Rethinking the Waste Hierarchy

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Recommendations

A number of specific recommendations for achieving cost-effective waste policies can be made based on both the US experience presented by Ackerman and Porter and on the analysis of European waste management presented by Pearce and Dijkgraaf & Vollebergh. The results of this project relate to both the target setting and the regulatory implementation of waste policy in the EU.

The main recommendations for future waste policies in the EU and Member States are:

- *The waste hierarchy must be considered a very general and flexible guideline for formulating waste policies.* What is environmentally desirable is not always a preferred solution, when considered from a socio economic perspective. The reason is that some environmental benefits may come at a comparably socially high cost. The marginal costs and benefits will vary depending on material and locality. It is recommended that social costs and benefits of new recycling schemes should be analysed and that a critical assessment be made on to determine if further steps are in fact socially desirable.
- Regulation by fixed target setting in waste policy is currently the preferred method of regulation at the EU level. Examples are the packaging waste Directive with fixed target rates for recycling. Instead of current strategies, *price-based policies are recommended,* since they have the ability to change the demand for disposal methods and potentially change the material used towards more sustainable choices. There are two reasons for this recommendation:
  - *There is no upper boundary to the costs* related to reaching the fixed targets.
  - *Fixed targets are bad at encouraging recycling*, since they have insignificant effects on demand for the environmentally desirable solutions and materials.

- Institutional factors and non-economic reasons can be very important in policymaking and may explain why the European Union prefers fixed target setting such as mandatory recycling targets. However, the empirical analyses in this report suggest *a risk of welfare losses due to fixed target setting policies in EU waste policies.*
- Since waste management is one of the single largest environmental expenses in EU Member States, the potential losses from inefficient policies are very substantial. Equally, *optimal waste policies can potentially free funds for other needs*.

A policy towards fairer pricing in waste policies would be recommendable and follow the tendency of the EU strategies for other environmental issues such as transport and energy.

Summary

There is an increasing need to couple environmental and economic considerations within waste management. Consumers and companies alike generate ever more waste. The waste-policy challenges of the future lie in decoupling growth in waste generation from growth in consumption, and in setting priorities for the waste management. This report discusses the criteria for deciding priorities for waste management methods, and questions the current principles of EU waste policies.

The environmental damage caused by waste depends on which type of management we choose. Also, the methods differ in price. Must we always opt for the solution that is least detrimental to the environment or should we also take into account the costs of the solution?

In this report, a number of economists offer their vision of the criteria that they think should underlie the future EU waste policies.

The basis for the discussion is the so-called waste hierarchy which has dominated the waste policy in the EU since the mid-1970s. The waste hierarchy ranks possible methods of waste management. According to the waste hierarchy, the very best solution is to reduce the amount of waste. After that, reuse is preferred to recycling which, in turn, is preferred to incineration. Disposal at a landfill is the least favourable solution.

Vollebergh and Dijkgraaf present the most controversial findings in the report, based on a socio-economic analysis of incineration and disposal. The authors argue that the net social cost of incineration by far exceeds the net social cost of landfill. This is because the financial costs of incineration are higher compared to landfill, while environmental costs are approximately equivalent for the two methods. Thus, the aggregate result indicates that landfill may be a better option than incineration. A literature review also shows that socio-economic analyses do not always support the ranking of methods in the waste hierarchy. For instance, the review suggests that recycling is *not* always the optimal solution.

The latter conclusion is supported by Pearce, who has examined whether the two EU waste Directives, the Landfill Directive and the Packaging and Packaging Waste Directive, pass a socio-economic "cost-benefit" test. Pearce concludes that the Packaging and Packaging Waste Directive does not pass the CBA test, and it is likely that the Landfill Directive will not pass, either.

