefully explained and sound rationale and be credible. A characteristic of some of the most important recommendations made in the international review is that these conditions are not satisfied.

A central recommendation is the setting of the residual waste levy for landfill, incineration and MBT, in terms of € per tonne. Here the international review completely fails to explain how its proposed levy structure was derived from the underlying research. Furthermore, an explanation is not at all obvious from that underlying research.

A major reduction in the volume of household waste will have important implications for residual waste treatment capacity. The international review recommends that the level of residual household waste per capita is to be halved over a 13 year period from 300 kg to 150 kg. England and Wales are cited as having similar targets, but they are taking 20 and 16/17 years, respectively. No explanation is provided as to why Ireland will be able to achieve these targets so much quicker and what the additional costs are of such a rapid reduction. Thus the target is not credible.

Pursuing the roadmap analogy a little further, the roadmap should provide guidance over a series of stages. In the short term, the most important goal of waste management policy is ensuring that Ireland meets the Landfill Directive targets for 2010, 2013 and 2016. Failure to do so will result in, potentially at least, large EU fines. Here the international review must be considered a failure. The international review does not set out the magnitude of the problem, except for 2010. It prefers not to forecast the likely magnitude of BMW for 2013 and 2016 on the grounds that it is extremely difficult and that there is considerable uncertainty.

However, Eunomia could have considered alternative scenarios to assist in guiding policy. Does it matter if biodegradable municipal waste ("BMW") grows at 1% or 4%? Which is more likely? The scenarios indicate a range of possible outcomes, depending on the assumptions made. Suppose, for example, under all reasonable scenarios there is a large volume of residual waste for 2013 and 2016. If this was the case then it would make sense to invest in (say) an incinerator since these have a large sunk capital component and need to operate at close to capacity to achieve maximum efficiency. If, on the other hand, the evidence was much more equivocal about what the level of residual waste, then it would make sense to invest in technologies that could be ramped up at short notice or perhaps waste could be exported for disposal.

The point of internalising externalities, as the international review quite rightly points out, is that once these external damages or benefits are incorporated into the price then appropriate decisions are made by public and private agents concerning which waste management technology to use. Hence, it is important that the methodology behind the international review's estimates of the externalities from landfill, incineration and MBT is well grounded and defensible. Unfortunately this is not the case. No account is taken of disamenities caused to households because of the presence of a waste facility. In estimating the externality attention should only be paid to *unpriced* externalities. If an externality, such as pre-combustion emissions from diesel, is already priced through the carbon tax - which was announced in the budget on 9 December 2009 budget, or through the ETS for CO_2 emissions, it should not be included in the residual waste levy. This is not only double regulation, but also double counting. The international review does not take this into account and hence its levies cannot be relied upon to send the correct price signals for selecting between alternative waste management technologies.

We turn now to the strategic or longer term issues. It was argued that the Section 60 policy direction to cap incineration and other matters should be considered in the light of the international review since it was to deal with the issue of mix of technologies. The international review did not provide any guide to the mix of technologies. It is completely silent on the issue of the merits of MBT over incineration, except to say that some countries seem comfortable with high usage of incineration. The practical problems of switching from incineration to the MBT, such as stranded assets if a 30% regional cap on incineration, were introduced are not addressed.

The international review argues that its recommendations will achieve the objectives of the Section 60 policy direction to cap incineration and other matters. At the same time, although not a recommendation, Eunomia suggest that all MBT plants are strategic and hence the planning process should be fast tracked for these facilities. These two policy strands are complementary in that the MSW that would have been disposed of by means of incineration will now (presumably) have to be sent to MBT plants if Ireland is to comply with the Landfill Directive targets and hence avoid large EU fines. However, the international review does not provide any guidance as to the feasibility, location, timing, nature, cost and legality of fast tracking the building of MBT plants, nor does it provide any credible evidence that its recommendations will lead to a limitation on incineration consistent with the Section 60 policy direction to cap incineration and other matters that at the same time conforms with the Landfill Directive.

In sum, while the international review sets forth some sensible general principles for guiding policy and makes some welcome recommendations with respect to household waste collection, producer responsibility and refundable compliance bonds for C&D, the international review must be considered a failure in respect of its proposals for setting residual waste levies, per capita targets for reduction in residual waste and guidance in the appropriate mix of waste technologies.

There are a number of reasons for this failure. *First*, the task set was too big and the terms of reference were too wide. *Second*, in the 2007 Programme for Government and Section 60 policy direction on incineration and other matters, constraints were placed on policy choices for no coherent or compelling reason. In particular, incineration was to be disfavoured and MBT encouraged. Making this a precondition in the analysis distorted the results from the outset.

The preference for MBT over incineration is most clearly demonstrated in terms of reference for the RIA on waste levies which stipulated that the levy rates should be set to avoid providing any “competitive advantage” to incineration and to encourage MBT. This is a highly debatable approach. The levy for a particular method of residual waste treatment should be set on the basis of its associated externalities after carefully reviewing and examining the evidence. This may or may not result in a levy for MBT which is less than that for incineration.

It may also explain the structure of the waste levies in the international review, which based in the relevant annex, should have been:

Landfill - **€86.54** to €129.14 per tonne

Incineration - **€36.20** to €58.11 per tonne

MBT - **€20.70** to €42.05 per tonne

In fact, only the figures in bold were selected. This of course could result in an MBT plant being built that should be paying €42.05 per tonne but instead is paying a much lower and incorrect price of €20.70 and thus only half the externalities are being internalised. This is, of course, inconsistent with the declared aim of the international review that externalities should be internalised.[3]

Third, there is no clear methodology set out to draw lessons from the international experience or to guide policy. Despite nods towards greater attention to costs and benefits these issues are conspicuous by their absence. *Fourth*, the international review is far too accepting of EU policies and targets which are a complex set of compromises that are not always in Ireland’s interests and which as such need to be questioned and challenged to ensure that they do reflect such interests.

The Minister for the Environment, Heritage and Local Government in launching the international review said that it:

- “ will create jobs in new waste industries”
- “will enhance competitiveness of the wider economy as a whole”
- is “considered research which is the essential foundation for good and robust policy;” and,
- “we have a blueprint for legislative, institutional, regulatory and organisational change to achieve a wholly sustainable approach to waste management” (DoEHLG, 2009, p. 1).

It may be true that if the international review’s recommendations and suggestions were implemented, jobs would be created in new industries – building MBT facilities. However, we must ask at what cost, given that these resources could have been used to create jobs elsewhere in the economy and that these MBT facilities will to a considerable degree be replacing incinerators that are or were to be built shortly. Apart from the compensation that might have to be paid to developers of these incineration facilities, it is

[3] As noted below there are difficulties with the approach used by Eunomia to estimate externalities.

extremely unlikely that Ireland would meet its Landfill Directives so that EU fines would be levied on Ireland. One would have thought in a period of financial stringency this not a good use of resources.

There is nothing to suggest that the proposals made in the international review will enhance international competitiveness. Rather the whole process of waste management policy over the past two to three years has generated unnecessary uncertainty that is likely to have increased regulatory risk and thus raised investment costs and biased technological choices, so damaging consumers and taxpayers. Moreover, by not achieving a least-cost mix of waste management facilities and practices, the proposals are likely to lead to higher prices for users of waste management services.

While the Minister is correct in stating that considered research is the essential foundation for good and robust policy, the international review does not fulfil that description. Although the international review provides a blueprint, it is very unlikely that it will lead to a sustainable waste policy.

1 Introduction

Ireland is at an important junction in refining and implementing its municipal waste management policies. While significant progress has been made in recent years in encouraging the use of recycling as an alternative to landfill, the State has to meet legally binding targets that will become increasingly challenging from 2010 onwards. Without additional investment in facilities and collection arrangements, as shown in Section 5 below, Ireland is unlikely meet these targets. Indeed, it is likely to fall well short, despite the recession.

Significant policy decisions are under consideration, both in an effort to meet the targets and in response to political preferences concerning waste management options. In this report we aim to contribute to the policy debate by setting out a roadmap for economically efficient waste policies. We will then apply this roadmap to assess specific options.

The specific options considered arise first from the government's proposed policy direction pursuant to Section 60 of the Waste Management Act, 1996 (Section 7 below). In particular, we assess the proposals intended to limit the use of incineration of municipal solid waste ("MSW"), with a cap of 30% being set. Second, we consider matters raised in the government-commissioned international review of waste management (Section 8). We also put forward our own suggestions for waste policy in Ireland in Section 6

Our first task in this report is to set out an economic approach to municipal waste policy: this is done in Section 2. We believe that the overriding objective of public policy should be to maximise societal welfare. This means that both societal benefits (including environmental benefits) and costs of policy options need to be considered. Targets set by government, whether or not they are themselves optimal, should be met using the least cost method.

Markets are typically an efficient way of achieving this aim. However, markets may fail for a variety of reasons. Prices, for example, may not reflect the environmental damage caused by a particular activity. This provides a rationale for government intervention. Any intervention needs to recognise the economic characteristics of MSW services, as set out in Section 3.

There are, or course, other ways of setting policy towards municipal waste management, for example:

- Rules of thumb are often used. Three are particularly important in MSW: the waste management hierarchy; the polluter pays principle; and proximity principle. Such rules invariably contain grains of truth, but if they are applied mechanistically can lead to costly policy errors.
- Setting targets for waste recycling, reuse etc. based on best practice elsewhere is another alternative. Benchmarking has a useful role in informing policy, but if it does not take local conditions into account it can give misleading results.
- *Ad hoc* policies driven by a preference for certain technologies such as MBT, but not for others, such as incineration, may be employed. Preferences such as this may be appropriate if supported by evidence, but can be inefficient or environmentally damaging if applied without regard to market circumstances.

These approaches, as discussed in Section 2 and, to a lesser extent, in Sections 3, and 6 through to 8 below, are not necessarily inconsistent with the economic approach. The polluter pays principle might be used, for example, to justify a tax on the activity causing environmental damage so that the price that the consumer pays reflects this reality.

It is, however, important to realise that a consistent, economically optimal approach has significant benefits. It creates regulatory certainty. This applies equally to private firms investing large sums in environmental infrastructure projects such as incineration and to firms entering markets such as kerbside collection. If the rules of the game – the regulatory regime - change in unanticipated ways such that unjustified costs are imposed on the private sector then infrastructure projects will cost more and markets will be less competitive. This is, of course, not to suggest that if fresh evidence becomes available or a given policy is not working well that policy change is not merited. For example, the proximity principle was applied too rigidly and was subsequently relaxed.

If the exacting BMW targets set by the EU can be met in a less costly manner by employing the economic approach then that should increase societal welfare. At the present time, with an ongoing recession and a budgetary crisis, resources are scarce and need to be husbanded carefully to ensure that they are used in an optimal manner.

2 The Role of Government

2.1 Introduction

The objective of public policy using the economic approach is to maximise societal welfare. However, this sounds a little like motherhood and apple pie. Who could be against such an objective? All the participants in the debate over future waste management policy would no doubt take the view that their policy was designed to maximise societal welfare. Hence more precision is required as to the meaning of the phrase from an economic approach.

The Pareto principle is often applied in the economic approach in deciding whether or not societal welfare is maximised: it is not possible to make somebody else *better off* without making somebody else *worst off*. Hence if a given environmental objective can be achieved at lower costs this enables resources to be released for other purposes so making somebody else better off.

Of course, often policies end up making some people better off and others worst off. However, the winners – in theory at least – could compensate the losers and everybody would be better off. Hence it is sometimes argued that if the winners could compensate the losers then that policy would improve welfare; they do not actually have to pay the losers.

Elegant economic models have demonstrated that competitive markets maximise societal welfare. This also accords with common sense. In competitive markets firms compete to satisfy consumer demand. Costs are minimised. If firms try to raise prices above costs entry of new firms will occur. Thus such markets ensure efficient production that supplies what consumers want at minimum cost. So far, so good.

However, the world does not always correspond to economists' models and assumptions. Markets can fail. This can occur for a variety of reasons which (in a municipal waste context) are identified in Section 2.2 below. Markets may not be competitive, so that firms with market power can raise prices above cost. Prices may not reflect certain environmental damage caused in the production or use of a product. As a result too much is produced and consumed, with adverse environmental consequences.

Market failure provides a rationale for government intervention. In Section 2.3 we consider various instruments that can be used by government to intervene to correct the market failure. These vary from taxes to subsidies to the creation of property rights to competition policy to command and control. However, there are costs to intervention. Taxes may have to be raised, administrative systems created, while there may be compliance costs by business and households. All these need to be considered in matching the market failure with the appropriate instrument.

As noted in Section 1 above there are other ways than the economic approach in which waste management policy can be managed such as rules of thumb including the waste management hierarchy, the polluter pays and proximity principles. These rules of thumb are often developed to create an easily understood characterisation of complex issues and processes for a wider audience. Such rules may also result in a greater support for environmental policy. In Section 2.4 below these alternative rules of thumb are considered and related to the market failure approach. In a number of instances there are clear links between the economic approach and rules of thumb. As a result it may be possible to square the economic approach with at least elements of these other approaches.

Before government implements a new policy it typically undertakes an assessment of the policy. There are usually screens or sets of rules that ensure only policies with substantial or significant impacts are subject to extensive assessments. The methodology consistent with the economic approach is cost-benefit analysis; this underpins the government's Regulatory Impact Analysis ("RIA") model. However, another review methodology is sometimes applied to proposed environmental regulations: Strategic Environmental Assessment ("SEA"). This latter methodology is used to evaluate the proposed Section 60 policy direction to cap incineration and other matters discussed in Section 6 below (throughout this report we favour using specific and consistent vocabulary. Therefore, "incineration" over "thermal treatment" or "waste to energy" is used). In Section 2.5 RIAs and SEAs are discussed. The final section, 2.6, compares the economic approach with the rules of thumb approach to waste management policy.

2.2 Market Failure

As noted above, we focus on two potential categories of market failure: market power and externalities. Each is defined and then its relevance to waste management policy discussed. However, these two forms of market failure should not be treated separately as they may interact with one another.

Market power is defined as the ability of a firm to raise price above marginal costs on a sustained basis. This drives a wedge between price and cost. Some consumers who would be prepared to buy the product or service if price were set equal to cost are unable to do so. The consumers that do purchase the product or service pay a higher price than if price were set equal to cost. Income is transferred from consumers to producers.

In MSW services, market power is most frequently considered to arise in the collection of household waste. There are good reasons for a single waste collection firm to serve a given geographical area. This reflects that there are economies of density, which refers to efficiencies associated with serving a large number of customers located close together. In other words, if there are (say) 100 households on a street then the collection costs per household are less if one truck collects all the household waste compared to a situation where two or more firms collect the same total volume. Refuse collectors and garbage trucks spend more time collecting waste and less time travelling between pick-ups.

Despite the fact that there are good economic reasons for a single firm providing household waste collection, this does not prevent market failure. The monopoly provider is able to raise price above cost without attracting entry with competitive products at lower prices. The incumbent monopolist already realises all the economies of density and thus the entrant is at a disadvantage.

Prices are meant to reflect costs. However, in some instances, not all costs may be included in the price. The factory that produces widgets may discharge at zero cost polluted water into a river killing fish downstream and lowering the quality of drinking water. Plastic bags may be given away freely by retailers to customers, who subsequently litter public places and beauty spots causing visual pollution and clean up costs for owners of such places. These costs are referred to as external costs or externalities.

The market failure reflects the fact that the external costs are not taken into account in the production of widgets or the use of plastic bags. In the former case the factory is using the water for free. If it had to treat the water that is pumped into the river so that fish would not die and the quality of drinking was not lowered, then the price of widgets would increase to reflect these costs. Equally, the user of plastic bags is not being charged for the use of public spaces and beauty spots to discard plastic bags. In both cases the fact that prices are too low means that too many fish die, the quality of drinking water is lowered and that the countryside is festooned in plastic bags.

The two forms of market failure may interact so that the sum is greater than the individual parts. The monopoly supplier of household waste may raise price above costs causing some consumers to discontinue using the service and illegally dump the waste, thus imposing external costs of the sort associated with the plastic bag example referred to above. It is an issue that we will return to below.

2.3 Instruments of Intervention

Governments have at least two instruments with which to address market power. *First,* competition policy. It is a breach of competition law for a firm to abuse a dominant position, where dominance refers to market power. Abuses consist of exploitative abuses such as excessive prices and exclusionary abuses such as predatory pricing or discriminatory discounts which are used to disadvantage a competitor. *Second*, market design. In some instances competition law may not be able to address market power. Markets can be improved by better design. In some cases government intervenes in a market to improve its working. In the case of household waste collection, for example, the market could be organised in a number of different ways: competition for the market, in which the local authority auctions off the right to conduct household waste collection; or the current system of what is called side-by-side competition, in which private and/or public firms compete with each other.

Governments have at least four instruments with which to address externalities. *First,* taxes/subsidies. A tax (where the price is too low) or subsidy (where the price is too high) could be introduced to more closely align the price of a good or service with the external costs (benefits) that it causes. For example, a tax could be imposed on plastic bags or widgets to reflect the value of the external costs imposed. Equally a subsidy may be imposed on household waste collection so the price is set equal to cost. In the waste management sector, economic instruments such as landfill levies have been employed for this purpose, and in Britain a landfill allowance trading scheme is used to control the amount of material sent to landfill. It may, of course, be difficult to estimate the value of the external costs, an issue addressed in more detail in Annex A below. *Second,* assignment of property rights. Market failure can result from the fact that nobody owns the right to clean water or to a litter free countryside; creating a right may resolve the situation. A property right to clean water or to a litter free countryside could be established under which the factory would pay the owners of the water to use it and the consumer to litter the countryside. Clearly enforcing water rights would be a lot easier than those relating to a litter free countryside.

Third, command and control. Under this system firms are subject to detailed regulation concerning their activities that generate externalities. The firm producing the widgets would be told what sort of machinery to use and what volumes of water to use to ensure that no fish die or drinking water quality is lowered. There would be frequent inspections. Plastic bags would be banned. *Fourth*, information. Producers and consumers make decisions based on the information in their possession. If they are made aware of the costs that they are imposing on others then their behaviour may change. If the factory owner is told that the polluted water is killing fish he may opt to pollute less and so on. However, as Thaler and Sunstein (2008) make clear, the way that such information is presented can have profound effects on the outcome.

Given that there are a wide array of different instruments to address any given market failure, the issue arises as to how to select the most appropriate one. Since, as noted above, the economic approach involves maximising societal welfare, the instrument selected should be that which minimises the net cost to society. If, for example, we can find cheaper ways to achieve a given standard of environmental quality the savings can be used for other goals and society will be better off.

The costs that need to be considered in selecting the policy instrument include both direct and indirect costs. Direct costs can be divided into public and private costs. The former consist of the costs of regulatory agencies and government departments in formulating, implementing and monitoring intervention, while the latter include the costs of complying with the intervention, which might involve decisions concerning investment, location and so on. Indirect or second round costs are harder to measure and refer to the costs which affect firms in terms of innovation and possible reductions in efficiency.

These direct and indirect costs are likely to vary by the instrument selected. For example, a command and control approach to ensuring that a factory does not pollute a river might involve specifying not only the type of machinery that should be used, but also the quality of water from each point emission of the factory into the river. In contrast, a tax depending on the amount of pollutants entering the river would leave the factory with the incentive to innovate to reduce the level of pollution, which would be absent in the command and control approach.

The success of any instrument in addressing a market failure may also be a function of the choice of instrument. Some instruments, for example, may secure a greater degree of compliance than others. In order to ensure that the environmental costs and costs of disposal are taken into account for large consumer appliances (white goods), a disposal charge could be made when the consumer takes the appliance to a bring or civic amenity centre or alternatively a tax covering the disposal charge could be included in the price of the appliance and the cost for taking the appliance to the bring or civic amenity centre would be set at zero. The second option is more likely to secure compliance than the former, since some consumers may decide simply to leave the appliance by the roadside instead of paying a disposal charge to the bring banks or civic amenity sites.

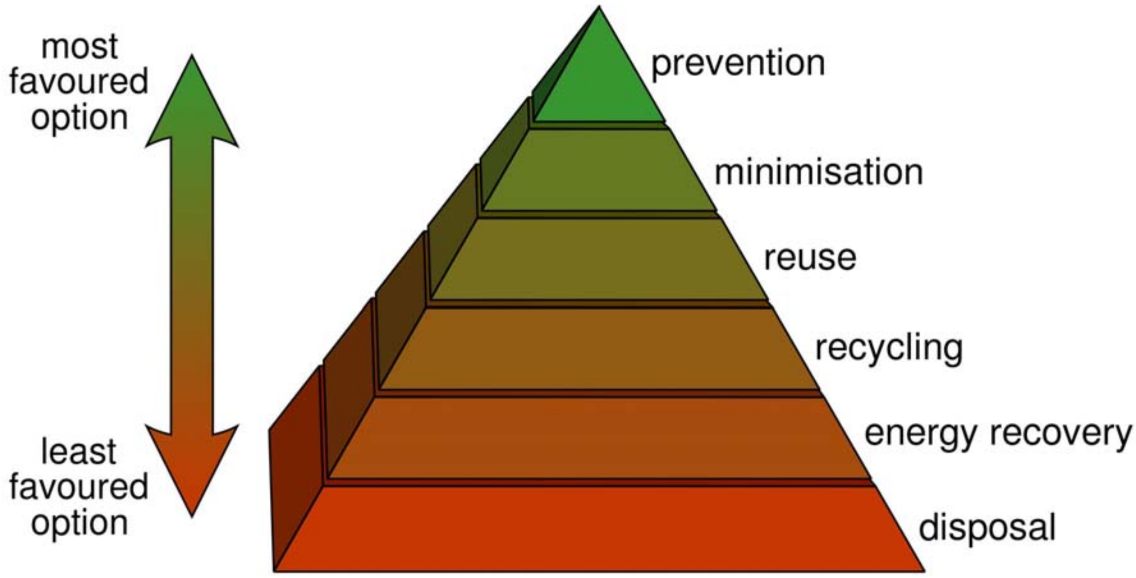
Distributional considerations may also need to be taken into account. In the case of household collection of waste, waivers may be given to those on low incomes. However, it could be argued that the tax and benefit system is the place for such considerations to be taken into account rather than micro-managing every policy for its distributional consequences. Nevertheless, if it is felt that those on low income would have the greatest difficulty complying with kerbside collection of waste, then there may be not only distributional but compliance reasons for waivers.

2.4 Alternative Approaches: Consistent or Inconsistent?

A number of principles or rules of thumb are used to guide waste policy, such as the waste management hierarchy and the polluter pays principle. Indeed, the waste management hierarchy underlines much of the reasoning and rationale behind the proposed Section 60 policy direction to cap incineration and other matters to be discussed in Section 7. It is thus important that some consideration be given to these rules of thumb and whether or not they are consistent with the economic approach.

A cornerstone of MSW policy is the waste hierarchy. Virtually every major government document starts by reference to the waste management hierarchy, illustrated in Figure 2.1 below. For example, in the 2004 *Taking Stock and Moving Forward*, the Department of the Environment state that under this hierarchy of options, "greatest emphasis [is placed] on waste prevention, followed by minimisation, re-use, recycling, energy recovery and, finally, the environmentally sustainable disposal of residual waste" (p. 3). More recently, the 2007 Programme for Government expressed a strong commitment to the waste management hierarchy (Department of the Taoiseach, 2007, p. 22).

Figure 2.1: The Waste Management Hierarchy



Source: Wikipedia

The objective of the waste hierarchy is first to create as little waste as possible by prevention and minimisation, and then, second, of what waste is created to extract as much value as possible through reuse, recycling and energy recovery, so that as little as possible is sent to landfill as residual waste.

The waste management hierarchy as a rule of thumb might be consistent with the economic approach if, on average, the benefit-cost ratios declined as one progressed down the pyramid. In other words, for every €1 invested in waste prevention the return would be, on average, (say) €10, while for every €1 invested in energy the return might be, on average, (say) €2. In making these calculations, of course, all of the costs including the externalities discussed above would need to be taken into account.

The critical issue is, of course, what rule should be employed to determine what option in the pyramid should be selected in a particular instance. Here we consider two approaches. *Option 1:* the waste hierarchy is treated as ranking preferred options so that prevention is ranked above minimisation, which is ranked above reuse and so on. Under this approach one does not move to an alternative lower level down in the hierarchy when considering how to treat a particular waste product, until all the technical possibilities have been exhausted, irrespective of cost, at the current level. Hence every prevention avenue is explored before waste minimisation is considered; every waste minimisation option is explored before reuse is considered and so on. The marginal costs at the different levels in the hierarchy are unlikely to be the same; indeed they are likely to decline as one moves down the hierarchy.

Option 2: the economic approach would treat the waste hierarchy not so much as a hierarchy, but rather as a menu of choices. Each option would bear a price reflecting the net cost of that particular option. This would take into account the externalities referred to above, which may vary by type of waste. It may be that the set of net costs is such that they increase in line with the waste hierarchy, but there can be no *a priori* guarantee that that will be the case.

Hence the waste hierarchy may be consistent with the economic approach. However, if the hierarchy is interpreted in a rigid technical manner with no attention to costs then it is unlikely to be consistent with the economic approach.

The polluter pays principle is another rule of thumb. This, defined according to the OECD, is, "the principle according to which the polluter should bear the cost of measures to reduce pollution according to the extent of either the damage done to society or the exceeding of an acceptable level (standard) of pollution."[4] This principle is very similar to the idea of a tax to take into account the fact that a particular activity, such as widgets or plastic bags, generates externalities.[5] The polluter pays principle would be consistent with a lump sum payable by the polluter at the end of the year after all the pollution had been measured and costed or by a small per unit tax reflecting the ongoing damage caused.

The economic approach would not, however, be indifferent to the method by which the polluter pays principle is applied. The economic approach is concerned with influencing behaviour of the firm or consumer, so as to reduce pollution and hence improve the environment. In short, to correct a market failure. A lump sum payment by the polluter is unlikely to influence behaviour; in contrast a per unit tax will affect behaviour and cause less pollution. Furthermore, in some instances other instruments – as outlined above – may be more appropriate than the polluter pays principle.

The third rule of thumb, which is related to the polluter pay principle, is producer responsibility. Under this the principle the producer of a product is responsible in whole or in part for a product once it reaches the end of its life. In other words, the consumer no longer has finished with a product and it becomes waste. Producer responsibility is designed so that producers have incentives to:[6]

- uses fewer resources;
- reduces or eliminates the use of hazardous substances or materials in the manufacture of the product;
- uses greater amounts of recyclate in the manufacture of the product;
- minimises waste from the product;
- can be reused; and,
- can be more easily treated/dismantled and recycled.

Equally, since the costs of disposal of the product will be included in the price of the product consumers will take that cost into account in their purchasing decision.

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[4] See http://stats.oecd.org/glossary/detail.asp?ID=2074. Accessed on 6 July 2009.

[5] In some instances the pollutee pays rather than the polluter. For example, downstream residents from a factory that is polluting their drinking water may decide to band together and pay the polluting factory to stop polluting. The polluter pays principle ignores this option, which was first suggested by Coase (1960).

[6] The list is taken from the explanation of producer responsibility on the Defra website: see http://www.defra.gov.uk/ENVIRONMENT/waste/topics/producer-responsibility.htm. Accessed 31 August 2009.

Producer responsibility is consistent with the economic approach. The producer is frequently best placed in terms of knowledge, information and incentive to make the appropriate decision with respect to reducing waste, using fewer resources etc. However, in order for the producer to make the correct choice between landfill, recycling etc the right price signal needs to be sent to guide the producer. Furthermore producer responsibility might work much better in some situations than others, principally where there are low transaction costs in terms of operating the scheme. Finally, it should be noted that there are many different variants of the producer responsibility principle, so that great care is needed in the design and implementation of such schemes (OECD, 2006).

The fourth rule of thumb is the proximity principle. According the European Environment Agency, the proximity principle "implies that waste should generally be managed as near as possible to its place of production, mainly because transporting waste has a significant environmental impact."[7] On the face of it this seems like common sense, but again the issue revolves around how it is interpreted. The economic approach would interpret the proximity principle as stating that there is a market failure in that the transportation of waste is leading to environmental damage that is not taken into account by the operators of the trucks. This market failure could be addressed through a tax, such as a carbon tax,[8] and a congestion charge. Private agents can then organise transportation in the most efficient manner. An alternative approach, which has been employed by some countries in the past, is to mandate that all waste has to be dealt with in a given geographical area. However, this proved inefficient and the size of the geographic area had to be increased (Ley *et al*., 2002). Nevertheless, this relaxation of the constraint does not, for example, deal with the issue of congestion.

2.5 Regulatory Impact Analysis vs. Strategic Environmental Assessment

In evaluating environmental – and other public projects or interventions – the economic approach involves a careful weighing of the costs and benefits of the project. The framework is applied to everything from an incinerator to a composting plant. The costs and benefits are estimated and discounted using some test discount rate. All costs are considered including the sort of external costs discussed above. Of course, in some cases it might not be possible to quantify all of the relevant costs and benefits. Furthermore the longer the time horizon the more likely it is that the costs and benefits will become more uncertain – subject to a greater margin of error. Such an approach can be applied in the context of a Regulatory Impact Analysis ("RIA"), which is defined in Box 2.1. The RIA approach requires the specification of credible alternatives and the framing of the question(s) to be addressed such that the result is not predetermined. As will be discussed in Section 8.7 below, the RIA on the structure of waste levies is flawed because the terms of reference appear to have undermined the potential usefulness of the work.

[7] See http://glossary.eea.europa.eu/EEAGlossary/P/principle_of_proximity. Accessed on 6 July 2009.

[8] For a discussion of a carbon tax see Tol *et al.* (2008).

Box 2.1: **A Definition of a Regulatory Impact Analysis**

> Regulatory Impact Analysis is a tool used for the structured exploration of different options to address particular policy issues. It is used where one or more of these options is new regulation or a regulatory change and facilitates the active consideration of alternatives to regulation or lighter forms of regulation. It involves a detailed analysis to ascertain whether or not different options, including regulatory ones, would have the desired impact. It helps to identify any possible side effects or hidden costs associated with regulation and to quantify the likely costs of compliance on the individual citizen or business. It also helps to clarify the costs of enforcement for the State.

Source: Department of the Taoiseach (2009a, p. 3)

In contrast, the methodology adopted in evaluating the Section 60 direction on incineration discussed in Section 7 below uses the Strategic Environmental Assessment methodology. The aim of this methodology is to “minimise the significant environmental impact of a proposed action” (Scott & Marsden, 2003, p. 2.). The methodology consists of a comparison of the proposed action as compared with a limited number of variations, in order to minimise the adverse environmental aspects of the proposed action. Costs are not considered at all as a careful reading of the synthesis report on SEA prepared for the Environmental Protection Agency (“EPA”) makes clear (Scott & Marsden, 2003). Lest there be any doubt, Eunomia (2009, p. 15) in conducting the SEA on the Section 60 policy direction with respect to incineration, state, “in any case, strictly speaking, the issue of costs is not a matter for the SEA.”

The two different approaches are likely to yield different conclusions. A particularly stark example is contained in the Eunomia report on the Section 60 policy evaluation on incineration. One of the six objectives of the Section 60 policy direction is that trucks that access waste through built up areas should upgrade as soon as practicable to Euro V and Euro VI standards, which become operative in 2009 and 2014, respectively. This policy involves replacing 6/7 of all trucks in 2009 and 6/7 of all trucks in 2014 compared to no policy intervention, since 1/7 of the trucks are replaced by depreciation for any given year (Eunomia, 2009, Table 38, p. 109).[9] There are some gains in terms of lower emissions from the earlier switching to these standards. Eunomia (2009, Table 45, pp. 115-116) estimates the net present value of the benefits per vehicle to be between €1,833.64 and €55.34, depending on the year that the truck entered the fleet. However, while a RIA would compare these benefits with the costs of retiring vehicles sooner than otherwise would be the case, the SEA makes no such attempt. Furthermore a RIA would also give consideration to the fact that if garbage trucks are retired at a much earlier age than otherwise would be the case, this extra cost would need to be financed. One way would be for household collection charges to increase, leading to more illegal dumping and degradation of the environment (O’Callaghan-Platt & Davies, 2007). Again the SEA does not take into account such effects, and as a result is likely to present a biased and excessively optimistic picture of the environmental effects of a proposed policy change. In other words, the SEA considers only gross not net environmental benefits.

[9] Eunomia is comparing for each year from 2009 onwards on the assumption that trucks are replaced every seven years (no policy change) with the introduction of the new standard trucks that year, given the age profile of the existing trucks.

2.6 Conclusion: Two Solitudes?

In this section we have set out the basis of the economic approach in considering the role of government with particular reference to waste management. Market failures were identified and measures to address them discussed as well as the use of the RIA as a way of assessing policy. It is this approach that will inform the discussion in the rest of the report. At the same time we have also considered the approach used in the environmental sphere when considering waste management. This consists of various rules of thumb such as the waste hierarchy, polluter pay and proximity principles, while the SEA approach is used to assess environmental policy.

While the rules of thumb can be made to be consistent with the economic approach, it appears that the way that these rules are interpreted by policy makers suggests that this is not the case. This is most sharply brought into relief in the case of the contrast between RIA and SEA where the latter pays no attention to costs, only benefits, and even then it is only gross not net benefits. The use of SEA requires a prior step that spells out the policy rationale and makes some attempt to explain how the benefits exceed the costs. If this is not done it is difficult to understand the basis for the policy. It is an issue that we will return to in Section 7 below. Of course, rules of thumb also have a useful role in communicating policy messages to a wide audience, but this is not a justification for using them as a formal guide for policy implementation.

3 The Economics of Waste Management

3.1 Introduction

The purpose of this section is to set out the salient characteristics of the waste management system for MSW, which as noted above, consists largely of waste generated by households and service sector firms. These characteristics are selected to inform an economic approach to MSW policy in Ireland. No doubt an engineer or a soil scientist would select a different set of characteristics. Even if the same set were selected, they might be analysed in a different way. Thus this discussion will inform the debate in later sections of the paper on regulatory choices and policy intervention.

The section begins with a simplified description of the waste management system as it existed in the 1990s, the current situation and some indication of future trends in Section 3.2. The remainder of the section will then discuss various aspects of the system, including:

- Vertical relationships & complementarities (Section 3.3);
- Economies of scale, scope and density (3.4);
- Cost structures: sunk, fixed and variable costs (3.5);
- Regulatory risk (3.6); and,
- Subsidiarity (3.7)

The section concludes with a roadmap for economically efficient waste management policy in Section 3.8.

3.2 A Simplified Description of Waste Management: Times Past, Present and Future

In the 1990s the waste management system was relatively straightforward compared to subsequent developments. In all cases waste is generated by households. However, in the 1990s all waste was placed in a single container – a bin, a bag and so on – for kerbside collection and then deposited in landfill. In 1995, for example, 95.7% of all household waste was sent to landfill (DoEHLG, 1998, p. 3). There was minimal reuse or recycling of waste. The waste management system was to a very large degree operated, owned and controlled by local government, with some private sector involvement in collection and, to a much lesser extent, landfill (Barrett & Lawlor, 1995, pp. 18-19).

By 2009 the structure has changed dramatically. No longer does the household place all its garbage in a single container for kerbside collection.[10] Instead, the household separates waste for kerbside collection into three streams which are placed in three different coloured bins:

- Green bin for dry recyclables such as paper, cardboard, steel and tin cans, clean plastics and Tetra Pak;
- Brown bin for food and garden waste; and,

[10] The description in this paragraph is based on the system currently present in Dublin. For details see: www.dublinwaste.ie. Accessed 2 July 2009.

- Grey/black bin for all other waste.

The household may also take refuse directly to bottle banks which typically provide separate large containers for a variety of materials such as: glass; paper, cardboard, plastic Tetra Pak; and, textiles. There are also bring centres and civic amenity sites that accept such items as white goods, batteries, waste oils, electronic and electric waste such as computers. It is likely that the range of waste that can be recycled will increase through time as technology improves.[11]

Household refuse is no longer taken almost exclusively to landfill. Instead, the separation of waste into various streams by the householder means that each stream can be dealt with separately without necessarily requiring additional complex processing:

- Dry recyclables can be reprocessed and sold as newsprint, paper, glass and so on.
- The food and garden waste can be used for composting.

In each case there are separate post-collection plants to process these materials so that they can be reused. In some cases the recycling is conducted in Ireland; in others, the waste is exported to be reprocessed, such as plastics to the UK.

Nevertheless there is still a residual of household waste that cannot be recycled. At least two alternatives exist:

- Incineration. Although there are no incinerators dealing with municipal waste currently in operation in the State, several are proposed. The one planned for Dublin, for example, is capable of taking 600,000 tonnes of rubbish and will generate enough energy to potentially heat 60,000 homes annually and provide electricity to the national grid for another 50,000 homes; and,
- Mechanical biological treatment or MBT, which is defined as a "generic term covering a range of technologies for dealing with residual waste. The common theme is the integration of mechanical sorting and feedstock preparation alongside biological treatment of some part of the waste, either using aerobic processes or anaerobic processes." (Eunomia, 2008, p. 13). It should be noted that MBT is a pre-treatment and not a disposal technology and that a significant proportion of the output from an MBT facility needs to either be landfilled or incinerated.

The final option for residual household waste that is not diverted into one of the above streams is landfill.

In terms of future developments, European-wide and Irish policy continues to favour further movement up the waste hierarchy away from landfill. However, the degree to which MBT or incineration will be used to divert waste from landfill is likely to be influenced not only by their relative cost-effectiveness but also by interventions such as the proposed Section 60 policy direction to cap incineration and other matters analysed in Section 7 below. In addition, of course, there are multiple interests at every stage of the process.

The waste management system has changed dramatically between the mid-1990s and 2009, primarily because of regulatory changes, some inspired by the European Union, others by policy makers in the State, combined with ongoing technical change. Many of

[11] Recycling will, of course, also be affected by economic, policy and behavioural changes, some of which may also influence the rate of technical change.

these policy changes were motivated by the concepts and rules of thumb discussed in Section 2 such as the waste hierarchy. Some of the technical change is driven by regulatory change, such as setting ambitious targets for recycling and reuse. No longer is the waste management system the almost exclusive preserve of the public sector, with the private sector playing an increasingly important role. However, it is the public sector that sets the policy context and framework within which the private sector operates.

3.3 Vertical Relationships & Complementarities

The system of waste collection, separation, post-collection processing and landfill is a vertically related set of activities. The output of the two or three bin collection system is an input to post-collection processing, including composting, recycling, MBT and incineration installations. The waste generated from each of these installations is then an input to landfill. In such a vertically related chain, service and other choices made at one level of the market affect the costs and benefits of options at other levels (e.g. collection and processing/disposal). If these vertical complementarities are sufficiently strong, waste management may need to be treated as a consistent end-to-end system. Supply choices need to take into account vertical links to reach the least cost supply mix.

For example, if household waste is separated at source into three bins as compared to all waste being placed in a single bin, this would improve the economics of materials recycling and composting facilities. The inputs from the source segregated collection system are of higher and more reliable quality. Paper, for example, is dry and is not contaminated with liquid and other waste streams. On the other hand, if most post-collection processing was conducted through an MBT facility, there may be less need for segregation of waste prior to collection and, potentially, a lower cost of collection.

3.4 Economies of Scale, Scope & Density

Economies of scale, scope and density are of critical importance in understanding the economics of waste management, with reference to the latter having already been made in Section 2. These concepts can be defined as follows:[12]

- Economies of scale refer to the phenomenon where the average costs per unit of output decrease with the increase in the scale or magnitude of the output being produced by a firm. For example, in the case of waste, the average costs per unit of waste processed decrease with the increase in the tonnage of waste processed.
- Economies of scope exist when it is cheaper to produce two products together (joint production) than to produce them separately. In the case of waste, it might be cheaper to co-locate waste processing facilities rather than to operate them separately.
- Generally, economies [of density] wherein unit costs are lower in relation to population density. The higher the population density, the lower the likely costs of infrastructure required to provide a service. One example would be the costs associated with collecting waste in urban vs. rural areas.

These concepts are also important in considering regulatory options and policy.

[12] The first two definitions were taken from OECD Glossary of Terms, with the example concerning waste added. For details see: http://stats.oecd.org/glossary/. The last definition, with some modification of the example, is taken from: http://www.encyclo.co.uk/define/Economies%20of%20Density. All accessed on 8 July 2009.

As already noted in Section 2 above, collection is cheaper when a truck picks up all the bins on one street, rather than occasional bins (economies of density). Together with significant congestion externalities, economies of density in collection suggest that welfare may be improved if only one operator serves each local market. Competition *for* the market, i.e. franchising, may be used to prevent such local monopolies from pricing higher than marginal cost. Furthermore, there may be economies of scope when kerbside collection of multiple different bins – green, brown, black/grey – is delivered by a single entity.

Efficient modes of service provision may differ between rural and urban areas. Rural areas are less densely populated, so if collection services are delivered there in the same way and to the same standard as in urban areas, they are likely to be significantly more costly per household.

Processing may also be subject to economies of scale, whereby some technologies enjoy lower unit costs the larger the flow of material through them (e.g. incineration). Centralised waste processing or disposal is obviously not subject to economies of density, since the waste is aggregated before it arrives at the plant. (This should hold, to a greater or lesser extent, for landfills, incinerators, materials recovery facilities ("MRFs") and composting plants).[13] However, it may be subject to economies of scale. A few big processing/disposal centres are probably cheaper to run and perhaps regulate than many small ones. Home composting is an exception as it cuts the cost of collection and processing. Of course, concentrating processing at a few sites increases the need to transport waste from point of collection to the processing plants, all things equal. Because most waste materials are bulky and of relatively low value, there tends to be a high ratio of transport costs to value of waste material. The optimal concentration of processing will balance economies of scale and transport costs. See Callan and Thomas (2001) for a literature review of work conducted on these economies and empirical work that indicates that economies of scale, scope and density all exist in MSW markets.

3.5 Costs Structures: Sunk, Fixed & Variable Costs

The cost structure of an activity can have significant implications for the effects of policy towards it. In this sub-section we define the main variables that describe the cost structure from an economic point of view before discussing how prevailing cost structures in the waste management sector have implications for optimal policies.

Collection, processing and disposal activities vary in terms of their cost structures. The relative importance of fixed and variable costs is likely to differ, with the ratio of fixed to variable costs being highest for landfill, incineration and MBT and lowest for collection with, in between, paper recycling and composting. The ratio of fixed to variable costs may vary with the level of utilisation. In MRFs as the number of shifts increases then the ratio of fixed to variable is likely to decrease; in contrast, for an incinerator that has to run 24/7 such opportunities do not arise.

Sunk costs from a regulatory point of view as we shall see below are of particular importance. Sunk costs are defined as follows by the OECD:

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[13] Note that references to MRFs in this document relate to "clean" MRF, which takes segregated waste as an input.

> Sunk costs are costs which, once committed, cannot be recovered. Sunk costs arise because some activities require specialised assets that cannot readily be diverted to other uses. Second-hand markets for such assets are therefore limited.
>
> Sunk costs are always fixed costs, but not all fixed costs are sunk.
>
> Examples of sunk costs are investments in equipment which can only produce a specific product, the development of products for specific customers, advertising expenditures and R&D expenditures. In general, these are firm-specific assets.[14]

In the waste management sector, examples of sunk costs would make up a high proportion of the costs incurred by most waste processing facilities. Once built, an MSW incinerator cannot easily be sold and moved to a different market. In contrast, household collection vehicles would generally be expected to have very low if not zero sunk costs, since they can easily be resold to other operators.

A concept related to sunk costs is asset specificity. Some assets can, once constructed, only be used to supply one specific customer or set of customers. For example, once a capital-intensive waste processing facility has been built, the cost of transporting bulky, relatively low value waste means that it can only economically serve customers in a certain geographical catchment area. The problem faced by developers of infrastructure subject to asset specificity is that once they have built the assets, their customer gains a degree of local market power over them. In effect, the customer could insist on paying only marginal cost for use of the facility, rather than allowing the project developer to recover its capital cost as well. The developer would have no choice but to accept this, thereby losing most of their equity in the project. Because developers know they will face this risk, they are unlikely to invest in the first place, and facilities will be under-supplied. However, as we shall see in Section 3.6 below, mechanisms may be developed to overcome these problems.

Installations also vary in their length of life.[15] A large scale incineration plant might be expected to last 25 years or so, but with regular investment in maintenance/life cycle, 40 years is possible. While there is limited experience with MBT, 20-25 years appears to be the expected norm. MRFs with regular replacement of mechanical parts can go on for many years, but it has been suggested that 15-20 years is a reasonable estimate. Process providers are generally willing to guarantee the life of composting installations for 15 years. In contrast a collection vehicle has a life of about 7 years.

These cost characteristics have important implications for policy. The most obvious one is that in sectors with long-lived assets, choices made today will determine and restrict options for years to come. If long-lived assets are subject to an uncertain rate of technical progress, selecting a standard today can have the effect of foreclosing future choices.[16] If this is so, a higher social discount rate should be applied to such choices than if the technology were static. In effect, the decision maker should take into account the potential value of waiting to see how quickly the technology improves.

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[14] OECD, Glossary of Statistical Terms. For details see: http://stats.oecd.org/glossary/detail.asp?ID=3317. Accessed on 8 July 2009.

[15] The data in this paragraph are based on information supplied by Juniper Consulting Services Limited. For details see: http://www.juniper.co.uk/about/Company%20profile.html. Accessed 12 July 2009.

[16] This is known as a “real option” in the economics literature.

3.6 Regulatory Risk

A second important issue for policy is that investors in projects with high sunk costs or asset specificity will demand a higher cost of capital or will refrain from investing in the presence of regulatory uncertainty. This is known as a "hold-up problem", because investment may not occur even where there is demand and potential supply. Investors in large capital projects with a high element of sunk costs are naturally concerned that as far as possible the policy environment remains stable and predictable over the life of the project. The more certain and predictable is the policy setting, the less risk and uncertainty is associated with the project. The chance that the policy setting may change in unpredictable and unanticipated ways is referred to as regulatory risk.[17] The degree of regulatory risk has important implications for the cost of waste projects, the quantity of investment in waste management assets and the composition of the waste infrastructure.

If a potential investor is considering whether to commit significant non-reversible (i.e. sunk costs) sums to construction of waste facilities, uncertainty about the future regulatory environment will raise the cost of the required capital. In effect, any rational investor will impose a risk premium. This will significantly inflate the cost of the infrastructure, because, as we noted earlier, most of the relevant costs are made up of fixed capital rather than variable inputs. The overall system will be more expensive than it needs to be.

A second likely effect of policy and regulatory risk is that less capacity will be built than would be socially optimal. Waste infrastructure providers do not bear the cost of under-provision: consumers, with a lower quality environment, and taxpayers, through higher taxes, ultimately do.

Finally, the presence of policy risk is likely to skew the mix of supply technologies away from an efficient mix, biasing it in the direction of technologies with low fixed costs and high variable costs. Such a technology bias will tend to be increasingly inefficient in the context of rapid increases in demand, precisely the regime we expect to observe when Ireland recovers from the present recession.

Investors will try to insure against regulatory risk and asset specificity by various contractual arrangements. There are a range of contracts and other arrangements that try to address the issue. A good example is "take or pay" contracts. These typically are contracts between the local authority and a firm building an incinerator or MBT or composting facility or other similar installation, under which the local authority undertakes to provide a certain quantity of waste at a certain price or undertakes to purchase a certain percentage of the capacity of the installation. If the local authority fails to fulfil its side of the bargain, the contract specifies that the local authority will pay the installation owner compensation. This is a credible commitment by the local authority which reduces regulatory risk by pre-committing the authority to purchase the services provided by the facility at a price that covers its average costs, rather than offering only marginal cost after the facility has been built. It also has the advantage that the local authority is best placed to bear such a risk since it is responsible for the waste management policy in its locality and is thus in a position to make such a commitment.

[17] One definition of regulatory risk is: "Exposure to financial loss arising from the probability that regulatory agencies will make changes in the current rules (or will impose new rules) that will negatively effect the already-taken trading positions." Taken from BusinessDictionary.com, which can be found at: http://www.businessdictionary.com/definition/regulatory-risk.html. Accessed 8 July 2009.

The installation owner may not be best placed to manage demand risk. Contracts of this type are common in energy markets as a way to finance generating capacity at a lower cost of capital than would be available otherwise.

It is therefore a matter of concern to potential investors that the Programme for Government contains a clause in which it commits the State to preventing take or pay clauses from being employed in future waste projects, so that neither the State nor the local authorities will be exposed to such financial risk (Department of the Taoiseach, 2007, p. 22). Removing the scope to use contracts of this kind is likely to increase the capital costs of infrastructure. Incineration facilities can have a capital cost of tens of millions. A 1 per cent increase in the cost of capital due to increased regulatory risk or inappropriately transferred market risk translates into an extra €100,000 interest per year for every €10 million in costs.

3.7 Subsidiarity

Provision of waste management services in Ireland is complex; there is a mixture of public and private provision in supply in both the collection and processing segments of the value chain. A range of public policy measures are applied at EU, national and local levels. Yet the economic markets for waste services tend to be local or regional in scope, as dictated by the cost structure of providing these services. This implies that some, perhaps most, regulatory decisions affecting the supply of services should be tailored to local or regional conditions.[18] Although it is beyond the scope of this report, this raises the issue whether current boundaries are the most appropriate. In any event, the boundaries should not be artificially drawn.

3.8 Synthesis: A Roadmap for Economically-efficient Waste Management Policy in Ireland

Section 2 discussed the role of government in waste management. It is necessarily rather abstract. Much of Section 3 has therefore been concerned with the salient economic characteristics of the waste management system. Any government intervention needs to take into account these characteristics since they provide the rationale for intervention and assist in choosing the optimal set of instruments.

Drawing on the previous two sections, we can now set out a "roadmap" for maximising societal welfare through public policy towards waste management.

1. There are market failures in waste management, so government intervention is merited and should be directed at addressing them.
2. Since economic markets for waste services and most market failures in the sector are regional or local in their incidence, policymaking should allow for local

[18] This is an application and extension of an EU concept, which is defined by the Commission as follows: "The principle of subsidiarity is defined in Article 5 of the Treaty establishing the European Community. It is intended to ensure that decisions are taken as closely as possible to the citizen and that constant checks are made as to whether action at Community level is justified in the light of the possibilities available at national, regional or local level. Specifically, it is the principle whereby the Union does not take action (except in the areas which fall within its exclusive competence) unless it is more effective than action taken at national, regional or local level. It is closely bound up with the principles of proportionality and necessity, which require that any action by the Union should not go beyond what is necessary to achieve the objectives of the Treaty." For details see: http://europa.eu/scadplus/glossary/subsidiarity_en.htm. Accessed on 10 July 2009.

variations. These regions should conform to economic regions, which may not always coincide with existing administrative regions. It seems unlikely that applying the same collection and processing arrangements across all of Europe, or even the whole area of a given country, would be efficient.

3 The main source of market failure in this sector is externalities. Policy should, where possible, use economic instruments to address externalities. In particular, levies and taxes are to be preferred to command and control policies because they allow the price and cost of externalities to be equated and they can readily be adjusted over time as market conditions change, which facilitates higher efficiency. Command and control rules are prone to set implicit prices on externalities that are far out of line with their true cost, particularly when they are applied across a broad range of areas with different cost conditions. Furthermore, as noted in Section 2 above, command and control mechanisms are likely to inhibit technical change.

4 The few externalities from waste management that have significant effects beyond the regional level, in particular emissions of greenhouse gases such as methane and CO_2, are best addressed using instruments that ensure consistency across sectors and countries rather than sector-specific measures. For example, the EU Emissions Trading System controls CO_2 emissions across Europe. Such measures are to be preferred to sector-specific command and control measures.

5 A second source of market failure is the potential for market power. Along with some externalities (such as road congestion) which are difficult to price, it may be proportional to address market power problems by reserving exclusive rights for waste authorities and encouraging them to keep costs low by permitting competition *for* the market, rather than *in* the market. Thus further use of franchising and contracting-out of services should be considered.

6 Regional administrations setting policy for waste collection and processing in their areas should take into account vertical complementarities in waste management processes. This can be done by comparing the wider societal costs and benefits of alternative end-to-end waste management systems, e.g. with different mixtures of source segregation and bulk processing, and selecting the set of options with the highest net benefits.

7 Given the prevalence of asset specificity, long-lived infrastructures and sunk costs in waste processing, all levels of government should aim to avoid creating unnecessary policy and regulatory risk. Regional administrations should consider using contractual arrangements that situate regulatory and demand risks with the parties that can most efficiently bear them and should be free to avoid hold-up problems by credibly committing to contractual conditions in advance.

4 Recent Developments in Municipal Waste Management Policy

4.1 Introduction

In Section 4.2 we briefly outline developments in Ireland's municipal waste management policy from the mid-1990s up to the change in policy emphasis contained in the 2007 Programme for Government, which presaged the Section 60 policy direction placing a cap on incineration. The section then turns to policy with respect to the particular stages of the waste management system: collection (4.3); recovery (4.4); disposal and incineration (4.5); and illegal disposal (4.6). The section is completed with a discussion of some trends in policy (4.7).

4.2 Background

Municipal waste policy in Ireland has been characterised by rapid changes in the last fifteen years. The system itself, the actors within it and the attitudes toward it have all evolved considerably. Prior to the mid-1990s, the waste management sector was still governed by the Public Health (Ireland) Act 1878. In the main, the local authorities (LAs) provided collection and disposal services but, owing to the low priority of waste management, low-cost, low-technology solutions were the norm. In the early 1990s, the vast majority of municipal waste was landfilled. There was little or no recycling and no biological treatment or municipal waste incineration facilities, and so Ireland was totally reliant on landfills. Many of these were also below modern environmental standards.

However, the mid-1990s witnessed the first recognition of modern Ireland's waste problems; the Waste Management Act 1996 was the first legislative measure to address them. This period also saw a change in the funding structure of LAs; sewage and water charges were abolished but waste collection charges retained. Overall, the funding of local government remains highly centralised compared to other OECD countries.

The first major policy statement, *Waste Management: Changing Our Ways* (DoEHLG, 1998), proposed a regional approach to waste management, demanded the formulation of waste plans and introduced into Irish policymaking the now well-rehearsed principles of waste management: the waste hierarchy; the polluter pays principle; and the proximity principle. It also set out concrete targets for waste management. The most notable (and due by 2013) are: a diversion from landfill of 50% of household waste, as measured by weight; a 65% reduction in BMW sent to landfill; and a recycling rate of 35% of municipal waste. Only the latter has been achieved to date.

In terms of EU-directed targets, and the subsequent need for national policy realignment, the EU Landfill Directive (1999) set specific limits on the tonnage of biodegradable waste that can be accepted at landfills (Table 4.1 below). These targets are the most imminent of all waste policy targets in Ireland: by 1st January 2010, Ireland may only landfill a maximum 75% of the biodegradable waste generated in 1995, i.e. a maximum of 967,443t can be landfilled. By 2007, Ireland was over this 2010 target by 34% (EPA, 2009a).

Table 4.1: Targets for biodegradable waste diversion from landfill (per Directive 1999/31/EC)

Target Year	Landfill Directive Target (%)	Landfill Directive Target (tonnes)
2010[19]	75% of quantity generated in 1995	967,433
2013	50% of quantity generated in 1995	644,956
2016	35% of quantity generated in 1995	451,469

Source: EPA (2009a, Table 14, p. 14)

Targets and strategies have been central to environmental policymaking, but it was only by the Waste Management (Amendment) Act 2001 that economic instruments like the landfill levy and the plastic bag tax were used for the first time to address the management of waste. This Act also removed the power to adopt waste plans from locally elected members to LA managers. The Protection of the Environment Act 2003 changed the executive powers of LAs further and made the introduction of an additional economic instrument, unit-based charges, much easier in 2005.

The entry of the Green Party to Government in 2007 led to a greater focus on waste policy and a change in emphasis on some of its particulars. The 2007 Programme for Government included a statement that MBT facilities should be introduced "as one of a range of technologies" and also made clear that incineration was not to be a favoured option (see Section 7.3 below for further details on the Programme).[20] This is in stark contrast to the most up-to-date overview of waste policy from the DoEHLG, *Taking Stock and Moving Forward* (2004), which anticipated significant use of incineration. When also considering that the majority of regional waste management plans identified incineration as necessary in the medium to long-term, it remains to be seen how such sentiments emanating from central government will affect waste policy in Ireland in the immediate future.

4.3 Waste Collection

Until 1996, waste collection was undertaken predominantly by local authorities (LAs). The Waste Management Act 1996 allowed for greater private sector involvement; by 2007, they operated in all 34 LAs (EPA, 2009a). By 2008, 19 of the 34 local authorities did not collect any waste stream at all (OECD, 2008). This level of private sector involvement without any municipal oversight makes Ireland unusual, and, in fact, the recent OECD report on the Irish public sector recommended that the licensing functions of the LAs be removed to the regional/national level (for the purposes of facilitating effective competition).

Charging for domestic solid waste collection was abolished in 1977, and from this period on the costs of collection was covered by a block grant from central government. LAs were once again allowed to charge for waste services from 1983, but did not do so for many years due to public opposition. In 2003, the DoEHLG gave LAs greater power to

[19] The Landfill Directive (1999/31/EC) allows Ireland to avail of a derogation under Article 5 of the Directive which postpones the 2006 and 2009 targets for 4 years (EPA, 2009).

[20] It was the Programme for Government document that called for the international review of waste policy which is due in July 2009. For details see Department of the Taoiseach (2007).

charge for waste services.[21] Thus, by 1st January 2005, all domestic waste collectors were obliged to implement a pay-by-use ("PBU") charging system (which upholds the polluter pays principle).[22] The PBU system is implemented in three ways: by volume (normally a differentiated charge by bin capacity); by bin/bag-tagging; and by weight. Tagging has proven most popular: 82% of LAs use it (due to ease of implementation and issues of equity). However, in most LAs a combination of the PBU systems is used. Furthermore, the collectors themselves are also mixed with 41% of LAs reporting both public and private collection operators.

The above results, from an EPA-commissioned study by O'Callaghan-Platt and Davies (2007), which is also discussed in Section 5.2 below, describes the PBU system as a success, but one which is not reaching its full potential. Although PBU areas saw a decline in waste presentation - 227 kg per person in 2003 to 219kg in 2005[23] – many collectors used volume-based charging, a system that can yield similar (ineffective) results to the flat-rate charging of waste collection in the past. The average reduction found in this study is quite low by international standards. And by the end of 2006, six LAs still had waste collectors operating a flat-rate charge in parts of their functional area (O'Callaghan-Platt & Davies, 2008a).

An effective PBU system sends a more accurate pricing signal to the producers of waste than a flat rate, and encourages them to reduce their waste generation, but the continued use of a flat rate or even bin-size based charging weakens this incentive to change household and business habits. In O'Callaghan-Platt and Davies' survey, it emerged that volume-based charging is used in 62% of LAs' functional areas, and that the price difference between a 140l and 240l bin is quite small, thus explaining in part the weak effect of the PBU system in Ireland. It is notable that this form of charging is not used by any public waste collectors. This fact also illustrates the lack of control LAs have over collection practices under the current permitting system (O'Callaghan-Platt & Davies, 2008b).

In a case-study conducted by Scott and Watson in 2006, the effects of weight-based charges in West Cork County Council were examined. As expected, this form of PBU charging produced a marked reduction in the amount of waste being collected. Per household reductions in waste were greatest for those which recycled most. (Of course, this result was in part caused by the extension of recycling facilities in the area at roughly the same time as weight-based charging was introduced). By directly questioning households, it emerged that the introduction of this form of collection charging did substantially and positively affect recycling rates. It also emerged from such questioning that the overwhelming majority preferred the pay-by-weight system to payment by other means.

In terms of source-segregation, further changes can be seen in the manner of collection, for example, in 2002 Green Bins (dry recyclables) were introduced in the Dublin Region, with brown bins (organic waste) following in 2006 (Phelan, 2007). This latter service was expected to continue rolling out until the end of the current Waste Management Plan period in 2010. However, the recent decision to cancel the tenders for two biological

[21] By the Protection of the Environment Act 2003.

[22] This system had first been mooted in 1998, in *Waste Management: Changing Our Ways*. This form of charging has been used internationally since the 1970s.

[23] In areas without PBU, waste presentation actually increased over the same period.

treatment plants for brown bin waste at Ballyogan and Kilshane (due to value-for-money concerns) places this plan in doubt (Gartland, 2009).[24] Their impact has been noteworthy: between 2006 and 2007, the amount of waste collected by the brown bins in the Dublin Region increased from 2,667t to 4,617t (EPA, 2009a). The greatest impact of brown bins, however, has been in Galway City Council and Waterford's County and City Councils, whose combined collection levels for food and garden waste almost double that of all the other LAs put together (EPA, 2009a). The roll out of brown bins has not been notable amongst private waste collectors to date.

4.4 Waste Recovery

Recycling infrastructure has improved in recent years. The number of bring banks totalled 1,960 in 2007, an increase of over 500 from 2001. The recovery rate of municipal waste reached 36% in 2006 and remained at this rate in 2007. O'Callaghan-Platt and Davies (2007) have shown that the county council with the lowest waste presentation was also the one which provided the most frequent recycling collections.[25] Their work suggests there has been an upward trend in recycling, regardless of waste collection charge type, and that recycling infrastructure may be just as important as economic instruments like PBU in reducing waste presentation. The case-study of West Cork County Council by Scott and Watson (2006), mentioned in the previous section, also confirms this important relationship.

One particular component of the waste stream is growing rapidly in Ireland: waste electrical and electronic equipment ("WEEE"). It has only recently become a target for recovery efforts. The WEEE Directive was brought into force in August 2005. This Commission directive aims to reduce the amount of WEEE generated in member states. It is based on the principle of producer responsibility, making them responsible for the financing of the collection, treatment, recovery and disposal of WEEE. By 2007, Ireland had exceeded all but one of the recovery targets (that related to large household appliances). The Directive is currently under review, with the likelihood of greater producer responsibility in this area and greater harmonisation across countries possible in the future.

4.5 Waste Disposal and Incineration

Historically, the majority of Irish municipal waste has been sent to landfill. As of 2007, 29 authorised landfills accepted municipal waste (the same number as in 2006 but down significantly from 48 in 2001 and 120 in the late 1990s). Four of these are run by private operators, all under the Greenstar Group (EPA, 2009a). The reduction in the number of landfills is due to greater regulation of the sector, with all landfill operators now obliged to obtain a waste licence from the EPA.

The quantity of biodegradable municipal waste disposed of to landfill increased by 4% compared to 6% to 1,475,000t in 2007, moving Ireland further from the first Landfill Directive target due in 2010 by a surplus of 508,000t (EPA, 2009a). Despite the

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[24] "The cost to Dún Laoghaire-Rathdown of a 15-year design, build and operate contract for a 20,000 tonne per annum facility did not compare favourably with the costs of nearby composting services contracts, the council said" (Gartland, 2009). Such commentary adds weight to the analysis on costs and regulatory risk in section 3

[25] The LA in question, Galway City Council, actually used a flat-rate charge, but focused heavily on providing recycling facilities in its functional area.

multiplicity of local, national and European targets, this figure has been on the increase since 2003, as will be illustrated in Section 5.3 below. The increase can be attributed to growth (economic and population), changing consumption patterns and better reporting methods.

The cost of disposal is relatively high in Ireland: in 2008, disposal at landfill averaged €132 per tonne (€20 levy included), although there is variability across the country and reductions can be negotiated (Forfás, 2009). By contrast, the landfill gate fee cost less than €10 per tonne in 1996 (EPA, 2005). Charges at landfill showed a steep upward trend nationally until 2004, but have since declined (despite the increase in the landfill levy in 2008). As noted, however, there is considerable variability in charges nationwide: the average fee (including levy) in 2007 by province ranged from €120 in Ulster to €145 in Leinster.[26]

There have been changes in waste disposal policymaking (if not policy implementation) since the mid-1990s; almost all regional waste management plans include incineration as a solution for dealing with waste. Although work has begun on the incinerator in Carranstown, Co. Meath, and pre-construction work has begun at Poolbeg, incineration as a means of reducing waste disposal will not be operational in Ireland until the end of 2011 (Forfás, 2009). Needless to say, incineration remains a highly contentious and unresolved issue with the public, and, as demonstrated by the Section 60 policy direction, amongst politicians too.

4.6 Illegal Disposal

The high cost of landfilling and fears over decreasing landfill capacity in 2001 were some of the factors that encouraged illegal behaviour around this period. Since the late-1990s, there have been a few high-profile cases of illegal dumping in Ireland, for example, the large-scale dumping that took place in Co. Wicklow from 1997-2002, and the illegal movement of waste to Northern Ireland in the 2000s. The Irish government has recently committed itself to removing a quarter of a million tonnes of this illegal waste from Border areas of the North (Keenan, 2009). Both cases are indicative of the change in waste management costs and enforcement in Ireland, with both now operating at a much higher level than previously.

There has been a decrease in organised illegal dumping in recent years (due to better enforcement); but fly-tipping and backyard burning have reportedly increased, with the former being a bigger issue in urban areas and the latter in rural parts of the country. 80% of LAs have reported backyard burning as a significant issue (EPA, 2005). Backyard burning is a particular problem due to the dioxins it can emit; almost 73% of the dioxins emitted to air in Ireland come from the uncontrolled, low-temperature burning of waste (Scott & Watson, 2006). There is some evidence to suggest that this practice increased with the introduction of weight-based waste collection charges, and, interestingly, that the number of burners is roughly equal across all levels of education and income (ibid.). At the same time, people broadly welcomed weight-based charging when questioned on it, which suggests that there is a gap between what one says and what one does.

The EPA has identified the household sector as the most significant in terms of uncollected waste. Only 80% of households are served by kerbside collection services, although it cannot be ascertained whether this is by choice or due to lack of service in

[26] Based on information provided by IBEC.

their areas (EPA, 2009a). Where waste goes uncollected, the EPA assumes that a large part of this is disposed of by illegal means, mainly backyard burning. However, exact figures on illegal disposal methods are hard to estimate, as not all LAs monitor this effectively.

4.7 Conclusions

Several trends can be derived from the above discussion:

- Waste policy has become more sophisticated: landfill is no longer considered a sufficient response; collection methods and charges have become more varied and more aimed at reducing and recovering waste; instruments like the plastic bag tax have successfully been introduced; enforcement of waste management laws is more stringent.
- Waste policy has become more complicated. A sophisticated system can become complicated when there are too many horizontal actors (i.e. private and public) and too many vertical actors (i.e. local, national and European), too many targets (e.g. Landfill Directive) and too many policy agendas (e.g. Green Procurement, WEEE and so on. At the same time there has been a re-working of the delivery mechanisms for waste management planning in Ireland with the regionalisation of waste management that has, for example, made incineration a viable option. These changes in planning can, if properly structured, improve policy delivery, despite the increase in complexity (Boyle, 2003).
- Waste management has become more costly. When waste management consisted solely of landfilling, it was a lot cheaper to implement. With the increase in the stages of waste management – be they bring banks or biological treatment facilities – costs have increased. The fact that LA funding is still controlled in large part from the centre and that Ireland does not have domestic rates makes outsourcing of costly waste collection, recovery and disposal schemes to private companies more attractive.
- Waste policy is more politicised. Public consciousness on the issue has increased, and acceptance of unsustainable options like landfilling has declined. Similarly, incineration has faced a sustained amount of public and political opposition in Ireland, which may, in part, be exacerbated by multi-member constituencies. Nonetheless, the fact that illegal dumping and backyard burning continue suggests an “attitude/behaviour gap” persists among the public.

5 Demand for Municipal Solid Waste Services in Ireland

5.1 Introduction

In Section 5.2 we outline recent quantitative evidence on factors influencing the demand for municipal solid waste ("MSW") management services in Ireland. This is obviously a partial analysis, since not all waste demand parameters have been estimated for Ireland. A complete analysis would have to consider the costs of supply options. That would require substantial new research and is outside the scope of this report.

Also in this section, to illustrate the background against which future policies will have to operate, we present baseline estimates of how future MSW and biodegradable municipal waste ("BMW") arisings are likely to be managed in Ireland (5.3) and for the Dublin region (5.4) if current trends continue. These results provide an indication of the distance to target that the government faces with respect to Landfill Directive limits on landfill of BMW. They also illustrate the potential impact of planned new incineration facilities on the disposition mix and the likelihood that the targets set out in the Section 60 policy direction to cap incineration and other matters might be binding.

5.2 Evidence on Factors Driving MSW Demand in Ireland

Demand for MSW services is a *derived demand*, in the sense that households and services firms generate waste in the course of consuming and producing other goods and services, rather than for the sake of producing waste itself. This implies that the main drivers of waste arisings should relate to consumption and production activities, and indeed the international literature indicates that this is so.[27] Some of the main drivers of MSW arisings are listed below, with the direction of effects in parentheses:

- Number of households (positive);
- Persons per household (positive, with rate declining as size increases);
- Household disposable income or expenditure on goods and services (positive);
- Output of the services sector (positive);
- Real prices of goods that are complementary with waste generation, e.g. goods supplied with packaging (negative);
- Value of material for reuse (negative);
- Cost of managing waste, for those facing per unit prices (negative); and,
- Less easily observable factors that may vary by person or firm, including household preferences for different sorts of goods, attitudes towards the environment, the nature of goods being produced by firms, etc.

The list above helps explain why waste generation *per se* tends to be hard to reduce through public policy. Numbers and sizes of households tend not to be the subject of policy; incomes and service sector output are seen as desirable since they increase societal welfare; and waste generation tends not to be very sensitive to the prices of goods or cost of managing waste. Only educational and informational interventions offer much

[27] Surveys of behavioural evidence are given in Jenkins (1993), Choe and Fraser (1998) and Kinnaman (2003).

prospect of affecting total arisings, and there is little published research into their effectiveness in an Irish context. In sum, it is likely that policies to make large reductions in arisings would involve heavy societal costs.

Because waste generation is so costly to reduce, most public policy interventions focus on waste disposition instead. As discussed in Sections 3 and 4 above, these types of interventions can be, and are, made at different points in the waste services supply chain.

Starting with the collection stage, there is plentiful evidence that policy can affect the way waste is presented, offering great potential to affect its final disposition at reasonable cost (we refer to how waste is ultimately managed as its "disposition"). Here are some of the factors that can affect the extent of waste processing activity within households and firms, leading to changes in the way material is presented for collection:

- Availability, frequency and convenience of mixed waste collection services;
- Availability, frequency and convenience of recycling options, e.g. kerbside collection of recyclables, bring banks, producer/retailer responsibility schemes;
- Relative prices of each option for dealing with waste including the cost of time incurred by households and firms and direct costs of services for those facing per unit prices;
- The mix of housing types in an area, which can affect the scope for particular options such as home composting or multiple bin systems;
- Direct regulation of alternatives, to the extent that agents are adequately informed about them and that they are enforced effectively, e.g. bans on fly tipping and backyard burning; and
- Here too, household and firm preferences, environmental attitudes and education about options can have important effects.

It is important to remember that work undertaken within households and firms prior to waste presentation is not costless to society; it simply lacks a visible price in cases where no market transactions are involved. Economically-optimal waste policy will aim to minimise the total costs to society, taking into account both "hidden" costs and visible ones.

An increasing amount of research has been conducted recently into the behavioural parameters governing generation of waste and its presentation by households and firms in Ireland. Estimates exist for many of the key drivers listed above, and for others estimates are available from international research.

Waste per household is generally understood to rise with the **number of persons in the household**, but there are economies of scale in household waste generation. As additional people are added to a household, the incremental amount generated by each additional person falls. Estimates in Scott and Watson (2006) indicate that weight presented rises by 0.49% for every 1% rise in the number of persons in a household.

Waste generated by households in Ireland appears to be unusually sensitive to **household income**. Curtis *et al.* (2009) use econometric modelling to estimate this parameter and find a one-to-one relationship between percentage changes in real disposable income and total waste arisings. Most estimates in the international literature find a figure of less than 0.6 for this relationship.[28] A high sensitivity of waste generation to income implies that waste will be strongly pro-cyclical with respect to economic activity. A key question for future empirical research is whether this relationship will persist in Ireland, or whether the income-sensitivity of household waste quantities will fall towards the levels observed in other rich countries. There is some evidence, e.g. in Lyons *et al.* (2009), that changes in consumption patterns adjust with a lag to changes in personal sector incomes. If this is so, Irish consumption patterns may still be adjusting to the country's relatively recent achievement of high income status and the sensitivity of waste quantities to income might be expected to fall. On the other hand, the unexpectedly sharp fall in household and services waste quantities from 2007 to 2008, as the current recession took hold, may suggest that the responsiveness of waste to income in Ireland remains very high.

A number of studies have found, unsurprisingly, that **availability of kerbside collection of dry recyclables** reduces mixed waste arisings. In the case of Curtis *et al*. (2009), the reduction is about 15%. Their evidence is less clear on the extent of diversion from **introducing a third bin for compostable waste –** a "brown bin"; indicative results from the same study suggest that about 22% of household waste in an urban area and 10% of waste in a rural area might be diverted. However, we would have more confidence in estimating these effects if household level data from brown bin pilot schemes were made available to researchers.

Turning to the sensitivity of mixed waste arisings to the price of collection, the effectiveness of weight-based, and to a lesser extent volume-based prices, in diverting household mixed waste into segregated streams is well established in Ireland and internationally. Introduction of **weight-based pricing** in Ireland appears to reduce mixed waste quantities by about 45%, and a subsequent one percent increase in the charge per kilogramme reduces waste by about 0.27% on average.[29] **Volume-based pricing** (e.g. tag a bag schemes) have a weaker effect in reducing mixed waste quantities, yielding about a 0.15% decrease in response to a one percent increase in the charge per unit volume according to international research.[30]

There is thus a considerable weight of evidence that changes to collection arrangements can encourage significant amounts of source segregation, leading in turn to substantial diversion of material from landfill. Of course, as noted earlier, these measures should only be taken if their benefits outweigh the costs, including costs from externalities and increased illegal disposal. Another important lesson from recent research is that some collection arrangements are much more effective at encouraging diversion than others.

There are also potential complementarities between measures. For example, increasing the **landfill levy** may have an indirect effect in encouraging source segregation if it is passed on to households through effective pay by use charges. However, if households do not pay by use or only face limited pay by use tariffs, such as differing tariffs by bin size, a change in the landfill levy is unlikely to have much effect through this channel. This example is illustrated in EPA (2008).

In practice, it is unclear whether current Irish regulatory structures can increase the share of waste collectors using effective per unit charging schemes. Research by O'Callaghan-Platt and Davies (2007, 2008c) indicates widespread non-compliance or weak compliance

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[28] This statistic is often referred to as the income elasticity of demand.

[29] .See Curtis *et al.* (2009) for a summary of the evidence on this.

[30] Scott and Watson (2006).

with existing measures to encourage effective pay-by-use charging. If less effective charging mechanisms remain common, price effects will not operate fully in waste collection. Any signals from disposal-side pricing (e.g. from landfill levies) will continue to have a muted effect on household behaviour.

The final segment of the value chain for MSW management involves processing of waste after collection. Here the economic choices are made by waste processing authorities and firms, rather than by the households and companies in the services sector who generated the waste. In this report, we do not attempt to model these choices explicitly: data for Ireland are too scant, and much further research is required. Instead, we limit ourselves to providing some baseline projections for waste arisings and disposition based on available evidence on the behaviour of households and firms, the current state of collection arrangements, and simple assumptions on the near-term availability of disposition infrastructure. These baseline projections give an indication of the magnitudes involved and suggest how much still has to be done to achieve national targets.

We focus on the BMW component of municipal waste in the remainder of this section, because of the environmental and regulatory importance of this waste stream. However, we also provide some analysis of likely non-BMW flows from the household and services sectors. BMW made up about 65% of Ireland's municipal waste in 2007.

5.3 Projected BMW Arisings – National

The top line in Figure 5.1 below shows baseline projections from the ESRI/EPA ISus model for the total amount of BMW generated in Ireland, assuming that the economy follows the "world recovery" scenario from the ESRI's recent report *Recovery Scenarios for Ireland*, Bergin *et al*. (2009). The figure also shows the projected shares of services (or commercial in EPA terminology) and household (or residential in EPA terminology) sector BMW.

Our national projections take into account EPA *National Waste Report* data up to 2008. Use of the 2008 data as a baseline makes a significant difference, because there was a substantial fall in reported waste quantities between 2007 and 2008. The precise estimates given here should be treated with caution, both because of the uncertainty surrounding behavioural parameters and because there have been substantial changes in the data collection process and the characterisation of waste in recent years.

ISus is a satellite model of the ESRI's Hermes macroeconomic model, and it has been developed by the ESRI to project environmental pressures into the future. Other parts of the ISus model are described in O'Doherty and Tol (2007) and Fitz Gerald *et al*. (2008).[31] Baseline data is mostly obtained from successive National Waste Reports ("NWR"), in particular EPA (2008, 2009a, 2009b).

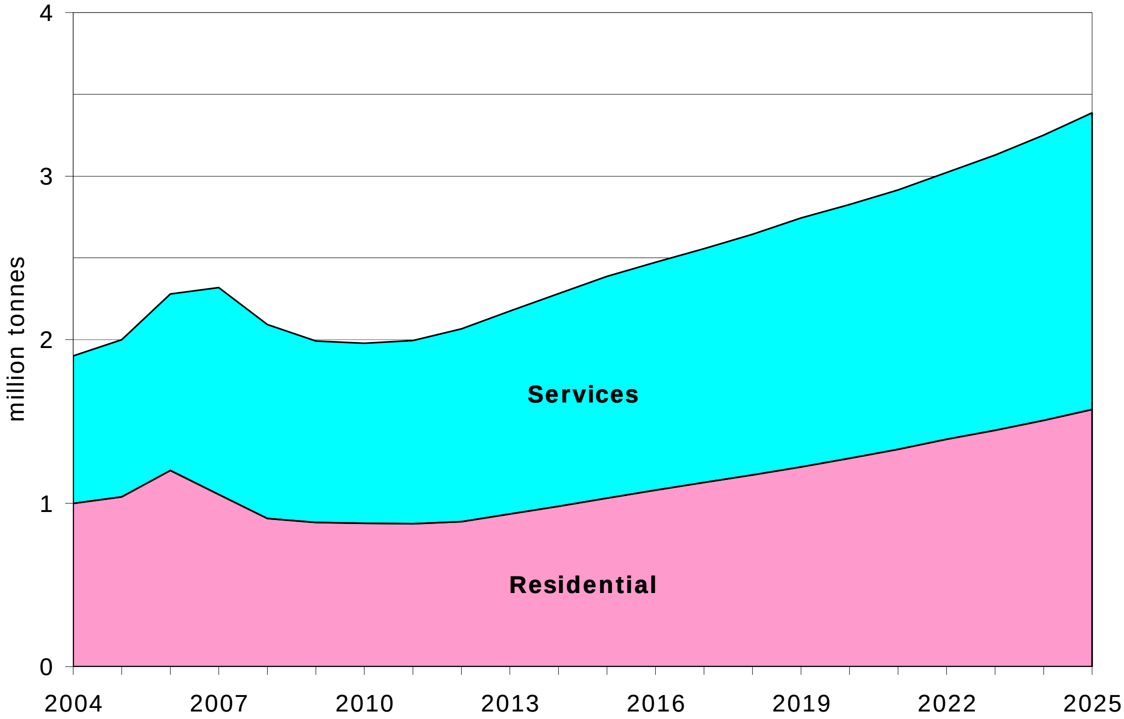
The ESRI's macroeconomic projections imply about a 50% increase in the number of households and a 25% rise in population from 2006-2025. This demographic boost will lead to a substantial medium term increase in waste arisings, all things equal.

The recession is expected to stall BMW growth between 2008 and 2012, as household incomes and service sector output soften and net immigration falls. Although the

[31] Detailed documentation on the ISus model is available at: http://www.esri.ie/research/research_areas/environment/isus/

household and service sectors are likely to respond slightly differently to recessionary conditions due to differing underlying behavioural parameters, the medium term pattern is similar.[32] When the recovery gets underway, high rates of growth in arisings are likely to resume for a period, reinforced by a rebound in household incomes and service sector output. We estimate that the average growth rate of BMW between 2012 and 2025 will be about 4% per annum.

Figure 5.1: Actual and projected BMW arisings for Ireland by sector (data through 2008 are actuals)



Source: EPA NWR actuals and ISus model adjusted for the effects of the recession

We have also modelled how BMW disposition might evolve in the absence of further policy measures. Proposed policy measures, including those set out in the draft Section 60 policy direction to cap incineration and the international review, are not taken into account here. The results are shown in Figure 5.2 below.

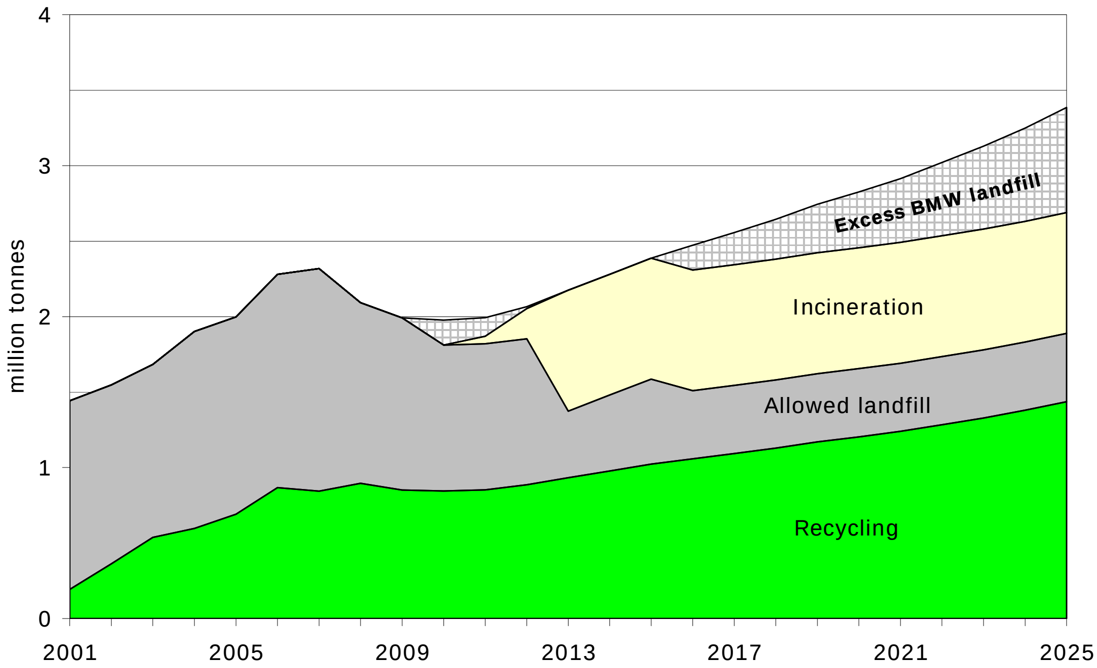
Here we assume that households' and commercial firms' current propensities to segregate recyclables and to present mixed waste material for disposal remain constant and that there are no significant changes to pay-by-use charging arrangements or rates.[33] We also abstract from future investments in waste processing infrastructure, with the only change taking place in the period being the introduction of the two incineration facilities that already have planning permission, Carranstown in Meath and Poolbeg in Dublin. When these plants might come on stream is a matter of conjecture, but for the purposes of illustration we assume that full availability of Carranstown starts in Q4 2011 and Poolbeg in 2013. Predicted socioeconomic and demographic changes over the period are taken into account, including changes in the numbers of households, persons per household,

[32] Note also that there seems to have been some waste reclassified from commercial to residential in the 2007 NWR data, leading to offsetting changes in the two series.

[33] The segment labelled 'Source segregated recycling' also includes any existing waste processed in MRFs.

household incomes and service sector output. This picture may therefore be seen as a "no-action" baseline.

Figure 5.2: Actual and projected BMW arisings and disposition for Ireland (data through 2008 are actuals)



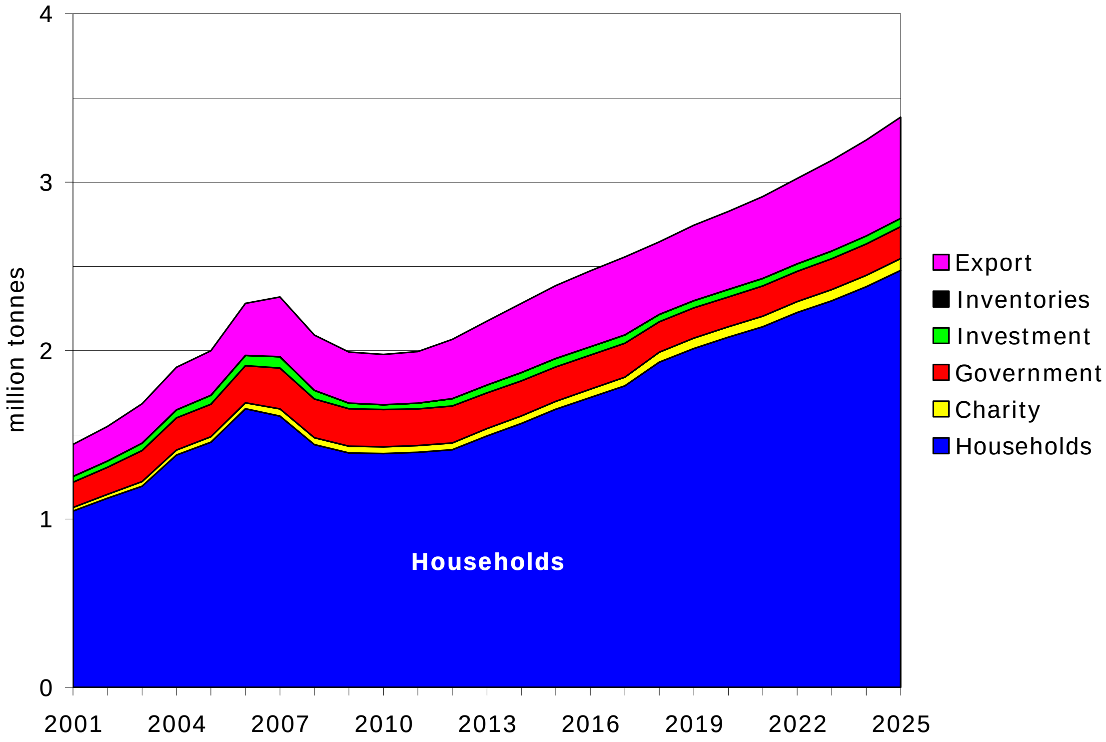
Source: EPA NWR actuals and ISus model adjusted for the effects of the recession

As noted in Section 4 above, the Landfill Directive sets limits on the quantity of BMW sent to landfill in Ireland from 2010 onwards, with the limit declining further in 2013 and 2016. This is shown in the narrowing "Allowed landfill" segment. The sum of the "Excess landfill" and "Incineration" segments represents material that will require diversion in order to meet the Landfill Directive targets.

Taking into account 2008 data, our projections indicate that the current gap between landfill of BMW and the 2010 limit is relatively modest. Once planned incineration capacity comes on stream, this gap should close until about 2015. After that, an increasingly stringent target is likely to combine with robust post-recession growth in waste generation. This implies a substantial additional requirement for diversion, presumably to be met through some mixture of collection-side arrangements and post-collection processing infrastructure. The other possible option to help close the gap is to make substantial reductions in the total BMW generated, but we suspect this will be difficult and will generally involve higher marginal abatement costs than the other options.

Unlike some environmental emissions, for example methane, BMW generation in Ireland are primarily attributable to domestic demand. Figure 5.3 below breaks down BMW emissions by category of final demand. Service sector emissions are primarily attributable to final demand from households, although exports provide the second largest component.