Porter presents arguments against the use of regulatory instruments based on quantity, such as fixed recycling targets, in each EU Member State. For instance the Packaging and Waste Packaging Directive requires all Member States to recycle 22.5 percent of plastics. Porter is of the opinion that regulatory instruments based on price and demand for waste management options should determine the allocation of waste between recycling, incineration and landfill disposal. Porter discusses three price-based solutions and one of them is a tax that raises the costs of waste disposal to a level that also includes the negative effects on the environment and human health. According to Porter, one of the reasons that politicians prefer volume regulations to price regulations is that volume regulations are easier for policymakers to apply.

Ackerman makes the point that the volume of waste recycled should be determined by the population's willingness to pay for recycling. Ackerman also believes that Europe can learn from the best recycling systems in the US, Canada and Australia. Europe has many separate recycling schemes, whilst in some places in the US several types of materials are collected under the umbrella of the same recycling system. At the same time, Ackerman emphasises that local conditions such as population density are vital to the success of a recycling system.

The final chapter present reflections on the discussion that took place on the seminar held on 14 December 2004. This summarises the responses of the critics. It also provides a constructive response to the criticism and summarises some suggestions for further studies and actions.

Anbefalinger

Med udgangspunkt i erfaringerne fra USA, præsenteret af Porter og Ackerman, og analyserne af den europæiske affaldspolitik, præsenteret af Vollebergh og Dijkgraaf og af Pearce, kan der gives en række af specifikke anbefalinger til en mere omkostnings-effektiv europæisk affaldspolitik. Anbefalingerne til den fremtidige affaldspolitik i EU og medlemslandene er:

- ***Affaldshierarkiet skal betragtes som en meget generel og fleksibel vejledning ved udformning af affaldspolitik.*** Det er vigtigt at huske på, at den løsning der foretrækkes ud fra et snævert miljømæssigt synspunkt, ikke altid er den bedste løsning ud fra en bredere samfundsøkonomisk vinkel. Miljøgevinsterne kan komme ved en høj samfundsøkonomisk omkostning. De marginale miljøgevinster og -omkostninger vil variere afhængig af behandlingsmetode, materiale og lokalitet. Derfor anbefales det, at der udarbejdes samfundsøkonomiske analyser af nye genanvendelsessystemer. Politikere og embedsmænd bør foretage en kritisk vurdering af, om yderligere genanvendelse reelt er samfundsøkonomisk hensigtsmæssig.
- Regulering ved faste målsætninger i affaldspolitik er i øjeblikket den mest foretrukne reguleringsmetode på EU niveau. Eksempler er Emballagedirektivet med faste rater for genanvendelse. ***I stedet for den nuværende strategi, anbefales en prisbaseret politik***. En prisbaseret politik, som fx afgifter, kan ændre efterspørgslen efter bortskaffelsesmetoder og også ændre materialeanvendelsen hen imod mere valg af mere bæredygtige materialer. Der er to årsager til denne anbefaling:
    - ***Der eksisterer ingen øvre grænse for omkostningerne, der relaterer sig til at nå målsætningerne***
    - ***Faste målsætninger bliver anset som værende dårlige til at fremme genanvendelse,*** idet de har ubetydelige effekter på efterspørgslen efter de miljømæssigt ønskværdige løsninger og materialer.

- Institutionelle faktorer og ikke-økonomiske årsager kan være meget vigtige i udformningen af politik og kan forklare, hvorfor EU foretrækker faste målsætninger som f.eks. genanvendelsesrater. De empiriske analyser i denne rapport indikerer, at der er *risiko for velfærdstab som følge af faste målsætninger i EU's affaldspolitik*.
- Når affaldshåndtering er en af de største enkeltstående udgifter på miljøområdet i EU's medlemslande, kan de potentielle tab fra ikke-efficiente politiker være betydelige. Tilsvarende kan en *optimal affaldspolitik frigøre økonomiske midler til andre behov*.

En politik mod mere fair prissætning i affaldspolitik er meget anbefalelsesværdig og vil følge tendenserne i EU strategierne for andre miljømæssige problemstillinger som f.eks. transport og energi.