Figure 5.3: Actual and projected shares of final demand for BMW generation in Ireland (data through 2008 are actuals)



Source: ISus model adjusted for the effects of the recession. Decomposition by final demand use CSO input-output data

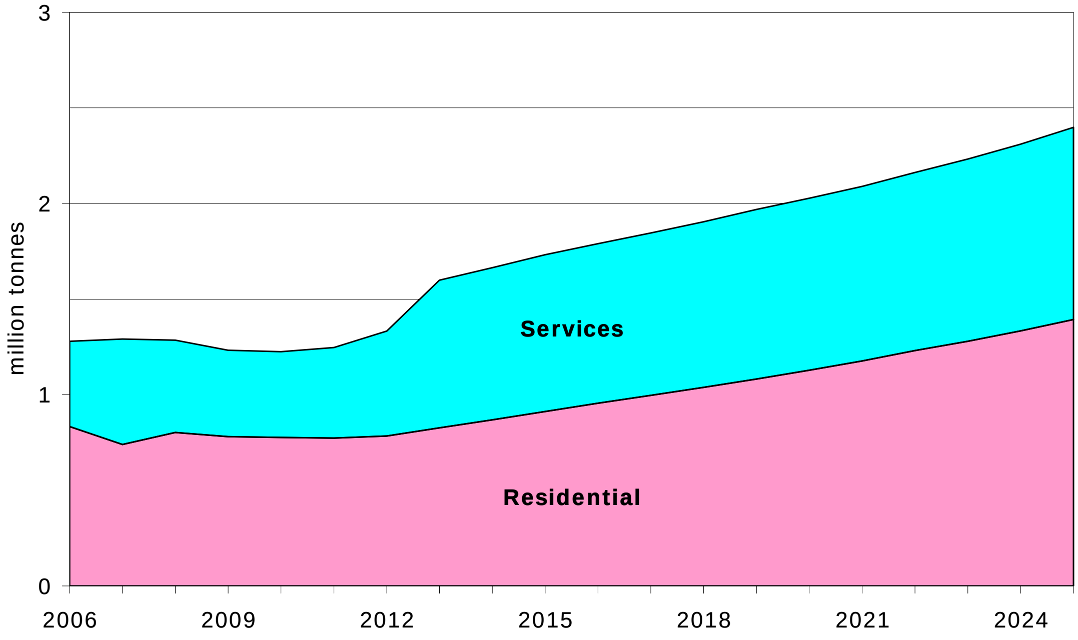
Together with the analysis of externalities arising from waste, the significance of the household sector in the causation of waste demand suggests that waste policy should be driven by domestic (or perhaps regional/local) considerations. Both causation of, and damage from, these emissions originate predominantly in domestic demand. This stands in contrast to transboundary pollutants such as greenhouse gases, which require a supranational policy response to achieve efficient mitigation.

Ideally, then, analyses of demand and supply for waste services would be carried out at a more restricted spatial level than that of the whole country. However, data limitations make this difficult. A few papers, in particular Tol *et al.* (2009) and Purcell and Magette (2009), have used imputation methods to disaggregate waste demand to small spatial areas in Ireland. EPA data are available on some aspects of waste collection at a county or regional level. In the next subsection we use estimates of BMW arisings for the Dublin region to repeat elements of the national analysis set out above.

To complement this picture, Figure 5.4 below shows the projected non-BMW component of MSW emissions for Ireland over the period from 2006-2025. This shows a broadly similar pattern as for BMW above. There is a noticeable spike in service sector non-BMW emissions in 2013, which is caused by the predicted quantities of bottom ash from incinerators.[34]

[34] Here we classify the waste output of waste management firms as attributable to the service sector of the economy, and we assume that incinerator bottom ash will not be classified as BMW (or as hazardous waste).

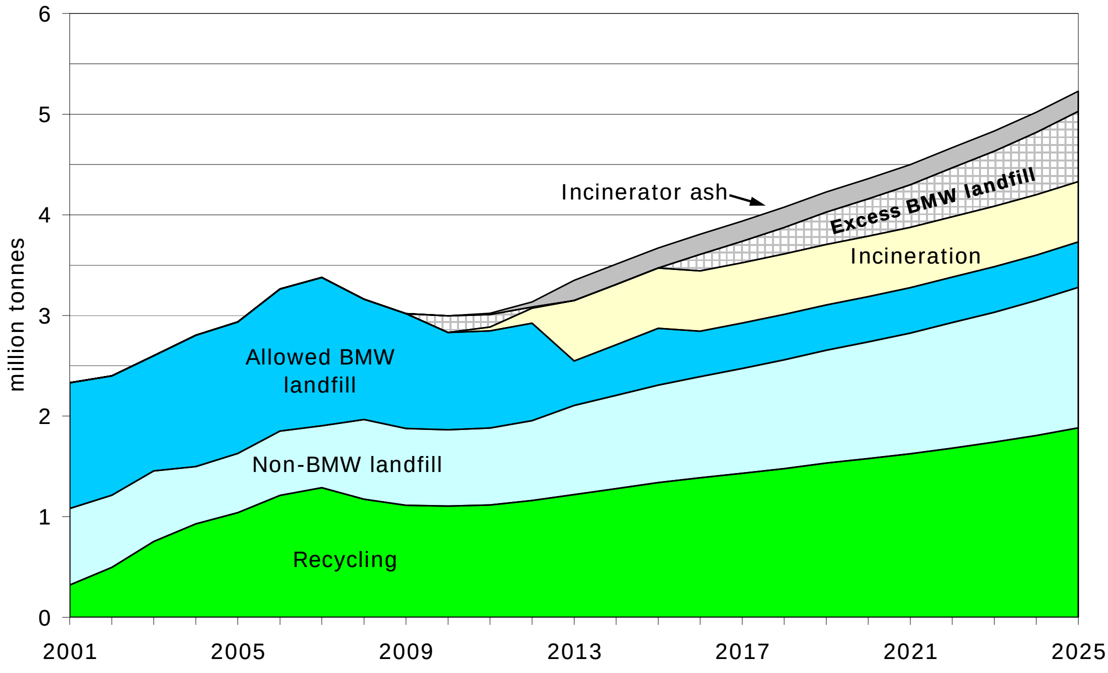
Figure 5.4: Actual and projected non-BMW municipal waste arisings for Ireland by sector (data through 2008 are actuals)



Source: EPA NWR actuals and ISus model projections adjusted for the effects of the recession

To illustrate the baseline for different waste dispositions in handling both BMW and non-BMW streams, we project all MSW in Figure 5.5 below.

Figure 5.5: Actual and projected MSW arisings and disposition for Ireland (data through 2008 are actuals)



Source: EPA NWR actuals and ISus model projections adjusted for the effects of the recession

As for BMW (reported above), our estimates suggest that MSW arisings will regain 2008 levels in about 2012, growing about 4% per annum on average thereafter. It would require an increase in planned incineration capacity over and above Carranstown and Poolbeg incinerators for the limit proposed in the draft Section 60 policy direction to cap incineration and other matters (30%, falling to 25% of MSW) to be reached at the national level. The relevant limits given our projections are set out in Table 5.1 below. However, with the addition of an incinerator in Cork and one in the South East, capacity would reach 1,000 thousand tonnes and might be binding in some future years.[35] As we shall see in the next section, the limit would likely be binding in some regions if it were applied at the regional level.

Table 5.1: Illustration of a national limit on incineration capacity for Ireland, 2010-2025

Year	Estimated MSW quantity (000s tonnes)	Proposed Incineration Limit	Limit quantity (000s tonnes)
2010	2,996	0.3	900
2011	3,021	0.3	910
2012	3,134	0.3	940
2013	3,348	0.3	1,000
2014	3,508	0.3	1,050
2015	3,670	0.25	920
2016	3,806	0.25	950
2017	3,937	0.25	980
2018	4,074	0.25	1,020
2019	4,226	0.25	1,060
2020	4,358	0.25	1,090
2021	4,498	0.25	1,120
2022	4,666	0.25	1,170
2023	4,832	0.25	1,210
2024	5,018	0.25	1,250
2025	5,228	0.25	1,310

Source: EPA NWR actuals and ISus model projections adjusted for the effects of the recession

5.4 Projected BMW Arisings – Dublin Region

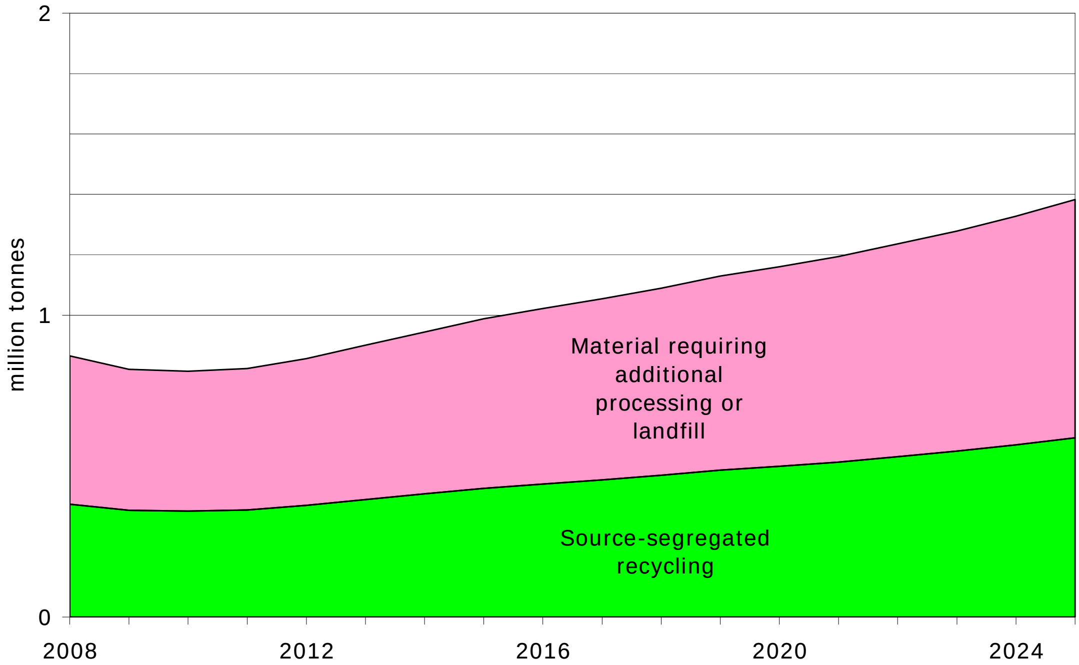
Waste collection can be analysed at a regional or even local level, but waste disposition is more problematic. Significant quantities of waste can and do cross regional boundaries for processing or disposal in other areas, which complicates the task of end-to-end accounting. In this sub-section we provide indicative projections for BMW and MSW arisings for the Dublin region, based on the assumption that national parameters for waste management parameters for households and commercial enterprises take the same values for individual regions as for the country as a whole.

Figure 5.6 below illustrates BMW arisings for the Dublin region. Since the target for diversion of BMW from landfill is a national one, we do not attempt to disaggregate it to

[35] These estimates for the capacity of the Cork and South-East incinerators are based on Eunomia (2009, p. 62)

regional level. Nevertheless, each region including Dublin will need to make a substantial contribution towards diverting waste if the target is to be reached.

Figure 5.6: Illustration of projected BMW arisings and disposition for Dublin region (data for 2008 are actuals)

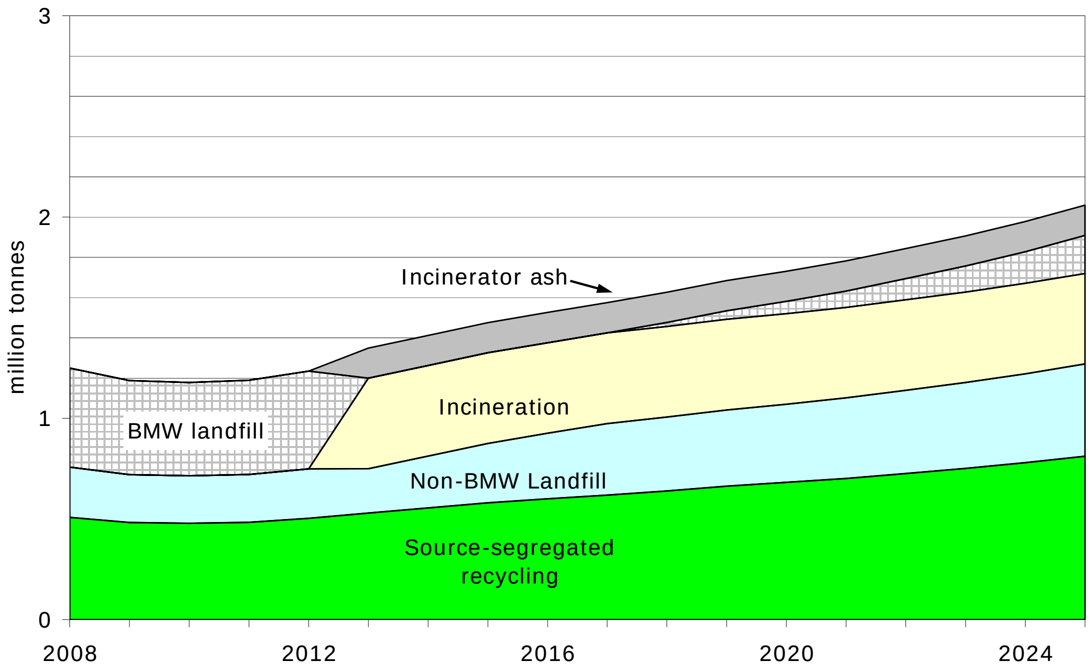


Source: ISus model adjusted for the effects of the recession and using RPS estimates of baseline Dublin BMW arisings.

Figure 5.7 shows projected volumes of MSW and includes the incineration facility that currently has planning permission (Poolbeg) as per the approach taken for our national projections in Figure 5.2 above. Assuming that Dublin's BMW is directed to this facility in whole or substantial part, it appears that Dublin will be able to more than meet its hypothetical share of the national diversion target during the period analysed. Indeed, some incineration capacity is likely to be available to take quantities of non-BMW municipal waste. Of course, if the planned facility were not available or were substantially delayed, other facilities would be required to meet the target.

It is also important to note that, particularly for a region with significant urbanisation such as Dublin, our assumption that collection arrangements will broadly remain static is probably too conservative. As noted earlier in this section, there is substantial scope to divert additional material to recycling by rolling out pay-by-use charging and a third bin where these options are economically viable, and their economic viability should be strongest in urban areas where economies of density are highest. Indeed, we note that such extensions to collection arrangements are already planned by the relevant local authorities.

Figure 5.7: Illustration of projected MSW arisings and disposition for Dublin region assuming the Poolbeg incinerator is used from 2013 onwards (data for 2008 are actuals)



Source: ISus model adjusted for the effects of the recession and using RPS estimates of baseline Dublin BMW arisings.

We can also attempt an indicative comparison of projected disposition in the Dublin region with the limits on the share of incineration set out in the proposed Section 60 direction (Section 7 below). According to our baseline projections, the share of MSW sent for incineration in the Dublin region would fall from about 44% in 2013 to about 29% by 2025. This suggests that, in Dublin at least, the limits could well be binding if applied at regional level. Table 5.2 below presents for estimates of the relevant limits for Dublin, based on our projections. The planned capacity of the Poolbeg incinerator at 600,000 tonnes is likely to be considerably higher than the proposed cap on capacity if the limit were applied a regional level.

Table 5.2: Illustration of a county-level limit on incineration capacity applied to the Dublin region, selected years between 2010 and 2025

Year	Estimated MSW quantity (000s tones)	Proposed Incineration Limit	Limit quantity (000s tones)
2010	1,178	30%	350
2014	1,412	30%	420
2015	1,475	25%	370
2025	2,059	25%	510

Source: ISus model adjusted for the effects of the recession and using RPS estimates of baseline Dublin BMW arisings.

6 Applying the Economic Approach to Waste Policy

6.1 Introduction

In this section we apply the economic approach to waste management policy. In Section 6.2 we detail some general principles that should govern waste management polices. Since this draws extensively on earlier parts of the report our discussion here is brief. However, we employ these principles in Sections 7 and 8 in respect of the Section 60 Policy Direction to cap incineration and other matters and the international review, respectively. For example, although we only outline the principle that should be employed in organising the market for household waste in this section, in discussing the international review's recommendations in this area we explore the practical implications in some depth.

Section 6.2 and Section 6.3 deal with two practical matters about which there has been much discussion. The former section addresses the question of appropriate mechanisms for meeting the targets set out in the Landfill Directive, while the latter sections deals with two interrelated issues - the appropriate mix of waste technologies and the principles involved in setting residual waste levies.

6.2 General Principles

At the outset it is important to stress that the objective of policy to maximise social welfare. Building on this government intervention in the waste management sector should aim to address market failures, in particular externalities and market power.

Economic instruments such as the landfill levy and plastic bag tax should be applied where possible, and command and control measures should be used only as a last resort. Externalities of waste management activities can be efficiently addressed by applying levies that increase the private costs of those choosing the disposition of waste equal to the social costs that their choices entail. We have noted that all of the activities in the waste management service value chain, including all options for final disposition of waste, generate some externalities. This implies that levies or equivalent economic instruments should be applied to all options in proportion to the marginal societal harm they inflict.

It may be proportional to address market power problems by reserving exclusive rights for waste authorities and encouraging them to keep costs low by permitting competition *for* the market, rather than *in* the market. Thus further use of franchising and contracting-out of services should be considered. However, in administering these policies a potential conflict of interest may arise as the local authority is not only a provider of such services but also the purchaser. Mechanisms can be put in place whereby the public operations are separated from the local authority so as to create a level playing field between public and private bidders for any tender.

Policymaking should apply a high degree of "subsidiarity", since the bulk of service costs and externalities are local or regional in their causation and incidence. Most of the relevant economic markets are also local or regional. Too much centralisation of policy will leads to a substantial risk of regulation being applied that is inappropriate for many areas. In particular, centralised command and control measures such as blanket rules on what practices and technologies may be used, to what extent, and in what places, are likely to be very costly to Irish society. Nevertheless, this should not prevent local or regional authorities learning from each other nor that co-operation between local or regional authorities should not take place concerning issues that have impacts across local or regional boundaries.

Nevertheless, we recognise that very challenging national targets have already been entered into under the Landfill Directive. Because they are challenging, they are likely to be costly to meet. At a time of national economic crisis, this makes it all the more important that measures to meet the targets be as cost-effective as possible. Any restrictions that are placed on service provision should pass a cost-benefit test. Regulatory Impact Analysis of new significant regulatory restrictions should be applied in line with published guidance.

Finally, management of policy and demand risk is key in this sector. To restate our earlier point: given the prevalence of asset specificity, long-lived infrastructures and sunk costs in waste processing, all levels of government should aim to avoid creating unnecessary policy and regulatory risk. Regional administrations should consider using contractual arrangements that situate regulatory and demand risks with the parties that can most efficiently bear them and should be free to avoid hold-up problems by credibly committing to contractual conditions in advance.

6.3 Meeting the Landfill Targets: Cap & Trade

The Landfill Directive targets are binding constraints. The levels of waste that can be sent to landfill are fixed for 2010, 2013 and 2016. Failure to meet these targets is likely to lead to large EU fines. There are currently a number of programmes in place which are designed to meet those targets. There is some uncertainty around both the volume of waste that will be generated and the success of various recycling and waste diversion schemes which are designed to reduce the flow of waste to landfill. While the landfill levy raises the price of using landfill and hence discourages its use, the levy is set based on the level of externalities not the price that will result in Ireland meeting the Landfill Directive targets. Hence the landfill levy might result in too much, or too little or just the 'right' amount of waste to meet the Landfill Directive target.

Since the Landfill Directive targets are known and fixing a landfill levy to meet those targets is a difficult task, while at the same time the conventional wisdom is that Ireland is going to have difficulty meeting the targets, then some thought needs to be given to use of an additional policy instrument to ensure compliance. We suggest that serious consideration be given to a cap and trade system that is currently employed in the ETS and similar environmental arrangements in the US.

Under this system the cap is set by the Landfill Directive for 2010, 2013 and 2016. Intervening year targets could be interpolated. Permits would be issued to use landfill for waste denominated in tonnes, perhaps by the EPA. These could be traded in order to ensure that they are used in the most efficient use is made of the permits. As with all such schemes a thorny issue concerns the allocation of the permits since these may become valuable property rights. Economists often argue that the permits should be auctioned off and the funds accruing to the State. This is not just on equity grounds, but also - at least under the ETS[36] - free allocation of permits based on existing levels of CO_2 has lead to inefficiencies. However, in practice permits are usually initially at least distributed free based on existing patterns of usage with the proportion that is auctioned

[36] For details see Matthes & Neuhof (2007).

off gradually increasing over time to reach 100%. We suggest a similar approach be adopted for waste permits for landfill.

Initial allocation would be pro-rated based on 2009 levels of waste disposal, so that firms could not engage in any strategic behaviour to effect the allocation. Subsequently the proportion of the permits that would be auctioned off would increase so that by 2016 it would reach 100%. There would, of course, have to be an allocation held back in the case of entry. Permits would be issued on a use it or lose it basis.

We also suggest this approach for another reason. Gaining acceptance of the use of waste permits for landfill is likely to be resisted. If the various programmes that are designed to meet the Landfill Directive by prevention, diversion and recycling combined with the impact of the landfill levy are effective then the Landfill Directive targets may be met and the permits will be of little value. It seems better to introduce such permits in the manner suggested above than risk considerable resistance to the idea by moving to 100% much sooner.

6.4 Residual Waste Levy & Technology Mix

Externalities are defined as the costs and benefits that arise when the social or economic activities of firms or consumers affect another group and these effects are not captured in the pricing, transfer and regulatory systems– i.e. they are external. For example, a farmer pollutes a river but does not compensate a downstream fish farm that is adversely affected. The issue thus becomes how to internalise or take into account these externalities such that societal welfare is maximised. We consider different instruments that might achieve this task, before deciding on the methodology for estimating residual waste levies. Attention is then devoted to the issue of technology mix.

Two Solutions: Levies or Command and Control Regulation

The problem of externalities is commonly addressed by bringing the private costs in line with the social costs. Levies or taxes can be imposed on the externality-generating activity, set at a rate reflecting the damage caused. In the above example, a levy would be placed on the farmer equal to the damage caused on the fish farm. A common alternative is to set permissible levels of emissions or discharges, often tied to a particular technology. The permissible level set by command-and-control regulation may be chosen to ensure that the externality-generating activity does not cause significant adverse environmental damage. In the above example, the amount of fertiliser that the farmer could apply might be limited. While there are pros and cons to each approach they are usually considered substitutes, not complements. Policy makers should select either levies or direct regulation to attain a given objective, but not both since they are both trying to set the same objective but by different means.

Consistency of Approach Across Different Sources of Externalities

It is important that there is consistency across the policies used to address different sources of externalities (e.g. MBT, cement kilns, and so on). In other words, two sources of externalities in a similar situation should be treated in a similar manner; and, any given source of externality should not be subject to several forms of instruments of intervention all designed to achieve the same objective. If there is not such consistency then there is a danger that government intervention may not only prove to be needlessly expensive, but it may also not achieve its objective. Furthermore, this would violate the basic principle of treating like case alike, while arbitrary deviations from that principle also hamper government accountability.

Command and Control Regulation: Integrated Pollution Prevention Control and Waste Licences

Licensing is the main instrument used to regulate externalities of the kind generated by waste treatment facilities (but also from manufacturing activities). This involves specifying allowable levels of emissions and monitoring compliance. For pollution from large scale production operations such as pharmaceutical facilities, cement kilns and power plants an Integrated Pollution Prevention Control ("IPPC") licence is issued by the EPA which is "designed to [ensure] … that emissions from the activity do not cause a significant adverse environmental impact."[37] Waste operations are also subject to EPA licensing to ensure that the operation "will not cause environmental pollution."[38] Each Waste Licence the EPA issues contains emission limits that the waste facility must adhere to. This type of command and control regulation sets the level of pollution that is designed, implicitly at least, to be the right or correct or socially acceptable level. It has been selected instead of imposing a levy or tax reflecting the level of externality. There is often a considerable range of values for a given externality, so it is understandable why command and control regulation is preferred.

What Role Levies?

This does not mean, however, that there is no role for levies in relation to waste treatment facilities. For example, the EU Emission Trading Scheme (ETS) sets a price for emissions of CO_2 via a cap and trade system. We recommend levies that reflect two sources of externalities from waste management. First, methane. Although waste licenses set permissible levels for licensed facilities, via landfill gas concentration limits, this gas is such an important contributor to global warming that a differentiated levy should be imposed across all the waste treatment options that are considered here. Second, the disamenity impact of each waste treatment option is likely to vary depending on the nature of the activity and other factors such as the housing density in the immediate vicinity of the waste treatment. As a result disamenity costs for each waste treatment type should be estimated so that an appropriate levy may be applied. Note, however, that if facility operators pay direct compensation to local residents in some form, this may address the disamenity externality without a need for a separate levy.

It could be argued that if a levy is to be imposed on methane that one should also be imposed on CO_2. However, CO_2 from paper, textiles, wood etc. is, firstly, largely carbon neutral and, secondly, accounted for at source. Hence it is not appropriate that CO_2 from these sources should be subject to a levy. However, what about CO_2 from fossil fuels? Incinerators are subject to the ETS and hence any CO_2 emissions from the fossil fuel component of their inputs will be taken into account, while for landfill non-biodegradable materials such as many plastics do not give off CO_2. For MBT, fossil fuel use will be captured under the forthcoming carbon tax, plastics are either recycled or sent to landfill and CO_2 from plant materials is largely carbon neutral, so again no process CO_2 is emitted. Hence we suggest there is no need to include CO_2 emissions in waste management levies.

[37] As set out on the EPA website: http://www.epa.ie/whatwedo/licensing/ippc/. Accessed 5 October 2009.

[38] As set out on the EPA website: http://www.epa.ie/whatwedo/licensing/waste/. Accessed 5 October 2009.

Recommended Levies for Landfill, Incineration and MBT.

The difference in the optimal levies for MBT, incineration and landfill revolves around methane emissions and disamenities. We have assumed that CO_2 emissions and other emissions to air will continue to be regulated through other instruments, the ETS and the emission limits already specified in each Waste Licence. The levy rates per tonne of waste implied by the analysis are summarised below and their derivation presented in Annex A below.

Landfill	€44.24 to €54.89	(Table A.10 below)
Urban Incineration	€4.22 to €5.07	(Table A.17 below)
Rural Incineration	€0.42 to €0.50	(Table A.17 below)
MBT	€0.92 to €1.45	(Table A.23 below)

It should be noted, however, that these levy rates may underestimate the total costs of using incineration and MBT, since use of these options is likely to involve additional externality-related levy payments due to the production of outputs from incineration and MBT that are sent to landfill. However, a countervailing factor is that we have not taken into account any direct payments by facility operators to local residents in compensation for disamenities. Landfill externalities not already regulated elsewhere give rise to an implied levy over double the current (2009) landfill levy rates. Both the methane and disamenity components are higher for landfill than those for the other technologies we have examined due to the high levels of biodegradable waste currently landfilled in Ireland and the impact of housing densities on disamenity calculations.

Technology Mix

We are neutral concerning the technological mix for dealing with residual waste management in terms of an a priori preference for landfill, incineration and/or MBT. There is no reason to prefer one technology over another, except on the basis of the private costs and benefits and the proposed residual waste levy that is set out above.

7 Applying the Economic Approach to Waste Policy: the Section 60 Policy Direction to Cap Incineration and Other Matters

7.1 Introduction

In this section we focus on specific issues that arise in the context of the proposed Section 60 policy direction to cap incineration and other matters. The nature of the proposed Section 60 policy direction is explored in Section 7.2. The question of how the Section 60 policy direction increases regulatory risk is dealt with in Section 7.3, while whether the SEA or RIA methodology is appropriate to evaluate the policy direction is the subject of Section 7.4. The feasibility of the 70% recovery target for MSW is assessed in Section 7.5, while Section 7.6 deals with the appropriate choice of instrument to deal with externalities. Section 7.7 evaluates another target in the Section 60 policy direction, the cap on incineration. In Section 7.8 the consequences of meeting some targets, but not others is explored. Section 7.9 discusses the impact of Section 60 on competitiveness.

7.2 Proposed Section 60 Policy Direction

The Minister for the Environment, Heritage and Local Government requested comments on a proposed Section 60 policy direction that would put a cap on incineration capacity, expressed as a percentage of MSW arisings, and other matters. The policy direction is addressed to local authorities. LAs are required to have regard of the policy direction with respect to their regional waste management policies. In order to inform the policy making process a SEA was undertaken by environmental consultations, Eunomia. According to the tender document, contract commencement with respect to the SEA was 27 March 2009, with completion on 22 May 2009. The 300 page SEA is dated June 2009. The deadline for responses to the consultation on the proposed Section 60 cap on incineration was 17 July 2009, subsequently extended to 31 July 2009. To date no decision has been made on whether or not to issue a Section 60 policy direction, but in the October 2009 *Renewed Programme for Government* there was a reaffirmation of the goal of placing a cap on incineration, having regard to the recommendations of the international review, which is the subject of Section 8 below.

The proposed Section 60 policy direction is aimed at achieving six policy objectives which are set out in Box 7.1 below. The waste hierarchy, discussed in Section 2 above, is central to the first three objectives. The cap on incineration is seen as vital in ensuring that waste that could be treated higher up the waste hierarchy is not drawn to incineration. Other objectives reflect the reduction in emissions from trucks accessing waste facilities in built-up areas to the ongoing monitoring of pollution.

Box 7.1: Proposed Section 60 Policy Direction: the Declaratory Version

The Minister for the Environment, Heritage and Local Government recently arranged for environmental consultants to prepare an Environmental Report, according to Strategic Environmental Assessment requirements, to inform the policy making process for waste management in respect of a proposed Section 60 policy direction to achieve the following objectives:

1) to ensure that incineration capacity does not reach a level such that waste is drawn to incineration which could have been dealt with by prevention, reuse, recycling, composting/anaerobic digestion of source segregated biowaste, MBT or other methods higher up the waste hierarchy

2) to ensure that the waste hierarchy is complied with in that local authorities, as waste management authorities, do not direct holders of waste to deliver it to lower elements in the waste hierarchy, thereby preventing them acting in support of waste management options at the bottom of the hierarchy

3) to ensure that the waste hierarchy is complied with in that local authorities, as waste management authorities, could direct holders of waste to deliver it to higher elements in the waste hierarchy, thereby encouraging them to act in support of waste management options at the top of the hierarchy;

4) to minimise the air pollution arising from trucks accessing waste facilities in built-up areas;

5) to ensure appropriate monitoring of air pollution in the vicinity of major waste facilities;

6) to reduce air soil and water pollution from incineration and comply with the Stockholm Convention

In advance of finalising the Strategic Environmental Assessment comments on the report's recommendations are invited from relevant stakeholders and any other interested parties.

Source: Department of the Environment, Heritage and Local Government (DoEHLG)

One difficulty with responding to the objectives as set out in Box 7.1 is that they lack specificity as to the means that should be used to meet them. As a result it would be difficult to determine whether or not the objectives of the policy direction are being complied with by a local authority. For the same reasons it would be difficult to comment on the proposed Section 60 policy direction to cap on incineration except in generalities, such as targets are fine provided that the costs and benefits have been carefully thought through or that a target set too high may not be binding while one set too low while binding might be very costly to attain.

It is perhaps for the former set of reasons that the six objectives have been fleshed out with quantification and, in other cases, greater precision as to their meaning. The results are presented in Box 7.2 below. They were included in the terms of reference for the

SEA study and were adopted by Eunomia when they conducted a SEA of the proposed Section 60 direction.

Box 7.2: Proposed Section 60 Policy Direction: Giving Substance to the Objectives

… the Minister has been considering a series of interim policy actions which would inter alia accelerate progress towards meeting the Landfill Directive targets.

Policy direction

1. The Agency [EPA] and local authorities in the discharge of their statutory functions, should ensure that incineration capacity is limited to ensure that waste is not drawn to incineration which could have been dealt with by recycling or other methods higher up the waste hierarchy. To this end, the aggregate capacity of licensed incinerators should not exceed 30% of municipal solid waste arisings, and from 2015 should not exceed 25% of municipal solid waste arisings. [Alternative text subject to SEA: The Agency and local authorities in the discharge of their statutory functions, should ensure that the aggregate capacity of licensed incinerators should not exceed 30% of municipal solid waste arisings in any region and from 2015 should not exceed 25% of municipal solid waste arisings].
2. A local authority should not attach a condition to a permit to direct that waste be taken to a landfill or incinerator.
3. A local authority may attach a condition to a permit to direct waste to a facility higher in the waste hierarchy including facilities for or which enable its reuse, recycling, treatment by means of anaerobic digestion or for the production of compost.
4. Local authorities and the Agency should ensure that trucks going to waste facilities in, or accessed through, built-up areas should be of Euro V standard, in order to reduce air pollution in those areas and as soon as practicable after the adoption of the Euro VI standard insert a conditions in permits or licences requiring trucks going to waste facilities in, or accessed through, built-up areas the subject of the permit or licence to be of this standard.
5. Local authorities and the Agency should ensure the carrying out of the monitoring of relevant pollutants in the vicinity of major waste facilities to ensure that predicted environmental quality levels are met and to establish data for epidemiological analysis
6. Local authorities and the Agency should ensure that input to incineration is controlled to prevent or reduce the potential for emissions of persistent organic pollutants, heavy metals or other pollutants, in order to reduce air, soil and water pollution and to comply with the Stockholm Convention.

Source: DoEHLG

Objective 1 places limits on the importance of incineration, with a 30% cap as a share of MSW arisings to 2014, followed by a lower cap of 25% from 2015 onwards. Objectives 2 and 3 limit the ability of a local authority in issuing a waste collection permit to direct waste to particular levels in the waste hierarchy. Objective 4 brings forward the replacement of trucks sooner than would otherwise be the case, while objective 5 is concerned with monitoring and objective 6 is concerned with specifying inputs into incineration consistent with the Stockholm Convention.

The purpose of this section of the report is to comment on the proposed Section 60 policy direction to cap incineration and other matters. In part this will consist of examining the SEA conducted by Eunomia. However, only in part, because there are wider issues raised by the policy direction that need to be addressed. These issues go to the heart of policy making and if not taken into account have the potential to needlessly raise the costs of waste management policy for no apparent gain. At a time of ongoing budgetary financial stringency such issues are of particular relevance.

7.3 MSW Policy Making: Increasing Regulatory Risk

As noted in Section 3 above, if policy revisions and change increases regulatory risk then this will increase capital costs and bias investment in other ways. While these costs, like many regulatory costs, are hidden and are hard to find on a balance sheet, they are still no less real. In order for these costs to be minimised it is important that, to the maximum extent possible, the policy environment should be stable and credible. This does not mean, of course, that policy cannot or should not change. New research may change through fresh light on the costs and benefits of a particular recycling option; technical change may lower the costs of (say) one form of waste disposal thus favouring it over other options. Policy clearly needs to be able to react to such new developments.

There appears to have been a radical change in the way in which incineration is viewed in Government environmental policy, with the Section 60 policy direction placing a cap on incineration being the culmination of that policy shift. The 2007 Programme for Government committed the government to the: (i) abolition of take or pay clauses in future projects for waste facilities in which the State or local authorities were participants; and (ii) the landfill levy should not be altered so as to give a competitive advantage to incineration.[39] Subsequently in the Strategy Statement of the DoEHLG for 2008-2010, this is elaborated upon as follows:

> In terms of waste, a suite of policies developed by the Department over the last decade has provided the framework for the implementation of national policy. This has led to considerable success, particularly in terms of recycling. The new Programme for Government indicates a further development of waste and resource policy in the direction of sustainability, in particular, to move away from mass burn incineration towards alternative technologies and to minimise waste going to landfill, *subject to the outcome of the review of the waste management strategy*. This major international review being undertaken by the Department will address how best to implement waste prevention and minimisation, and the emergence of new technologies in waste management. (DoEHLG, 2008b, p. 13, emphasis supplied).

A number of observations can be made about these developments, combined with the Section 60 policy direction to cap incineration and other matters.

First, despite no new evidence being cited or changes in technology, policy is being changed in such a way as to discourage the use of incineration and a move towards alternative technologies with MBT being mentioned in the Programme for Government. In other words, a strong preference by government is being expressed against the use of incineration. While governments, like consumers, are entitled to change their preferences there is an obligation to explain the basis of that beyond increased sustainability.

[39] Department of the Taoiseach (2007, p. 22).

Second, incinerations and MBT are juxtaposed as substitutes. It appears that in practice they are, to some extent at least, complements. A recent European Environment Agency report which examined the waste management policies of selected member states concluded that:

> Mechanical-biological treatment is used as an alternative option to incineration to treat mixed municipal waste in Estonia, the Flemish Region, Germany and Italy. Mechanical-biological treatment is a pre-treatment method, whereby mixed household waste is mechanically separated into a high caloric refuse-derived fuel product and a residue, which is first digested or composted and then sent for landfilling or to dedicated incinerators. Capacity for mechanical-biological treatment has doubled or tripled in some countries, with Italy having by far the largest treatment capacity at 240 kilograms per capita. The countries studied that use this treatment option all use or are planning to use dedicated incineration and co-incineration of the refuse-derived fuel produced to generate energy. (EEA, 2009, p. 9).

The complementarities between MBT and incineration and MBT and collection arrangements have practical implications. In particular, if a three-bin collection system is widely employed, this will remove many of the recyclables that MBT is designed to extract, increasing the net cost of the system. In addition, the net cost of a system including MBT will include the costs associated with final disposal or incineration of the material left over after treatment.

Third, as noted above policy change should be made after careful examination of the alternatives and in the light of any fresh evidence or information. Thus the Strategy Statement quite correctly commits government to moving policy away from incineration *after* the completion of the international review, which is expected later in the summer. However, the Section 60 policy direction placing a cap on incineration is being made *before* the completion and discussion of the international review. This is surely the wrong way around. There should be no policy direction until *after* the policy review, otherwise there is a danger that not only will incorrect choices be made, but a perception may gain currency that the international review is largely irrelevant since policy choices have already been made.

Fourth, a consistent mantra of government is adherence to the waste hierarchy. In the Programme for Government, for example, reference is made to the government being committed to the waste management hierarchy. In the waste management hierarchy as discussed in Section 2 above incineration is placed above landfill. Hence it is inconsistent with that approach to put in place measures that appear to treat the two as much the same, rather than favouring incineration over landfill.

The net result of these four points is that regulatory risk will be substantially increased if the Section 60 policy direction placing a cap on incineration is enacted, particularly ahead of the release of the international review study. Furthermore future policy choices are being restricted by issuing the Section 60 policy direction that might well prove costly.

The impression of regulatory risk in Ireland arising from the Section 60 policy direction, particularly amongst potential international investors, is likely to impede the development of new facilities. This is further amplified by the credit crunch, which is seeing banks moving towards incineration both as the most proven technology and with technology counterparties which, in the main, have more significant balance sheets to post the necessary bonding for process guarantees. Any investment delay leads to a greater risk of failing to meet the Landfill Directive targets. On the other hand, it is recognised that incineration brings with it the risk of additional planning delay. This is an issue which would be addressed in a comprehensive RIA.

Thus, we think that there is merit in considering deferring the implementation of the Section 60 policy direction so that it can be considered in the context of the international review as envisaged in the DoEHLG's *Strategy Statement* for 2008-2010.

7.4 SEA: Is it the Right Methodology?

As discussed above in Section 2, the SEA methodology applied by Eunomia to the Section 60 policy direction, based on earlier EPA guidance, measures gross environmental benefits of a policy change. In other words, it compares the status quo on the assumption of no policy changes to a series of different policy options. These options are compared to the status quo. If gross benefits are increased between (say) option 1 and the status quo then there is an improvement.

As argued in Section 4 above, this approach is seriously flawed. First, no attention is paid to the costs of the options under consideration. In an era of severe financial constraints this is a major shortcoming. Second, partly because of the lack of attention paid to costs only gross not net benefits are considered. If these cost considerations are not taken into account then, as set out in Section 2 above, it is by no means clear that Objective 4 of the Section 60 policy direction, which concerns more rapid introduction of Euro V and Euro VI trucks, either leads to an improvement in the environment or that the benefits exceed the costs. A cost benefit study, which could be conducted as part of a Regulatory Impact Assessment, discussed in Section 2 above, would not suffer from these shortcomings.

To meet these concerns, implementation of the Section 60 policy direction should be deferred until a Regulatory Impact Analysis has been undertaken.

7.5 Setting Targets without Costs

Targets can be useful devices if carefully thought through and based on firm evidence, with consideration of the costs and the benefits. If this is not done there is a danger that policy will be needlessly costly and perhaps unattainable. This is not some academic debate. If Ireland does not meet its targets under the Landfill Directive then heavy fines are likely to follow. Furthermore, if certain options are closed off or severely constrained by the Section 60 policy direction, such as the use of incineration, given the long lead times then there may not be enough time to reach the landfill targets set by various EU directives.

The tender document for the SEA sets out the rationale for the cap on incineration by arguing that recycling rates can be increased to 70% of MSW. It takes current recycling rates of Member States for various waste streams. These typically show considerable variance. Recycling rates for wood, for example, varies from close to 100% for Ireland to around 35% for the Netherlands; for metal packaging from around 90% for Belgium to around 40% for Denmark. It is then assumed that Ireland will be able to meet and in some cases exceed the highest rates of recycling in the EU for each waste product. The results are reported in Table 7.1. Since these target rates will need to be met by 2014 in view of the earlier discussion, this implies very substantial rates of increase across virtually all categories. Eunomia (2009) in its SEA also argues that the 70% is feasible, but enters the caveat that the policy framework for meeting the targets is "lacking sufficient force" and that "recycling rates are plateauing, and there is no policy which devolved responsibility for meeting the targets to any entity with the capability of meeting the targets." (p. 32).

Table 7.1: Disposal and recovery of municipal waste: actual (2006) and target

Material	Quantity managed (tonnes)	Quantity landfilled (tonnes)	National landfill rate (%)	Quantity recovered (tonnes)	National recovery rate (%)	Target Recovery rate (%)
Wood	219,317	15,480	7.1	203,837	92.9	95
Glass	164,181	59,873	36.5	104,308	63.5	90
Ferrous	65,285	26,697	40.9	38,588	59.1	75
Paper and Card	1,063,841	475,285	44.7	588,556	55.3	90
Aluminium	36,020	22,707	63.0	13,313	37.0	75
Plastic	327,141	263,615	80.6	63,526	19.4	40
Other metals	15,365	12,896	83.9	2,469	16.1	60
Organics	779,015	723,671	92.9	55,345	7.1	70
Textiles	176,474	166,623	94.4	9,851	5.6	50
Other	253,672	213,773	84.3	39,899	15.7	-
Total	**3,100,310**	**1,980,618**	**63.9**	**1,119,692**	**36.1**	**69.77** **(2.163 million tonnes)**

Source: DoEHLG, Terms of Reference for Section 60 Policy Direction

Apart from these difficulties there is a more fundamental criticism of the methodology used to establish these targets. Member States vary across a whole range of dimensions such as productivity by sector, labour costs, urban/rural split, household type (apartment compared to a house), market size, policy interventions,[40] and so on. Differences in recycling rates are thus fully consistent with this general picture and no doubt reflect conditions particular to each Member State. Thus the idea that Ireland and presumably each Member State can achieve the highest recycling rate of the highest member state without taking into account these differences is extremely unlikely to be successful or cost effective.

Thus, we think that there is merit in the DoEHLG carefully specifying how and by what policy and other mechanisms Ireland will reach any recycling targets that it ultimately adopts.

[40] For example, in Germany and the Netherlands, paper is collected separately and thus more valuable.

7.6 There is More than One Way to Skin a Cat – But Do We Have to Use them All?

The second objective of the Section 60 policy direction is that the local authority should not attach a condition to a waste permit that directs waste to landfill or incineration.[41] Earlier the Programme for Government stated that the landfill levy was not to be changed in such a way to give a competitive advantage to incineration. Subsequently in circular No WPRR 04/09, dated 29 May 2009, the Minister of the Environment announced that there will be an increase in the landfill levy and the introduction of a levy on incineration.

A number of points can be made. *First,* as discussed above in Section 2 those who are best placed to bear risk should bear the risk. In the case of incineration it was pointed out that the local authority might be best placed to bear the "demand risk" associated with ensuring that waste is delivered to an incinerator. Contractual arrangements such as "take or pay" contracts can also help deal with the problems of regulatory risk and asset specificity arising from the high portion of sunk cost involved in building incinerators or other types of capital-intensive facilities. Removing such contractual options is likely to be economically inefficient and needlessly raise costs.

Second, as discussed in Section 2 prices do not always reflect externalities. One solution is to impose levies or taxes that take this into account. In the case of landfill and incineration therefore the levies should reflect these externalities that may not already be captured in the current price. However, it is inconsistent to then argue that the pricing should not give incineration a competitive advantage over landfill. If the externalities caused by incineration are lower than landfill, so be it; the levy should represent the relative costs, not some predetermined outcome with no empirical support.

Thus, we think that there is merit in setting any levies on landfill and incineration or MBT by carefully considering the externalities to which they give rise. Our own views on the appropriate levies were set out in Section 6 above, while in Section 8 below we comment on the levies proposed in the international review.

7.7 Should there be a Cap on Incineration?

Taking into account only the two incineration plants for which planning permission has thus far been granted, the Section 60 policy direction's proposed limits on incineration would appear not to be binding if the proposed limits were set on a national basis. However, if they were set on a regional basis then, as shown in Section 5, the limits would be binding for the Dublin region. However, this is likely to very much understate the effect of the Section 60 policy direction.

It should be noted that the very fact that there is a limit on incineration will affect behaviour; a lower level of capacity is likely to be installed. Absent the Section 60 policy direction, it is likely that a number of incinerators would be built in addition to Poolbeg and Carranstown. It appears that the regional waste plans of many authorities contain an element of incineration, with the planned Cork incinerator already having an EPA license. If the Section 60 policy direction were put in place then some of these plans may not go ahead. Some may be deterred because there of uncertainty over whether the cap will be breached; hold-up problems caused by the banning of take or pay contracts; and the rules concerning the landfill and incinerator levies. Hence the fact that the policy direction

[41] The issue of whether or not the local authority should have the power to direct waste is addressed in Section 8.12 below.

may not be binding, based on the analysis in Section 5, at the national level and only for one region, should not be construed to mean that the direction would not have the effect of preventing incineration plants from being built that otherwise would be absent the policy direction.

Turning to the incineration cap, it is, to use the words in Section 2, a command and control measure as opposed to an economic instrument such as a levy. Command and control mechanisms are likely to impose needless costs on the economy. As noted above, targets can work well if they are carefully thought through and firmly grounded in sound analysis. However, when a target lacks such a basis – as occurs with the incineration limits set in the policy direction – then they have the potential to increase the costs of meeting a given policy target. We use the term ‘potentially’ because the target may not be binding; however that would not be the case with respect to incineration limits for the Section 60 policy direction.

Much better in our view would be to ensure that the private cost of using each waste processing option will be set at a level reflecting the externalities it causes. Of course, setting such prices requires information as to the costs and benefits and an appropriate methodology which were set out in Section 6 above.

7.8 Targets: Aspirational or Credible - Implications

There are several sets of targets relating to MSW, some set at the EU level, others by the Member State. The attainment of each set of targets depends on a complex set of factors relating to technical feasibility, fines or other sanctions for not meeting the targets, whether or not the targets would have been met without the policy intervention and the economic incentives to comply with the targets. Since the strength of these factors is likely to vary from target to target, this means that some targets are more likely to be met than others. Where there are a series of inter-related targets, as is the case with MSW, it is important to consider whether or not a particular set of targets is likely to be met and the consequences if some targets are met while others are not.

In terms of waste management policy the most binding targets are those contained in the Landfill Directive which has been discussed earlier in report. Having credible policies in place to meet these targets is vital, not least because failure to meet these targets will result in Ireland paying substantial fines for non-compliance. However, recent MSW policy changes at the national level in Ireland may make these targets harder to meet.[42]

Next we consider the targets in the Section 60 policy direction concerning the cap on incineration and the corollary, a 70% target for recovery rate from MSW. The discussion above concerning the 70% recovery rate from MSW questions not only whether it is technically feasible but also whether the policies are in place to realise the target. In contrast, with respect to incineration there are credible policies to ensure that the 30% cap to 2014 and 25% thereafter will be realised due to a series of policies, discussed above, which actively discourage incineration:

- The landfill levy cannot be changed in such a way to give a competitive advantage to incineration;

[42] An additional landfill-specific objective, which was included in the 2007 Programme for Government, is also unlikely to be met: that of ensuring that no more than 10% of waste is consigned to landfill.

- Local authorities cannot enter into any future contracts to build incinerators that include take or pay clauses;
- Local authorities cannot attach a condition to waste collection permits that directs waste to landfill or incineration; and,
- An increase in regulatory risk associated with building an incinerator because of the unanticipated change in the policy environment concerning MSW in general and incineration in particular.

Thus it appears that the incineration caps – if all these policies are in fact implemented – will be complied with perhaps substantially undershooting the cap on a national if not Dublin basis, but that the recovery rate of 70% will almost certainly not be attained. Waste that would have gone to incineration and some of the waste that was scheduled for recovery will be diverted to landfill. Ireland may not comply with the targets set in the Landfill Directive, resulting not only in fines but also perhaps a lower quality environment. The latter implication was dealt in Section 6 above.

7.9 Effects on Competitiveness

Competitiveness is a broad concept concerned with the ability to compete successfully while at the same time raising living standards. Government has an important role to play in ensuring an appropriate framework within which investment and other decisions vital to economic development can be made that goes well beyond the night watchman functions of providing law and order and securing property rights. Consistent, credible and predictable policies are likely to create a framework that is conducive to economic development and, as such, contribute positively towards enhancing Ireland's reputation as a place to do business. Our findings suggest that the Section 60 policy direction on MSW policy and corollary policies will not contribute positively towards that reputation and will thus harm economic development and competitiveness, a conclusion strengthened by the observations in Section 8 below.

8 Applying the Economic Approach to Waste Policy: the International Review of Waste Management Policy

8.1 Introduction

An international review of waste management "plans, practices and procedures" was promised in the 2007 Programme for Government, with an undertaking to act on its conclusions (Department of the Taoiseach, 2007, p. 22). The international review is designed to "greatly assist in the determination of the best mix of technologies suited to Ireland's needs" according to the 2007 DoEHLG *Annual Report*.[43]

The technology theme was reflected in the comments on the international review in the DoEHLG's *Strategy Statement 2008-2010*,

> The new [2007] Programme for Government indicates a further development of waste and resource policy in the direction of sustainability, in particular, to move away from mass burn incineration towards alternative technologies to minimise waste going to landfill, subject to the outcome of the review of the waste management strategy. This major international review being undertaken by the Department will address how best to implement waste prevention and minimisation, and the emergence of new technologies in waste management (DoEHLG, 2008b, p. 13).

Subsequently, these themes were linked not only to best practice but also to the improved "delivery of national objectives and EU obligations." (DoEHLG, 2008c, p. 16).

The EPA in 2009 echoed the importance of the international review when it listed, under the priority actions required,

> Delivering the new waste policy on foot of the international review of waste management as quickly as possible – the international review is due for completion mid-2009 – to provide certainty and to allow for accelerated investment programmes that are necessary if organic waste is to be treated and landfill avoided (EPA, 2009a, p. vii).

The terms of reference for the international review are very wide ranging.[44] The objectives for the international review are set out in Box 8.1 below. As such the objectives elaborate and extend the DoEHLG references to the international review noted above. Not only is the international review to assist Ireland move towards a sustainable resource and waste policy, together with any appropriate changes to the current legal institutional and organisational arrangements, but it must also provide ways in which Ireland can meet or exceed national, EU and other international objectives and requirements. Thus the objectives contain a mixture of the strategic and tactical – in particular meeting the Landfill Directive targets.

The international review was undertaken by a consortium led by the UK-based Eunomia Research and Consulting, together with institutes and consultancies located in Ireland, Belgium, Germany, Austria and Italy.[45] Eunomia, of course, was also commissioned to

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[43] DoEHLG (2008a, p. 12).

[44] For details see DoEHLG (2008c, pp. 27-31).

[45] The terms international review will be used interchangeably with Eunomia *et al.* and the Eunomia Consortium.

conduct the SEA of the Section 60 policy direction to cap incineration and other matters considered in Section 7 above.

The international review, which was released on 19 November 2009, consists of a 78 page summary report which, in turn, is based on 65 annexes, a total of 1204 pages.[46] The summary report contains twenty-five recommendations[47] that are intended to provide a roadmap for future waste management policy in Ireland. The Minister for the Environment, Heritage and Local Government anticipates that he will bring "… a new policy statement to Government with a view to its publication in the New Year" (DOEHG, 2009).

Box 8.1: Objectives of the International Review of Waste Management Policy

The study should: -

- identify possible changes to policy at national level in order to assist Ireland to move towards a sustainable resource and waste policy including minimising the creation of waste and self-sufficiency in the reuse and recycling of materials, and
- examine the legal, institutional, and organisational arrangements currently in place and analyse potential changes, which could assist in achieving Ireland's policy goals, and meeting national and international obligations.

The overall policy goals of a sustainable resource and waste policy include: -

- *minimising waste generation and the hazardous nature of certain wastes*
- *minimising raw material use, especially non-renewable resources*
- *minimising energy use, especially non-renewable resources*
- *minimising pollution, including eliminating or restricting emissions of persistent organic pollutants in line with the Stockholm Convention*
- *protecting and promoting public health*
- *maximising economic benefit, including the revenue that can be gained from the waste resource, providing the above goals are met*
- *maximising opportunities for enterprise in reuse, remanufacturing and reprocessing*
- *a sustainable production and consumption approach*.

Given Ireland's economic situation and the lack of lock-in to existing waste technologies, Ireland is in a position to reach and lead best practice in resource recovery and waste minimisation policy. The study should approach the policy area on that basis. In addition, the policy must provide for meeting, and where appropriate exceeding, national, EU and other international objectives and requirements.

Source: Department of the Environment, Heritage and Local Government (DoEHLG) (2008c, p. 27, emphasis supplied).

[46] When account is taken of the RIA, released at the same time as the international review, the page count goes up to 1405. The RIA referred to the plastic bag tax and certain waste facilities. For details see AP EnvEcon (2008) for details.

[47] The recommendations are number 1 to 24, with the 25th having no number as such, but it is clearly a recommendation. For details see Eunomia *et al* (2009, p. 62).

Perhaps, not surprisingly in view of the wide ranging terms of reference for international review, the twenty-five recommendations refer to many aspects of waste management policy and practice. Greater source separation by households is recommended. A series of targets are set for recycling and residual household waste. A levy structure is proposed for residual waste, the income from which, it is recommended, should be used to finance priority areas under the aegis of the Environmental Fund. The collection of household waste should be the responsibility of local authorities that would decide the most appropriate collection method model: public sector; private sector; or joint public/private sector. Finally, the 10 regional waste management plans should be replaced by a single national waste plan, with ultimate responsibility resting with the Minister for the Environment, Heritage and Local Government.

It is the purpose of this section of the report to comment and evaluate not only on the international review's recommendations, but also the underlying analytical and evidentiary base. As with the discussion of the Section 60 policy direction to cap incineration and other matters considered in Section 7 above, the international review will be evaluated using the economic approach that has been developed in the earlier sections of this report. In other words, the purpose of the section is not so much to determine whether or not the terms of reference have been satisfied, although from time to time some observations in that respect may be made.

The section is structured into four parts. The first part looks at two methodological issues: first, the approach used by the international review to learn from the experience of waste management practice and policies in other jurisdictions (Section 8.2); and, second, the rule or approach used to decide how to select one policy over another (Section 8.3). The second part deals with what is missing from the international review. Here there are *two* large gaps: no discussion of choice of technology. References to MBT and incineration are conspicuous by their absence (Section 8.4); and, the lack of justification for the policy premise that underpins the international review (Section 8.5). The third part of the section reviews the twenty-five individual recommendations (Sections 8.6 to 8.14). The fourth and final part of the section evaluates whether the international review provides a roadmap for future waste management policy in Ireland (Section 8.15).

8.2 The International Experience: What Can it Tell Us?

A small open economy such as Ireland will inevitably have much to learn from the experience of other countries and regions. Waste management policies are no exception to this generalisation. However, in order to learn from other countries and regions at least three sets of issues need to be considered. We briefly outline these conditions and provide a practical illustration of their application in a recent European Environment Agency project. Given this as necessary background attention then turns to the international review, both in terms of the degree to which these three conditions are met and the lessons that international review draws from the experience of others. Of course, meeting these three conditions is not all within the remit of the Eunomia consortium since, for example, the terms of reference for the international review is the responsibility of the DoEHLG.

Clear Research Questions

In undertaking any research project, be it an international review of waste management experience or a review of the introduction of the brown bin by Dublin City Council, the policy issue or research question or purpose of the research should be clearly identified and defined. In other words, why are we examining the waste management policies of

Germany or Flanders? On what issues do we want to learn lessons from others? What questions or issues motivate the research?

This is not some academic or debating point. If the policy question is not well defined and clearly specified then the researcher will not know what is required, what they are looking for and what questions to answer. Amorphous, wide-ranging and ill-defined terms of reference are unlikely to lead to a satisfactory outcome. It is a like trying to nail a jelly to the wall.

A Common Framework

Countries and regions differ in a multitude of ways in terms not only economic, social and spatial characteristics, but also policies in terms of their range, nature, goals, implementation and success. The characteristics and policy outcome are, of course, likely to be related. A society, for example, that has a strong preference for a clean environment, is much more likely to efficiently separate household waste by stream and much less likely to burn household waste illegally. If such differences are not taken into account then inappropriate lessons are likely to be drawn.

Thus in drawing on the international experience careful attention needs to be given to developing a common framework or template that takes into account such characteristics and policies. If a series of country and region case studies are being undertaken then this framework would need to be carefully worked out so that it is applied consistently across the case studies such that lessons can be drawn.

If the literature is being surveyed all relevant factors – policy and non-policy – also need to be taken into account so that spurious lessons are not drawn. A similar caveat applies to quantitative analysis.

Relevant Country/Region Selection Criteria

Obviously an international study needs to select a sample of countries and/or regions from which to draw lessons. A set of criteria is needed. These should of course relate to the research question being asked. It might, for example, be appropriate to select countries and/or regions that resemble Ireland’s economic, social and spatial characteristics, but at the same time vary considerably along the selected policy question being considered.

If the literature is being surveyed then these factors would need to be taken into account in drawing lessons. Similar considerations would need to be taken into account if instead of the case study approach described above a large cross-country/region quantitative exercise were undertaken.

In the discussion in Section 6 above and Annex A below on setting levies based on externalities, for example, externalities estimates from other jurisdictions were used. However, the results for other countries were not simply read across unadjusted. Instead, to the maximum extent possible, allowance was made for differences between Ireland and (say) the Netherlands or the UK.

An Illustrative Example

The European Environment Agency has recently undertaken a study designed to learn from the experience of the EU. Research questions were posed; a methodology developed; a series of case studies were commissioned; and, a summary report issued (EEA, 2009). We use this project to illustrate the way it addressed the above three issues.

The research questions were clearly identified:

- To what extent has waste management practice changed in the last decade?
- How much of the change was due to the Landfill Directive (and other EU instruments)?
- What measures and institutional arrangements did countries introduce?
- Which measures and arrangements proved most effective in different national and regional contexts? (EEA, 200, p.7).

In order to ensure that the lessons could be drawn and that the results could be compared a common methodology was developed (EEA, 2009, Table 3.1, p. 22). This methodology, in line with the research questions, sought to explain the factors influencing the effectiveness of a policy of diverting BMW from landfill. These factors were divided into those relating to:

- BMW landfill policy (e.g. landfill tariffs/gate fees);
- Waste production and collection (e.g. BMW generation per capita);
- The landfill sector (e.g. landfill residual capacity);
- Incinerator sector (e.g. dedicated incinerator capacity); and,
- Material recycling and recovery sector (e.g. MBT capacity).[48]

These case study country/regions were: Estonia, Finland, Flanders, Germany, Hungary and Italy. These included areas with quite different records of diverting BMW from landfill as well as including some newer as well as more established Member States. Some quantitative analysis was also undertaken. The summary report drew lessons from these case studies

International Review: Clear Research Questions?

What is striking about the international review is that the terms of reference is not so much that they do not provide the basis for a well specified set of questions/issues that can guide the research. Rather the terms of reference provide the basis for far too many research questions and too wide a range of issues. Furthermore, it is not clear how the research questions/issues are related.

There are at least two sets of objectives set out in Box 8.1 above: strategic longer terms objectives relating to a sustainable resource and waste policy (text in italics in Box 8.1); and, more short term ones related to meeting various targets in, for example, the Landfill Directive (the last sentence of Box 8.1). We consider each in turn.

Achieving a sustainable resource and waste policy is broken down into at least eight[49] wide ranging goals varying from promoting public health, to minimising waste generation to maximising economic benefit. Formulating a research question around each goal is possible (e.g. what are the determinants and what policies are best suited to minimising waste generation and the hazardous nature of certain wastes?). However, this has to be done for at least eight goals, each one of which is likely to be a major research project in and of itself.

[48] For further details see EEA (2009, Table 31, p. 22). The methodology is explained in more detail in Mazzanti and Zoboli (2006).

[49] The objectives set out in Box 8.1 above imply that there are more than eight goals.

The short run goals involve meeting the various targets set down in, for example, the Landfill Directive. As set out in Table 4.1 above these targets relate to the level of BMW for 2010, 2013 and 2016. Here the research questions are clear: Will Ireland meet the Landfill Directive targets on unchanged policies? If not, what policies should be introduced in order to reach those targets? Irrespective of whether the targets will be met, what steps can be taken to ensure that the targets are exceeded? Given that in the short-run many characteristics and policies are fixed or can only be changed at substantial cost, it is likely that there will much less that the international experience can teach Ireland in meeting goals for 2010 and 2013, but rather more for 2016 and beyond.

Thus it appears the terms of reference can be used as the basis for a series of research questions that can be reasonably well specified. However, this does not mean that the terms of reference are not subject to criticism. On the contrary, there are at least five grounds for concern.

First, there are, arguably far too many research issues for a single project to address satisfactorily. As noted above, each question is a major research project in itself.

Second, the research questions are not ranked in importance, so that there is no guide to the researcher as to whether or not more or less time and resources are devoted to (say) “addressing a sustainable production and consumption approach” or “minimising raw material use, especially non-renewable resources.”

Third, as Eunomia *et al.* (2009, p. 3) remark it “is clearly not possible to simultaneously maximise and minimise across this [the sustainable resource and waste policy goals] range of parameters.” A ranking is needed in order to be able to say that goal x is superior to goal y. This is not contained in the terms of reference.

Fourth, the status of the goals set out in the terms of reference with respect to a sustainable resource and waste policy is not clear. They do not appear to be the result of any public debate or indeed draw on existing policy statements. If the goals had been the result of such a process then they would command support and perhaps as a result of that debate any conflicts among goals removed and a ranking of goals developed. Hence the wisdom of commissioning research on such a broad array of goals without such support can be questioned.

Fifth, the research questions represent a mixture of longer term strategic goals and shorter term aims. This raises the issue of the need to take into account any interdependencies between the two sets of research questions.

In sum, the research questions that flow from the objectives of the international review appear to be too wide-ranging, with no attempt to rank the objectives or provide guidance if the objectives conflict, while the providence of the goals is not clear.

International Review: A Common Framework

The summary report sets out the way in which the international review examined the international experience:

- Seeking to understand the current state of affairs in Ireland, and what relation it bears to existing policy;
- Understanding what the empirical evidence suggests is already possible in other countries, and elicit the key policy drivers which have influenced outcomes;

- Estimate how much further one could go - what might be possible - in the most favourable outcome (in other words, what 'best policies' might achieve where best technologies / outcomes occur); and
- Recommend changes to policies and institutional structures, in the light of a) to c), which will enable Ireland to deliver the most favourable outcome. (Eunomia *et al.*, 2009, p. 5, internal references omitted)

There is much to be said for such an approach. We consider each of the four elements in turn.

It is clearly important to have a firm understanding of the current state of affairs in Ireland with respect to waste management and its relationship with existing policy. This is the point of departure for the international review's recommendations.

In order to apply the lessons from the international experience, it is essential to understand the key drivers which have influenced outcomes elsewhere. Only then can these lessons be applied sensible to Ireland. For example, high recycling rates in country x may reflect a much higher population density than exists in Ireland, suggesting that if a similar policies to those in country x are applied in Ireland, they may not lead, other things equal, to the same outcome. If due recognition is given to such factors, as noted above, then valuable lessons can be learnt that permit sensible policies to be recommended that can subsequently adopted by government.

The next element of the approach involves applying the lessons from the international experience by estimating what might be possible in terms of the most favourable outcomes, using what are judged as the best policies in terms of technologies and outcomes. The reference to technologies no doubt reflects the reference to technologies in the objectives of the international review reproduced in Box 8.1 above and in various DoEHLG and EPA documents referred to in Section 8.1 above.

Finally, the international review will make recommendations based on these results of the findings and results of the first three steps. These changes will be both to policies and institutional structures. The reference to the institutional structure reflects the terms of reference of the international review to "examine the legal institutional and organisational arrangements" (Box 8.1 above).

There are a number of comments that could be made on this approach, which are mainly confined to the last two aspects of the international review's methodology. *First*, it is not clear why best policies need to be judged in terms of their technology *and* outcome. Surely the emphasis should be on the outcome in terms of, for example, "minimising waste generation and the hazardous nature of certain wastes." Policy, it could be argued, needs to be technology neutral as between (say) MBT and incineration, provided, of course, that the costs of using any technology to reach a certain target are appropriately taken into account. It may be that the reference to best technology is a reference to the preference for MBT over incineration discussed in Section 7 above. If this is the case then it should be mentioned explicitly rather than in the rather elliptical manner set out above. As we shall see in Section 8.4 below one of the shortcomings of the international review is the absence of any guide as to technology and infrastructure choices.

Second, it is not clear what is meant by the most favourable outcomes. Are they the rather general objectives set out in Box 8.1? Does the most favourable outcomes reflect or take into account the stated preference for MBT over incineration and if so how? Does a favourable outcome mean that the Landfill Directive targets can be met?

Third, there seems to be a missing link in the approach that is being adopted here between the international experience and applying it to Ireland. The inference seems to be that the recommendations will be formed by selecting the best technologies/outcomes based on the international record. However, what is needed is a rule to select among these possibilities so as to select the best option for Ireland. This is an issue that is addressed later in the summary report and discussed further below in Section 8.3.

In sum, it appears that the approach of the international review to learning from others asks some of the right questions, but there may be difficulties transferring that experience into practical policy recommendations. That step will be examined when we come to consider the recommendations of the international review below.

International Review: Relevant Country/Region Selection Criteria.

The international review appears to be primarily an examination of the existing literature and experience rather than an exercise in undertaking primary data collection and policy analysis.[50] The international review conducted a survey of the existing literature on the international experience with respect to a wide range of instruments, from recycling targets to landfill and incineration levies; statistical analysis; and specific policies. However, this does not mean the Eunomia consortium did not have to make a decision about which countries and/or regions to study more closely. It is not at all clear how the international review selected certain countries/regions for closer study and attention as evidenced by their inclusion in the annexes.

International Review: What are the Lessons?

As noted above the international review summarises the lessons from the international experience in the summary report in four pages (Eunomia *et al.*, 2009, pp. 31-34). The major lessons are set out in Box 8.2 below. These are, perhaps not surprisingly very general lessons given the broad mandate of the international review in terms of the objectives and goals set out above. In some cases it appears that it is not a lesson that is being set out, but rather a description of the experience or preference such as the discussion of incineration in the last lesson in Box 8.2.

Many of the lessons seem unexceptional such as need to combine policies effectively, while in a number of other cases the lessons to be drawn from the international experience are tentative with the use of words such as “appears to be”, “tend to be”, “probably have”, “might be regarded as leaders” and “seem comfortable.” As a result it is not clear how much reliance can be placed on the Eunomia consortium’s general conclusions drawn from the research on the international experience as a basis for policy.

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[50] One notable exception is the research on household waste collection costs. For details see Eunomia *et al.* (2009, Annex 64).

Box 8.2: The International Review: Lessons from Abroad

"There is a need to combine policies in effective packages" (p. 31)

"The studies [cross-country studies with regard to specific themes, or policies or waste streams] offer little by way of statistically conclusive evidence" (p.32);

"Producer responsibility schemes, probably have a marginally stronger effect on waste generation than others as a consequence of their funding mechanism" (p.32);

"As regards recycling and composting/digestion, key instruments tend to be:

* Producer responsibility mechanisms, especially where these incorporate high targets, and incentives for meeting these. ….;

* Targets for recycling, often incorporated within producer responsibility mechanisms;

* Mandates for separate collection of targeted materials. … ; and

* Policies which make residual waste treatment and disposal expensive, including:

- Landfill taxes and, as used in a growing number countries, incineration taxes; and,

- Landfill bans and restrictions;

With the exception of Germany and some North American cases, the use of landfill bans … is always accompanied by the deployment of landfill taxes" (p.33).

It is not possible to estimate the relative significance of the aforementioned policies but "[T]he basic principle of policy is simple:

1. The economics of source separation is heavily influenced by the avoided costs of disposal;

2. Positive instruments such as producer responsibility, mandates and targets are used to pull material into recycling and composting / digestion; and

3. Instruments such as levies and bans (the latter, having the effect of an infinite levy) are used to increase the costs of disposal and treatment of residual waste, so 'pushing' source separation" (p. 33);

"As regards residual waste, countries which might be regarded as leaders in this regard seem intent on either:

* Reducing the amount of waste landfilled; or

* Reducing the amount of waste which is landfilled without prior treatment.

The instruments used are the same as those used to support recycling efforts, namely, levies and bans on landfill and incineration, and in some cases, residuals treated by mechanical biological treatment (MBT)" (p. 33);

"It goes without saying that taxes and bans on *landfill* will tend to move waste away from landfill. They will not, in and of themselves, determine that waste will be recycled or composted / digested, let alone, prevented or re-used. The fate of material is determined by the relative cost of alternative management options and / or other mechanisms designed to increase prevention of waste, and re-use, recycling and composting / digestion of materials" (p. 33, emphasis in original)

"Some countries clearly focus more upon ensuring that waste is not landfilled, and seem comfortable with a high proportion of waste being incinerated. Others use a range of accompanying policies to ensure that waste is not simply shifted from landfill to incineration or they ensure that measures to encourage movement of waste into recycling are in place beforehand," (p. 33).

Source: Eunomia et al. (2009)

Conclusion

There do not appear to be strong unequivocal lessons from the Eunomia consortium's review of the international literature on waste management policy. It may reflect an over ambitious terms of reference combined with a series of predetermined govern-mandated technology options such as promotion of MBT and discouragement of incineration. It is, perhaps, more likely that the international experience will be more relevant to individual policies that are the subject of the twenty-five recommendations discussed below.

8.3 A Decision Rule for Recommendations: Costs, Benefits and Economic Implications

The international review conducted by the Eunomia consortium suggests that its recommendations can inform the choice of policies for Ireland. In discussing its approach to the international experience and "with a view to improving domestic policy," Eunomia *et al*. (2009, p. 5) state that their approach is based on what might possible, based on "what 'best policies' might achieve where best technology/outcomes occur." Best policies provide an opportunity set from which to select a set of recommendations and new policies. This is consistent with the apparent tentative nature of the lessons to be drawn from the international experience.

This raises the obvious question of what considerations or decision rule should be used when making recommendations for a better waste management regime. How should one option be selected over another? Two rules were discussed in Section 7 above: the SEA approach which was used to evaluate the Section 60 policy direction on incineration and other matters; and the cost-benefit approach favoured by the economic approach to waste management policy and which is part of the Regulatory Impact Analysis referred to in Section 7 above. The problem with the SEA approach is that it does not take into account the costs, only the benefits, while both are considered in cost-benefit analysis. Not taking into account costs when making recommendations in the current economic climate is an especially egregious error.

In this respect the international review marks a welcome break from the SEA methodology used to evaluate the Section 60 policy direction to cap incineration and other matters. Costs, cost effectiveness, economic implications and benefits are given more prominence. This is more consistent with the economic approach that we consider to be the appropriate methodology for guiding waste management policy and evaluating different policy options.

In the 'Introduction' of the international review's summary report, for example, it is stated that:

> Particularly in the current economic environment, there is clearly a need to understand, also, the economic implications of what is being proposed.[51] The analysis has not simply addressed matters from an environmental perspective. Some of the changes proposed are designed to improve the efficiency of the delivery of services, and reduce costs to households in particular. As such, the package of measures is designed to deliver cost effective improvements in the waste

[51] It is not at all clear why such considerations are relevant to the analysis of the international review but not the Section 60 policy direction discussed in Section 7 above.

> management system, whilst seeking to internalise some of the principle externalities (Eunomia *et al.,* 2009, p.1).

Furthermore, the summary report also notes that consideration needs to be given to the incentives and disincentives that are created by policy and policy instruments (Eunomia *et al.,* 2009, p. 6).

The international review correctly recognises the potential benefits of using economic instruments:

- The setting of differential levies based on externalities;
- The use of the tax on plastic bags; and,
- Refunded compliance bonds for construction and demolition projects.

However, as we shall see below, in some cases the international review recommendations are not fully consistent with the economic approach and should be amended to better serve the goal of maximising societal welfare.

The summary report argues that it would be inappropriate in the context of the international review to conduct a full cost benefit study of each recommendation until the proposal is well developed, since "the costs and benefits will depend upon the detailed design" (Eunomia *et al.,* 2009, p. 6). However, all too often this appears to be an excuse for presenting ill-thought out and ill considered ideas. There is frequently not enough discussion, rationale and justification for the recommendations.

Nevertheless, the international review argues that it has carried out pieces of analysis which assess:

- The likely environmental benefits of choosing one residual waste management method rather than another (see Annex 63 and Figure 2-1);
- As part of this, the potential greenhouse gas benefits to be derived from achieving the outcomes which are proposed (see Annex 63 and Figure 2-2); and
- Some key issues in respect of costs, particularly in respect of household waste management (see Annex 63) (Eunomia *et al.,* 2009, p. 6).

In the case of the first two pieces of analysis the international review presents two diagrams in the summary report, which are based on Annex 63 (Eunomia *et al.*, 2009, Figure 2-1 & Figure 2-2, p. 7). However, the link between the annex and the two diagrams is unclear, while the fact that the units of the two diagrams are not specified makes them hard to interpret. The estimates of household waste collection costs are discussed further below.

Thus it appears that while the international review does not go as far as endorsing the economic approach to deciding which recommendations to make, it has nevertheless decided to take into account some key costs and various environmental benefits in making recommendations. This is a clear advance over the methodology used in the SEA analysis of the Section 60 policy directive to cap incineration and other matters.

8.4 MBT or the Dog that did not Bark: What is Missing in the Recommendations #1

As noted above the international review's summary report contains twenty-five recommendations. Before commenting on those recommendations, it is important to flag a couple of significant omissions from the summary report. In developing a framework to draw lessons from the international experience the Eunomia consortium do not only have to pay attention to the multitude of research questions discussed above but also give regard to certain technology issues. The DoEHLG make reference to the international review making a contribution to the best mix of technologies for Ireland. Various government documents have made clear a preference for certain technologies. For example, in the waste management section of the Programme for Government - which is reproduced by the international review in its summary report (Eunomia *et al.*, 2009, Box 1, p.2) – there is a commitment to “the introduction of Mechanical Biological Treatment (MBT) facilities as one of a range of facilities” (Department of the Taoiseach, 2007, p. 22).

In this context it is surprising that the summary report contains little or no coverage of the lessons that have been learnt internationally about the pitfalls and potential issues associated with MBT-led approaches. Moreover, MBT systems vary enormously in their suitability. The variation in MBT is recognised by the international review in that it estimates externalities for six different MBT technologies. (See Table 8.2 below for details).

For the international review to be regarded as robust it needs to have considered international experience in the context of MBT being a preferred technical option; in particular we would have expected it to identify the constraints, if any, that this commitment has on the international review, the extent to which the results/findings from those countries and/or regions in which MBT is not the preferred technical option may be distorted or not relevant, and equally, consider in detail the lessons from those countries in which MBT is the/a preferred technical option. This would also helpfully address concerns expressed in some quarters that the international review has in fact been written with this commitment in mind.

While government statements demonstrate a clear preference for MBT, there is also a clear preference against the use of incineration as a waste management technique. In the Section 60 policy directive for a cap on incineration and other matters, analysed in Section 7 above, the growth of incineration was limited by placing a cap on its share of MSW. In addition various complementary measures designed to increase the cost and reduce the viability of incineration as a waste treatment option have been advocated by the DoEHLG.

Just as it is surprising that the summary report contains little or no coverage of MBT, it is also surprising that there is little or no coverage of the lessons that can be learnt about the pitfalls and potential issues associated with waste management systems that impose arbitrary limits on the share of one technology, as is proposed for incineration.[52] Incineration does vary considerably in its importance by country and region so there is no reason that lessons could not have been learnt from that experience.

In Table 8.1 we present the importance of incineration as a method of disposal of MSW, for EU Member States for 1996, 2001 and 2006. This shows that at the overall EU level incineration has been increasing in importance, accounting for 14% of all MSW in 1996 and 19% in 2006. Thus a policy of placing less reliance on incineration is inconsistent with overall EU trends.

[52] The international review in an annex does comment on the disadvantages of a blanket ban on landfill, however (Eunomia *et al.*, 2009, Annex 56, p.841).

Turning to individual Member States it is clear that there is no break in the share of incineration in accounting for MSW that suggests 30% or 25% are in any sense justifiable or 'natural' limits on the share of incineration in MSW. Furthermore, Table 8.1 suggests that if Ireland wishes to draw lessons from the international review with respect to incineration then it should look to Greece, Romania, Lithuania, and Bulgaria, all of which have zero incineration, rather than Denmark or Sweden or the Netherlands..

The closest that the summary report comes to commenting on incineration is in the lessons from the international experience where it is stated that: "[S]ome countries clearly focus upon ensuring that waste is not landfilled, and seem comfortable with a high proportion of waste from households being incinerated" (Eunomia *et al.*, 2009, p. 34). In Vienna, for example, has an incinerator located in the city centre (see Figure 8.1 below).

Figure 8.1: Incinerator, Vienna, 2009

Source: photo by John Fitz Gerald

No reference is made to the observation made in Section 7 that incineration and MBT normally exist side by side, implying that they are complements and not – implicitly at least – substitutes. Indeed, the DoEHLG acknowledged in 2005 that incineration and MBT are complements (Comptroller & Auditor General, 2006, p. 77).

Table 8.1

The Importance of Incineration in Accounting for Municipal Waste, EU Member States, Selected Years, 1996, 2001, 2006

Member State	% municipal waste incinerated 1996	% municipal waste incinerated 2001	% municipal waste incinerated 2006
Panel A: EU-15			
Belgium	34	34	33
Denmark	50	57	55
Germany	16	21	22
Ireland	0	0	0
Greece	0	0	0
Spain	5	6	7
France	35	33	33
Italy	6	9	12
Luxembourg	52	42	38
Netherlands	30	32	34
Austria	30	32	34
Portugal	0	22	32
Finland	0	9	9
Sweden	38	38	47
UK	7	7	9
Panel B: Accession Countries			
Bulgaria	0	0	0
Czech Rep	0	13	10
Estonia	0	<1	<1
Cyprus	0	0	0
Latvia	0	1	<1
Lithuania	0	0	0
Hungary	7	8	8
Malta	0	0	0
Poland	0	0	<1
Romania	0	0	0
Slovenia	0	0	<1
Slovakia	10	10	12
EU-27	14	16	19

Source: Eurostat (2008, Table 10.4, p. 417)

Thus a major omission from the international review's summary report is the lack of lessons or commentary on either MBT and/or incineration as part of the mix of technology and infrastructure in Ireland's waste management policy. As noted above in Section 8.3, the DoEHLG's *Strategy Statement for 2008-2010* envisaged that the switch away from incineration towards alternative technologies would await the outcome of the international review of waste management strategy, which was going to address "how best to implement … the emergence of new technologies in waste management." In that respect the international review's summary report is clearly a failure. It thus further undermines the case for the Section 60 policy direction to place a cap on incineration.[53]

It could, of course, be argued that the international review claims that the effects of its recommendations will be to achieve the objectives on the limitation on incineration envisaged in the Section 60 policy direction to cap incineration and other matters (Eunomia *et al.* 2009, p. 60), while, although not a recommendation Eunomia *et al.* (2009, p. 40) suggest that all MBT plants are strategic and hence the planning process should be fast tracked. These two policy strands are complementary in that the MSW that would have been disposed of by means of incineration will now (presumably) have to be sent to MBT plants if Ireland is to comply with the Landfill Directive targets and hence avoid potentially large EU fines. However, the international review – as we shall see below – does not provide any guidance as to the feasibility, location, timing, nature, cost and legality of fast tracking the building MBT plants nor does it provide any credible evidence that its recommendations will lead to a limitation on incineration consistent with the Section 60 policy direction to cap incineration and other matters that at the same time conforms with the Landfill Directive.

8.5 Certainty & Stability: What is Missing in the Recommendations #2

As noted in Section 3 above and illustrated with respect to the Section 60 policy direction to cap incineration and other matters discussed in Section 7 above, if policy revisions and changes increase regulatory risk then this will increase capital costs and bias investment in other ways. Thus it is important to reduce regulatory risk to a minimum.

The summary report of the international review rightly recognises that uncertainty surrounding waste management policy and its implementation is costly. On the current uncertainty surrounding the delay and outcome of the High Court judgment concerning household waste collection in Dublin, for example, the summary report states:

> The delay in these legal rulings is unfortunate … The absence of legal certainty on some key matters affecting the market, which such delays reflect, has a paralysing effect on the market, since would-be investors and developers crave the certainty which they hope the resolution of such cases will bring. Paralysis is undesirable at any time, but it is particularly unwelcome a year before the first of the Landfill Directives targets has to be met by Ireland (Eunomia *et al.,* 2009, p. 4).

However, it also recognises that,

> For the same reason, it has been argued that this review – and the awaiting of its outcomes – has contributed to this paralysis. There is clearly some truth to this, but one of the objectives of the review is to recommend how clarity might be given to policy to enable matters to develop more swiftly, and more cost effectively (Eunomia *et al.* , 2009, p. 4).

The summary report raises the important issue that policy is not static and that it will continue to change for a variety of reasons. In this respect the summary report comments:

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[53] Of course, if the issue of incineration and/or MBT had been dealt with in the international review then given the discussion in Section 8.3, some reference would be made to the costs of the switch from one to another which do not form part of the SEA analysis discussed in Section 7 above.

> Several stakeholders we have spoken to have highlighted the lack of certainty occasioned by this review. It should be said, however, that some strong signals have already emerged from Government, not least in the Programme for Government, and in the objectives for this review. Some stakeholders clearly seek to draw too close a parallel between the terms 'certainty' and 'stasis'. If there is one certainty, it is that the management of materials will continue to improve over time, as it has done in the past. Some premium, therefore, needs to be placed upon flexibility in the overall approach to waste management, the aim being to ensure that operational practice is not ossified for decades hence. The emerging paradigms are those of resource efficiency, sustainable materials management and sustainable consumption and production (Eunomia *et al.*, 2009, pp. 9-10).

The challenge is, of course, to manage policy change in such a way that minimises regulatory risk.

Nobody is arguing that policy should not be subject to change. However, what is being argued is that such changes as do occur, particularly on the scale proposed in the 2007 Programme for Government and the Section 60 policy direction to cap incineration and other matters, there needs to be a clear and compelling articulation of the reasons for the change and that the change proposed is the optimal response to the problem identified. If, for example, new evidence emerged that a particular method of disposal was extremely harmful to human and animal health then that might justify limiting if not banning its use. However, no such justification has been offered for the change in waste management policy referred to above.[54] Thus the changes seem arbitrary and are likely to increase regulatory risk.

This suggests some sort of pre-announced review mechanism for waste management policy is needed, independent of ad hoc reviews such as the international review. We would suggest that it should be clearly articulated, including the date of such reviews, well in advance and that consideration be given to the provision of 'grandfather rights' so as to reduce regulatory risk. Such a mechanism would have the advantage of avoiding the rather abrupt changes in policy embodied in the 2007 Programme for Government and related subsequent documents.

The international review compounds the problem of uncertainty, despite providing targets covering the period to 2016 to which the Landfill Directive applies, by not thinking through and explaining the basis on which important recommendations and suggestions are made. In relation to residual landfill waste levies, for example, although a structure is presented in the summary report, no explanation is provided as to how it is derived. The reader has to infer the derivation from the relevant annex. This is to put it mildly unsatisfactory. If important recommendations are not explained in a credible manner then considerable doubt is placed on the credibility of the policy making process and thus investors are likely to demand a risk premium when investing in assets in Ireland especially where there is a large sunk cost component, whether it is an incinerator or an MBT facility.

Overall it would appear that the international review has taken place in a context that has given little consideration to the need to derive policies and practices that provide a stable

[54] This is not, it should be noted, a criticism of the international review, but rather of the policy-makers who advocate change but fail to provide the rationale.

framework for investment decisions given that the capital for the new infrastructure – whatever that might be – will have to come from the private sector and is likely to be long lived with a large sunk cost element. The international review exacerbated this situation.

8.6 Counterfactuals, Context, Separating Waste Streams & Meeting the Landfill Directive Targets

The international review, as noted above, "must provide for meeting and where appropriate exceeding, national, EU and other international objectives and requirements" (Box 8.1 above). The summary report echoes this when it states that a "key requirement of this study has been to understand how policy might need to be changed to ensure Landfill Directive targets are met" (Eunomia *et al.,* 2009, p. 15). To meet this requirement, as noted above, the international review needs to answer questions such as:

- Will Ireland meet the Landfill Directive targets on unchanged policies?
- If not, what policies should be introduced in order to reach those targets?
- What steps can be taken to ensure that the targets are exceeded?

To answer these questions requires the development of a counterfactual, an understanding of current waste management policies and what is required in terms of policy in order to meet the Landfill Directive targets if, on unchanged policies, the targets are likely to be missed with possible consequent fines imposed by the EU on Ireland. The approach adopted by the international review to reviewing policies, outlined in the summary report (Eunomia *et al.,* 2009, pp. 5-6) and discussed above, should have put the Eunomia consortium in an excellent position to deal with these issues.

Meeting the Landfill Directive Targets: the Current Situation

The international review points out that in 2007 Ireland was landfilling 519,000 tonnes more than it would be allowed to in 2010 under the Landfill Directive. Although there have been improvements in recycling these have been more than offset due to the overall growth in BMW (Eunomia *et al.,* 2009, p. 12). The international review also points out that there has been, as of 2007, little progress in recycling of organic waste and textiles, while large quantities of paper and card remain uncaptured. The review argues that there is a need to capture these waste streams for recycling, reuse, anaerobic digestion and composting.

In terms of existing targets used in Irish waste management policy, the international review makes a number of criticisms in terms of their lack of clarity and failure to always assign clear responsibility for the attainment of a target. As a result the international review argues that there "is a clear need for targets which:

- reflect the potential for improvement which exists in Ireland, taking account of experience in other countries, and the specific context of Ireland;
- are, as far as possible, targeted at specific actors; and
- are, where actors are targeted, backed either by incentives, or sanctions for non-compliance, so as to ensure targets are indeed met (Eunomia *et al.,* 2009, p. 16).

Obviously as the summary report states the specific actors have to be those with responsibility for meeting the targets.

Meeting the Landfill Directive Targets: 2010

The international review forecasts the level of BMW in 2010 will be 143,000 tonnes more than the target under the Landfill Directive for that year. This estimate is based on certain assumptions about the fall in the volume of waste in 2010 relative to 2007 and increases recycling and reuse rates for certain waste streams.[55]

Four options are considered in the summary report (Eunomia *et al.*, 2009, p. 36) as to how the 143,000 tonnes of residual waste should be dealt with: incineration; prepared for Solid Recovery Fuel ("SRF") and exported; prepared for SRF and co-incinerated in cement kilns; and, stabilised prior to being landfilled. Although the international review does not endorse any option for 2010, the discussion favours preparing the residual waste as SRF and either exported or used in cement kilns. Incineration was felt to be problematic as it was not at all clear that the Carranstown plant would be operational for any of 2010, while it was not clear that there were sufficient incentives for stabilisation prior to being landfilled. In terms of meeting the landfill targets for 2013, the international review favours increased "source separation of dry recyclables, and putrescible/organic wastes where these wastes are treated through anaerobic digestion" (Eunomia *et al.*, 2009, p. 37).

Meeting the Landfill Directive Targets: 2013 and 2016

The international review presents a series of six recommendations that correspond to its discussion of the current situation with respect to meeting the Landfill Directive targets.[56] These relate to greater separation of waste streams by households, the reduction of household waste per capita, combined with commercial and C&D recycling targets. These recommendations are reproduced in Box 8.3 below. Sometimes there is surrounding text commenting on, for example, who should bear responsibility for implementing the recommended action. This has not been included in the box.