Resumé

Der er et stigende behov for at tænke miljø og økonomi sammen på affaldsområdet. Øget forbrug hos forbrugere og virksomheder efterlader stadig mere affald. Udfordringen i fremtidens affaldspolitik er dels at afkoble væksten i affaldsmængden fra væksten i forbruget og dels at prioritere indsatsen over for behandlingen af affald. Denne rapport diskuterer, hvilke kriterier der skal tages i brug, når vi skal prioritere metoderne til behandling af affald, og rapporten stiller spørgsmålstegn ved de gældende principper i EU's affaldspolitik.

Skaderne på miljøet afhænger af hvilken behandlingsmetode vi vælger. Der er også forskel på, hvad metoderne koster. Skal vi altid vælge den løsning, der giver mindst skade på miljøet, eller skal vi også tage hensyn til, hvad det koster?

En række økonomiske eksperter giver i denne rapport deres bud på hvilke kriterier, de mener, bør lægges til grund for EU's fremtidige affaldspolitik.

Baggrunden for diskussionen er det såkaldte affaldshierarki, som har domineret affaldspolitikken i EU siden midten af 1970'erne. Affaldshierarkiet rangordner mulige metoder til håndtering af affald. Helt grundlæggende siger affaldshierarkiet, at mindre affald er den allerbedste løsning. Herefter er genbrug at foretrække frem for genanvendelse, der igen er at foretrække frem for forbrænding. Deponering på losseplads er den dårligste løsning.

Vollebergh og Dijkgraaf præsenterer i rapporten det mest udfordrende resultat baseret på en samfundsøkonomisk analyse af forbrænding og deponering. I analysen argumenterer forfatterne for, at de finansielle omkostninger ved forbrænding er langt større end omkostningerne ved deponering. Det sammenholdes med at prisen på miljøskaderne er omtrent den samme for de to metoder. Det samlede resultat tyder derfor på, at deponering kan være bedre end forbrænding set ud fra et samfundsøkonomisk synspunkt. Et litteraturstudie viser også, at samfundsøkonomiske analyser ikke altid understøtter affaldshierarkiets rangordning af metoder. F.eks. tyder studiet på at genanvendelse *ikke* altid er den bedste løsning.

Denne konklusion understøttes af David Pearce, der har undersøgt om to EU direktiver på affaldsområdet, Emballagedirektivet og Deponeringsdirektivet, består en samfundsøkonomisk "cost-benefit" test. Pearce konkluderer at Emballagedirektivet ikke består CBA-testen, og der er stor sandsynlighed for at Deponeringsdirektivet heller ikke består.

Richard Porter argumenterer i rapporten imod anvendelsen af mængdebaserede reguleringsinstrumenter, som f.eks. faste genanvendelsesrater i hvert EU medlemsland. Emballagedirektivet kræver blandt andet, at alle medlemslande skal genanvende 22,5 procent plastik. I stedet mener Porter, at reguleringsinstrumenter baseret på pris og efterspørgsel bør afgøre, hvilken mængde der skal genanvendes, forbrændes og deponeres på losseplads. Porter diskuterer tre prisbaserede løsninger. En af dem er en afgift der øger prisen for bortskaffelse af affald til et niveau, som også inkluderer de negative indvirkninger på miljøet og menneskers helbred. Ifølge Porter er en af årsagerne til at politikere foretrækker regulering på mængden frem for på pris, at mængdereguleringer er lettere at anvende i praksis.

Endeligt argumenterer Frank Ackerman for, at mængden af affald, der genanvendes, bør afgøres af befolkningens betalingsvilje for genanvendelse. Ackerman mener også, at europæerne kan lære af de bedste genanvendelsessystemer i USA, Canada og Australien. Europa har mange separate genanvendelsesordninger modsat nogle steder i USA, hvor flere materialer indsamles under samme genanvendelsessystem. Samtidig argumenterer Ackerman for, at lokale forhold som f.eks. befolkningstæthed er helt afgørende for, om et genanvendelsessystem har succes.