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[55] These assumptions are as follows: "a net fall of 8% in municipal waste by 2010 relative to 2007 figures; Ireland's recycling of paper and card achieves a 60% capture rate in 2010 (the rate in 2007 was just under 55%); the capture of organic wastes reaches 30% in 2010 (the rate in 2007 was 9%, but there has been significant progress in rolling out 'brown bin' collections, partly in response to Circular WPPR 17/08). It should be noted that this might be considered an estimate which lies at the optimistic end of the 'realistic'spectrum …; and the capture of textiles increases to 15% by 2010. Once again, this might be considered to be at the end of the optimistic end of the realistic spectrum" (Eunomia *et al.*, 2009, p.35).

[56] Eunomia *et al.* (2009) could object that they do not explicitly state that these recommendations are actually related to the meeting of the Landfill Directive target for 2013 and 2016. However, the targets relate to that period and as Eunomia *et al.* (2009, p. 15) acknowledge a key requirement of the international review is dealing with how the Landfill Directive targets are met.

Box 8.3 Eunomia Consortium Recommendations to Reach Ireland's Landfill Directive Targets: 2013 & 2016

R1. Legislation requiring that all collectors who collect household waste provide, within their service offering to households (…), the following:

A) A collection of paper and card for recycling at least fortnightly, and no less frequently than the collection of refuse;

B) A collection of textiles for recycling at least monthly;

C) A collection of food waste at least weekly;

D) A collection of food waste at least weekly;

E) A collection of plastic bottles at least fortnightly, and no less frequently than the collection of refuse;

F) Either a collection of glass containers at least fortnightly, and no less frequently than the collection of refuse, or a network of bring banks with a density of at least 1 per 600 inhabitants (pp, 37-38).

R2. Legislation to ensure that all household waste recycling centres are equipped with facilities for the separate collection of garden waste, food waste and textiles (p. 39).

R3. The Waste Management (Food Waste) Regulations, currently in preparation, will require commercial producers to avail themselves of a food waste collection service (p. 39).

R4. The following targets for household waste are proposed:

1) Less than 250kg per inhabitant by 2011;

2) Less than 200kg per inhabitant by 2014;

3) Less than 175kg per inhabitant by 2017; and

4) Less than 150kg per inhabitant by 2020.

These targets would apply to each local authority responsible for waste management.

In order to give local authorities an incentive to ensure that improvement in this regard is delivered, we propose the following incentive mechanism, adapted from an approach used in Wallonia in Belgium: 1) Where a local authority exceeds its allowance, a levy will be applied to the total excess residual waste; 2) We suggest that the levy is applied at the level of €50 per tonne of excess residual waste. The levy rate should be announced in advance, but could be made flexible to deliver the right magnitude of incentive; and 3) The revenue received will be refunded back to all authorities who are below the target level. The refunds will be in proportion to the over-performance against the target (measured in total tonnes per authority) (pp. 42-43).

R5. Commercial waste recycling rates should reach

1) 55% in 2010: 2) 60% in 2011; 3) 65% in 2014; 4) 70% in 2016 (p. 43).

R6. The target rates for the recycling of construction and demolition waste are:

75% in 2010; 80% in 2012, 85% in 2014, 90% in 2016 (pp. 44-45).

R7. We propose that for:

* New residential development of 10 houses or more;

* New developments other than above, including institutional, educational, health and other public facilities, with an aggregate floor area in excess of 1,250 m^2;

* Demolition/renovation/refurbishment projects generating in excess of $100m^3$ in volume, of C&D waste;

* Civil Engineering projects producing in excess of 500m3 of waste, excluding waste materials used for development works on the site;

A site waste management plan would be mandatory, and that the plans should demonstrate that the following recycling targets will [be] met: 80% in 2011; 85% in 2012; 90% in 2014; 92% in 2016.

Bonds would be payable by developers, to be determined in the detailed working up of these proposals (p. 46).

Source: Eunomia et al. (2009).

The recommendations are developed in the summary report in terms of the method of implementation (e.g. a Section 60 policy direction or Waste Permit Regulations); accompanying measures, where appropriate; and assignment of responsibility for implementing the recommendation. There are occasional references to the international experience.

Comment #1: Why Stop the Counterfactual at 2010?

The Landfill Directive targets relate not only to 2010, but also 2013 and 2016. Hence it is essential, indeed necessary, to estimate on existing or unchanged policies, what is the likely level of BMW that will be diverted to landfill so as to be able to determine the extent to which Ireland will exceed or meet the Landfill Directive targets. However, the international review does not attempt to estimate the level of waste likely to be diverted on unchanged polices except for the single year 2010.

The international review's explanation for this omission is as follows:

> It is extremely difficult to estimate the quantity of BMW in the coming years, especially in the period under examination where the economic situation and trajectory are quite different to those which have been in place in recent years, and where there is no data which gives a hint as to the likely consequences. There is considerable uncertainty around these figures (Eunomia *et al.*, 2009, p. 35, footnote 3).

However, this statement is not entirely consistent with the fact that there are two quite different sets of forecasts *are* included in the report and are used for different purposes. In Annex 63 Eunomia estimates the growth of household and non-household municipal waste (Eunomia *et al.*, 2009, Annex 63, Table 63-35, p. 1032) up to 2016, while in the RIA released at the same time as the international review uses the forecasts that are provided by the ESRI/EPA ISus model,[57] which are based on the same methodology we have discussed in Section 5 above. The forecast waste growth rates used in the Annex 63 analysis are very low by historical standards (1-2% per annum) and do not seem to be based on an analysis of the likely growth in the number of Irish households or other recognised drivers of waste arisings. It is unclear why neither of these forecasts was used in the analysis discussed in the main report, or if there was a genuine disagreement about

[57] See AP EnvEcon (2009, p. 129) for details.

likely waste growth among the study consortium, why the different growth paths could not have been used as scenarios in each model.

Of course, there is some uncertainty as to what are 'unchanged' policies. The Minister for the Environment, Heritage and Local Government has as yet to make a final determination on the Section 60 policy direction to cap incineration and other matters. The delay – just over a year – for the High Court to deliver its judgment concerning cases that bear on the local authority's power to direct household waste and who owns the waste (Eunomia *et al.*, 2009, p. 4).[58] However, that is not an argument for stopping at 2010, but rather estimating residual BMW under different counterfactual scenarios.

These scenarios are likely to be very useful in guiding policy. The scenarios indicate a range of possible outcomes, depending on the assumptions made. They can then be used to inform policy. Suppose for example under all reasonable scenarios there is a large volume of residual waste. If this was the case then it would make sense to invest in (say) an incinerator since these are large sunk capital investments which need to operate at close to capacity to achieve maximum efficiency. If, on the other hand, the evidence was much more equivocal about what the level of residual waste, then it would make sense to invest in technologies that could be ramped up at short notice or perhaps waste could be exported for processing.

Without an estimate of the level of residual BMW in excess of the Landfill Directive targets for 2013 and 2016, it is difficult to see how the international review can set credible targets for residual waste per household as well as recycling targets. The targets that are set may be too high, too low or just right. Thus there is a disconnect between the goal of meeting the Landfill Directive targets for 2013 and 2016 and the various recommended targets set out by the Eunomia consortium.

Comment #2: Are the Targets Credible?

It is, of course, important that any targets are credible. In the present context the term credible can be used in four senses.

- First, are the targets credible in that they will enable Ireland to comply with the Landfill Directive targets for 2010, 2013 and 2016?
- Second, are the targets credible in that they will enable Ireland to exceed the targets set down in the Landfill Directive?
- Third, are the targets credible in that they are cost-effective?
- Fourth, are the targets credible in that they can be readily achieved?

The international review – for reasons set out immediately above – does not attempt nor does it provide the basis for answering either of the first two questions, while no reference is made to cost-effectiveness at all. Thus the answer to the first three questions is a firm no. Hence we will confine our discussion of credibility to the fourth question.

The international review recommends a **substantial increase in the number of different types of household collection**. At present households have at most three types of collection: brown; black/grey; and, green. The international review proposes to expand the range so that now the number will increase to seven, as set out in R1 in Box 8.3.[59] Is

[58] The last day of the case was 8 December 2008; judgment is expected on 21 December 2009.

[59] R1 does not include the black/grey bin, plus category D) may include more than one different collection.

this feasible? There is no reference to the costs of increasing the number of collections; household resistance which has characterised the roll-out of brown bins in some areas;[60] and, there is no reference to the benefits of such increased separation. Furthermore, it is a one-size-fits all recommendation, that takes no account of the fact that while more bins might be cost-effective in an urban area such as Dublin, it may not be appropriate for a rural area such as Donegal or Mayo, where it is even more likely that collection costs will rise to such an extent that it is uneconomic. Finally, of the seventeen countries for which materials requiring separate collection are presented by Eunomia *et al.* (2009, Table 6-1, p. 38) only four have separate collection for all the streams mentioned in R.1.[61] Hence there is little evidence on which to judge its credibility.

Turning now to the recommendations that make **explicit quantitative targets, R4 sees the amount of residual household waste per inhabitant falling from 300 kg per in 2007 to 150kg in 2020.** Such a target is based on the experience of Flanders where "the overwhelming majority of local authorities … are meeting this target" (Eunomia *et al.,* 2009, p. 42), the fact that the 150kg target is proposed for Wales while England has committed itself to a 50% reduction in residual household waste per person. Two points can be raised about this proposal, which suggest that it is not credible.

First, no account appears to have been taken of differences between countries and regions in household size and income, both of which are important determinants of residual waste per inhabitant. Research for Ireland, for example, concluded that: "There are economies of scale with respect to numbers in the household … The amount of waste per head decreases with increasing numbers in the household though at a diminishing rate" (Scott & Watson, 2006, p. 39). Furthermore, over time the average size of households in Ireland is expected to fall from 2.8 persons in 2010 to 2.6 persons in 2015,[62] increasing the amount of household waste per inhabitant, other things being equal, thus making the targets even more difficult to hit as time progresses. Equally the nature of the housing stock may affect the amount of residual household waste. The presence of gardens or allotments, for example, is likely to encourage greater use of composting. Other factors that are likely to influence the residual household waste per capita include the degree of waste separation provided by the collection system and its pricing. It is precisely these sought of factors that were pointed out above need to be taken into account in drawing on the international experience and were ignored in this case.

Second, while it is true that England has made reducing by 50% residual household waste per person a target, they intend to achieve reductions at a slower rate than Ireland. In the *Waste Strategy for England 2007* the following schedule appears:[63]

[60] In one area of Dublin the local residents gathered all the brown bins together, loaded them onto a truck and then deposited the bins at Dublin City Council waste depot. Some claimed that they composting at home. Great for community solidarity, but not waste policy. Based on discussion with Dublin City Council.

[61] Over half have no separate collection for textiles, while 6 have no separate collection for biowaste.

[62] Based on the demographic projections underlying the world recovery scenario in Bergin *et al.* (2009). Data kindly provided by the authors.

[63] DEFRA (2007, Table 8.3, p. 109).

Year	Household waste after reuse, recycling and composting per person: Actual and Targets, England, 2000-2020
2000	450 kg
2005	370kg
2010	310 kg
2015	270 kg
2020	225 kg

However, the 50% reduction in residual waste per person is to be achieved over a 20 year period whereas the international review suggests that Ireland should achieve a similar percentage fall over a 13 year period. Equally it is true that in the case that Wales[64] is considering reaching a target of 150kg per person, but this is to achieved by 2025, a period of 16/17 years, from a level in 2008/09 of 319kg per person, again over a longer time horizon than is suggested by the Eunomia report for Ireland. Is the target for Ireland credible? The international review does not address this issue, but in the absence of any evidence to the contrary the attainment of the target should be treated with considerable scepticism.

In order to achieve the residual household waste per person target an **incentive mechanism is recommended** whereby local authorities that exceed its allowance pay a levy in excess of its allowance, which would be redistributed to those local authorities below target. The idea is adapted from an approach used in Wallonia, Belgium. Eunomia's openness to new ways of enforcing targets is to be welcomed. However, some details need to be provided before a judgment can be made as to its efficacy.

Before commenting on the proposal itself we, albeit crudely, attempt to see who the winners and losers from the proposed incentive mechanism might be. Assume that those local authorities above the average in terms of waste per capita paid a levy to those below the average. Then according to Figure 8.2 the residents of high waste per capita local authorities such as Roscommon, Mayo and Louth, would pay a levy to low waste per capita local authorities such as Donegal, Limerick and Cork. Alternatively the critical level above which payments are paid to those below could be set at (say) 350 kg, but the redistribution would be similar.[65]

First, at the present time individual local authorities co-operate through various regional waste management plans. This should result in better co-ordination and waste management. However, a reward/penalty system that is a zero sum game is likely to alter the incentives to co-operate. If local authority x knows that by not co-operating with local authority y that it will disadvantage y so that y exceeds its allowance then local authority x may be tempted not to co-operate. This would potentially undermine the current co-operative administrative arrangements. This suggests that the regional level might be more appropriate.

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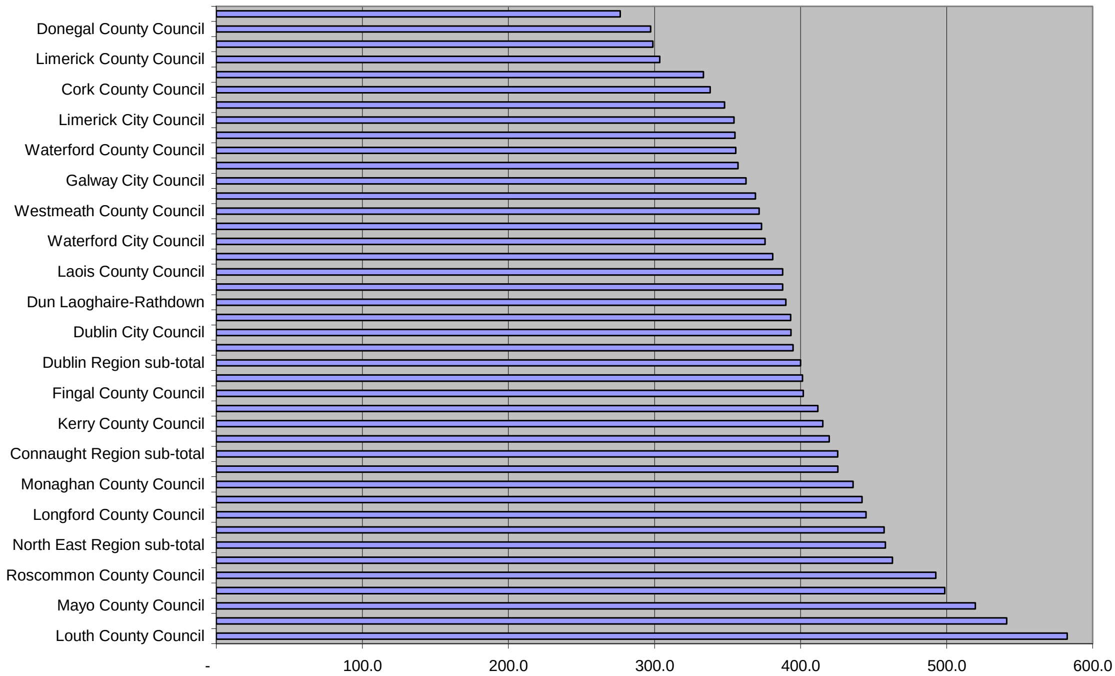
[64] Welsh Assembly Government (2009, p. 30) for the target for 150 kg; the level recorded for 2008/09 was kindly provided by the Waste Strategy Branch, Department for Environment, Sustainability and Housing, Welsh Assembly, Cardiff.

[65] Such an approach is consistent with the example used in Eunomia *et al.* (2009, Annex 59, pp. 896-899).

Figure 8.2: Per Capita Waste by Local Authority, Kilograms, 2006



Source: EPA (2007, Appendix B, pp. 62-63) and Population Census, 2006.

Second, it is not clear how the allowances would be set. There are good reasons why residual household waste per person might differ between different local authorities, just as this number differs between countries for reasons set out above. These reasons are outside the control of the local authority and hence would need to be taken into account in setting the allowances. Apart from the fact that the data may not be available to conduct such an exercise it is likely to be, to say the least, a contentious task and time consuming task, with considerable lobbying by local authorities to ensure that the allowance is such to favour them.

Third, if local authority's behaviour is to be influenced by the levy system then it will need to be predictable and stable for a number of years for the same reasons that regulatory risk should be minimised. However, the Eunomia proposal states that the levies "should be announced in advance, but could be made flexible to deliver the right magnitude of incentive" (Eunomia *et al.*, 2009, p.42). Flexibility is the antithesis of certainty and given the problems identified above with the scheme there will be no doubt be quite a lot of tinkering which would undermine the levy.

The last two recommended targets relate to recycling rates refer to commercial waste and C&D. On the **targets for commercial waste recycling** the international review states that there is "clearly considerable room for improvement" and as a result see recycling rates increasing to reach 65% by 2014. On **C&D recycling targets** the international review is a little more tentative due to the absence of the reliable data and, as a result, the recommendations are couched in terms of levels that should be aspired to (Eunomia *et al.*, 2009, p. 44). It is not clear why targets for recycling of C&D waste are required at all, given the presence of economic instruments such as the landfill levy.

As with meeting the target for residual household waste per person, the international review has a **novel mechanism to achieve compliance with the C&D targets**. The mechanism is refundable compliance bonds. For construction projects above a certain minimum size or generating a certain volume of waste, the developer would be required to fulfil certain recycling targets from 2011 to 2016. A site management plan would be mandatory. In order to ensure compliance a bond would be posted by the firm responsible for the project. On meeting the recycling targets the bond would be refunded in whole or in part.

Such a bond is in principle a sensible idea. The incentives are structured correctly. The developer is incentivised to comply with the targets agreed in the mandatory plan under pain of losing the bond. However, the bond pre-supposes that the recycling targets are necessary, feasible and can be met in a cost-effective manner. If the costs of meeting the targets are high then that will be reflected in the cost of new build. Hence before adopting such a measure the cost of meeting the targets needs to be investigated particularly in view of the tentative nature of the recommendation concerning C&D.

Comment #4: Targets & Levies - How Do They Relate?

The international review seeks to meet the targets in the Landfill Directive. This means reducing the flow of residual BMW that goes into landfill. There are a variety of mechanisms that can be used to meet these targets. The international review recommends two approaches: first, the various targets for recycling and reuse which have been discussed above; and second, levies on residual waste, which are discussed below. The international review recognises, correctly, that these two instruments can be used to achieve the same objective.

In the discussion of commercial waste recycling targets the summary report states that, "[S]ignificant under-performance against these rates in later years would be expected to lead to a broadening of the range of materials which businesses were obliged to separate at source, or, in exceptional circumstances, re-consideration of the levies …" (Eunomia *et al.,* 2009, p. 43). Furthermore in the final section of the summary report, 'Sequencing and Dependencies,' the international review again recognises the interdependencies between the target/levy approach when says, for example in relation to C&D waste recycling targets, "These targets are most likely to be met if there is a commitment to introduce the residual waste levy as proposed (or similar – Recc 8) …" (Eunomia *et al.,* 2009, p. 65).

This suggests that the international review sees targets and residual waste levies as possible alternative ways of achieving the same objective(s). However, what is missing is a discussion of the relative merits of each instrument and whether it is necessary to employ both instruments. If properly structured levies or targets meet the Landfill Directive requirements why have both? It is an issue that was discussed in Section 6 above, and where we suggested that the solution was that to set levies reflecting externalities and to impose quotas on the level of BMW sent to landfill based on the Landfill Directive targets. Such quotas would ensure that the targets were met, while allowing this to take place using the least-cost combination of collection and processing arrangements.

Comment #5: Choice of Technology – MBT

In considering the implementation of R1 to R3 the international review calls for a fast tracking of procedures to approve biowaste treatment facilities. The review considers "every biowaste treatment facility is highly strategic" (Eunomia *et al.*, 2009, p. 40).[66] Furthermore in fast tracking such facilities An Bord Pleanala should pay no attention to any of the Regional Waste Management Plans to the extent that they imply a limit on such capacity. Given the importance of these facilities to the international review it would have been helpful if the review had estimated how many were likely to be needed, where they might be cited and at what rate they might be built. Furthermore, given the likelihood of intense local opposition to such plants, how credible or proportionate is a proposed mechanism that will severely curtail the consideration of local preferences and concerns in the planning process? For example, the green waste composting facility at St Anne's Park in Raheny, which did not even carry out all the functions of an MBT plant, was forced to close in 2004 because of local opposition. Cork County Council and Cork City Council have failed due to local councillor and resident opposition in the past 10 years to find a site for a high specification MBT plant.[67] None of these issues are considered in the international review.

8.7 Externalities & Residual Waste Levy

The international review's recommendation that residual waste levies should be set based on externalities is to be welcomed. It is consistent with the economic approach presented above. It is a positive step forward and is consistent with the remit of the international review that states the review should "identify possible changes to policy at the national level in order to assist Ireland move towards a sustainable resource and waste policy" (Box 8.1 above).

The proposed levy structure is set out in Box 8.4. Levies are to be increased from 2010 to 2012 and then it appears that the levies remain unchanged. The levies for each residual waste technology are specified in terms of the price per tonne of waste that enter the waste facility. However, in the case of incineration there is some ambiguity as to what exactly is being proposed by the international review. It appears at first glance that various non-GHG emissions are to be taxed separately, with for example NH_3 priced at €9.15 per kg and so on. However, at the bottom of the list is written 'As €99.00', which could be interpreted as the summation of these non-GHG taxes on incineration. However, as we shall see below, it would appear this should be €9.99, so that the levy for incineration is €35.90 per tonne (i.e., €26.00 + €9.90). The reason for the uncertainty is that no attempt is made either in the summary report or in the relevant annex to set out the derivation and reasoning behind the levy structure recommended by Eunomia.

The international review's rationale for the use of residual waste levies is:

> With the exception of Germany, all European counties [sic] with high recycling rates make use of a levy on landfill to increase the costs of residual waste disposal. In Germany, a restriction on landfilling has a similar effect. A growing number of countries also make use of levies on incineration. Few such levies are based on an assessment of environmental externalities. This study has sought to do this with residual waste treatments (Eunomia *et al.*, 2009, p. 47, internal references omitted).

The summary report also makes the point that the residual waste levy will give "additional certainty to the market that landfilling of untreated waste is likely to be the

[66] It is not clear why this argument, if valid, should not also be applied to proposals for incinerators and landfill.

[67] Based on information provided by Dublin City Council and RPS.

most expensive destination for most municipal wastes and residual (non-inert) construction and demolition wastes in the future" (Eunomia *et al.*, 2009, p. 47).

Box 8.4: Eunomia Consortium Recommended Residual Waste Levy, 2010-2012

R8. We propose a levy structure set out as in Table 6-2. The aim is to respect the principle of seeking to ensure that the environmental externalities are respected.

Table 6-2: Proposed Structure for a Residual Waste Levy

	Proposed levy rates, 2010 2011 2012
Landfill (residual MSW not meeting the stability threshold under the EPA Pre-treatment Guidelines)	€40/t €60/t €85/t
Incineration / Advanced Thermal Treatment	€10 /t €20 /t €26 /t plus non-GHG pollutant related taxes (per kg of pollutant, all years) NH_3 € 9.15 VOCs € 2.50 $PM_{2.5}$ € 52.00 SOx € 17.30 NOx € 13.60 Cd € 26.00 Cr € 21.00 Hg € 7,400.00 Ni € 2.60 Pb € 740.00 Dioxin €46,000,000.00 As € 99.00[9.90][1]
MBT processes	€5/t €12/t €20/t
Landfilling of Stabilised Biowaste, Standard Landfill	€5 /tonne sent to landfill €15 /tonne sent to landfill €25 /tonne sent to landfill
Landfilling of Stabilised Biowaste, Dedicated Cell	€0/ tonne sent to landfill €0/ tonne sent to landfill €5/ tonne sent to landfill
SRF to incineration	As for incineration, but expressed per tonne SRF
SRF to cement kiln	£0

1. This appears to be a misprint. It should be €9.90. For details see text.

Source: Eunomia et al. (2009, pp. 48).

We consider the international review's treatment of residual waste levies by asking four questions:

- Is the objective correctly specified?
- Is the appropriate methodology used?
- Are the levies set using a sensible decision rule?
- Is the revenue from the residual waste levy allocated properly?

We have grave concerns over the international review's methodology, decision rule and, to a lesser extent, the hypothecation of revenues from the residual waste levy to the Environment Fund. We also have concerns about the RIA on waste levies that was released with the international review and so make a couple of brief comments on it. However, at the same time, we welcome the emphasis in the international review of pricing externalities and the guidance offered in awarding and allocating the disbursement of funds from the Environment Fund. The latter provide at least a focus for debate and discussion.

Comment #1: Is the Objective Correctly Specified?

The objective of setting residual waste levies according to the international review is to internalise the externalities from waste management. In other words, by putting a price on the environmental and other damage (i.e. externalities) caused by incineration, landfill or MBT that is not already borne by those generating the waste, levies can influence the decision as to use of these methods of residual waste management. If, for example, the externalities generated by landfill are much greater than either incineration or MBT this will encourage greater use of the latter two methods of dealing with residual waste and less use of landfill. Such an approach is consistent with the economic approach advocated in this report and set out in Section 6 above and Annex A below.

Comment #2: Is the Appropriate Methodology Used?

The methodology used by the international review to estimate the residual waste levies is set out in Annex 63. The methodology can be summarised as follows:

- Unit damage costs that yield a 'high' and a 'low' estimate (Eunomia *et al.,* 2009, Annex 63, pp. 979-981). These estimates are drawn from various sources some of which are discussed in Annex A below.
- Certain externalities are omitted from consideration, including emissions to water and land, land use and transport. Disamenities are omitted because of insufficient data, with the result that "[N]one of the impacts [e.g. odour, nuisance] assessed can be said to have been estimated with a high level of certainty" (Eunomia *et al.,* 2009, Annex 63, p. 982).
- Avoided emissions from electricity generated from residual waste management leads to a corresponding reduction in the externality. In other words, if, for example, an incinerator generates electricity it is assumed this displaces the generation of electricity from a power station, with the result that certain emissions are avoided (Eunomia *et al.,* 2009, Annex 63, p. 985).
- The use of diesel in the treatment of residual waste management is assumed to give rise to certain external damage costs arising, for example, from pre-combustion emissions. However, it is not clear exactly what externalities

associated with diesel need to be taken into account. Nevertheless, the impact is increase the estimate of the externality (Eunomia *et al.,* 2009, Annex 63, p. 986).

- Emissions avoided through recycling of material recovered and recycled from the residual waste management process lead to a corresponding reduction in the externality. In other words, materials are recovered from the waste treatment process are retrieved and recycled. There are emissions avoided since these recycled materials lessen the demand for new or virgin material and consequently there are emissions avoided (Eunomia *et al.,* 2009, Annex 63, pp. 986-987).

We have three sets of criticisms of this approach by Eunomia. First, is that disamenities are omitted; second, double regulation may lead to double counting; and, third, there is inconsistent treatment of emissions across pollution sources. Each is considered in turn.

First, as noted above Eunomia do not consider the issue of disamenities. However, it is not at all clear that the estimates of disamenities set out in Section 6 above and Annex A below are any more or less reliable than the other environmental damage estimates used in estimating externalities. It should be noted that our results agree with the expectations of Eunomia *et al.* (2009, Annex 63, p. 982) that disamenities are likely to be higher for incinerators (but only to the extent they tend to be located in densely populated urban areas) and lower for MBT plants (to the extent they tend to be in rural areas). This pattern could change, particularly if giving enhanced planning support to MBT plants led to them being developed in urban areas.

Also, the number and scale of facilities has a bearing on disamenity effects. The disamenity effect appears to be mainly fixed in that the disamenity relates to the fact that a person is living next to a waste residual treatment facility rather than its throughput. If this is the case, then in policy terms when considering the total disamenity costs of disposing of (say) 600,000 tonnes of waste then the comparison might be between one incinerator and (say) 10 MBT plants each processing 60,000 tonnes. It is not at all clear that the former would be greater than the latter.

Second, the issue of double regulation which may lead to double counting both of negative and positive externalities. In determining the appropriate treatment of a given externality by means of a residual waste levy, attention needs to be given as to whether or not that externality has already been taken into account or addressed by another policy instrument such as a tax or a regulation. We are only interested in *unpriced* externalities. If the externality is already *priced* then it is already internalised.

In considering the appropriate waste residual levy attention needs to be paid to the fact that:

- CO_2 emissions from a large point emission such as a power plant are already taken into account through the EU emission trading scheme (ETS). Hence the price of electricity which an incinerator or other waste treatment facility receives for the electricity it sells to the grid already includes a component equal to the value of its avoided carbon emissions.[68] If the waste facility is then credited, when estimating the residual waste levy, with the value of the avoided emissions caused by the electricity it generates, the facility will have been compensated twice. The waste levies should thus not include a deduction for the value of CO_2 avoided in power generation;

[68] For details see Single Electricity Market Committee (2008, p.1).

- A carbon tax, which was announced in the December 9 2009 budget and is part of the Programme for Government (Department of the Taoiseach, 2008, p. 2), will apply outside the sectors covered by the ETS, means that the externality caused by the use of fuels such as coal, peat, petrol and diesel has been taken into account prior to its being purchased for use as a fuel. The price of the fuel is higher than would otherwise be the case due to the carbon tax. As a result there is no need to include in the residual waste levy allowance for the fact that the diesel used in waste facilities creates CO_2 emissions. If the externality is included in the residual waste levy then it is being counted twice; and,
- The emission avoided through recycling may have already been taken into account if the inputs of the original product had been covered by either the ETS or been the subject of a carbon tax in Ireland or some other EU Member States (soon to include Ireland). However, for products recycled from China or India where the equivalent of the ETS or a carbon tax are not present, then it would be appropriate in designing a residual waste levy to take into account the emissions avoided. However, this is likely to be neither a straightforward nor easy task.

It thus appears that Eunomia *et al.* (2009, Annex 63) in setting the residual waste levy have not taken into account the fact that some of the externalities – both positive and negative – have already been priced. Hence there appears to be double counting of externalities with the result that the residual levies designed by Eunomia cannot be relied upon until the double regulation/double counting has been removed.

Third, it is important that there is consistent treatment of externalities across different sources of emission. A tonne of CO_2 causes the same damage irrespective of whether it is from an incinerator or a power plant. Distortions arise if different approaches and prices are used. If the evidence suggests that the correct price is x for an externality then that should be applied everywhere. There are several cases where the international review needs to be consistent.

IPPC licenses issued by the EPA to incinerators, power generation plants, cement works, aluminium smelters and so on are designed to ensure that the operation of these facilities will not cause environmental pollution. Equally in the case of landfill or an MBT facility the Waste Licence fulfils a similar function. In order to achieve this objective the IPCC license sets maximum levels of permissible emissions. What is being proposed by the international review is that emissions below these maximum levels be subject to a residual waste levy, but only for a particular subset of facilities subject to IPCC control – residual waste management facilities. There is no attempt to justify not only this double regulation, but why these particular facilities merit such treatment. One suspects that non-CO_2 gases emitted by an incinerator such as NO_x would be quite small compared to, for example, a coal-fired power plant. Furthermore investment and other decisions would be distorted by such asymmetric regulation and perceived regulatory risk might be increased, as discussed in Section 7 above.

Comment #3: Are the Residual Waste Levies Set Using a Sensible Decision Rule?

The international review's proposed residual waste levy structure is set out in Box 8.4. Although the previous discussion raised considerable doubts about methodology used to estimate the externalities that are the basis for the residual waste levy, it may nevertheless be the case that the international review's decision rule for selecting the schedule of residual waste levies could be applied to externalities appropriately measured. However, what is striking about the international review is that there is no justification or decision rule for the recommended tariff schedule of residual waste levies. This applies both to the international review's summary report (Eunomia *et al.*, 2009, p. 47) and the discussion of externalities in the supporting material (Eunomia *et al.*, 2009, Annex 63). This is an extraordinary omission, given the importance of the residual waste levy to waste management policy.

Nevertheless, despite the fact that there is no discussion of the way in which the recommended residual waste levies are set by the international review, it is possible to infer a rule. In Annex 63 summary tables of the externalities are presented using a high and low external set of costs. It appears that the relevant estimates in setting the levy are the former and these are reproduced below as Table 8.2. Our discussion covers only the first three options in Table 6-2, which is part of Box 8.4.[69]

Table 8.2
International Consortium Estimates of Externalities, by Residual Waste Management Technology, High External Costs, Per Tonne

Residual Waste Management Technology	Climate Change Externality	Air Quality Externality	Total Externality
Landfill (50% gas capture)	€80.66	€5.88	**€86.54**
Landfill (20% gas capture)	€123.62	€5.52	€129.14
Incineration (meets WID)[1]	€26.24	€31.87	€58.11
Incineration (out-performs WID)	€26.24	€9.96	**€36.20**
MBT aerobic biodrying / incineration	€22.94	€15.38	€38.20
MBT aerobic stabilisation	€18.82	€3.55	€22.37
MBT aerobic biodrying / cement kiln	€9.16	€11.53	**€20.70**
MBT aerobic biodrying (50% gas capture)	€26.67	€15.38	€42.05
MBT aerobic biodrying (out-performs WID)	€22.83	€8.03	€30.85
MBT aerobic stabilisation (50% gas capture)	€35.84	€3.55	€39.39

1. WID= Waste Incineration Directive
Source: Eunomia et al. (2009, Annex 63-17, p. 1010, emphasis supplied).

As can be seen the table presents estimates for the externalities for landfill, incineration and MBT and in each case there are several estimates under various assumptions. It appears that the international review has selected the lowest priced total externality option for each of landfill, incineration and MBT (Table 8-2, in bold). It is not clear on what basis the lowest priced option was selected. If the option selected is the actual choice made by providers of landfill, incineration and MBT, then this might be an appropriate way of structuring the levy. However, there is nothing to indicate whether or not this is the case. The recommended levies in Table 6-2 are to be phased in over the period 2010 to 2012, although the rate of increase differs across each of the three residual waste treatment methods for no apparent reason. It is not clear why in the case of incineration Table 6-2 includes a whole series of non-GHG pollutant related taxes which seem to add up to €9.90.[70] To avoid market distortions, these emissions should either be taxed consistently across all emitters (both in the waste and industrial sectors) or not taxed at all and regulated consistently via IPPC and waste licences.

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[69] Cement kilns despite emitting CO_2 and other pollutants has a proposed levy of €0.0. It is not clear why.

[70] There appears to be a misprint in Table 6-2 where instead of using €9.96 from Table 8-2, the international review has moved the decimal point one place to the right to yield €99.00.

The purpose of setting the levy is to encourage the internalisation of externalities. The greater the level of externalities the higher the levy should be, so as to encourage the use of technologies that result in less externalities. The proposed structure does not assist in this regard in that for landfill, incineration and MBT there is no incentive not to use a technology that is less polluting *within* each of these three waste management technologies since a single price is set irrespective of the level of externality. For example, irrespective of the type of MBT facility built the residual waste levy is €20 per tonne, despite the fact that the levy for three of the six MBT technologies in Table 8.2 indicates that the levy should be about twice that level. To avoid this problem there should be a range of prices which reflect the particular technological choice adopted. This would have the desired effect of not only setting the levy to reflect the level of externality but also that it would provide an incentive to use less polluting choices. By not following such a rule the international review's recommended levy structure is not entirely inconsistent with its declared objective of setting the levy such that the externality is internalised.

Comment # 4: Is the revenue from the residual waste levy allocated properly?

An Environment Fund has been established under the Waste Management (Amendment) Act 2001 to fund various activities relating to the environment and waste management. Major expenditure has been devoted to waste management including recycling projects, recycling operational costs, local authority enforcement initiatives and EPA R&D (Eunomia *et al.,* 2009, Annex 60, Table 60-1, p. 909). The Fund is financed by revenues from the levies on plastic shopping bags and landfill levy. The latter two are hypothecated taxes.

The international review takes the position that the revenues raised by the residual waste levy should be channelled into the Environment Fund. The review estimates that in 2011/2012 the levy will yield €70 million (Eunomia *et al.*, 2009, p. 45) although no source for this estimate is provided nor how it is derived. This is a substantial increase over the annual income of the Fund. In 2006 the Fund had an income of €52.13 million, of which only €40.37 million was spent (Eunomia *et al.,* 2009, Annex 60, Table 60-1, p. 909). Combined with existing revenues, the residual waste levy is a substantial increase in the size of the Environment Fund.

In this context the international review makes a couple of recommendation on method by which the Environment Fund should award projects and the subject matter of the projects which are presented in Box 8.5.

Box 8.5: Eunomia Consortium Recommendations on Environmental Fund

R 14. Consequently, we recommend that disbursements from the Environment Fund should always consider the following:
a) Where possible, funds are awarded in a competitive manner, and that there are criteria for evaluating the success of bids, and hence also, for evaluating the benefits of the revenue spend;
b) In cases where there is no competition, local authorities should only be the sole beneficiary in cases where disbursements are not used to support any activity in which the public sector and private sector are in direct competition (for example, cleaning up old landfills, or in civic amenity site provision);
c) In all cases where it is intended that the local authority may be a beneficiary, and where the private sector also provides the service under consideration, then the relevant funds should be made available to all relevant parties, preferably on a competitive basis, to ensure that disbursements deliver value for money;
This ought to ensure that the revenues from the Environment Fund should not distort competition in future (pp. 53-54).

R16. Appropriate purposes for the use of money from the Environment Fund are given in Section 74 (9) of the Waste Management Act. These are wide-ranging. The approach to awarding funds does not seem to be strongly targeted at addressing priority areas. It also fails to recognise adequately the market context in which the support is given.

It is clear that the short-term priority for waste management in Ireland is the treatment of source segregated biowastes. We propose that, subject to the levy rates under [R8] …, being implemented, a fund of at least €10 million is earmarked for 2010 and 2011 and 2012 to support the capital cost of the treatment to source segregated organic waste from the municipal stream. The fund needs to be developed in line with State Aid rules. The aim should be to set up a competitive scheme and invite bids for the fund, outlining clearly the evaluation criteria which will be used to assess bids. One of these should be to demonstrate how it is anticipated that the organic waste will be sourced. This Fund should be open to private and public sector alike.

Another priority area which should be considered for support under a competition is waste prevention. Quality initiatives from businesses and local authorities should be funded, as well as work in delivering the National Waste Prevention Programme (though there are reasons why it might be favourable for long-term commitments to be made from central government regarding the NWPP).

Other funding needs clearly exist, such as dealing with legacy sites, and it is worth noting that Austria used revenue from its landfill levy to support the clean up of old sites. Consideration should be given to supporting clean-up, though care needs to be taken to ensure such funds are well spent.

The activities supported by the Environmental Fund should be critically appraised periodically to ensure that the Fund is delivering value for money. In this respect, all awards ought to be made with a clear understanding of what the objectives are, how they will be delivered, the additionality (i.e., the change which will be delivered over and above the current situation) and what (in the case of grant-supported projects and initiatives) is the ‘exit strategy’ to ensure they are sustained beyond the period of the funding award. (p. 55, internal references omitted).

Source: Eunomia et al. (2009).

A number of points can be made about these two recommendations their relationship with the Environment Fund. *First*, it is not clear that the volume of environmental research should be a function of the revenue raised from the various levies mentioned above. Just because the residual waste levy has increased does this necessarily mean that the optimum size of the Environment Fund has also gone up? Suppose that there were readily available and cheap alternatives to plastic shopping bags and landfill resulting in these levies raising little revenue, does this mean that the optimum level of research is virtually zero?

Each year the Fund has not spent all of its income. While in the early years this is understandable, in 2006 the underspend was 22% of revenue. In this regard the international review notes that "if levies increase, and, … the total revenue generated increases, questions should be asked as to whether the size of the fund might simply generate waste in the use of revenues" (Eunomia *et al.*, 2009, Annex 60, p. 919). While there is no suggestion that the current expenditure by the Environmental Fund is being wasted, given the underspend and the large increase in the Fund there is a danger that that might occur.

Second, the recommendation concerning the method by which funds are awarded through competitive means, with well defined criteria for both awarding the contract and evaluating the benefits are sensible and to be welcomed. The recommendation that funding should result in additionality and that there should be an exit strategy are also to be welcomed. These recommendations are made on the basis of a review of the international experience which is presented in Annex 60.

It is not clear, however, the extent to which the practice of the Environment Fund is consistent with these recommendations. It appears from the international review's description of the current operation of the Environment Fund that it is somewhat opaque, but Eunomia did not examine the performance of the Fund in any detail (Eunomia *et al.*, 2009, Annex 60, pp. 907-912).

The international review also makes the sensible suggestion that the Environmental Fund should not be used as an ongoing source of local government activities since there are more appropriate revenue streams (Eunomia *et al.*, 2009, pp.55-56).

Third, we are not in a position to judge whether the current research priorities of the Environment Fund are appropriate or whether the one proposed in the international review are superior. Nevertheless, it would have been useful if the international review had explored the implications of these priorities for current funding priorities.

Comment # 5: Can You Expect Sensible Answers When the Terms of Reference for the Waste Levy RIA Are Pre-Determined?

A Regulatory Impact Analysis ("RIA") discussing some aspects of the waste levy was published with the international review. We make only a brief comment on this part of the report, because despite the authors' efforts to make a worthwhile technical contribution, the terms of reference set by the Department of the Environment, Heritage and Local Government appear to have undermined the potential usefulness of the work.

By design, the RIA was intended to focus on a restricted set of issues:

> …a principal goal of this particular RIA was to consider the possible structures, levels and implementation for any revised levy proposal in the context of the defined Department objectives and constraints. To that extent, the focus of this particular RIA

is different to that of more traditional RIAs that are designed to evaluate the impact of a wide range of specific alternative policy proposals (AP EnvEcon, 2008, p. 28)

All analytical work is subject to some limitations in scope and focus, however two highly restrictive assumptions were placed on the analysis from the outset:

1) …any proposed change to the levy applying to landfill must be structured with an appropriate differential for the incineration of waste such that no competitive advantage is given to incineration from any proposed change.
2) …a differential for the landfill of suitably stabilised biowaste is to be suggested that would, in conjunction with the change in the landfill levy, support and encourage the development of Mechanical Biological Treatment (MBT) plants within Ireland. The boundaries outlined by the Department in the initial briefing form the overriding boundaries of all assessments within this RIA (AP EnvEcon, 2008, pp. 30-31)

It is unfortunate that the scope of the waste levy RIA was constrained in this way. The stipulations that the levy rates should be set to avoid providing any "competitive advantage" to incineration and to encourage MBT rather than set to manage externalities is highly debatable (for reasons discussed elsewhere in this report). If one does not accept these premises, the findings of the RIA are of limited usefulness.

A less constrained RIA applying data and techniques such as those used by the authors of this one might have made a very useful contribution; for example by identifying how various possible policy measures could contribute to meeting the Landfill Directive targets, at what cost and with what benefits to society.

In sum, the recommendations in the international review with respect to the residual waste levy are severely flawed. The methodology used to derive estimates of externalities includes externalities that have already been *priced*. The residual waste levy should only deal with *unpriced* externalities. The structure of the recommended residual waste levy does not provide incentives for the deployment of technologies that generate lower rather than higher levels of externalities within a given waste technology and perhaps across technologies, surely a perverse outcome. The proposed levy structure in Section 6 does not suffer from these shortcomings.

8.8 Producer Responsibility: Levies and Targets

Producer responsibility, as noted in Section 2 above, is related to the polluter pays principle. Producer responsibility makes the producer responsible in whole or in part for the product once it has reached the end of its life and becomes waste. The producer is thus provided with appropriate incentives to minimise the cost of disposal, through recycling, reuse etc. In minimising these costs it is important that the right price signals are sent to the producer in terms of reusing and recycling and residual waste charges reflect the appropriate externality imposed.

There are five recommendations made in respect of producer responsibility and the related issue of polluter pay, as asset out in Box 8.5. R9 reflects the view of the international review that where the producer is responsible 100% for dealing with the product at the end of its life then that is more efficient compared to a situation where it is responsible (say) 80% and local authority for the remainder. In other words, it is easier to internalise the externality. Transaction and administrative costs are likely to be lower. If the producer is not responsible at all then the costs of disposal are spread very broadly, even on consumers who do not purchase the product at all, and the result is that the externality may not be internalised as efficiently.