I det sidste kapitel præsenteres refleksioner over diskussionen der fandt sted på seminaret afholdt den 14. december 2004. Der gives en opsummering af den kritiske respons. Kapitlet giver også konstruktive svar til kritikken og opsummerer nogle forslag til yderligere studier og tiltag.

Foreword

Why is there a need to rethink the waste hierarchy? The idea behind this project originates at the Danish Environmental Assessment Institute. This institute is an environmental assessment organisation, funded by the Danish Government but independent of political influence, and using a combination of best-available evidence and socio-economic analysis to give advice on the effectiveness of environmental policy.

The aim of the project is to influence the current approach to waste management in the EU, and to encourage a move towards the wider use of a social cost-benefit approach in order to raise the social benefits of waste policies and lower the costs.

The objectives of the project are to deliver information on the advantages of the social cost-benefit approach in EU waste management to EU Governments, the EU Commissioner, environmental NGOs, citizens in the EU Member States, and environmental economists.

The approach involved collaboration. The Environmental Assessment Institute asked leading economists in the field of the economics of waste to contribute. The contributors are:

- Frank Ackerman, Tufts University, author of “Why do we recycle?”
- Richard Porter, The University of Michigan, author of the “The Economics of Waste”
- David Pearce, University College London and Imperial College London
- Elbert Dijkgraaf and Herman Vollebergh, Erasmus University Rotterdam (SEOR and the Rotterdam School of Economics)

This background report is the second output of the project. The first output was an open seminar held on 14 December 2004 at the Environmental Assessment Institute in Copenhagen. At the seminar the contributors to the project presented their results. The presentations were followed by a discussion between the authors and the audience. In chapter 6, Ackerman presents a personal view of the proceedings that incorporates the *pros* and *cons* expressed by delegates at the seminar on the socio-economic approach.

1 Rethinking the waste hierarchy?

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1.1 Introduction

Since the mid-1970s, the waste hierarchy has played an important role in the European Union waste policy. The hierarchy has been used as a waste policy guidance principle on how to prioritise methods for waste treatment and disposal. Scientific studies, such as life-cycle analysis, show that the waste hierarchy provides useful advice on how to determine priorities for waste handling methods according to impacts on the environment. On the other hand, socio-economic studies indicate that the hierarchy may fail to provide general guidance on how to prioritise waste handling methods according to social desirability, i.e. according to social costs and benefits.

Box 1 - The waste hierarchy

The waste hierarchy is a principle which prioritises different waste management options according to their environmental desirability. A common interpretation of the waste hierarchy in the European Union is:

1) source reduction,
2) reuse,
3) recycling,
4) composting,
5) incineration with energy recovery,
6) landfill with energy recovery,
7) incineration without energy recovery, and
8) landfill without energy recovery.

In many countries, the waste hierarchy is used as the guiding principle in waste management policies and strategies.

The basis for the ranking of options in the waste hierarchy is not very clear. However, the main message from the hierarchy is that waste prevention is generally better than waste disposal. Disposal of waste by incineration or landfill clearly has environmental effects in terms of emissions to air and chemical waste generated during the treatment. Even recycling is associated with environmental effects from transportation, energy use, and other residuals that occur in relation to the recycling process. Incineration, landfill and recycling also have positive environmental effects because the energy produced from incineration and landfilling can displace the energy produced from other sources, e.g. from coal-fired electricity plants. Similarly, recycled material can displace extraction of new materials from natural resources.

Both the direct and indirect environmental effects should be considered when attempting to rank the options for waste treatment. The waste hierarchy may take into account the environmental effects, but the relevant question in relation to policy development is not whether to choose recycling, incineration or landfilling. The relevant question is how much waste we should recycle, incinerate or landfill, respectively.

From an economist point of view, social cost-benefit analysis (CBA) is a well-known method that can be used to determine if it is socially desirable to increase or decrease the percentage of waste that should be recycled, incinerated or landfilled. The CBA method allows the direct and indirect environmental effects to be weighed against the positive and negative economic impacts of choosing a specific method. In this perspective, the method can be used to assess if we achieve a more cost-effective waste policy.

The CBA method is capable of providing guidance on the social desirability of specific waste policies. Nevertheless, it is also important to inform about the uncertainties related to CBA studies. In the literature review presented in this report, Dijkgraaf and Vollebergh identify uncertainties in most of the CBA studies carried out so far. They point out that many of the studies have not been published in refereed international scientific journals. So raising questions about the quality of the studies. A number of studies are not very clear about the shadow prices used to monetarise emissions. Not many studies account for local and national circumstances that may prevent generalisation of the results. The definition of private costs is often unclear and this may affect outcomes. The benefits of landfilling are not always properly defined. Finally, the benefits estimated for incineration are sometimes too large.

Researchers face a challenge to reduce these uncertainties in their analyses especially when the CBA method is used to advise policy-makers on the social desirability of specific waste policies. The uncertainties also establish a need for harmonising the methodological approach and presenting the results and assumptions in a transparent way. Methodological harmonisation provides more consistency between the studies and transparency signals openness and willingness to discuss assumptions.

The aim of this report is to provide an understanding of the social costs and benefits of the current European Union waste policy and the underlying principles of the waste hierarchy. There are four contributions in the report from five experts in the economics of waste. Together they present persuasive arguments to rethink the waste hierarchy as the basis for policy recommendations in the EU and in the Member States. There are a number of clear recommendations that all contributors support and some issues where different arguments are put forward. Section 1.3 will draw together the main findings of the authors.

What can economics tell us about the principles of the waste hierarchy and the European Union policy on waste disposal? The authors of this report all present specific recommendations for achieving a more cost-effective waste policy based on both the US experiences presented by Ackerman and Porter and on the analysis of European waste management by Pearce and Dijkgraaf & Vollebergh.

1.2 Background

Every day we produce and consume goods which leave behind a certain amount of waste. The management of waste is becoming more and more technologically advanced, but the treatment still has unwanted environmental consequences. When the generation of waste increases, so do the impact on environment and the expenses for waste management. Public and private funds to cover these expenses are a limited resource and thus we have to consider how to allocate this resource in the best possible way. Since waste management in many EU Member States is the single largest environmental expense, policy analysis to help improve economic soundness can potentially release significant sums for competing needs.

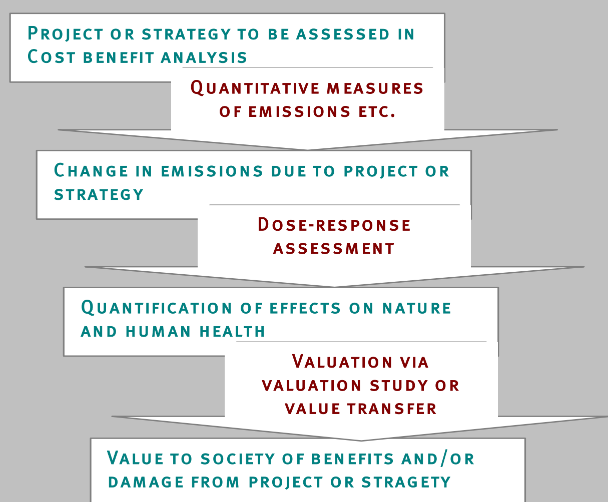
Government intervention affects almost every type of waste we generate today. Regulation of the market for waste disposal is needed due to the occurrence of externalities when waste is disposed of. Externalities occur when the activity of one person influences the activity or welfare of another person. For instance, when waste is landfilled it generates noise, smell, and is an eyesore to neighbours. Incineration of household waste may generate air pollution that is toxic to human health and can influence the functioning of ecosystems. The presence of externalities results in what in economics is termed a "market failure". That means that since we as individuals would prefer not to suffer illness and damage nature around us, society as a whole would be better off by limiting these externalities to an acceptable level. But doing something about this requires collective action. The central issue related to market failures is the problem of how to regulate these situations. This is a three-part task. Firstly, how much waste should there be? This is the issue of how much waste reduction there should be. Secondly, given a certain amount of waste, what is the optimal mix of disposal and recycling etc.? Finally, given this optimal mix, what is the optimal set of regulations/instruments for securing this solution?