Producer responsibility schemes are likely to encourage greater recycling and less residual waste. The greater used of production responsibility schemes may also alleviate the problem of non-users of household waste collection services (Eunomia *et al.*, 2009, p. 49). As a result the international review recommends that the targets for relevant recycling targets can be raised, which is reflected in R10. R11 seems to be no more than bringing forward certain proposed revised EU targets before they are actually implemented, while R12 largely reiterates and strengthen steps already announced in the National Strategy on Biodegradable Waste (Eunomia *et al.*, 2009, p. 52). R13 is a call for wider implementation of polluter pay type measures of which the plastic bag tax is considered a stellar example. It is considered that such measures might be particularly suitable for items "which are deemed to lead to excessive generation of waste in the face of obvious alternatives" (Eunomia *et al.*, 2009, p. 52).

Box 8.5: Eunomia Consortium Producer Responsibility & Polluter Pays Recommendations

R9. We propose, therefore, that a policy principle is established: Where producer responsibility measures are in place, the principle is established whereby producers will be expected to be fully financially responsible for delivering the services required to meet their obligation. Sanction should be applied in the event of non-compliance, with compliance being the subject of adequate enforcement checks (p.50).

R10. The targets under Article 11 of the Waste Management (Packaging) Regulations be increased such that in future, the minimum recycling target increases to 75% (up from 63.6% in 2007) by 2013.

In lieu of the fact that these targets will require active participation by all concerned, we suggest that the *de minimis* thresholds under Article 5 of the Waste Management (Packaging) Regulations should be abolished (p. 51).

R11. The recommendation, therefore, is to update the Waste Management (Waste Electrical and Electronic Equipment) Regulations 2005 in line with the proposal for revised targets from the EU (p. 52).

R12. We recommend that producer responsibility initiatives are extended to cover newspaper and magazines as well as junk mail and other forms of direct marketing (p. 52).

R13. It is recommended that consideration is given to the wider implementation of product levies, principally targeting disposable products in widespread use. Suitable targets – recognising that alternatives are available – would be, for example, razors and cutlery. Here alternatives are readily available.

For some items, depending upon the feasibility of establishing take-back, or separate collection networks, it may be useful to consider such levies as a supporting measure in the context of producer responsibility – obligated producers would be subject to the levy if they fail to meet the targets set under the producer responsibility scheme. This approach has been used in the context of ecotax legislation in Belgium (p. 52, internal references omitted).

Source: Eunomia et al. (2009).

Comment #1: Fleshing Out The Principles

Producer responsibility and 'polluter pays' are two tools or policy instruments that the policy maker can use in order to meet certain environmental objectives. The international review is correct to draw attention to these instruments and to indicate where they might be most appropriately be deployed. In the case of the levy it argues that if the item is leading to excessive waste generation and there are obvious alternatives then a levy as in the plastic bag type tax make obvious sense. However, this is only half the story. What happens if the obvious alternatives are costly?

The economic approach would place emphasis on the benefits and costs of imposing the tax. While the gross benefit of the tax is the reduction in waste for the item which is taxed, what is important is the net benefit that will include any increase in waste attributable the use of alternatives. These benefits will need to be offset against the costs of the using the alternatives. If these are sufficiently high then they may outweigh the benefits.

It could of course, be argued, that this is implicit in the discussion of the levy, but given the general low level of awareness of the economic approach in framing policy there can be no presumption that that is correct. Hence it is important to be explicit about such issues.

Comment #2: Targets, Costs & Relevance

A point made repeatedly in this report is that in setting targets account needs to be taken of costs and benefits as well as feasibility in meeting targets. However, such a discussion is absent from the discussion surrounding R10 and R11, apart from the claim that the 65% target under R11 "is likely to be challenging, though achievable" (Eunomia *et al*., 2009, p. 52). To some extent this approach is taken because the targets are taken from the EU and hence in some sense exogenous. However, that is too easy an explanation. Attention needs to be paid as to whether for Ireland such targets make sense. All too often it appears that EU climate change and environment targets and policies are not made using the economic approach, but based on political considerations and expediency that may not necessarily be in Ireland's interests.[71] Earlier intervention in the process may ensure that policies are adopted that better suit Ireland's interests.

8.9 Going Green: Should Ireland Adopt Green Procurement Policies?

Green procurement is a method by which the buyer of goods and services compels contractors to meet certain necessary conditions or specifications in order to be awarded a contract or order for business. While the buyer can be public or private, it would appear that green procurement is more prevalent in the public sector with the EU setting certain non-binding targets for Member States. Ireland has been somewhat remiss in the development of national action plans for green procurement, although recent ministerial statements suggest that this will soon be rectified.[72] This resolve will no doubt be strengthened by the international review's recommendation that:

> **R17.** The development of a National Action Plan for Green Procurement needs to be progressed with a matter of urgency (Eunomia *et al.,* 2009, p. 56).

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[71] See Helm (2009) for a trenchant critique of EU climate policy.

[72] This paragraph is based on the discussion in Eunomia *et al.* (2009, p. 25-26)

The international review provides some illustrative examples of what it refers to as "simple changes" to the procurement process that will affect the waste sector:

Examples of simple changes which will affect the waste sector are:

- Introducing recycled content specifications for suppliers of goods;
- Setting targets for the use of recycled materials in state-supported construction projects;
- Specifying the use of waste derived composts in projects involving gardening / landscaping etc.;
- Specifying that biowaste treatment facilities deliver products which comply with compost standards, and are part of a quality assurance scheme (when available);
- Requesting that suppliers furnish information regarding environmental policies; and
- Specifying that all suppliers and tenderers print documents on both sides of paper, and use 100% recycled paper (Eunomia *et al.*, 2009, p. 56).

Others have different targets on their green procurement agenda with the European Investment Bank, for example, setting targets for 2009-2010 which includes, "[O]rganic food produced locally during the appropriate season will be on offer in the restaurant" (EIB, 2009, p. 2). The European Commission, according to its website, "proposes a political target of 50% green public procurement to be reached by Member States by the year 2010." [73]

Comment #1: Rationale, Market Failure and Costs

Government intervention, as discussed above, should be based on a sound rationale, such as market failure. The intervention by government should then address that market failure. In the case of green procurement policies it is difficult to determine exactly what the problem is that warrants government intervention in the first place and secondly the use of procurement as the desired method of resolving the problem. Green procurement seems to reflect a view that governments can drive environmental change and improve environmental conditions since the state is such a large purchaser of goods and services. It is presented almost as risk free costless option. However, is this the case? Are public procurement prices likely to be higher as a result of Green Procurement? Will it lead to an excessive level of regulation – why does the government need to know the environmental policies of the firm or specify the meals served in works canteens? If it is economic to print on both sides using recycled paper, why wouldn't a firm already be doing it? What is the cost of monitoring compliance with these requirements? Is Green Procurement yet another example of using several instruments to achieve the same objective?

Certainly the international review provides no rationale for green procurement, accepting the policy at face value and recommending its promotion and adoption by government. It could, of course, be argued that the policy comes from the EU and that Ireland has to adopt the policy. But as noted above green procurement is a voluntary not a mandatory policy so Member States have a choice. Furthermore, if green procurement does not have a sound rationale then the international review can play an important role in articulating that and influencing the debate at the EU level.

[73] http://ec.europa.eu/environment/gpp/consultation_en.htm. Accessed 2 September 2009.

Thus it is suggested that before R17 is adopted that the rationale for the policy be clearly articulated, whether there are less costly ways of meeting the same objectives, how green procurement interacts with other government policies, particularly those that are aimed at achieving the same or similar objectives and that the benefits outweigh the costs.

8.10 Guilty Until Proven Innocent: The Burden of Proof for Ash Residues from Incineration

Incinerator bottom ash is the residual from solid waste material from the combustion process in MSW incineration facilities. The ash normally contains a small amount of ferrous metals. The ash can be processed and the contaminants removed in order for the ash to be used as an aggregate in, for example, bulk fill, pavement concrete and asphalt.

The international review argues that there is increasing evidence that bottom ash is eco-toxic. Thus the Eunomia consortium argues, based on this evidence, that there should be a presumption that "bottom ash is hazardous until it has been shown otherwise" (Eunomia *et al.*, 2009, p. 46). Although bottom ash can be treated prior to use "the effects of some approaches in use are not obviously wholly beneficial" Eunomia *et al.*, 2009, p. 47). As a result the international review recommends

> **R18**. Bottom ash should be treated as hazardous until it has been demonstrated otherwise. The EPA will need to develop protocols for sampling and testing of bottom ash and the assessment of H14 ecotoxity to demonstrate whether the waste is hazardous or not. This should be reflected in operating licences for facilities producing such ash.

Comment #1: It Isn't Necessarily So

The proposed position on bottom ash is inconsistent with the position adopted in most European countries; where, in general the objective is to maximise the beneficial use through recycling bottom ash rather than landfilling. The default position of initially categorising **all** bottom ash as hazardous would be likely to make establishing markets for recycled non-hazardous bottom ash much more challenging. In our opinion, constraining the market opportunity for recycling in this way is inconsistent with the waste hierarchy and could make the cost of incineration unnecessarily high.

As far as we have been able to establish, only in Hungary and Austria is bottom ash categorised as hazardous and, according to data produced by Confederation of European Waste to Energy Plants ("CEWEP"), the incineration facilities in these countries represent circa 2% of total population of incinerators in Europe.

Eunomia (2009, p.147) contains the following comment with regards to bottom ash:

> The above discussion, therefore, highlights the increasing body of evidence supporting the view that if bottom ash is not always hazardous, the presumption should certainly not be that it is not hazardous.

This is, in our opinion, a more pragmatic position than the proposed automatic presumption that bottom ash is hazardous until it has been demonstrated otherwise; not least because there seems to be general agreement that the ecotoxicity of bottom ash is a function of feedstock.

One possible approach to bottom ash classification would therefore be the adoption of a classification process for bottom ash tied to the accepted wastes and any relevant operational parameters within the IPPC permit. This could be developed from a robust analysis of bottom ash from existing facilities.

An alternative approach, which is based on regular testing of samples and classification of samples, may also be considered. However, the potential uncertainty associated with this approach may make raising finance for such facilities challenging.

Thus we suggest that if the bottom ash can be demonstrated to be non-hazardous based on waste feedstocks within certain input characteristics, the operator should be able to continue to assume that they remain non-hazardous until the waste feed mix changes beyond the agreed criteria. If some are found to be hazardous, then a more rigorous testing regime should be adopted.

8.11 Waste Planning: Decentralisation to Centralisation

At the present time waste policy in Ireland is characterised by broad goals and parameters being set by the DoEHLG, based often on EU Directives and guidelines. Implementation is through a series of 10 regions.[74] For example, there is a Dublin Region and a Cork Region. A number of benefits are expected to flow from these arrangements according to the international review including the realisation of economies of scale, efficiency of collection services which are often local or regional in scope as well as the development of an integrated waste management approach (Eunomia *et al.*, 2009, p. 17). Each region is responsible developing a Regional Waste Management Plan ("RWMP").

However, there are other benefits as alluded to in Section 6 above. Typically a one size fits all policy does not work because of variations across the country. For example, in dense urban areas such as Dublin a three or even four bin collection system might make sense, whereas in a rural area such as Mayo or Donegal, a single bin system combined with a network of bring banks and civic amenity centres may be more appropriate in view of the differing costs of collection.

Furthermore there can be experimentation and learning from the experience of other regions, which is much less likely under a centralised one size fits all. For example, Dublin City Council ran a pilot project before rolling out its brown bin across the city ran a pilot scheme. As a result of this pilot scheme, the brown bin itself was redesigned so that it was less smelly and so more acceptable to householders.[75] If this experience can be shared with others then that learning will inform their decisions without the need to repeat the pilot scheme.

The Eunomia consortium record that there have been criticisms of the current administrative arrangements for implementing waste management policy. Some (unnamed) stakeholders felt the current system was ineffective; some (unidentified) investments were made allegedly despite rather because of the RWMPs. Others suggested that a national plan should be introduced, perhaps through the creation of a waste management authority. Apart from these rather vague assertions there are more substantive comments on the current administrative structure.

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[74] For further details of the plans and the regions see: http://www.environ.ie/en/Environment/Waste/WasteManagementPlans/. Accessed on 3 August 2009.

[75] Based on discussions with officials of Dublin City Council.

The City and County Managers Association ("CCMA") is cited, for example, as calling for the need for greater "national co-ordination, both in framing of sustainable policy and in the development of the necessary infrastructure across the regions" (Eunomia *et al.*, 2009, p. 18). The OECD (2008, p. 328) also calls for greater co-ordination. RWMPs are chastised by the Eunomia consortium for not always filing their annual implementation reports and even those that are filed do not always fully comply with what is required (Eunomia *et al.*, 2009, p. 18). Nevertheless it is admitted that national policy "has not always been entirely clear about the nature of the policies it expects regions to implement" (Eunomia *et al.*, 2009, p. 18), while the OECD (2008, pp. 327-328) notes that the restrictive interpretation of the proximity principle by the DoEHLG, although recently relaxed, inhibited co-ordination.

Based on these observations the international review makes a sweeping recommendation centralising waste management policy in Ireland. More specifically:

> **R19.** We propose that the RWMPs be replaced with one national waste plan. This would require a change in the Waste Management Act. We propose that the process of changing the law, and developing the Plan, should be initiated as a matter of urgency as soon as a decision has been taken as to which of the policy recommendations made in this document are to be taken forward. In the interim, whilst a National Plan is being developed around these policies, these policies would then guide the implementation of waste management at the local level in the intervening period. Planning decisions should reflect the policies being implemented.
>
> The ultimate responsibility for developing a National Waste Plan would rest with the Minister. A Ministerial Programme Board chaired by the Minister and including representatives from DoEHLG, EPA, IBEC, IWMA, Cre, CCMA, the CIF, An Taisce and environmental NGOs, would be responsible for steering the Plan. The Minister would, however, still make the executive decisions (Eunomia *et al.*, 2009, pp.57-58)**.**

The only text surrounding this recommendation is the rather elliptical statement that where it makes sense local authorities should be encouraged to work together. No reference is made to any of the 65 annexes or the international experience to justify or inform the recommendation, nor co-ordination mechanisms that fall short of the centralisation of waste policy *and* its implementation in the hands of central government. Finally, the international review does not discuss the impact of the removal of a substantial part of the second most important aspect of local authority responsibility, environmental protection, on local democracy.[76]

Comment #1: The Recommendation Has No Clothes

The recommendation abolishing the RWMPs and replacing them with a National Waste Plan ("NWP") lacks rigour, clarity, and a rationale. It is not clear that anonymous and untested statements by stakeholders (which may be vested interest or pressure groups) should be given any credence whatsoever. No investigation appears to have been undertaken to establish the veracity of these statements.

[76] The OECD (2008, p. 320) states, "Waste management is a major part of the second biggest expenditure item, environmental protection, which forms 19.2% of total local expenditure."

Furthermore, it should be noted that the CCMA called for national co-ordination in relation to the provision of major infrastructure, it did not call for the abolition of RWMPs in favour of a NWP, nor did the OECD (2008, Box CS3.3, p. 336). The CCMA were concerned that the lack of a co-ordination mechanism between the regions, private operators and other stakeholders, raised “the potential for overcapacity and duplication between regions or between public/private sector exists” (CCMA, 2007, p. 25).

The international study could have investigated whether these claims had any validity and, if so, what might be appropriate solutions – are there, for example, legal or other barriers that prevent greater co-ordination or contracts to be drawn up across RWMPs? Could the DoEHLG or the EPA take a lead role in convening such a forum, particularly in view of the apparent good interaction that already exists? What decisions in particular might be suitable for co-ordination – incinerators or landfill, MBT plants? (CCMA, 2007, p.6). In other words, where are there externalities between regions that might merit coordination? Eunomia did none of these things. Finally, reference is made to the fact that some of the RWMPs do not always file reports on time or in the required detail. Moving to a NWP is a disproportionate response, given the importance of local factors in the economics of waste management (see Section 3 above). In sum, the Eunomia consortium does not establish a credible case for the abolition of the RWMPs and their replacement by a NWP.

Comment #2: Assigning Responsibilities between National & Regional Waste Management Plans

Nevertheless, let us assume for the sake of argument that there is a case for a NWP, the issue arises as to what functions presently discharged by local authorities under the RWMPs should be assumed at the national level and what should remain with the local authority.[77] At the present time the major responsibilities of local authorities under the RWMPs are, according to the DoEHLG, as follows:[78]

> All local authorities have now reviewed their Regional Waste Management Plans. It is evident from the plans that local authorities have been guided by the various policy statements on waste. In particular, these plans make provision for the development of an integrated waste management infrastructure, including
>
> - "kerbside" collection of recyclable materials in urban areas;
> - "bring" facilities for recyclable materials in rural areas;
> - civic amenity sites and waste transfer stations;
> - biological treatment of "green" and organic household waste;
> - materials recovery facilities;
> - recycling capacity for construction and demolition waste; thermal treatment facilities; and
> - residual landfill requirements.

Under the recommended approach we will presumably we will now have one NWP which deals with all these issues.

[77] It may, of course, be the case that the issue of the motivation or rationale for the NWP would inform what responsibilities it would assume.

[78] These are taken from the DoEHLG’s website. For details see: http://www.environ.ie/en/Environment/Waste/WasteManagementPlans. Accessed 4 September 2009.

The Eunomia consortium is silent not only on which responsibilities should be assigned to which level but also what criteria should be used to make the decision. It does not discuss the degree to which in essence policy is already national in that the DoEHLG sets the framework and if the recommendations of the international review are implemented with respect to the pricing of externalities then these prices are set for Ireland as a whole. Instead, the Eunomia consortium sidesteps the issue completely by recommending all responsibilities should be assigned to the national level. This is a major shortcoming.

Comment #3: Some Thoughts on Assigning Responsibilities

In Section 6 above we set out the sorts of factors that need to be taken into account in allocating responsibilities between different levels of administration. To repeat, policymaking should apply a high degree of "subsidiarity", since the bulk of service costs and externalities are local or regional in their causation and incidence. Most of the relevant economic markets are also local or regional. The problem identified by the CCMA may mean that some thought needs to be given to appropriate co-ordination mechanism, since there may be negative externalities being generated by existing arrangements, but that requires identifying the problem and then the appropriate solution. The international consortium does neither.

Too much centralisation of policy will leads to a substantial risk of regulation being applied that is inappropriate for many areas. In particular, centralised command and control measures such as blanket rules on what practices and technologies may be used, to what extent, and in what places, are likely to be very costly to Irish society. In rural areas for example it may not be appropriate to have household waste separated into several streams for collection, whereas this may be appropriate for an urban area with a much more dense settlement pattern. Equally, while an MBT plant might be appropriate for a rural area, it ma be less suitable for an urban area, where incineration might be preferable. There will be a loss of experimentation if all policy emanates from the centre. Finally, centralisation may not be able to adequately take into account complementarities between (say) particular residual waste collection technologies and the sorting, collection and disposal of household waste

In sum the Eunomia report provides no compelling reasons for moving to a NWP and abolishing the RWMPs.

8.12 Household Waste Collection: Irish Exceptionalism – Is it Justified?

In the penultimate block of recommendations relate to the issue of household waste collection. The recommendations are predicated on certain findings which may be summarised as follows:[79]

- Household waste collection costs in Ireland are "astronomically high given the quality of the service (in terms of coverage and frequency) to which this applies" (Eunomia *et al.*, 2009, p. 20).[80] This appears not to be due to high input costs or

[79] Much of the analysis is based on a County and City Managers' Association position paper on waste management, much of which is devoted to household waste collection (CCMA, 2007).

[80] This statement is based in Annex 64 of the international review, which suggests savings in the order of 20-40%. (This is derived as follows: Eunomia *et al.* 2009, Annex 64 estimate the savings from switching from competition in the market to competition for the market as €80 per household (Table 64-9, p. 1048), while the current charges per household are between €220 and €420 per household per annum (p. 1038)). This estimate of the savings does not seem out of line with the empirical evidence from elsewhere which is

rurality, but rather the "issue is far more fundamental and relates to the interplay of factors" (Eunomia *et al.,* 2009, p. 20). The international review finds it surprising that there has been no meaningful investigation into this issue. Its analysis is to be found in Annex 64 of the international review.

- The international review that the "issue of costs cannot be completely divorced from the matter of configuration of the waste collection market" (Eunomia *et al.,* 2009, p. 20). The current system for household waste collection is characterised by the public and private sector operators competing on the same route. Such arrangements are unusual by international standards, where either the local authority provides the service or it is tendered for a 5 to 7 year period. The international review could only identify Poland and Kosovo within Europe as having household waste collection arrangements similar to Ireland.
- The powers of the local authority in relation to the regulation of household waste are somewhat ambiguous. For example, since households do not have to avail of household waste collection this creates difficulties and uncertainties if a local authority want to tender for household collection services to private operators, since the size of the market cannot be determined with precision as a basis for estimating costs for a tender. Although not stated it appears that Ireland is unusual in that households can opt out of household waste collection.
- Household waste charges by local authorities may have to cover more than the cost of providing the collection service, since there are some waste services that do not cover their costs, and it is not clear whether or not DoEHLG and other funding is sufficient. In other words, there may be a cross subsidy from users of household waste collection services to pay for local authority civic amenity and other services. If this is the case, it would result in reduced usage of the publicly provided collection service as prices rise to and households switch either to lower priced privately provided services or decide not to avail of any household waste collection service. The international review felt that "it has become very clear to us that some form of local government taxation, or precepting of funds (which exists in most other jurisdictions) would be a desirable counterpart to the provision of quality local waste management services" (Eunomia *et al.*, 2009, p. 23). According to the Commission on Taxation (2009, p. 428), "Ireland is one of the few countries which does not impose a tax on domestic property to fund local government**.**"
- The power of local authorities to direct waste to certain facilities may result in those facilities being able to charge monopoly prices as well as possibly imposing significant costs on private operators.

Thus it would appear, in part at least, problems in household waste collection come about because of Irish exceptionalism. Based on this analysis a number of recommendations are made by the international review which are presented in Box 8.6.

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summarised in Competition Authority (2005, Box 1, pp. 41-42). Hence the claim that costs are 'astronomically high' is perhaps exaggerated.

Box 8.6: **Eunomia Consortium Household Waste Collection Recommendations**

R20. We recommend that the Waste Management Act is amended such that household waste collection is made the responsibility of local authorities. Household waste may be collected only by the local authority itself, or by an enterprise acting on its behalf. As under the existing Section 33 of the Waste Management Act, local authorities would be entitled to deliver services jointly …. We suggest that this change makes clear that the law will enter into force at a later date (we suggest 2014). It is important to i) legislate for this early and ii) allow time to elapse before the law enters into force (p. 58).

R21. We recommend that the Waste Management Act also makes clear that no local authority should enter into a contract for waste collection, treatment or disposal for a quantity of waste which exceeds the quantity which it, or a contractor acting on its behalf, can reasonably expect to be in direct control of. We suggest that this change makes clear that this aspect of the law will apply to all contracts extending, in the case of collection, more than 2 years beyond the entering into force of the above clause, and in the case of treatment, more than 7 years beyond the entering into force of the above clause. The distinction between 'contracting' and 'a facility's capacity' needs to be made clear.

For the avoidance of doubt, we recommend that the Waste Collection Permit Regulations be amended such that it be made clear that commercial waste cannot be directed to a *specific* facility of any type, and that as far as residual commercial waste is concerned, the treatment / disposal of the material would be expected to be determined by the market and the relevant legislation in place (the levy described [in Box 8.4 above] ... is unlikely to make landfilling an attractive option). This measure should apply as soon as possible after drafting (p. 60).

R22. We recommend that once [R20 & 21 have] …been implemented, all households will be required to use, and pay for, the waste collection service provided by the local authority or its contractor (subject to waivers etc.) (p. 61).

R23. Legislation should be implemented to make clear that backyard burning of waste is illegal (p.61).

Source: Eunomia et al. (2009)

The international review anticipates that R20 should partially resolve the issue of high collection costs of household waste, since there will be competition *for* the market, not as at the moment, competition *in* the market. As a result collectors will, for example, be able to take advantage of economies of density. The purpose of R21 is to bring clarity to the market. One of the implications, according to the international review, is that where local authorities have or expect to enter into contracts to supply a facility (e.g., an incinerator) in excess of what the local authority[81] collects, either directly or on its behalf, then these contractual arrangements will have to be revisited. Furthermore the international review is of the view that this recommendation combined with others will have the same effect on incineration as that envisaged under the Section 60 Policy Direction to cap incineration and other matters considered in Section 7 above.

R22 is linked to R20 in that if a household has to avail of a collection service then it makes it possible for the local authority to:

- Devise tenders which are more attractive to potential bidders;

[81] Or Regional Waste Management areas.

- Plan more coherently for treatment capacity for household waste;
- Implement waiver mechanisms for less well-off households in possession of more complete knowledge regarding the revenue implications of the total service (Eunomia *et al.*, 2009, pp. 60-61).

In 2007, 20% of households did not avail of household collection service in Ireland, which is more of a rural than an urban feature (EPA, 2009, p. 11**).** Apart from Turkey where the percentage is around 30%, other countries for which there is data, the corresponding percentage is less than 10%, with in the case of the US, 0% (OECD, 2008, Table CS3.1, p. 232). R23 is designed to bring clarity to the situation with respect to the law on backyard burning of waste, which may be substantial given the significant number of households that do not avail of waste collection services. However, as Eunomia *et al.* (2009, p. 61, fn 73) acknowledges regulations are being introduced that prohibit "disposal within the curtilage of a dwelling."

These recommendations are broadly speaking consistent with the economic approach. However, R20 needs to be refined so as to exclude the possibility of a public sector monopoly in household waste collection, while R21 seems inconsistent with the overall thrust of developing integrated regional waste management plans. R22 is a natural corollary of R20. However, it is not at all clear that the results of R21 will have the same impact on incineration as the Section 60 Policy Direction to cap incineration and other matters. It is to these issues that attention now turns.

Comment #1: Household Waste Collection: Competitive Tendering, not a Public Sector Monopoly.

There are good reasons why there should be only one supplier of household waste collection in any geographic area; in particular the ability to realise economies of scale, scope and density. Since it is generally recognised that there are these economies the issue becomes how to select the firm – public or private or some combination of the two – that should be given responsibility for collecting waste. The normal mechanism is a tender process when the tender guarantees the winner the exclusive right to collect household waste over a number of years for a defined set of households. Part of the bid is a schedule of collection charges, with the lower the level the greater the probability that the firm will win the tender. In other words we have competition *for* the market, rather than competition *in* the market. Under these arrangements, if the public sector provider is the incumbent, then it will like any private sector firm have to bid for the tender. To avoid possible conflicts of interest the public sector operator can be turned into a quasi-firm, sometimes referred to as a direct labour organisation, and compete with private sector operators.[82] It would also be necessary to ensure that local authorities were treated no more favourably than the private sector with respect to the application of VAT.[83]

In the discussion surrounding R20 international review clearly envisages competition for the market. However, R20 is silent on the issue of how the local authority selects who should collect the waste. Indeed on one interpretation of R20 it appears that the local

[82] This was the solution that was adopted in the UK to resolve the possible conflicting roles of the local authority as both a purchaser and provider of household waste collection services when it introduced compulsory tendering procedures. The OECD (2008, p. 334) also points out this potential conflict of interest.

[83] This is the object of R15 Eunomia *et al.* (2009, p. 54).

authority could award the contract to itself without any tendering process. If the efficiency savings that the international review are seeking are to be realised then the overwhelming message from the literature cited by the international review is that a tendering process is required in order to award the right to collect household waste, with any incumbent public (or private) sector operator given no advantage. Indeed, one of the reasons that tendering – or as it was referred to, compulsory competitive tendering – was introduced in the UK was a concern that public sector monopoly providers had become high cost and inefficient, a verdict that was verified by subsequent developments.

In introducing tendering it is important that there is clarity and certainty over the size of the market that is to be the subject of the tender and that enforcement of prohibitions against alternatives such as backyard burning and fly tipping is effective. Hence R22 and R23 are important for the successful implementation of competition for the market.

Comment #2: Should the Local Authority (or Regional Waste Management Plan) Direct Waste?

The issue of whether or not the local authority should be able to direct waste is an interesting issue. Eunomia *et al.* (2009) take the view that the local authority should not be able to contract for waste in excess of what it collects or what is collected on its behalf. As such the local authority cannot issue instructions that waste be directed towards a particular facility, such as a MBT or incineration plant. Since the local authority is responsible for household waste collection, the recommendation applies to commercial waste. Here, the international review argues that decisions will be made by the market and the relevant legislation. However, we know from the discussion above that markets may not always work well in providing certain kinds of waste infrastructure because of the hold-up problem. In part this recommendation reflects the need to "bring order to the current system and to clarify where the freedom of the market begins and ends" (Eunomia *et al.*, 2009, p. 60).

Before agreeing with the recommendation, it is useful to ask why might a local authority (or local authorities acting together with respect to a RWMP) want to direct waste to a particular facility. Recall that the local authority is responsible for preparation and administration of the regional integrated waste plan. It is therefore in some sense not like every other participant in the process and hence there may be grounds for it to be given such powers. But that does not answer the question of why it should have the power.

The local authority, for reasons discussed above, may want to enter into take or pay contracts with incinerators or other waste residual infrastructures. If the volume of waste is in excess of what the local authority collects then the local authority will have to find additional sources of waste. The recommendation appears to prevent that option and thus prevents potentially useful mechanisms from being adopted with the result that less efficient arrangements will be used.

We would suggest that the local authority be able to: (i) contract with other waste collectors in order to be able to enter into take or pay contracts or for such other purposes to meet the overall objectives of the regional waste management plans; and (ii) that they have the right to direct waste. In directing waste the local authority would have to satisfy certain conditions: first, the direction relates directly to the goals of RMWPs; that the direction is least restrictive of competition; third, that the direction is proportionate; and, fourth, account would need to be taken of any prior commitment that the owner of the waste may have made, particularly an investment in infrastructure with high sunk costs.

Eunomia *et al.* (2009, 23-24; p. 59) are concerned that giving the power to local authorities will result in competition problems and excessive prices. More specifically Eunomia point out that if the local authority has the power to direct privately collected waste to a particular facility then that facility is in a monopoly position. The owner of a landfill – typically the local authority – will charge the waste collector a monopoly price for using the landfill; equally if waste is directed to an incinerator or an MBT plant, these facilities will pay very low prices consistent with their position as a single purchaser. In support of their contention Eunomia *et al.* (2009, p. 24) refer to CCMA (2007) complaining that private operators switch between landfills and thus drive the price down. This is indeed a potentially serious problem. In the latter case if there is extensive excess landfill capacity then it is makes sense that the price of landfill should drop and that landfill – taking into account all the externalities and the Landfill Directive targets – be used more than other disposal methods. The question is though should these potential competition problems result in local authorities not having the power to direct waste, given that there are, as noted above, good reasons for local authorities to have such a power. In this context the proposed safeguards set out above should ensure that the power is used by local authorities so that these competition problems do not arise.

Comment #3: Section 60 Policy Direction to Cap Incineration and Other Matters by the Backdoor?

The international review claims, in discussing R21, that if this and the other twenty-four recommendations are implemented in full then this will,

> render the need for measures to limit incineration capacity, envisaged under the Section 60 Policy Direction, more or less redundant. Incineration would be no more, or less, constrained than other residual waste treatments, but emphasis would remain on shifting up the waste management hierarchy, leading to the achievement of significant environmental benefits (Eunomia *et al.,* 2009, p. 60).

The other recommendations that would appear to be particularly relevant to reducing the role of incineration are: R8, on proposed residual waste levies; R18, on the treatment of bottom ash; R19, on the abolition of RWMPs to be replaced by the Minister; and, although not a formal recommendation, provisions for planning law to facilitate the rapid roll out of MBT plants through the bypassing of normal planning procedures by designating all such plants as strategic.

Apart from the above statement there is nothing in the international review that links recommendations with the prediction that incineration capacity will be lower absent the implementation of these recommendations. In the final section, for example, of the international review, ‘Expected Results’, (Eunomia *et al.,* 2009, p. 73-78) there is no attempt to quantify the impact of the recommendations on incineration capacity nor in the relevant Annex, 63. Furthermore, in recent discussions of the Poolbeg incinerator it appears that Dublin City Council have only undertaken to provide 300,000 tonnes[84] of waste to the incinerator which has a capacity of 600,000 tonnes. The other 300,000 tonnes is the responsibility of the incinerator owner and there is no reason to assume that the ability of the incinerator owner to find the other 300,000 tonnes will be affected by the recommendations cited above. The possible exception is the residual waste levies, but as noted above it is not clear what exactly is being proposed by the international review in this respect due to the ambiguity in the structure of the levy for incineration and

[84] This is the waste that Dublin City Council collects or is collected on its behalf.

the lack of any discussion as to how the levy was determined. Hence it is possible that the residual waste levy on incineration will adversely effect the viability of this option. However, there is nothing in the international review that addresses this issue apart from the assertion above concerning the Section 60 policy direction to cap incineration and other matters.

8.13 Renewable Energy Incentives

The penultimate recommendation in the international review concerns the mechanisms for the use of biogas from anaerobic digestion to “ensure that they are aligned with the relative environment benefits” (Eunomia *et al.*, 2009, p. 61). The international review takes the view that for Ireland the best use for biogas would be as vehicle fuel. Reference is made to Annex 63 as supporting this view. The recommendation following from this very brief discussion is that:

> **R24** Incentive mechanisms need to be examined to ensure that they are correctly aligned with the options for use of biogas, in terms of their environmental performance. This needs to be addressed swiftly to ensure that projects which are in, or are approaching, development are incentivised to operate in the most beneficial manner.

The justification for this recommendation is somewhat cryptic. Although there is a discussion in Annex 63 of ‘Combustion if the Biogas in a Gas Engine’, it is not entirely clear how this relates to the recommendation. Hence the international review needs to flesh this recommendation out in order for a judgment to be made as to its worth.

8.14 Concern with the Welfare of Others: Border Issues

The final recommendation centres on the argument that certain recommendations including: R8, residual waste levies; R1-R7 & R9-R13 designed to increase recycling and the targets under producer responsibility; and the EPA’s approach to pre-treatment, will create the incentive for greater export of waste. The international review takes the view that Ireland needs to align its policy with international best practice. Thus:

> **R25.** It is recommended that the proposals in Annex 65 are considered with a view to ensuring that exports of waste which do take place are legitimate, and that they will not give rise to problems when exported (p. 62).

The 76 page Annex 65 contains fifteen recommendations on not only the export of waste but also the importation of waste. The summary report makes no attempt to summarise or justify the particulars of these recommendations, while the justification for the recommendations is not always clear. Why for example should Ireland “steer transfrontier waste shipments to the highest quality treatment” or “prohibit … the import for disposal of hazardous wastes, waste collected from households, or residues arising from incineration of household waste, in order to safeguard the treatment capacity (existing or to be developed) for Irish waste” or “that waste cannot be exported if a higher quality of waste treatment is available in Ireland” (Eunomia *et al.* 2009,. Annex 65, pp. 72-73)? Some of these recommendations would certainly interfere with trade and thus may fall foul of WTO rules, while not always appearing to be in the interests of Irish residents.

8.15 Conclusion: Does the International Review Provide a Roadmap for Waste Policy?

The purpose of the international review is to provide a roadmap for waste management policy in Ireland. A fresh beginning so to speak; a blueprint for change. A roadmap normally implies clear well reasoned and explained directions towards a well defined objective. In this section we consider that despite some positives, that the international review does not provide a roadmap for the way forward. We analyse the reasons for these failures and comment on the lessons drawn by the Minister for the Environment, Heritage and Local Government on the international review for future policy.

Moving in the Right Direction

The international review provides a number of important pointers concerning the way forward. It pushes a number of the right buttons in designing a waste management policy for Ireland. The international review quite correctly acknowledges the need to consider the economic implications of what is being proposed and that policy should be cost-effective and efficiently delivered. The international review also champions the principle that waste management policy should internalise any externalities. This principle is of paramount importance in the environmental area. Here there are a number of important *unpriced* externalities that need to be *priced* and hence internalised.

Apart from affirming these important general principles or considerations in framing waste policy, the international review makes twenty-five recommendations aimed at providing clarity to policy so that implementation can move forward swiftly. Many of these recommendations are to be welcomed. Basing residual waste levies on externalities is a useful application of the general principle set out above. The refundable compliance bonds to ensure compliance with C&D targets is a novel approach that merits serious consideration.

The recommendations concerning the introduction of competition *for* the market rather than competition *in* the market for household waste collection provide the basis for the realisation of substantial increases in efficiencies in collection, with householders benefiting through lower collection prices. The recognition that in applying producer responsibility, producers are to be responsible for full financial responsibility for delivering the services required to meet their obligations is likely to make this a much more effective policy instrument.

A Lack of Clarity

For a roadmap to be successful, however, it must provide unambiguous instructions, which can be easily understood. In the context of the international review this means that the recommendations must be clear, have a carefully explained and sound rationale and be credible. A characteristic of some of the most important recommendations made in the international review is that these conditions are not satisfied.

A central recommendation is setting of the residual waste levy for landfill, incineration and MBT, in terms of € per tonne. Here the international review fails completely to explain how its proposed levy structure was derived from the underlying research. Furthermore, an explanation is not at all obvious from that underlying research.

A major reduction in the volume of household waste will have important implications for residual waste treatment capacity. The international review recommends that the level of residual household waste per capita is to be halved over a 13 year period from 300 kg to 150 kg. England and Wales are cited as having similar targets, but they are taking 20 and

16/17 years, respectively. No explanation is provided as to why Ireland will be able to achieve these targets so much quicker and what the additional costs are of such a rapid reduction. Thus the target is not credible.

Short-term Issues: Meeting the Landfill Directive Targets and Other Matters

Pursuing the roadmap analogy a little further, the roadmap should provide guidance over a series of stages. In the short term, the most important goal of waste management policy is ensuring that Ireland meets the Landfill Directive targets for 2010, 2013 and 2016. Failure to do so will result in, potentially at least, large EU fines. Here the international review must be considered a failure. The international review does not set out the magnitude of the problem, except for 2010. It prefers not to forecast the likely magnitude of BMW for 2013 and 2016 on the grounds that it is extremely difficult and that there is considerable uncertainty. Curiously, two contradictory sets of forecasts are used for different purposes in annexes to the report, and they are neither compared nor used in the main report.

However, Eunomia could have considered alternative scenarios to assist in guiding policy. Does it matter if BMW grows at 1% or 4%? Which is more likely? The scenarios indicate a range of possible outcomes, depending on the assumptions made. Suppose, for example, under all reasonable scenarios there is a large volume of residual waste for 2013 and 2016. If this was the case then it would make sense to invest in (say) an incinerator since these have a large sunk capital component and need to operate at close to capacity to achieve maximum efficiency. If, on the other hand, the evidence was much more equivocal about what the level of residual waste, then it would make sense to invest in technologies that could be ramped up at short notice or perhaps waste could be exported for disposal.

The point of internalising externalities as the international review quite rightly points out is that once these external damages or benefits are incorporated into the price then appropriate decisions are made by public and private agents concerning which waste management technology to use. Hence, it is important that the methodology behind international review's estimates of the externalities from landfill, incineration and MBT is well grounded and defensible. Unfortunately this is not the case. No account is taken of disamenities caused to households because of the presence of a waste facility. In estimating the externality attention should only be paid to *unpriced* externalities. If an externality, such as pre-combustion emissions from diesel, is already priced through the carbon tax - which was announced in the budget on 9 December 2009, or through the ETS for CO_2 emissions, it should not be included in the residual waste levy. This is not only double regulation, but also double counting. The international review does not take this into account and hence its levies cannot be relied upon as sending the correct price signals for selecting between alternative waste management technologies.

Strategic Issues: Waste Management Technology Mix and Other Issues

Turning now to the strategic or longer term issues. In the discussion in Section 7.3 above of the Section 60 policy direction to cap incineration and other matters it was argued that there was merit in considering the implementation of the Section 60 policy direction in conjunction with the international review which was to deal with the issue of mix of technologies. This proved a vain hope. The international review did not provide any guide to the mix of technologies. It is completely silent on the issue of the merits of MBT over incineration, except to say that some countries seem comfortable with high usage of incineration while the practical problems of switching from incineration to the MBT, such as stranded assets if a 30% regional cap on incineration were introduced, are not addressed.

The international review claims that the effects of its recommendations will be to achieve the objectives on the limitation on incineration envisaged in the Section 60 policy direction to cap incineration. At the same time, although not a recommendation, Eunomia suggest that all MBT plants are strategic and hence the planning process should be fast tracked for these facilities. These two policy strands are complementary in that the MSW that would have been disposed of by means of incineration will now (presumably) have to be sent to MBT plants if Ireland is to comply with the Landfill Directive targets and hence avoid large EU fines. However, the international review does not provide any guidance as to the feasibility, location, timing, nature, cost and legality of fast tracking the building of MBT plants, nor does it provide any credible evidence that its recommendations will lead to a limitation on incineration consistent with the Section 60 policy direction to cap incineration and other matters that at the same time conforms with the Landfill Directive.

In sum, despite while the international review sets forth some sensible general principles for guiding policy and makes some welcome recommendations with respect to household waste collection, producer responsibility and refundable compliance bonds for C&D, the international review must be considered a failure in respect of its proposals for setting residual waste levies, per capita targets for reduction in residual waste and guidance in the appropriate mix of waste technologies.

Why Did the International Review Fail?

There are a number of reasons for this failure. *First*, the task is too big and the terms of reference were too wide. *Second*, in the 2007 Programme for Government and Section 60 policy direction on incineration and other matters constraints were placed on policy choices for no coherent or compelling reason. In other words, incineration was to be disfavoured and MBT encouraged. Starting from what appears to be an ill-thought premise and then write a coherent report is a difficult task even for the best of researchers.

The preference for MBT over incineration is most clearly demonstrated in terms of reference for the RIA on waste levies which stipulated that the levy rates should be set to avoid providing any "competitive advantage" to incineration and to encourage MBT rather than set levies to manage externalities.

It may also explain the structure of the waste levies in the international review, which based in the relevant annex, should have been:

Landfill - **€86.54** to €129.14 per tonne

Incineration - **€36.20** to €58.11 per tonne

MBT - **€20.70** to €42.05 per tonne

But were in fact the figures in bold were selected. This of course could result in an MBT plant being built that should be paying €42.05 per tonne but instead is paying a much lower and incorrect price of €20.70 and thus only half the externalities are being internalised. This is, of course, inconsistent with the declared aim of the international review that externalities should be internalised.[85]

Third, there is no clear methodology set out to draw lessons from the international experience or to guide policy. Despite nods towards greater attention to costs and benefits there are conspicuous by their absence. *Fourth*, the international review is far too accepting of EU policies and targets which are a complex set of compromises that are not always in Ireland's interests and which as such need to be questioned and challenged to ensure that they do reflect such interests.

Taking the International Review Forward: the Official Reaction

The Minister for the Environment, Heritage and Local Government in launching the international review said that it:

- " will create jobs in new waste industries"
- "will enhance competitiveness of the wider economy as a whole"
- is "considered research which is the essential foundation for good and robust policy;" and,
- "we have a blueprint for legislative, institutional, regulatory and organisational change to achieve a wholly sustainable approach to waste management" (DoEHLG, 2009, p. 1).

While it is true that if the international review's recommendations and suggestions were implemented that jobs would be created in new industries – building MBT facilities. However, at what cost, given that these resources could have been used to create jobs elsewhere in the economy and that these MBT facilities will to a considerable degree be replacing incinerators that are or will be built shortly. Apart from the compensation that may have to be paid to developers of these incineration facilities, it is extremely unlikely that Ireland would meet its Landfill Directives so that EU fines would be levied on Ireland. One would have thought in a period of financial stringency this not a good use of resources.

There is nothing to suggest that the international review will enhance international competitiveness. Rather the whole process of waste management policy over the past two to three years has generated unnecessary uncertainty that is likely to have increased regulatory risk and thus raised investment costs and biased technological choices, so damaging consumers and taxpayers. Moreover, by not achieving a least-cost mix of waste management facilities and practices, the proposals are likely to lead to higher prices for users of waste management services.