The advantage of cost-benefit analyses is that they indicate if the benefits are far above the costs of a specific target or a specific regulation or vice versa. This is done by translating the environmental benefits into social benefits through "valuation" of the benefits. Despite diverse approaches to valuation, most agree that cost-benefit analysis, done properly, provides good support for social decision-making (see Box 2). In spite of this, such analyses are seldom influential in the choice of specific waste strategies in the EU.

Box 2 – Measuring environmental effects in cost-benefit analysis

A central issue in a discussion of cost-benefit analyses and their application as guidance for waste strategies is the question of measuring the benefits of recycling. The first step in the process is to determine what the relevant environmental effects of the project or policy are. The next step is to quantify the effects and through dose-response assessments quantify the effect on nature and human health. Thirdly, the value of the change in nature and human health is valued in terms of its value to society. The three steps are illustrated in the arrows in Figure 2.

Figure 2. Steps necessary in valuation of environmental benefits or damage



Source: Mod. version of Navrud & Bergland (2001): Value transfer and environmental policy, EVE Policy Research Brief, Cambridge Research for the Environment.

A number of effects are today quite routinely valued in environmental assessment by looking at alternative abatement costs or nationally agreed value-estimates for the damage to society. This goes for most kinds of air pollution, greenhouse gasses and human health effects. However, for a number of environmental effects the technical and natural science data are not complete enough to get to the penultimate level in Figure 2, where the effects on nature and human health of a given change in technology can be quantified. In other cases the valuation studies are not sufficient to give accurate valuation estimates. In these cases one can choose to either use the best available estimates, though uncertain, for the benefit/forgone damage, or not include the effect in the cost-benefit analysis and describe it qualitatively. These weaknesses of cost-benefit analysis are acknowledged by economists, but it does not automatically follow that the uncertainties mean the method should be dismissed entirely.

1.2.1 The waste hie rarchy and the European Union

As early as the 1970s, the waste hierarchy principles were introduced into the European policy on waste disposal. They were introduced during discussions in the Council and formulated in the first Directive on waste in 1975 (Box 3).

Box 3 – The Waste Hierarchy in EU legislation

Council Directive of 15 July 1975 on waste (75/442/EEC) is the starting point of the EU waste policy. Article 3 of that directive states that: Member States shall take appropriate measures to encourage:

a) firstly, the prevention or reduction of waste production and its harmfulness,

b) secondly:

(i) the recovery of waste by means of recycling, re-use or reclamation or any other process with a view to extracting secondary raw materials, or

(ii) the use of waste as a source of energy.

The text of article 3 does not really put an order of priority on the different waste management options, but it would be natural to deduce that prevention of waste generation ranks highest, because it is placed first in the enumeration.

The list of options for waste management was written in the Council Directive (75/442/EEC), but not in the Commission's proposal C 142 of 1974. The Commission's proposal suggested only, in Article 4, that Member States should promote the regeneration and re-utilisation of waste. This indicates strongly that it was the Member States in their Council discussions who decided to make these different options of waste management statutory obligations by putting them into EC legislation.

The closest answer to the question of where the hierarchical principles came from, is that during their discussions in the Council the Member States introduced these policy principles. The only "proof" being that the Commission's proposal did not contain them, but the final directive did. Which of the Member States suggested the principles and the arguments during the discussions in the Council is unclear, but the environmental basis for the hierarchy seems widely accepted in most Member States.