While the Minister is correct in stating that considered research is the essential foundation for good and robust policy, the international review does not fulfil that description for reasons set out above. Although the international review provides a blueprint it is very unlikely that it will lead to a wholly sustainable waste policy.

[85] As noted above there are difficulties with the approach used by Eunomia to estimate externalities.

9 A Concluding Note

It is important to get waste management policy right. Significant investment decisions need to be made in assets with a high element of sunk costs, whether it they be an incinerator or an MBT or a landfill facility. An efficient and cost-effective waste management system will enhance Ireland's competitive position. A stable predictable policy environment is essential for investment decisions in assets that have life spans measured in decades rather than years. Hence any new policy needs to tread carefully, particularly where there has been an accepted well developed policy in place for some years and on which investment and other decisions have been made. This is not to say that change should not take place, but to caution that careful consideration needs to be given to change.

The government has undertaken to "adopt and implement a new Waste Policy following the completion of the" international review (Department of the Taoiseach, 2009b, p. 24). There are number of useful lessons from the international review that should, in our view, guide any new policy. These include:

- Consider the economic implications of what is being proposed;
- Policy should be cost-effective and efficiently delivered;
- Externalities should be internalised; and
- Household waste collection should be based on competition *for* the market not competition *in* the market as is currently the situation.

These are unexceptional lessons which are consistent with the current waste management policy and other studies of waste management which, for example, have called for competition *for* the market (e.g., Competition Authority, 2005).

Nevertheless there is much in the Section 60 policy direction to cap incineration and other matters and the international review that in our view is not likely to increase societal welfare, which we take to be the object of policy. Unjustified arbitrary limits on incineration and consequent expansion of MBT have no place in waste management policy. The international review's setting of residual waste levies is flawed, suffering from both double regulation and double counting, with the result that the levies are much higher than is appropriate. There is much else that is wrong which we discuss extensively in Sections 7 and 8, while putting our own views as to the way forward in Section 6 above.

It seems to us that all policy measures should answer the question set out in the government's better regulation agenda – is it necessary? In the case of the move to reduce the role of incineration and increase the role of MBT that basic question has not been asked. This is unfortunate. Waste management policy, which made no such assumption until a couple of years ago, is suddenly focussed on this single untested objective which determines waste management policy. While there is much discussion about a lost generation of young people who cannot get jobs in the recession, we think that the last couple of years have set back the development and implementation of waste management policy based on an untested premise.

Annex A: Externalities

1.1 Introduction

Externalities are defined as the costs and benefits that arise when the social or economic activities of one group of actors (people/firms) affect another group of actors and the effects are outside the pricing, transfer, and regulatory systems i.e. "external" (Eshet *et al.*, 2005). An often cited example is when a firm pollutes a river but does not compensate downstream river-users for this damage.

In the absence of a normal market signal, i.e. a price, we must use other methods to construct a price for such external damages (or external benefits). The approach to reaching an economic valuation of, for example, emissions of methane from a landfill or high odour levels from an MBT plant, involves identifying the source of the externality, identifying how the externality is dispersed, identifying who bears the burden or boon arising from its existence and, finally, calculating in money terms these costs or benefits. This is known as the impact pathway approach. Estimates of external benefits and costs are rare for Ireland; available estimates are unrelated to waste. In all instances in this report we therefore draw on the results of studies undertaken elsewhere, which are then adjusted or adapted to take into account local conditions.

Externalities can occur at all stages of the life cycle of waste: at generation of waste, at collection of waste, at transport of waste, at recovery of waste, and at disposal of waste. Before discussing these externalities further, two things should be borne in mind: 1) uncertainty in identifying the environmental and health impacts of the waste life cycle, and, 2) unresolved methodological issues in measuring these impacts. Both issues are symptoms of the general problem of uncertainty in economic analysis. Sometimes estimates and forecasts will be wrong. Decision makers will always be faced with imperfect information; it is impossible to adopt an optimum strategy *a priori.*

In this annex we employ a common framework to estimate externalities for landfill (in Section 1.3), incineration (Section 1.4) and MBT (Section 1.5). It can also be used to estimate externalities for composting plants, material recovery facilities and other waste treatment operations referred to in the main body of the text of this report. The various externalities of all three will be discussed, as will the factors that affect them, and the final prices we have estimated for them. These three forms of waste treatment were selected because it is around these waste treatment options that the policy debate in Ireland has taken place. A common methodology is used across all three forms of waste treatment, which is set -out in Section 1.2, while a conclusion is presented in Section 1.6.

1.2 Externalities and waste policy

As noted in Section 2 above the effects of externalities are commonly addressed by bringing private costs in line with social costs via a pricing mechanism or by applying direct regulation. In the former case taxes or levies reflecting the damaged caused are imposed on the externality generating economic activity; in the latter case permissible maximum levels of emissions of (say) CO_2 are set, often related to a specific technology. While there are, as noted above, pros and cons to each approach, they are usually considered to be substitutes rather than complements.

It is important that there is consistency across different sources of externalities (e.g. MBT, cement kiln, and so on). In other words, two sources of externalities in a similar situation should be treated in a similar manner; and, that any given source of externality should not be subject to several forms of instruments of intervention all designed to achieve the same objective. If there is not such consistency then there is a danger that government intervention may not only prove to be needlessly expensive, but it may also not achieve its objective. Furthermore, this would violate the basic principle of treating like cases alike, while arbitrary deviations from that principle also hamper government accountability.

The major way in which externalities of the kind generated by waste treatment facilities but also from manufacturing activity are regulated is through licensing which sets allowable levels of emissions and monitors compliance. For pollution from large scale production operations such as pharmaceutical facilities, cement kilns and power plants an Integrated Pollution Prevention Control ("IPPC") licence is issued by the EPA which is "designed to [ensure] … that emissions from the activity do not cause a significant adverse environmental impact."[86] Waste operations are subject to EPA licensing to ensure that the operation "will not cause environmental pollution."[87] This type of direct regulation sets the level pollution that is designed, implicitly at least, to be the right or correct or socially acceptable level. It has been selected instead of imposing a levy or tax reflecting the level of externality. As we shall see in the discussion below there is often a considerable range of values for a given externality, so it is understandable why direct regulation is preferred.

This does not mean, however, that there is no role for levies in relation to waste treatment facilities. For example, the ETS sets a price for CO_2 through a cap and trade system. In this annex levies are recommended reflecting two sources of externalities. First, methane. Although waste licenses set permissible levels via landfill gas concentration limits, it is nevertheless the case that this is such an important source of global warming that it was felt that a levy should be imposed across all the waste treatment options that are considered here. Second, the disamenity impact of each waste treatment option is likely to vary depending on, for example, the housing density in the immediate vicinity of the waste treatment. As a result disamenity by waste treatment type will be estimated. Third, it could be argued that if a levy is to be imposed on methane that one should also be imposed on CO_2. However, CO_2 from paper, textiles, wood and other organic materials is largely carbon neutral and, secondly, accounted for at source. Hence it is not appropriate that CO_2 from these sources should be subject to a levy. However, what about CO_2 from fossil fuels? If incinerators are sufficiently large we presume they will be subject to the ETS and hence any CO_2 emissions from fossil fuels will be taken into account, while for landfill the plastics do not give off any CO_2. For MBT and smaller incinerators, fossil fuel use will be captured under the forthcoming carbon tax, plastics are either recycled or sent to landfill and CO_2 from organic materials is largely carbon neutral, so again no process CO_2 is emitted. Hence there is no need to take into account CO_2 in terms of a levy.

Although we suggest that not all externalities need to be included in levies for reasons set out above, nevertheless in the discussion below we do present estimates of a broad set of externalities. We do this for a number of reasons. First, we wish to illustrate the range of plausible estimates for a given externality. It is the range, combined with uncertainty surrounding the estimate, which explains, in part at least, the preference for direct regulation. Second, should the preferred approach of dealing with externalities proposed

[86] As set out on the EPA website: http://www.epa.ie/whatwedo/licensing/ippc/. Accessed 5 October 2009.

[87] As set out on the EPA website: http://www.epa.ie/whatwedo/licensing/waste/. Accessed 5 October 2009.

in this report be rejected in favour of levies then these estimates will, we believe, provide a useful starting point for policy-makers. Third, the discussion of the derivation of the pricing of externalities does, we believe, provide useful insights into the channels by which MSW and its treatment alternatives affect the environment, human health and so on.

1.3 The externalities of landfill

Introduction and Irish context

More research has been conducted on landfill externalities than on those of other waste processing and disposal options. The main external cost landfill creates is through the production of landfill gas, the end product released from the degradation of biodegradable waste. Its main components are greenhouse gases; by volume it is made up of approximately two-thirds methane and one-third carbon dioxide (DEFRA, 2004a).[88] Some 64% of biodegradable waste is landfilled in Ireland, making landfill a substantial contributor to the production of methane (EPA, 2009a; Smith *et al*., 2001).

An important feature of landfill emissions is the timescale over which they arise. The majority of emissions from biodegradable waste materials typically take place over a period of some 20 years following disposal. However, generation of landfill gas and leachate will continue at a lower rate for many years. This is an important point because the poor management of landfill in the past will impose considerable costs on the present and future, even if all current landfills are well-designed and meet EU and EPA requirements.

Greenhouse Gas Emissions

In common with the approach suggested in the international review, this report values the climate change impact of carbon dioxide at its **current market price in the Emissions Trading Scheme** ("ETS"), with prices taken from www.eex.eu), where such prices are available, and at an assumed medium term price for period where there is currently no trading price. Methane emissions are valued by multiplying the CO_2 price by the global warming potential ("GWP") of methane, currently set at 21 by the Intergovernmental Panel on Climate Change ("IPCC").[89]. This approach is consistent with the Irish government's treatment of GHG emissions in public expenditure appraisals (Department of Finance, 2009) and assessment of projects by state enterprise agencies (Lyons and Tol, 2008). It has the benefit of ensuring consistency of GHG pricing across sectors (i.e. between waste sector and ETS sector emissions) and with other European countries.

This consistency is important to avoid distortions arising from policy. For example, using an ETS-based price ensures that the reductions in fossil fuel carbon emissions brought about by generating energy from incineration or biogas are valued at the same price as the

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[88] There are two stages of the report published by the UK's Department for the Environment, Food and Rural Affairs ("DEFRA") that will be referred to frequently in this section. DEFRA (2004a) will refer to the first stage of that report, which reviewed the health and environmental impacts of landfills and incinerators. DEFRA (2004b) will refer to the second stage, which put an economic value on these impacts

[89] Global warming potential ("GWP") is a measure of how much a given mass of greenhouse gas is estimated to contribute to global warming. It is a relative scale which compares the gas in question to that of the same mass of carbon dioxide (whose GWP is, by definition, 1). GWP is calculated over a specific time interval; in the main text, a GWP of 21 for methane refers to its impact over a 100-year period from the date of emission.

emissions from waste facilities. In effect, generating energy from waste reduces the need to use ETS permits for fossil fuel generation, saving the relevant cost. Using the same GHG prices ensures that we are making a like-for-like comparison among options based on waste and fossil fuel inputs.

There are alternatives to our approach, and for completeness we provide a brief overview of the options below. The valuation of global warming damages is extremely complex (Tol, 2005; Anthoff *et al.*, 2009). In an extensive report commissioned by DEFRA on the externalities of landfill and incineration in 2004 (DEFRA, 2004b), a range of £9.5 – 38/tonne of CO2 was used (2003£); this is known as the social cost of carbon ("SCC"). The SCC is defined as "the discounted value of damage associated with climate change impacts that would be avoided by a marginal reduction in carbon emissions along an arbitrary trajectory" (Anthoff *et al*., 2009, p. 1).[90] In other words, it is the global marginal damage cost of climate change. The range used in DEFRA (2004b) was based on previous work conducted by the ExternE project ('Externalities of Energy', a European Commission programme that monetises pollution impacts from energy and transport). A number of points can be made about this earlier work. It based its range on marginal damage costs rather than global warming potentials (GWP). This is an important distinction: whilst the GWP of methane is 21, its impact (its marginal damage) is not necessarily 21 times greater than carbon dioxide - as both gases decay differently over time and marginal cost ratios are discounted, unlike GWPs. Therefore, by using marginal damage costs a more accurate reflection of the absolute and relative external costs of these greenhouse gases emerges. Secondly, the ExternE work was based on an economic model that largely excluded adaptation, which, if included, would lower the marginal cost estimate.

The ExternE work also used equity weights, which increases the marginal damage costs. Equity weights reflect the regional impact of climate change as damage done to the worst-affected regions is given a higher weighting in the calculation; countries in such regions are generally poor and heavily dependent on climate-sensitive sectors like agriculture. This decision - to equity weight the results - highlights just one of the ethical assumptions on which pricing carbon rests. Choices have to be made to reflect the value placed on inequity aversion and other factors such as valuing risk aversion and valuing the future.

Having established a cost per tonne of GHGs emitted, the next step is to estimate the quantity and timing of GHG emissions associated with each tonne of waste landfilled and to work out their external cost. Doing this required several assumptions. The assumed amount of degradable organic carbon[91] ("DOC") in the average tonne of material sent to landfill is based on the latest figures from the Environmental Protection Agency (McGettigan *et al*. 2009). These figures assume that the components of MSW turn to DOC at set rates (i.e. 15% of organic waste converts to DOC, 40% of paper converts to DOC) and also that these rates are constant across the lifetime of the waste. Once this

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[90] Since 2007, however, DEFRA has recommended using a "shadow price of carbon" ("SPC"). This different terminology and pricing methodology gives a slightly lower estimate of the cost of carbon dioxide. Their SPC is actually determined by an assumed emissions level - whereas a true shadow price would be determined by the intersection of marginal damages and marginal abatement costs. The UK's SPC is currently under review as it has received much criticism since its introduction.

[91] The portion of organic carbon present in such solid waste as paper, food waste, and yard waste that is susceptible to biochemical decomposition. EIA energy glossary:

http://www.eia.doe.gov/glossary/glossary_d.htm. Accessed 8/10/09.

figure is calculated a number of other parameters are needed in order to get the potential methane emissions from a tonne of landfill. The DOC is assumed to be 95% managed, 60% of it is assumed to dissimilate, 50% of this is assumed to give rise to methane and the managed methane conversion factor ("MCF") is assumed to be 1 with the unmanaged conversion factor 0.4. Again these parameters are assumed constant across the life of the waste. The methane emissions follow a lifetime emission path as developed under two stage first-order model (Christensen, Cossu, and Stegmann, 1996). The GWP of methane is assumed to be 21. As has already been discussed, methane's impact is not necessarily 21 times greater than carbon dioxide, but the discrepancy between the science of climate change and the policymaking of climate change is defended on grounds of consistency; this report adheres to the Irish government's treatment of GHG emissions in public expenditure appraisals (Department of Finance, 2009). Finally the carbon emission pricing is based on futures contracts traded on the European Energy Exchange up to 2012 which is the last available contract. Post 2012 carbon price assumptions are based upon Department of Finance (2009).

Table A.1 below illustrates the calculations for a hypothetical tonne of waste. Based on the assumptions discussed above, the methane-related GHG externality is €33.60 per tonne.

Table A.1: Methane generation from a hypothetical tonne of waste sent to landfill

	Methane Lifetime Distribution %	Methane Tonnes Emitted per Year	Converted to Carbon	CO2 price – current €	CO2 price – PV €	Total cost – PV €
Year 0	0.000	0.0000	0	13.4	13.4	0
Year 1	0.060	0.0048	0.102	13.7	13.3	1.348
Year 2	0.164	0.0132	0.278	14.3	13.4	3.712
Year 3	0.181	0.0146	0.306	15.1	13.6	4.171
Year 4	0.079	0.0064	0.134	16.8	14.6	1.954
Year 5	0.056	0.0045	0.095	17.9	15.1	1.431
Year 6	0.055	0.0044	0.093	39	31.7	2.955
Year 7	0.049	0.0040	0.083	39	30.7	2.543
Year 8	0.044	0.0035	0.075	39	29.6	2.207
Year 9	0.044	0.0035	0.075	39	28.6	2.132
Year 10	0.039	0.0031	0.066	39	27.6	1.826
Year 11	0.034	0.0027	0.058	39	26.7	1.538
Year 12	0.034	0.0027	0.058	39	25.8	1.486
Year 13	0.034	0.0027	0.058	39	24.9	1.436
Year 14	0.029	0.0023	0.049	39	24.1	1.183
Year 15	0.026	0.0021	0.044	39	23.3	1.025
Year 16	0.023	0.0019	0.039	39	22.5	0.876
Year 17	0.017	0.0014	0.029	39	21.7	0.626
Year 18	0.013	0.0010	0.022	39	21	0.462
Year 19	0.010	0.0008	0.017	39	20.3	0.343
Year 20	0.010	0.0008	0.017	39	19.6	0.332
TOTAL						**33.60**

Source: see text.

A final issue worth noting is that GHG emissions from landfill are affected by management practices. Table A. below illustrates this point with estimates from DEFRA (2004a).

Table A.2: Greenhouse gas emissions, by landfill management type

	Scenario A: Emissions wholly released to atmosphere via fugitive releases	**Scenario B: Emissions wholly released to atmosphere via landfill gas flaring**	Scenario C: Emissions wholly released to atmosphere via gas generating engine (i.e. energy utilisation)	**Scenario D: 75% gas captured by flaring and 25% released untreated to atmosphere**	Scenario E: 75% gas captured by generating engines and 25% released untreated to atmosphere
Methane (g/T MSW)	75, 000	**400**	2,000	**19, 000**	20, 000
Carbon Dioxide (g/T MSW)	130, 000	**220, 000**	350,000	**200, 000**	300, 000

Source: DEFRA (2004a, pp. 91-94).

Table A.2 above highlights the trade-off that must be made when choosing how to manage landfill. For example, flaring the landfill gas prior to release (Scenario B) produces more carbon dioxide than the alternative of doing nothing (Scenario A), but produces about 1/200th the amount of methane produced in Scenario A. The columns highlighted in bold are the ones most likely to prevail at a national level in Irish waste management. However, due to the concentration in the Dublin region of landfills that recover biogas, Scenarios C and E may be more useful when considering the Dublin region alone.

Leachate

Leachate is generated from landfills containing residual solid waste. It enters the local soil, groundwater and/or surface water, and can impact on human health and the local ecosystem. Leachate quantity and especially quality varies over time. The quantity of leachate generated depends mainly on the net precipitation and the type of landfill cover used. During the initial phases in the lifetime of a landfill, leachate typically contains very high concentrations of organic carbon, ammonia, chloride, potassium, sodium and hydrogen carbonate, whilst concentrations of heavy metals and specific organic compounds are relatively low (COWI, 2000). Leachate composition data is only available for certain landfills for a period of about 30 years; however, leachate generation may continue for several hundred years, and the long-term impact of this is unknown. Even if the impact pathway of leachate could be analysed in a satisfactory manner, there is no clear solution for the choice of time horizon and discount rate (Rabl *et al.*, 2008).

Dose response functions for the water environment are difficult to establish and generalise.[92] Some studies have been undertaken with respect to leachate but these (conducted in the 1990s in the US) have been of limited scope and the conclusions are not transferable. Aside from the difficulty of establishing the damage caused by leachate, the available valuation studies are also unsatisfactory. Valuation studies have frequently used

[92] Dose-response functions measure the relationship between exposure to pollution as a cause and specific outcomes as an effect. They refer to damages/production losses incurred in the current year, regardless of when the pollution occurs. Definition accessed on 09/09/09 at: http://stats.oecd.org/glossary/detail.asp?ID=6404

a clean-up costs approach, for example CSERGE *et al.* (1993). But this approach is not recommended for valuing external costs as it is not a true measure of the value of damage. At best it can provide a lower estimate of the expected range of values. The DEFRA study calculated a value of £1/tonne landfilled (£2003, ecosystem impact only) based on the CSERGE study mentioned above, but did not recommend using it.

A paper by Rabl *et al*. (2008) created a theoretical comparison between a dose of pollutants from leachate and a dose of the same pollutants from ordinary drinking water. The pollutants chosen were lead and arsenic (which have relatively high unit damage costs and an unlimited time horizon in leachate). Using regulatory limits for both leachate and drinking water to construct the argument, the paper concluded that the impact of leachate would be no worse than the impact of ingesting pollutants in normal drinking water. Thus, the impacts of leachate in a properly conforming landfill are negligible and do not need to be priced.

The Landfill Directive requires that all landfills receiving non-hazardous waste (which includes MSW) must have a barrier system to contain leachate. It is hard to gauge how many Irish landfills fully comply with this. In 2005, the Office for Environmental Enforcement ("OEE") published a report on non-compliance which detailed five prosecuted offences of inadequate landfill gas (LFG) and leachate management (EPA, 2005). This problem - failure to provide and maintain leachate and surface water management infrastructure – has not disappeared from the Irish waste management scene, despite the OEE's efforts. For example, Cavan County Council was prosecuted in 2007 for inadequate leachate management.[93]

Although Rabl *et al*. (2008) argued that leachate costs can be ignored, the crux of his argument rested on landfills adhering to regulations. The previous paragraph would suggest that this assumption cannot be taken as a given in an Irish context, even though landfills are typically operated by the local authority. However, considering the difficulties with establishing a monetary value and the need to transfer modelling from the UK in order to get a level of leachate, it is excluded from our analysis. This decision is consistent with other recent studies that do not value the external cost of leachate (Bartelings *et al*., 2005; and Dijkgraaf and Vollebergh (2004)).

Air Pollutants

The majority of landfill's emissions to air come from the greenhouse gases, but air pollutants also need to be considered. Many types of volatile organic compounds ("VOCs") have been identified in landfill gas.

The way in which pollutants react in the atmosphere to form secondary matter (such as nitrates and sulfates) makes it difficult to model the effects of air pollution with 100% accuracy. For example, ozone is formed from the interaction between nitrogen oxides and VOCs, but different modelling of this chemistry can lead to different answers for the amount produced of all three substances and thus their monetary cost.

Again, the way in which landfill is managed will contribute significantly to the amount of air pollutants it produces, for example if landfill gas is utilised for energy, releases of nitrogen oxides will be quite substantial. If it is not treated in any way, VOC emissions will be particularly high. Different scenarios are again provided in Table A. in order to

[93] Accessed on 18/09/09 from:http://epa.ie/whatwedo/enforce/prosecute/2007/name,23472,en.html

illustrate this. The columns highlighted in bold are the ones most likely to prevail in an Irish waste management context at a national level.

Table A.3: Air pollutant emissions, by landfill management type

Air Pollutant (g/T MSW)	**Scenario A: Emissions wholly released to atmosphere via landfill gas flaring**	Scenario B: Emissions wholly released to atmosphere via gas generating engine (i.e. energy utilisation)	**Scenario C: 75% gas captured by flaring and 25% released untreated to atmosphere**	Scenario D: 75% gas captured by generating engines and 25% released untreated to atmosphere
Nitrogen Oxides	**100**	900	**75**	680
Total Particulates	**8**	7	**6.1**	5.3
Sulphur Dioxide	**120**	70	**90**	53
Total VOCs	**1.7**	No data	**7.6**	6.4
Dioxins[94]	**74 ng TEQ/T**	190 ng TEQ/T	**55 ng TEQ/T**	140 ng TEQ/T
Cadmium	**No data**	0.1	**No data**	0.071
Nickel	**No data**	0.013	**No data**	0.0095
Arsenic	**No data**	0.0016	**No data**	0.0012
Mercury	**No data**	0.0016	**No data**	0.0012

Source: DEFRA (2004a, 91-94).

There are three recent studies which provide values for the externalities associated with environmental, agricultural and health damage: DEFRA (2004b); CAFÉ (2005); and Be-Ta MethodEx (2007). These are described in Table A.4, which also contains a discussion of the disadvantages and advantages of each for the purposes at hand. Rather than using one of these sources to measure externalities from air pollution, each is used for specific pollutants, which is noted in Table A.5. This discussion also applies to the economic valuation of air pollution from incineration and MBT.

[94] Dioxins are produced by incomplete combustion processes. They are resistant to biodegradation and accumulate in food (the main exposure pathway is by ingestion, not inhalation). The toxicity of each dioxin is indicated by its Toxic Equivalent Factor ("TEF"). 1 is the most toxic (assigned to the dioxin TCDD). On the assumption that the effect of various dioxins is additive, the TEF value for each is obtained and multiplied by its concentration. Thus a final Toxic Equivalent Quantity ("TEQ") is obtained (Guisti, 2009).

Table A.4: Overview of three major studies valuing externalities

Title	**DEFRA (2004b) – 'Valuation of the external costs and benefits to health and environment of waste management options'.**	**CAFE (2005) - 'Damages per tonne emission of PM2.5, NH3, SO2, NOx and VOCs from each EU25 Member State (excluding Cyprus) and surrounding seas'.**	**Be-Ta MethodEx (2007) – Version 2, February 2007, as part of the MethodEx Project for the European Commission DG Research.**
Author	Enviros Consulting Limited in association with EFTEC.	AEA Technology Environment	Mike Holland, EMRC.
Description	This was the second report in a study commissioned by the UK Department for Environment, Food and Rural Affairs (DEFRA) that sought to put an economic value on the health and environmental impacts of landfill and incineration (the scientific evidence for which was identified and assessed in the first stage of the study, DEFRA 2004a). It looked at the primary and secondary literature on externalities and from these chose appropriate money values and ranges. The information of most interest to our present work was actually a separate piece of original analysis on air pollutants included in this report that was carried out by AEA Technology Environment on behalf of DEFRA.	The aim of the Clean Air for Europe Programme (CAFE) is to establish a long-term, integrated strategy to tackle air pollution and to protect against its effects on human health and the environment. The work conducted here was part of the ExternE Research Project run by the European Commission.[95] It followed on from previous ExternE work, but made new assumptions regarding health impact assessment and also used outputs from the EMEP model for the year 2010 for assessment of pollution chemistry and dispersion.[96] The 'impact pathway methodology' for valuing external costs was used:[97] emissions and their sources were identified, their dispersion was calculated, exposure-response functions were applied in order to estimate health and crop damage (an example of such a function might involve the cases of asthma due to an increase in ozone), and finally this damage was priced.	This work also followed on from previous ExternE work. The objectives of the MethodEx project are to advance best practice in external cost assessment, and extend the original ExternE analysis where possible. Be-Ta MethodEx provides look-up tables for damage costs per tonne of emission for regional and global air pollutants, including trace metals and some organics. Compared to CAFE 2005, it made different assumptions about health impact assessment and particle damage. MethodEx used a discount rate of 3%; CAFE used a rate of 4%.
Advantages	The greatest advantage of the results obtained from their valuation of external	Whilst the modelling was not specific to a waste context, ***Ireland-specific***	It has the same advantages as CAFE 2005. Furthermore, it valued a

[95] ExternE (the Externalities of Energy) is a research project series that has been run by the European Commission since 1991. It has involved more than 50 research teams in over 20 countries, and its results and methodologies are well recognised sources for externalities estimation.

[96] The EMEP model collects emissions data using a 50km by 50km resolution, measures air quality using updated chemistry and meteorology, and models the depositions of air pollutants.

[97] This methodology was first developed within the ExternE project series but is widely used in other studies, including DEFRA (2004b).

	costs is that they were derived from ***specific modelling*** to reflect the exposure of populations around landfills and incinerators to emissions.	valuations for air pollutants were provided i.e. Irish data on emissions and dispersion were used. Furthermore, they used ***PM2.5*** as the metric for particulate matter, whereas DEFRA 2004b used PM10; PM2.5 is considered a better metric with which to quantify the impact on mortality of air pollution (COMEAP, 2007).	***greater range*** of air pollutants than the other studies.
Disadvantages	The report's lack of detail on pollutant modelling may make ***transferring*** its results to an Irish context difficult; the variety of sources the AEA Technology analysis used may also hinder transfer.	The exposure-response functions used were based on ***EU-averages***. The modelling was not specific to a waste context.	The exposure-response functions used were based on ***EU-averages***. The modelling was not specific to a waste context.

Source: see table.

Table A.5: Sources of externality valuations to be used for air pollutants

Study	**Results Used**
DEFRA (2004b)	None
CAFE (2005)	Primary air pollutants: NOx, SO2, VOC, PM2.5, NH3
Be-Ta MethodEx (2007)	Dioxins, heavy metals.

Source: see table

DEFRA (2004b) and Be-Ta MethodEx (2007) valuation of air pollutants are similar i.e. MethodEx's values (corrected for currency and year differences) fall into the DEFRA range.[98] As DEFRA's modelling was more site specific to waste management (e.g. it looked at point emissions from incinerators), this is a reassuring result. This makes the use of the MethodEx figures (which were based on more general modelling of emission dispersion and European averages for health and crop response functions) reasonably legitimate.

[98] With the exception of particulate matter, where the results aren't comparable as DEFRA analysed PM10 whilst ExternE and CAFE analysed PM2.5.

Box A.1: Valuing Mortality: VSL vs. VOLY

There is ongoing debate about how mortality should be valued. Two methods can be used:

VSL (value of a statistical life) can be defined as the willingness-to-pay (WTP) to avoid the risk of an ***anonymous premature death***. The aggregation of individual WTP values over a whole population leads to the VSL. For example, if the average person would be willing to pay €100 to reduce the probability of dying by 1 in 10,000, then a population of 10,000 individuals would be willing to pay €1 million to prevent one member of that population from dying prematurely. The €1 million figure is the VSL. Because it is not possible *ex ante* to identify the person whose life will be saved, this prevented mortality is considered a statistical life.

VOLY (value of a life year) can be defined as the willingness-to-pay (WTP) to avoid ***changes in life expectancy***. VOLY assumes that the WTP varies inversely with age. It is often calculated by adjusting the VSL figure. For example, a very basic conversion of VSL into VOLY involves dividing the VSL figure by the discounted years of remaining life expectancy. It should be obvious that the VSL figure will always be higher than the VOLY figure. Alternatively, VOLY can be calculated directly using specially-designed WTP surveys.

The willingness-to-pay (WTP) approach has its basis in the assumption that changes in individuals' economic welfare can be valued according to what they are willing (and able) to pay to achieve that change. According to this assumption, individuals treat longevity like any consumption good and reveal their preferences through choices that involve changes in the risk of death. The ExternE values for VSL and VOLY were calculated by following the contingent valuation method – this is a survey method where people are asked directly to state their WTP in certain hypothetical (or contingent) situations; it is a useful valuation method for those goods and services that are not traded in markets, like mortality risk reduction. Neither VSL nor VOLY is a measure of the intrinsic worth of life, but a way of valuing a change in risk.

Source: see text.

However, the MethodEx figures are not without some concerns. In terms of mortality valuation (which, after the costs of climate change, are the most crucial externalities to calculate correctly), MethodEx uses the median VOLY. The median is the correct descriptive statistic to use (as the distribution of income can be assumed to be highly right-skewed in a sample involving mortality valuation, thus making the mean figure less desirable as a statistic). But VOLY is not necessarily a better measure of mortality valuation than VSL. In fact, VSL, in our opinion, is preferred as VOLY assumes a relationship between mortality valuation and 'life remaining' that may not be strictly true i.e. VOLY relies on the argument that someone with, say, 40 years of life remaining and facing an immediate risk would tend to value remaining life more than someone with, say, 5 years of remaining life. To us, this line of thinking is neither obvious nor convincing; the *scarcity value of time itself* should not be ignored. [99] In any case, the VOLYs calculated by MethodEx (based on the 'NewExt' study in 2004) are calculated by adjusting the underlying VSLs. Such computations are not without their own issues

[99] It must be noted that this reasoning is not the current orthodoxy on the VSL vs VOLY debate. See Rabl (2003) for an opposing view.

either; e.g. the choice of discount rate used in adjusting the figure may be contentious and the calculation creates a simplistic relationship between the two measurements.

The MethodEx study follows roughly the same methodology as the CAFE study. The key difference arises over the harmfulness of different types of particles. The advice given to CAFE by the WHO was not to differentiate as the empirical basis for doing so is quantitatively weak so, for example, in their analysis NOx is, on average, just as harmful as PM2.5.[100] This method, according to CAFE, would also reduce the possibility of double-counting impacts. MethodEx considered the issue of separate impacts too significant to avoid, despite the literature on the subject being inconclusive. The other main difference is that CAFE did not specify which type of mortality valuation is preferred – they provide a range of results based on mean/median VSL and mean/median VOLY – whereas MethodEx explicitly favours using the median VOLY.

In conclusion, it is considered appropriate to use CAFE for our valuation of the primary air pollutants as we favour using the median VSL to value mortality; from the range of results they published, those that calculated the ***particulate matter mortality valuations*** by the ***median VSL*** will be used. These values do not, however, fall into the DEFRA 2004b range; in fact, they are of a magnitude 2-5 times higher. Yet this difference is less likely to be derived from the modelling of emissions and more a reflection of the unit values used for valuing mortality, as DEFRA's mortality values were also calculated using VOLY. Although it is regrettable that figures derived from a waste-specific context cannot be used to obtain the monetary cost per pollutant, the alternative still holds up well, most particularly in analysing the health impacts of air pollutants: CAFE's exposure-response functions are more up-to-date.

MethodEx analysed a wider range of pollutants than CAFE; they also provided monetary valuations for greenhouse gases, dioxins and heavy metals. Their valuations of dioxins and heavy metals will be used alongside the CAFE primary air pollutant results. This does, of course, raise the issue of ***inconsistency*** across figures used in terms of valuation methodology (MethodEx used the median VOLY). But the context in which these figures will be used must be remembered: the level of emissions of heavy metals in a waste management context is low compared to emissions of the primary air pollutants. With regard to dioxins, the MethodEx figures are the only reliable ones available at present.

The monetary values provided in the Table A.6 below must be viewed with caution. The science behind the interaction of air pollutants with each other, with human health and the environment is constantly evolving, and new values based on new knowledge and updated techniques will emerge. Economists often use the willingness-to-pay approach to get values for non-market goods, but, like the science of air pollution, it is reliable but not perfect. These figures are 'best estimates' based on what we know now about pollutant pathways, and how, in particular, we choose to value mortality.

[100] But note that relative PM2.5 risk from specific sources, e.g. from traffic or power plants, could be higher or lower in the analysis.

Table A.6: Damage costs for air pollutants

Air Pollutant	€/tonne (2000 €)	Relative Values (Normalised with PM2.5=1)[101]	Source
Arsenic (As)	80,000	3.64	BeTa MethodEx 2007 Note: For these pollutants, constant damages unrelated to site of emission are assumed, given that exposure is via multiple pathways, and food is transported over long distances. Therefore, Ireland-specific figures are unnecessary.
Dioxin[102]	37,000,000,000	1,681,818.18	
Lead (Pb)	600,000	27.27	
Mercury (Hg)	6,000,000	272.73	
Cadmium (Cd)	21,000	0.95	BeTa MethodEx 2007 Note: Cadmium, Chromium IV and Nickel are, according to current knowledge, carcinogenic only through inhalation, so Ireland-specific values (as provided here) are necessary.
Chromium (typical mix of CR species)	17,000	0.77	
Chromium IV	130,000	5.90	
Formaldehyde	66	0.003	
Nickel	2,100	0.10	
Ammonia (NH3)	4,000	0.18	CAFE 2005 Note: the range chosen was that which valued mortality by the median VSL.
Nitrogen Oxides (NOx)	5,600	0.25	
Particulate Matter (PM2.5)	22,000	1	
Sulphur Dioxide (SO2)	7,500	0.34	
Volatile Organic Compounds (VOCs)	950	0.04	

SOURCE: see table

There are no Irish data on the levels of air pollutants emitted per tonne of waste. Therefore it is necessary to transfer values from elsewhere. Values from the UK were chosen here, as they provide a credible example of the emissions pathways of air pollutants in an Irish context. These values were derived from scientific research in this area, emissions monitoring results from UK landfills in the late 1990s and early 2000s, and guidance produced by the UK's Environment Agency. Combined, this data can be taken to represent emissions arising from a landfill containing municipal solid waste. They are summarised in Table A.7 below.

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[101] Fine particulate matter (PM2.5) is considered the main culprit amongst public health experts for increased morbidity and premature mortality in an air pollution context. Therefore, as health impacts constitute the largest part of these externality estimates, it is chosen as the base here.

[102] Whilst the unit cost per tonne of dioxin emissions is very large, it must be borne in mind that its emission level in a waste management context will be quite low.

Table A.7: Summary of external costs of air pollutants per tonne of waste landfilled, by management type

	Scenario A: Emissions wholly released to atmosphere via landfill gas flaring	Scenario B: Emissions wholly released to atmosphere via gas generating engine (i.e. energy utilisation)	Scenario C: 75% gas captured by flaring and 25% released untreated to atmosphere	Scenario D: 75% gas captured by generating engines and 25% released untreated to atmosphere
Total external cost of air pollutants (2000 €)	1.64	5.74	1.24	4.34

Source: DEFRA (2004a), BeTa MethodEx (2007), CAFÉ (2005) and supplementary Table S.1.

Scenario C and D are the most plausible in Table A.7 as, in reality, a combination of emissions pathways is likely at landfill i.e. emissions will occur both by active landfill management and fugitively such as when gas escapes through cracks in the landfill's capping material. It is interesting to see that the lowest external cost in an air pollution context involves part of the landfill gas not being treated at all (Scenario C). However, it must be borne in mind that air pollutants make up only a small proportion of landfill's emissions to air; particularly, if Scenario C were to prevail, the methane emissions would be considerable.

We suggest that Scenario C should be used as a lower estimate and Scenario A as a higher estimate in a national context as only a minority of Irish municipal landfills are producing gas for electricity generation. All municipal landfills, however, have – or should have – flaring systems. However, due to the number of landfills in the Dublin region that extract biogas (Ballyogan, Baleally, Dunsink, Friarstown, Arthurstown), Scenarios B and D should be considered as the range if a Dublin-only analysis of air pollutant externalities were required.

Health

In the epidemiological literature, the link between human health and waste management is quite controversial. The main point to be taken is that the research has not proven that the negative externalities of landfills are harmful. It is unknown, however, if this is due to the improved technology of landfills or is due to the limitations of environmental epidemiological studies.

In Table A.8 below we list some of the possible health effects of landfill activity and the associated contributing emissions.

Table A.8: Health effects and contributing emission, landfill facilities

Health Effect	**Contributing emission**
Eye irritation	volatile organic compounds
Bronchitis	particulate matter, sulphur dioxide
Increased susceptibility to respiratory infection	sulphur dioxide
Asthma attacks	nitrogen dioxide

Reduction in oxygen-carrying capacity of blood	carbon monoxide
Effects on the central nervous system	lead, manganese, carbon monoxide
Effects on the immune system	lead, dioxins, mercury, polycyclic aromatic hydrocarbons, benzene, polychlorinated biphenyls, organochlorine compounds (including vinyl chloride), nickel, chromium, toluene
Reproductive effects	arsenic, benzene, cadmium, chlorinated compounds, lead, mercury, polycyclic aromatic hydrocarbons, polychlorinated biphenyls
Cancer	polycyclic aromatic hydrocarbons, arsenic, nickel, chromium, vinyl chloride, benzene

Source: DEFRA (2004a, p. 18)

The majority of these health effects are characterised by a threshold of effects i.e. a level of exposure below which no adverse health effects would be expected. The severity of any effect is likely to increase as the dose increases. However, in the case of some carcinogens, the possibility of a threshold of no-effect is less. In this case, the severity of the effect is not related to the dose, although the likelihood of the effect occurring is.

These health effects would only be expected to arise if exposure was sufficient. The scientific evidence suggests that exposure at sufficient levels does not normally arise from the existence of a landfill. Saffron *et al.* (2003) conducted a major review of the literature on the health effects of waste facilities and concluded that the evidence for adverse health outcomes was 'insufficient'. This conclusion applies to incinerators and composting facilities, as well as landfills. A similar conclusion, although not from this particular study, can also be reached for MBT (see the section on MBT for more detail on this).

In the economic valuation of air pollutants used in this report, however, the impact of such pollutants on human health is included in the modelling of their impact pathway.[103] This is the norm in the literature that evaluates the externalities of emissions to air. See the general section on air pollution externalities for greater detail on this. Although the health evidence is inconclusive, the point mentioned above on the *threshold of effects* coupled with the actual (relatively low) level of these pollutants emitted from landfill means that including health impacts in the calculation of these pollutants' impact pathways should not unduly bias the monetary values we will use.

[103] See definition of impact pathway in Table A.6 above.

Disamenities

Disamenities in a waste management context can be defined as the localised physical and psychological impacts of waste management activities. They include visual intrusion, noise, dust, odour, wind-blown litter, vermin and the perception of increased health risks. For the purposes of this analysis, the noise and congestion impact of waste trucks accessing the waste facilities is excluded. Landfill disamenities are, by and large, considered to be fixed externalities (Cambridge Econometrics, 2003). This means that the externality exists because the waste facility exists; it is not dependent on the levels of waste that the facility manages (if this were so, they would, like air emissions, be characterised as variable externalities). This is worth consideration as the other externalities of waste can be expected to decline in the future as greater emphasis is placed on reducing waste and/or waste technology improves and thus emissions fall.

In the disamenity literature, Rabl *et al.* (2008) distinguishes between the costing of disamenities and other externalities, such as greenhouse gas emissions. Amenity impacts are limited to the population in the immediate vicinity of the landfill (or any other waste facility) and are extremely variable across sites. According to Rabl *et al.* (2008) the appropriate internalisation of disamenity externalities is by negotiation with the local population before the project is approved. In the absence of such a Coasian solution, we must resort to putting a price on landfill's disamenities.

One of the most common ways of doing this is by studying house price changes in the areas surrounding waste sites: this is known as the hedonic pricing method. Its underlying assumption is that the price of a good is a function of its underlying attributes, including environmental ones. The distance from a site such as a landfill is considered as a proxy for environmental exposure, and, controlling for socio-economic and physical characteristics, it is expected that there is an inverse linear relationship between distance and the change in house price (Eshet *et al*., 2005). One important assumption involved in using this technique is that the variation in house prices is caused solely by the existence of disamenities and not by other types of externality such as, for example, air pollution.

The research in disamenity externalities is gradually improving (Bartelings *et al*. 2005, Cambridge Econometrics, 2003). However, whilst secondary literature flourishes there have been few major original studies since 2003, and none has ever been carried out in an Irish context. All major studies on waste externalities highlight the danger in transferring disamenity values across studies (Cambridge Econometrics 2003, Eunomia 2007, COWI 2000). But observed local resistance to landfill and the evidence of hedonic pricing studies suggest that its disamenities are significant. Therefore, with due caution and by highlighting all serious reservations, this report will attempt to place a value on the disamenity impact of landfill in Ireland.

The most recent and adaptable study in this area was conducted by Cambridge Econometrics (2003). The study analysed 11,300 landfill sites in Britain (of which, 6,100 were operational) and 592,000 mortgage transactions (containing information on house prices, housing characteristics and location) during the period 1991-2000. Regressions were performed on a county-by-county basis to help separate different property markets across Britain. The study found firm evidence of a statistically significant fixed disamenity impact at the aggregate British level, though this is statistically significant only within a half-mile of a landfill site. However, there is significant variation across individual regions, for example Scotland saw house price reductions of 40% within a 0.25 mile radius of the landfill, compared to a British average of 7%.[104]

A "life cycle" effect also exists; disamenity costs tend to be highest around the opening of new sites and level off after some time when local residents adjust to the presence of the landfill. For properties at distances up to or equal to 1 mile from a landfill site in England and Wales, house prices were about 10% lower for sites that have opened within the last ten years relative to sites that have been operational for 20 – 30 years. However, this result was a separate piece of analysis and not incorporated into the final result (here, taken to mean the **7% reduction in house prices within 0.25 miles of a landfill and the 2% reduction in house prices within 0.25-0.5 miles**).

Another recent study, Bartelings *et al.* (2005), used benefit transfer from this Cambridge Econometrics study in order to achieve appropriate Dutch values.[105] They took the headline result from the UK - £2.50 to £3.59 per tonne of waste (in 2003£) – and adjusted it for currency, income and housing density differences. See Bartelings *et al.* (2005, p. 91) for details. The figures thus adjusted, they found that the disamenity costs per tonne of landfilled waste in the Netherlands might be 39 percent to 50 percent larger than in the UK. Considering the up-to-date and comprehensive nature of the 2003 study, transferring the UK values is recommended in this report too, although our methodology will be different to the work of Bartelings *et al.* (2005).

However, before this is done, the results of another analysis, Walton *et al.* (2006), should be remembered. This study performed a meta-analysis of six hedonic pricing studies in the US. A preliminary benefits transfer exercise was conducted using the meta-function developed and applied to a site in the North East of England (Garrod and Willis, 1998). It was found that the transfer value underestimated the value estimated in this 1998 stated preference study for the same site. This point - that transferring values does not produce robust results - should be noted.

In transferring the Cambridge Econometrics results, the following methodology was applied:

1. Current average house prices (from the Permanent TSB/ESRI House Price Index) were disaggregated into county averages using the most recent data on relativities (for 2006). We then calculated the effects of a 7% and 2% reduction in prices for each county.
2. Using the An Post Geodirectory and GIS software,[106] we identified the number of active residential addresses located within specified distances of the boundaries of

[104] The authors explain the very high Scottish figure by acknowledging the high incidence of landfills near urban agglomerations. In other words, landfills in Scotland are located in high household density areas. The Scottish housing market is also more stable than other regions – so the disamenity impact may be more readily observable. If this is to be taken as given, the average price reduction for Britain which we shall use *underestimates* the disamenity effect. However, the Scottish figures were not deemed usable in an Irish context as the Irish frequency distribution of houses around landfills is more similar to the British than Scottish equivalent.

[105] Benefit transfer is a common procedure where data from previous studies (focusing on different regions and times) is used, but adjusted for, *inter alia*, currency and income differences.

[106] The Geodirectory version was July 2008, and ArcGIS v.9.3.1 was used.

existing landfills.[107] We multiplied the number of houses by the average estimated reductions calculated in the first step.

3. A total fixed disamenity value was thus estimated to be €215,000,000.
4. Adjusting for waste landfilled and the rental yield on the fixed (or capital) value, this produced an annualised disamenity value of €10.64-21.29 per tonne. The details are shown in Table A.9.

Table A.9: Disamenity Due to Landfill, per tonne, Ireland, 2009

Distance from landfill	**% reduction in price**	**Number of Houses**	**Fixed Disamenity Value (2009€)**
0-0.25 miles	-7.06	6,466	120,000,000
0.25-0.5 miles	-2.00	17,984	95,000,000
Total		24,450	215,000,000

	Disamenity (2009€) / tonne landfilled [108]
Per tonne total disamenity	106.00
Annual value, 10% assumed rental yield	10.64
Annual value, 5% assumed rental yield	21.29

Source: see text.

A brief note on comparing the disamenity values of waste processes

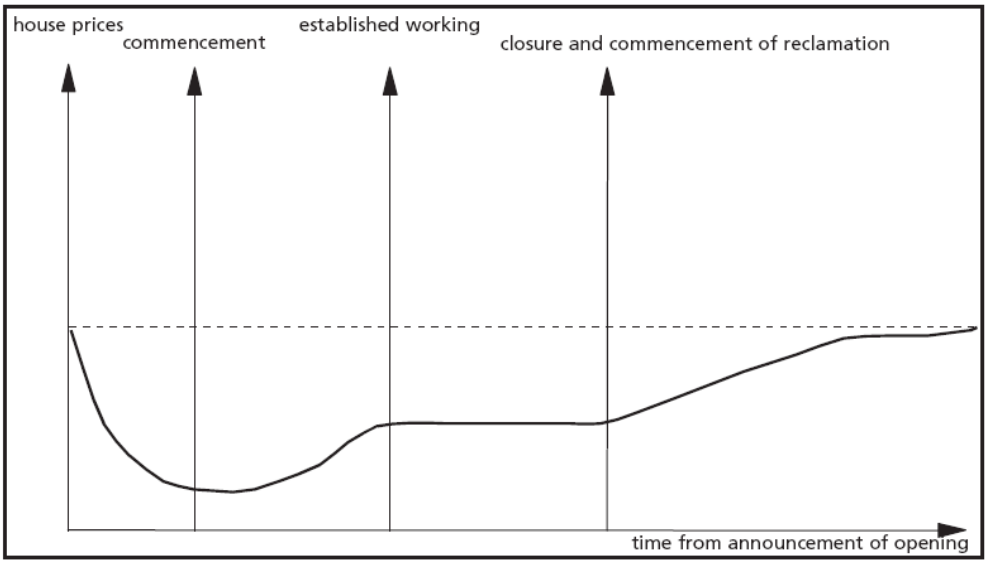
To enable the comparison between landfilling, incineration and MBT, external costs are calculated on a per unit basis i.e. per tonne of waste managed in each facility. This step in the analysis is not obvious for the disamenity effect as, in reality, the disamenity is not primarily determined by the amount of waste processed by the landfill, incinerator or MBT plant, but by the sheer existence of the site i.e. it is, as mentioned previously, a fixed externality. To facilitate comparison, however, the disamenity value is assumed to be proportional to the total amount of waste processed (Bartelings et *al.,* 2005). This is not necessarily a realistic assumption, but is a requirement for further comparative analysis.

Studies of incinerators and landfills have both come to the same conclusion on life cycle effects. Essentially, their impact (on house prices) does not remain constant over their lifetime; instead, the relative reduction in house prices is observed to decline with the age of the landfill site (Cambridge Econometrics, 2003) or the incinerator (Kiel and McClain, 1995). It can be assumed that the life cycle effect also applies to MBT.

[107] We omitted residential addresses classified as vacant.

[108] Taken from EPA (2009a) as 2,014,797 tonnes of municipal waste landfilled in 2007.

Figure A.1: The Life Cycle Effect



Source: Cambridge Econometrics (2003)

A summary of the external costs of landfill

The result of the above analysis in terms of the appropriate levy for refuse sent to landfilled based on the methodology outlined in Section 1.2 above is presented in Table A.10 below. This shows that the external cost of methane per tonne of waste is far more important that the disamenity effects. Bearing in mind that the cost of methane per tonne of waste is a variable externality and the disamenity effect is considered a fixed externality, there is scope for policymakers to affect this comparison as management of landfills and MSW change – and thus methane emission levels change - into the future.

Table A.10: Pricing the External Costs of Landfill, Per tonne, Ireland, 2009

Externality		**Price (2009€/tonne waste landfilled)**
Greenhouse Gases	Carbon Dioxide	(omitted since biodegradable waste removes CO_2 from the atmosphere)
	Methane	€33.60
Leachate		0
Air Pollutants		Omitted, regulated through emission limits in waste licence
Disamenities		€10.64 – 21.29
Total		€44.24 - 54.89

Source: see above

1.4 The externalities of incineration

Introduction and Irish Context

Ireland can expect to see incinerators operating in a waste management context starting in 2011 (at Carranstown, Co. Meath). Incineration's main objective is to reduce the various environmental and health concerns related to biodegradable waste and to minimize the volume of waste to be disposed of in landfills (Calabro, 2009). The main externalities associated with incineration are well-known. They include emissions to air, water and soil, as well as the energy recovered during combustion. With regard to negative externalities, the main concern is over the by-products of the combustion process, particularly the emissions to air.

In the literature, the crucial factor, which can tip the balance in favour of incineration as a favoured waste management option over landfill, is energy recovery. Energy recovery is taken to mean the energy that can be created by utilising the heat from incinerating waste; the savings from energy recovery include the avoided emissions that would have been produced had this energy come from another source such as, for example, a coal plant. Recent comparative studies of the external costs of incineration and landfilling have demonstrated that the choice between both options hinges on their energy recovery capacity e.g. Dijkgraaf and Vollebergh (2004), Rabl *et al*., (2008). In general, the net external costs (total external costs less the savings from energy recovery) have been declining in magnitude as incineration technology improves and greater appreciation and technical knowledge of their energy recovery develops. However, **this report does not consider the avoided emissions from other sources due to incineration's energy recovery to be an externality at all.** This process, as mentioned previously in the report, has already been internalised by the Emissions Trading Scheme and does not need to be calculated here.

Emissions to air can be controlled using various treatment processes that remove particulates and gases before the remainder of the incinerator's flue gas is emitted to the air via the smokestack. Flue gas cleaning processes produce residues that are considered hazardous and need to be treated prior to disposal. The incineration process also generates bottom ash requiring disposal to landfill and/or use e.g. as road construction material. Contaminants in the residual solid waste disposed to landfill can possibly be leached and lead to emissions to soil and water.

The difficulty for the present report lies in transferring values for incineration's external costs that have been calculated in different regions and at different times. Not all will be easily applicable to an Irish context. Furthermore, the levels of emissions to air per tonne of waste incinerated that we choose will have to be hypothesised as no municipal waste incinerators are operating yet in Ireland. As no figures exist yet, we shall use the Directive on Incineration limits as the emission levels to which to apply the prices of both greenhouse gases and air pollutants.

Greenhouse Gases

As with landfilling, the value put on the external costs of greenhouse gases will be determined by the current market price where available, and an assumed medium term average thereafter. The same difficulties associated with valuing the cost of carbon dioxide (and methane), as outlined in the externalities of landfilling section, apply here.

For illustrative purposes, however, values from DEFRA (2004a) are also shown in order to highlight the difference that various types of incineration can make to greenhouse gas emissions. The results in the first column of Table A.11 are based on UK operational incinerators. The data comes from the UK Environmental Agency, and it must be noted that a degree of uncertainty accompanies them.

Table A.11: Greenhouse gas emissions, UK incinerators & Waste Incinerator Directive Limits

	Emission Rates to air using operational UK incinerators[109]	**Emission Rates to air assuming incinerators operate at Waste Incineration Directive limits**[110]
Methane (g/tonne MSW)	19	0.6
Carbon Dioxide (g/tonne MSW)	100, 000	No limit

Source: DEFRA (2004a, Table 2.20, p. 68; Table 2.2, p. 69).

Assuming that the UK figures in Table A.11 are a reasonable proxy for emissions from an Irish incinerator, we can calculate the externality effect of GHGs from incineration. We use the same CO_2 price assumption as for given earlier for landfill (a current price of €13.40 per tonne of CO_2, and maintain the assumed methane GWP of 21. This implies €0.0053 per tonne of MSW externalities associated with methane emissions and €1.34 per tonne of MSW externalities associated with CO_2 emissions. The former figure is sufficiently small to ignore in our analysis. CO_2 emissions are covered under the ETS already, and in any event we presume that the component arising from combustion of renewable feedstocks (e.g. organic waste, paper, wood and some textiles) should be treated as carbon-neutral. We therefore omit CO_2 externalities from our final estimates as well.

The type of waste, as well as the type of incinerator, will also influence greenhouse gas emissions; for example, the composition of the waste is more important than the treatment process with respect to carbon dioxide emissions (COWI, 2000).

Air Pollutants

Impacts from air pollutants include adverse health effects from particulates, dioxins, heavy metals, VOCs, NO_x, CO and SO_2. Effects on the ecosystem and fauna arise from the same pollutants. Lower agricultural yield, forest and building damage can occur from emissions of acid gases and NO_x, with particulates also causing damage to buildings.

The magnitude of these impacts will heavily depend on whether the waste sent to incineration has been pre-sorted. In such an instance, the damage caused will be much lower (DEFRA 2004a, p. 220). The most effected method of presorted recyclables is by source separations of different materials into dry and wet fractions. Pre-sorting waste maximises the quantity and quality of recyclable materials and removes some of the more dangerous dioxin precursors and metals from the waste entering the incinerator. Stack

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109 The data comes from eleven incinerators using energy recovery. The level of energy recovery is not provided.

110 The Waste Incineration Directive (2000/76/EC) was introduced in 2000 in order to prevent or reduce, as far as possible, air, water and soil pollution caused by the incineration or co-incineration of waste, as well as the resulting risk to human health.

height also has a substantial impact on the local concentration of air pollutants. Moreover, the lower the temperature of the flue gases, the more concentrated the pollutants are in the local range (Dorland *et al.,* 1997).

Contaminants typically found in the flue gas residue include particulates, dioxins, heavy metals and their compounds, acid gases (SO_2, HCl, HF), NO_x, CO_2, and volatile organic compounds (VOCs). These contaminants are emitted into the atmosphere via the smokestack, although their concentrations can be reduced using flue gas treatment processes.

Again, using DEFRA (2004a) to illustrate possible levels of air pollutants from incineration, it can be seen that nitrogen oxides are the main contributor to air pollution in an incineration context (Table A.12).

Table A.12: Air pollutant levels per tonne of incinerated waste, UK incinerators & Waste Incinerator Directive Limits

(g/tonne MSW incinerated)	**Emission Rates to air using operational UK incinerators[111]**	**Best estimate of emission rates to air assuming incinerators operate at Waste Incineration Directive limits[112]**
Nitrogen Oxides	1,600	1,100
Total Particulate Matter	38	60
Sulphur Oxides	42	280
Total VOCs	8	60
Cadmium	0.005	0.3
Nickel	0.05	0.6
Arsenic	0.005	0.6
Mercury	0.05	0.6
Dioxins[113]	400 ng TEQ/T	560 ng TEQ/T

Source: DEFRA (2004a, Table 2.20, p. 68; Table 2.2, p. 69*)*

Representative data on air emissions from incinerators is difficult to obtain. If there is any doubt, it will be assumed that they are equal to the limit values of the Directive on Waste Incineration (EC/2000/76). In reality, emissions may be lower than these limits (cf.Table A.12).

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[111] The data comes from eleven incinerators using energy recovery. The level of energy recovery is not provided.

[112] The Waste Incineration Directive (2000/76/EC) was introduced in 2000 in order to prevent or reduce, as far as possible, air, water and soil pollution caused by the incineration or co-incineration of waste, as well as the resulting risk to human health.

[113] Dioxins are produced by incomplete combustion. Dixons are resistant to biodegradation and accumulate in food (the main exposure pathway is by ingestion, not inhalation). The toxicity of each dioxin is indicated by its Toxic Equivalent Factor (TEF). 1 is the most toxic (assigned to the dioxin TCDD). On the assumption that the effect of various dioxins is additive, the TEF value for each is obtained and multiplied by its concentration. Thus a final Toxic Equivalent Quantity (TEQ) is obtained (Guisti, 2009).

When modelling the emissions (g/tonne waste), and if using Waste Directive Limits, Rabl *et al.* (2008) gives more accurate values than DEFRA – reproduced below in Table A.13.

Table A.13: Assumptions for incineration of waste, based on Directive on Waste Incineration

Pollutant	Mg / Nm^3 [114]	g/tonne waste
PM	10	51.5
SO2	50	258
NO2	200	1030
CO2		861800
As	0.014	0.072
Cd	0.0406	0.21
Cr (VI)	0.00065	0.0033
Hg	0.05	0.26
Ni	0.169	0.87
Pb	0.11	0.57
Dioxins	1.00E-07	5.15E-07

Source: Rabl et al. (2008, Table 4, p. 155).

[114] Limit values of the flue gas concentrations in the Waste Incineration Directive, assuming 5,150 Nm3 per tonne waste.

Table A.14: External costs of air pollutants from incineration

Air Pollutant	€/tonne emission (2000 €)	€/gram emission (2000 €)	g/tonne waste[115]	€/tonne waste (2000 €)
Arsenic	80,000	0.08	0.072	0.00576
Dioxin	37,000,000,000	37000	5.15E-07	0.019055
Lead	600,000	0.6	0.57	0.342
Mercury	6,000,000	6	0.26	1.56
Cadmium	21,000	0.021	0.21	0.00441
Chromium IV	130,000	0.13	0.0033	0.000429
Nickel	2,100	0.0021	0.87	0.001827
Nitrogen Oxides (NOx)	5,600	0.0056	1100[116]	6.16
Particulate Matter (PM2.5)	22,000	0.022	51.5	1.133
Sulphur Dioxide (SO2)	7,500	0.0075	258	1.935
Volatile Organic Compounds (VOCs)	950	0.00095	60[117]	0.057
Total				**11.16**

Source: see text

Disamenities

There are several options for estimating a disamenity value for incineration.

A study on the potential impact of the Poolbeg incinerator on local house prices was conducted in 2005 by the real estate adviser CB Richard Ellis Gunne as part of the Environmental Impact Statement ("EIS").[118] It found that the Poolbeg plan had no impact on residential property prices in the surrounding area and that, in fact, prices in these areas rose faster in 2002-05 than the average Dublin price. The study did not control for a number of factors that are likely to influence house prices such as location of the property, its physical characteristics and the socio-economic characteristics of the area. CB Richard Ellis Gunne prepared an index of local house prices but did not conduct a proper hedonic pricing study.[119] In the absence of a more rigorous attempt at isolating the

[115] Based on Waste Incineration Directive (calculations in Rabl *et al.*, 2008).

[116] Level taken from DEFRA (2004a)

[117] Level taken from DEFRA (2004a)

[118] For details see Appendix 17.1 of the EIS (Elsam, 2009). The EIS may be accessed at: http://www.dublinwastetoenergy.ie/uploads/archive/files/technical-summary.pdf. Accessed 30/10/09.

[119] A brief discussion of hedonic pricing is found in the landfill section above.

disamenity impact of incineration, their result of 'no impact' does not seem robust, particularly in light of the experience of Cambridge Econometrics (2003) who found that in volatile property markets such as the south of England, the disamenity impact was less observable than in more stable property markets such as in Scotland (see footnote 18). The Dublin property market of the early 2000s certainly qualifies as volatile.

Another option is to adapt the hedonic price function of Kiel and McClain (1995) – which is based on a US incinerator operating in the 1970-90s. Kiel and McClain (1995) is one of the few hedonic price studies of incineration available in the literature. It examined the impact of an incinerator in North Andover, Massachusetts, over the course of its lifetime. By looking at a dataset spanning 19 years and 2,593 house sales, they concluded that the impact of an undesirable land use on house prices is not constant over time. The impact of the incinerator studied could be divided into five stages: pre-rumour; rumour; construction; facility comes online; ongoing operations stage. Their study suggests that house prices increase by 1.7 – 2.3% per mile away from the incinerator (and only in the final three stages). Because of the nature of the regression they performed (double log regression), it was impossible to isolate a point when disamenity impact stops, but during the last three phases - the only stages where they found price changes to be statistically significant - the maximum distance effect is reached at roughly 3.5 miles (5.6km).

The Dutch study, Bartelings *et al.* (2005), used Kiel and McClain. Adjusted into Euros, and using some very specific geographic information on housing levels to make their figures more realistic, they found that starting at 5.5 km from the site, the house price drops by approximately €9,500 (2005 prices) with every kilometre approaching the incinerator. DEFRA (2004b) (the major study of the externalities of landfilling and incineration) also interpret Kiel and McClain. By adjusting for inflation and currency they calculate a value of £21/tonne of waste incinerated (2003£). However, they do not recommend using this value as Kiel and McClain's data is too old and too remote (from the US). The Kiel and McClain (1995) work should rightly be considered out-of-date and applying it to Ireland will not yield plausible results.

However, a recent study of disamenities associated with French incinerators was carried out by the contingent valuation method i.e. by surveying people's willingness to pay (WTP) to avoid the disamenity (Arnold and Terra, 2006). In a review of the literature on disamenities, Eshet *et al.* (2005) concludes that hedonic pricing methods are preferred to contingent valuation methods because they are based on observable market prices (in the housing market). But Walton (2006) found – and Eshet *et al.* (2005) acknowledges – that the HP results and CV results are generally consistent. Thus, this French study can be considered as a basis for finding disamenity values in an Irish context.

The authors of this study created surveys based on three different scenarios for local residents living near a modern incinerator. They concluded that the best range to use for WTP was €40-54 per tonne per household where the WTP was per household per year over ten years and referred to WTP to close the incinerator (the sample number equalled 465 people who lived within 2km of the incinerator). They multiplied this by the number of households in the affected zone to get a total cost for the disamenity (in their case €311,000-420,000 per year). This was divided by the amount of waste the incinerator dealt with (85, 000 tonnes) to get a final range of €3.7-4.9/tonne waste incinerated.

Considering the localised or dated character of the other options discussed above, we employ the results of the French study. To apply this result to Irish conditions, we use the range **€40-54 per household per year** and adjust for Irish house numbers and proposed incinerator capacity. The following methodology was undertaken:

1. The housing densities in the 2km radius around both the Carranstown and Poolbeg incinerator sites were found using the ArcGIS model, described above.
2. These results were applied to the €40-54 per household per year figure.
3. These were then adjusted for purchasing power parity and inflation in order to establish a 2009 Irish € equivalent range.
4. The original survey posed the question as payment per year over ten years. Therefore, the results from step 3 were discounted over ten years using both a 5% and 10% discount rate. Although the public sector discount rate in Ireland is currently 3.5%, higher rates were used as the household discount rate can be expected to be much higher than the social discount rate.

The results are presented in Table A.15 below. The disamenity level cost is much lower than that for landfill.

Table A.15: Disamenity Due to Incineration, per tonne, Ireland, 2009

	Low Range		High Range		Average	
Discount Rate	10%	5%	10%	5%	10%	5%
Total Fixed Disamenity (2009€)	2,230,000	2,670,000	3,010,000	3,610,000	2,620,000	3,140,000
Disamenity/tonne incinerated [120]	2.78	3.34	3.76	4.51	**3.27**	**3.92**

Source: see text.

Considering the importance of housing densities in calculating the disamenity effect, the average totals in Table A.15 were re-calculated on a facility-by-facility basis. As can be seen in Table A.16, the urban incinerator (Poolbeg) is far more costly than the rural one (Carranstown), but in both cases the disamenity effects are low compared to those of landfills.

Table A.16: Disamenity per incineration facility, Poolbeg & Carranstown, Ireland, 2009

	Poolbeg Average		Carranstown Average	
Discount Rate	10%	5%	10%	5%
Total Fixed Disamenity (2009€)	2,530,000	3,040,000	80,000	100,000
Disamenity/tonne incinerated [121]	**4.22**	**5.07**	**0.42**	**0.50**

Source: see text.

[120] We divided the total fixed disamenity by 800, 000, which is the combined capacity of the Poolbeg and Carranstown incinerators.

[121] We divided the total fixed disamenity by 600,000 for Poolbeg and 200,000 for Carranstown.

9.1.1 Residual Solid Waste

Residual solid waste from incinerators includes bottom ash and the more toxic residues, such as fly ash, from flue gas treatment. The flue gas treatment process within a plant will influence the toxicity of residual solid waste. Flue gas residues are problematic due to the high concentration of heavy metals and are therefore disposed of in controlled landfill sites designed for hazardous waste. At first glance, the external costs associated with disposing of toxic waste to a hazardous waste landfill may appear to be high, as the waste in question would pose significant risks to health, agriculture and the environment. But if it is properly stored, any externality associated with the hazardous content of the waste is entirely eradicated. It can also be argued that the other possible source of external cost, the risk of an accident occurring, will be internalised into the overall treatment costs of such a landfill. Therefore, the total external cost of chemical waste in a properly-run hazardous waste landfill will be very low (Bartelings *et al.*, 2005).

However, in constructing all our external costs, the focus has been on municipal landfills: consequently the discussion below is based solely on the costs of disposing of bottom ash (which can be disposed in municipal landfills).

The bottom ash produced from the combustion process comprises around 20-30% of the mass of the waste treated at an incinerator (Bartelings *et al.*, 2005); this is consistent with the projections of bottom ash for the Poolbeg Incinerator (Waste Licence Application - Dublin WtE 060706). According to the waste application licence for Poolbeg, 120, 000 tonnes of bottom ash are expected to be produced per year, if operating at full capacity. The licence issued to Poolbeg gives it the option of disposing of this ash to landfill if it wishes, but the Dublin Waste Management Plan 2005-2010 stipulates that the ash will be recycled. Fly ash from Poolbeg, however, will be exported for disposal. The waste licence for Carranstown allows it to dispose of the bottom ash offsite, but it is unclear if this means disposal in an Irish landfill or its exportation abroad.

Based on the uncertainty associated with the destination of their residual solid waste for both Poolbeg and Carranstown, we highlight a means of valuing this specific externality, i.e. valuing the landfilling of bottom ash, but do not include it in our final analysis, as, at the time of writing, it may or may not emerge as an external cost for Irish incinerators.[122]

Establishing a cost for residual solid waste is based on work done in the Dutch study Bartelings *et al*. (2005). In a best-case scenario, they valued bottom ash at zero, due to its generally non-hazardous nature. The high cost scenario for this ash was valued at the external costs of landfilling in the Netherlands, exclusive of the costs of greenhouse gas emissions (because bottom ashes do not contain organic materials).

Based on this methodology, and referring to Table A.10 in Section 1.3, the cost of landfilling bottom ash would be solely **based on the disamenity cost of landfill**.

A summary of the external costs of incineration

The results of the above analysis in terms of the appropriate levy for material sent to incineration applying the methodology set out in Section 1.2 is presented in Table A.17. As methane emissions from landfill are far higher than methane emissions from incineration, methane here forms no part of the levy, and our total levy for incineration is much less than the landfill equivalent. Indeed, since most emissions from an incinerator

[122] Recently there has been opposition to the export of incinerator ash. See Mc Donald and Gartland (2009).

are gases normally controlled through licence provisions, we suggest that the levy for incineration per se should be based solely on its disamenity impact. This figure is also far smaller than the comparable landfill figures (€10.64 - €21.29).

Table A.17: Pricing the External Costs of Incineration, Per tonne, Ireland, 2009

Externality		Price (2009€/tonne waste incinerated)
Greenhouse Gases	Carbon Dioxide	Omitted, regulated through ETS
	Methane	Omitted due to small scale
Air Pollutants		Omitted, regulated through emission limits in waste licence
Solid Residue		Omitted
Disamenities	Urban	4.22 – 5.07
	Rural	0.42 – 0.50
Total	Urban	4.22 – 5.07
	Rural	0.42 – 0.50

Source: see text

1.5 The externalities of Mechanical Biological Treatment (MBT)

Introduction and Irish context

The establishment of correct costing for the externalities of MBT is particularly difficult. The term MBT encompasses a wide range of technologies, making the use of a standard definition difficult. There is also limited information on the costing of MBT in different countries. Quantitative results are often based on assumptions or omissions which makes a true comparison with landfill and incineration practices difficult. The lack of MBT facilities in Ireland also implies that there are no location-specific data. Therefore, quantification of external costs remains largely incomplete and as such, any costing data must be viewed against a background of uncertainty. This section presents more details on the literature than both landfill and incineration due to this uncertainty factor. Throughout this discussion, MBT is defined as "the treatment of residual municipal waste through a combination of manual & mechanical processing and biological stabilisation, in order to stabilise and reduce the volume of waste which requires disposal".[123]

It is worth noting that when discussing MBT, it is useful to be as specific as possible i.e. state whether the MBT process is operating in stabilisation mode or in a mode producing fuel such as RDF/SRF (this latter process is known as biodrying). The outputs from both processes obviously differ physically – the former leads to stabilised biowaste, the latter

[123] Taken *verbatim* from "Municipal Solid Waste – Pre-treatment & Residuals Management: An EPA Technical Guidance Document" (EPA, 2009b).

to fuel - and also differ legally. Only the biostabilised waste will help in meeting targets for diversion of biodegradable waste from landfill. If the output prepared as RDF/SRF were landfilled, it would not be exempt, as it is not considered to be biostabilised, and would indeed produce significant greenhouse gas emissions. The form of MBT in operation also creates different life cycles for the waste in question. After stabilisation, the waste is usually landfilled. After the production of fuel, the next stage is usually co-incineration in a cement kiln, along with the landfilling of the residual left over from producing the fuel.[124] We note that co-incineration of waste in a cement kilns remains 'waste incineration' under the terms of the EU Waste Incineration Directive.

As mentioned, little work has been done on establishing the key externalities likely to be associated with the formation of an MBT facility. However, its externalities are presented in Table A.18 below.

Table A.18: Key externalities associated with MBT

Activity	Noise	Odour	Dust	Flora/Fauna	Soils	Water quality	Air quality	Climate	Building Damage
MBT	**	***	**	–	–	**	**	*	*

–	**No effect**
*	**Unlikely to be significant**
**	**Potentially significant in some cases, but impact controllable**
***	**Issue at sites falling below best practice, but impact normally controllable**
****	**Significant issue at all sites**

Source: DEFRA (2004a, Table 5.1, pp. 219-221)

Greenhouse gases and air pollutants

Air emissions are the externalities to which costs can be readily associated, although, unfortunately, not in a treatment-specific manner. Table A.19 provides estimates of the GHG and air pollutant emissions from the MBT process.

[124] For a general European context: waste treated via MBT is sent to landfill in Austria, to incineration in Germany and used as (low grade) compost in France, Italy and Spain (COWI, 2004).

Table A.19: Estimated emissions to air from mechanical biological treatment (grams per tonne of MSW)

Substance	**Grams/T**
Nitrogen Oxides	72.3
Total Particulates[125]	258
Sulphur Dioxide	28
Hydrogen Chloride	1.2
Hydrogen Fluoride	0.4
Volatile Organic Compounds	36
Methane	411
Dioxins & Furans	4.0×10^{-8}
Polychlorinated Biphenyls	No data
Carbon Dioxide	181,000
Carbon Monoxide	72.3
Ammonia	~120

Source: DEFRA 2004a (Table 2.9, p. 48) based on Draft Reference Document on Best Available Techniques for the Waste Treatment Industries, European Commission 2003.

DEFRA (2004a) suggests that air quality impacts are likely to be closest to those seen in composting operations, with the primary emissions coming from bioaerosols and VOCs resulting from the biological treatment of the waste. Cadena *et al.* (2009) examines gaseous emissions in a composting plant and found that 206g of VOC were emitted per tonne of MSW and 3900g of ammonia per tonne. These figures differ quite significantly from the levels of these emissions provided in the above table. **However, in the absence of knowledge on which MBT process will be most prevalent in Ireland (**i.e. in being unable to predict how big the "B" will be) **the levels provided in the above table will be used in our analysis**.[126] Nonetheless it should be borne in mind that if the MBT process in question is biostabilisation of the waste, then the Cadena *et al.* (2009) emission levels may be more realistic, due to the similarities between the biostabilisation process and composting.

Assuming that the UK figures in Table A.19 are a reasonable proxy for emissions from an Irish MBT plant, we can calculate the externality effect of GHGs from MBT. We use the same CO_2 price assumption as for given earlier for landfill (a current price of €13.40 per tonne of CO_2, and maintain the assumed methane GWP of 21. This implies €0.116 per tonne of MSW externalities associated with methane emissions and €2.43 per tonne of MSW externalities associated with CO_2 emissions. As for incineration, the former figure is sufficiently small to ignore in our analysis. As discussed in the case of landfill, CO_2

[125] No data available. But, according to the EPA, the emissions limit value for particulate matter for biodegradable waste composting is 50mg/m3. Assuming a value of 5,150 Nm3 per tonne waste, as per the assumption used for the Waste Incineration Directive, 258g of particulate matter per tonne of waste can be calculated. As it is based on emission limit values, it should be noted this figure is a high estimate.

[126] In the Eunomia (2008) report on MBT, they profiled five Irish facilities that could be classified as MBT. All five used mechanical processes (the 'M'), two also used biological processes (the 'B') and only one produced RDF.

emissions from processing of organic materials is broadly carbon neutral and thus does not give rise to significant externalities. Emissions from fossil carbon may be covered under the ETS if an MBT plant is sufficiently large or will be covered by the planned carbon tax to the extent that they arise from burning fuels covered by the tax. If there are any remaining significant externalities from fossil CO_2, they could be included in the levy. However, it does not seem likely that such emissions will be significant.

Another study, McLanaghan (2002) argues that the emissions from MBT are largely concentrated on carbon dioxide and water, and thus will have only a minimal impact on air quality. Another study, conducted by Lahl *et al.* (1998) in Austria, looked at examining the VOCs which could be emitted from an MBT facility: the levels are provided below. They argue that these levels of VOCs would indicate that the presence of an MBT facility does not increase these emissions, as the levels are similar to what would be found in suburban/urban areas.

MBT residue

As mentioned previously, difficulty arises from the fact that two very different choices are possible for dealing with the output from MBT, and that emissions from these processes (landfill or incineration/co-incineration) can vary. Smith *et al.* (2001) carried out a report for the European Commission detailing much of the information about incineration, landfill and MBT. The report admits the difficulties of acquiring the necessary data due to the concentration of MBT facilities largely in Austria and Germany.[127] Smith *et al.* (2001) model for MBT emissions assumed that after separation of metals, the waste was separated into a compostable fraction and a reject fraction. The compostable fraction was disposed to landfill and the reject fraction (the residue) was either diverted to landfill or incineration with energy recovered as electricity. The proportions of the reject fraction (residue) diverted are:

- Paper 20%
- Plastics 100%
- Textiles 50%
- Miscellaneous combustibles 50%
- Miscellaneous non-combustibles 100%

Table A.20 and Table A.21 below, taken from this report, show the greenhouse gas fluxes for the processing of the residue from the MBT process in landfill or incineration. Three different cases are analysed, which make different assumptions about the behaviour of the compostable fraction in the landfill.

[127] MBT was in fact first developed in Germany in order to biostabilise MSW to the degree that it would meet the requirements of the Landfill Directive.

Table A.20: MBT with landfill of rejects and recycling of metals (kgCO2eq/tonne MSW)

	Case 1	Case 2	Case 3	Mean 1 & 2
N20	0	0	0	**0**
CH4	97	171	97	**134**
Carbon sequestered	-364	-364	-99	**-364**
Transport CO2	5	5	5	**5**
Avoided energy & materials	-162	-162	-162	**-162**
Energy use CO2	22	22	22	**22**
Process CO2	0	0	0	**0**
Net flux	-402	-328	-138	**-365**

Source: Smith et al., (2001, Figure 14, p.35).

Table A.21: MBT with incineration of rejects and recycling of metals (kgCO2eq/tonne MSW)

	Case 1	Case 2	Case 3	Mean 1 & 2
N20	3	3	3	**3**
CH4	0	74	0	**37**
Carbon sequestered	-289	-289	-23	**-289**
Transport CO2	5	5	5	**5**
Avoided energy & materials[128]	-241	-241	-241	**-241**
Energy use CO2	22	22	22	**22**
Process CO2[129]	205	205	205	**205**
Net flux	-295	-221	-29	**-258**

Source: Smith et al, (2001, Figure 14, p.35), adjusted to exclude avoided energy (as not treated as externality in this report).

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[128] Due to the recycling of metals.

[129] These high figures are mainly due to plastics incineration.

Case 1. The highly stabilised compost is landfilled and the small amount of methane formed is oxidised before it escapes.
Case 2. Less completely stabilised compost is landfilled and 25% of the methane formed escapes, the rest is oxidised.
Case 3. The highly stabilised compost is used as a surface dressing for landfill site restoration and decays aerobically. Because further decomposition occurs under the aerobic conditions under which the compost is used, sequestration of carbon is limited to the much lower rate used for compost applied to agricultural land.
*Case 1 and 2 represent most likely fate of the compost in landfill.

Smith *et al.* (2001) states that MBT shows a lower greenhouse gas flux than the EU-average standard of landfilling when the compostable residue is landfilled to maximise carbon sequestration, but that the results would be comparable with fluxes from the highest standards of landfilling. The figures in bold in the final column of Tables 17 and 18 are the best estimate of what greenhouse gas emissions would occur.

As mentioned previously, the landfill of residue produced from the MBT process may still be counted as BMW if it is not sufficiently biostabilised, and so prices associated with landfilling must be taken into account. However, the fact the MBT produces a more compact and degradable residue will extend the economic life of the landfill, and when landfilled it also has a reduced capacity in the production of landfill gas and leachate. MBT prior to landfill reduces the landfill gas emission potential by 90% compared with untreated MSW (Smith *et al.*, 2001). As with the landfilled residue from incineration, its **external cost should be calculated as the external costs of landfilling, exclusive of the costs of greenhouse gas emissions when the residue is biostabilised or including these costs when it is not** (as discussed in the section on incineration and derived from Bartelings *et al.* 2005). This implies, from Table A in Section 1.3, which the cost of landfilling the biostabilised residue from an MBT process is equal to the disamenity effect alone and equal to the total external cost of landfilling (including methane) for a non-biostabilised residue.

9.1.2 Health

No significant health effects appear likely from MBT (DEFRA, 2004a). The impact of treatments on operator health appears to be under-explored, however, for all treatment processes.

Water quality and leachate

MBT facilities are not thought to represent a high risk to water resources (DEFRA, 2004a). However, drainage systems to sewers (or collection for transfer to sewage treatment works) are likely to be necessary to prevent contamination.

Biological treatment of waste prior to landfilling is expected to reduce the production of leachate, and that leachate which is produced is weaker than that of conventional landfilling. Thus the potential negative effects of leachate are expected to be reduced.

Disamenities

Costing the disamenities of MBT is the area in which perhaps the greatest difficulties lie. Disamenity effects are not always included in studies or, where estimation of disamenities have been carried out, it often concentrates on landfill. The Cambridge Econometrics (2003) study for the disamenity costs of living near a landfill site has no equivalent for MBT, and there appears to be no quantification in any study for the impact of the presence of an MBT facility on property prices. As such, Eunomia (2008) arrived at an average of the figures used for landfilling and incineration, reasoning that the literature is suggestive that its value would lie between these two extremes. Such considerations led them to derive a disamenity value of €9.28/tonne for MBT (compared to €4.25 for landfill and €14.30 for incineration).

Despite the lack of formal quantification, it seems likely that disamenity costs would deviate significantly from zero. For example, the Cork Region (City and County) has faced public opposition - sustained over a ten year period - to its plans for an MBT plant.[130] This behaviour may indicate an expected disamenity impact from MBT among those in the affected area. A high disamenity value can be expected the more the following factors are prevalent:

i) Poor management of the MBT process (allowing prevalence of odours)
ii) The scale is significant, contributing to visual pollution, as well as increased transport pollution and congestion. This presents the risk of double-counting when transport externalities are being considered.
iii) The proximity to housing.

Currently, it does not appear that any olfactometry tests have been carried out for MBT facilities. DEFRA (2004a) suggests that odours are likely to be present at MBT facilities due to the presence of putrescible waste (food and garden waste). McLanaghan (2002) suggests such odours are likely to be controllable through processes such as biofiltration.

Noise issues are likely to arise from the transport of waste due to on-site vehicles, mechanical processes and ventilation/fan systems. Noise limits at receptors are also suggested as 45-55dB (A) (daytime) and 35-45dB (A) (night-time) as for other industrial facilities (DEFRA, 2004a).

In this report, we calculate the disamenity value of MBT using both the incineration and landfilling methodology in order to produce a range. The MBT facilities used to gather information on housing densities are those profiled in Eunomia (2008). They consist of a mix of MBT processes. All five used mechanical processes (the ‘M’), two also used biological processes (the ‘B’) and only one produced RDF. Therefore, the disamenity values in Table A.22 must be taken as characteristic of a non-specific form of MBT.

[130] Gerry Walsh of Cork City Council to the Joint Committee on Environment and Local Government, 25th January, 2006. Accessed on 30/11/09 at: http://debates.oireachtas.ie/DDebate.aspx?F=ENJ20060125.xml&Ex=All&Page=3

Table A.22: Disamenity of MBT Using Two Different Methodologies: Incineration & Landfill, per tonne, Ireland, 2009

Panel A: Disamenity value using incineration methodology

	Low Range		High Range		Average	
Discount Rate	10%	5%	10%	5%	10%	5%
Total Disamenity	367,000	440,000	495,000	594,000	431,000	517, 000
Disamenity/tonne MBT managed [131]	1.03	1.23	1.39	1.67	1.21	1.45

Panel B: Disamenity value using landfilling methodology

Distance from landfill	% reduction in price	Number of Houses	Fixed Disamenity Value
0-0.25 miles	-7.06	86	1,610,000
0.25-0.5 miles	-2	92	477,000
Total		178	2,090,000

	Disamenity (2009€) /tonne MBT managed
Per tonne, 5% rental yield	0.59
Per tonne, 10% rental yield	1.00

Source: see text.

9.1.3 A summary of the external costs of MBT

The results of the above analysis in terms of the appropriate levy for material sent to MBT applying the methodology set out in Section 1.2 is presented in Table A.23 below. As with incineration, the main component of the levy derives from the disamenity effect of MBT although, unlike incineration, methane does contribute somewhat to the total external cost. In the table the range is the average of each of the two estimation procedures in panel A and panel B of Table A.22. Even if one were to use the highest estimate from either panel the disamenity would still be lower than either urban incineration or landfill. Again, the effect of varying housing densities in the disamenity calculation is significant.

[131] The level of MBT managed in Ireland (in tonnes) was taken as the waste limits in the EPA licences for the five facilities profiled in Eunomia (2008). It was 356,519 tonnes.

Table A.23: Pricing the External Costs of MBT, Per tonne, Ireland, 2009

Externality		**Price (2009 €/tonne waste processed)**
Greenhouse Gases	Carbon Dioxide	Presumably not included in ETS, but non-carbon neutral emissions dealt with through carbon tax.
	Methane	0.116
Air Pollutants		Omitted, regulated through emission limits in waste licence
Solid Residue		Dealt with through landfill or incineration (and dependent on its biostabilisation)
Disamenities		0.80 to 1.33
Total		0.92 to 1.45

Source: see text.

1.6 Conclusion

The difference in the optimal levies for MBT, incineration and landfill revolves around methane emissions and disamenities. We have assumed that CO_2 emissions and other emissions to air will continue to be regulated through other instruments, the ETS, the carbon tax and the emission limits already set in each waste licence. The levy rates implied by the analysis in this section are summarised below.

Landfill	€44.24 to €54.89	(Table A.10)
Urban Incineration	€4.22 to €5.07	(Table A.17)
Rural Incineration	€0.42 to €0.50	(Table A.17)
MBT	€0.92 to €1.45	(Table A.23).

It should be noted, however, that these levy rates underestimate the total costs of using incineration and MBT, since use of these options is likely to involve some additional externality-related levy payments due to the production of outputs from incineration and MBT that are sent to landfill. Landfill externalities not already regulated elsewhere give rise to an implied levy more than double the current (2009) landfill levy rates. Both the methane and disamenity components are higher for landfill than those for the other technologies we have examined.

SUPPLEMENTARY TABLES

Table S.1 Externalities – Detailed Workings

Air Pollutant	Damage Cost		Scenario A: Emissions wholly released to atmosphere via landfill gas flaring		Scenario B :Emissions wholly released to atmosphere via gas generating engine (i.e. energy utilisation)	
	€/tonne emission (2000 €)	**€/g emission**	**g/tonne waste**	**€/tonne waste**	**g/tonne waste**	**€/tonne waste**
Arsenic	80,000	0.08			0.0016	**0.000128**
Dioxin	37,000,000,000	37000	0.00000074	**0.00274**	0.00000019	**0.00703**
Mercury	6,000,000	6			0.0016	**0.0096**
Cadmium	21,000	0.021			0.1	**0.0021**
Nickel	2,100	0.0021			0.013	**0.0000273**
Nitrogen Oxides (NOx)	5,600	0.0056	100	**0.56**	900	**5.04**
Particulate Matter (PM2.5)	22,000	0.022	8	**0.176**	7	**0.154**
Sulphur Dioxide (SO2)	7,500	0.0075	120	**0.90**	70	**0.525**
Volatile Organic Compounds (VOCs)	950	0.00095	1.7	**0.001615**		
Total				**1.64**		**5.74**

Source: DEFRA (2004a), BeTa MethodEx (2007), CAFÉ (2005).

Air Pollutant	**Damage Cost**		**Scenario C:75% gas captured by flaring and 25% released untreated to atmosphere**		**Scenario D: 75% gas captured by generating engines and 25% released untreated to atmosphere**	
	€/tonne emission (2000 €)	**€/g emission**	**g/tonne waste**	**€/tonne waste**	**g/tonne waste**	**€/tonne waste**
Arsenic	80,000	0.08			0.0012	**0.000096**
Dioxin	37,000,000,000	37000	0.000000055	**0.00204**	0.00000014	**0.00518**
Mercury	6,000,000	6			0.0012	**0.0072**
Cadmium	21,000	0.021			0.071	**0.001491**
Nickel	2,100	0.0021			0.0095	**0.00001995**
Nitrogen Oxides (NOx)	5,600	0.0056	75	**0.42**	680	**3.808**
Particulate Matter (PM2.5)	22,000	0.022	6.1	**0.134**	5.3	**0.1166**
Sulphur Dioxide (SO2)	7,500	0.0075	90	**0.675**	53	**0.3975**
Volatile Organic Compounds (VOCs)	950	0.00095	7.6	**0.00722**	6.4	**0.00608**
Total				**1.24**		**4.34**

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