In conclusion there is no "hierarchy" introduced, in form of any legally binding commitment, in EC law. Even though the word "waste hierarchy" is not directly written in EC law the tendencies in new EU waste directives mirrors the principles of the waste hierarchy.

Source: Krämer, Ludwig (2004): personal communication. Head of unit "Legal questions and Governance", DG ENVIRONMENT, European Commission.

The word hierarchy does not appear directly in the legislation, but the waste management options are ranked according to the hierarchy. The objectives of the recent EU waste Directives also mirror the principles of the waste hierarchy. For instance, the aim of the EU Directives on packaging waste, batteries, electrical and electronic waste is to reduce the environmental damage of these waste streams by channelling them towards higher ranking waste management options. The Directives require Member States to introduce legislation on waste collection, reuse, recycling and disposal of these waste streams.

The goal of the waste hierarchy is to minimise the environmental effects of waste disposal. In general, the waste hierarchy states that waste prevention is better than waste disposal, and the principles are based on the premise that the disposal option which is generally least environmentally harmful is ranked highest.

According to Pearce (see below) the problem with the hierarchy is that the basis for the ranking has never been clear and appears to have emerged as a 'consensus' ranking, although it is equally unclear to which parties the consensus is supposed to apply.

Both recycling and disposal of waste by incineration or landfill generate environmental effects during the treatment. Recycling is associated with environmental effects from transportation, energy use and other residuals that occur in relation to the recycling process. Incineration, landfilling and recycling also have positive impacts on the environment because the energy produced from incineration and landfill can replace the energy produced from other sources, e.g. from coal-fired electricity plants. Similarly, recycled materials can replace extraction of new materials from natural resources.

Both the direct and indirect environmental effects are important to consider when attempting to rank the options for waste treatment. It is uncertain whether both the direct and indirect environmental effects are taken into account in the waste hierarchy. Furthermore, it is uncertain how the specific environmental impacts, e.g. carbon dioxide and nitrogen emissions, are weighted and translated into one comparable scale. Even if all the environmental effects were taken into account and weighted in the waste hierarchy, the relevant question in relation to policy development is not whether to choose between recycling, incineration or landfilling. The relevant question is how much waste we should recycle, incinerate and landfill.

Pearce states that it is still possible to derive a hierarchy based on economic analysis of the baseline mixes of management options. The ranking can be calculated by measuring the (marginal) net social cost of each management option. *Net* social cost refers to the financial costs plus the environmental costs of the option less any associated benefits. The management option with the lowest net social cost will be the most preferred option, and so on.

Pearce also notes that the ranking may change as the quantity-mix of waste going to the various options changes. This is not widely understood. The idea of a waste hierarchy that is fixed and immutable is therefore erroneous. Furthermore, the hierarchy will not necessarily be the same for every location: both financial and environmental costs will vary according to where waste is sent.

Overall, Pearce concludes that the conventional hierarchy is shown not to be excessively inaccurate. However, the general outcome of a social cost analysis is that the hierarchy will vary by waste type and perhaps by country as well. The answer to this problem is that policies should be location-specific. This conclusion therefore runs counter to the requirement of environmental harmonisation within the EU.

Environmental impact assessments, such as life-cycle assessments, often support the general ranking of the waste hierarchy. However, life-cycle assessments lack an overall weighting of the specific environmental impacts. The description of the impacts in these analyses does not allow a direct comparison with the costs involved. This is problematic because there are notable social budget constraints on the choice of the best environmental policies, just as for any other kind of public choice. A guideline such as the waste hierarchy, which to a large extent is supported by life-cycle assessments, and which is solely based on what is termed “environmental desirability” may therefore not be appropriate (See Box 4).

Box 4 – Life-cycle assessment and social cost assessments

When assessing an environmental problem in the preparation of policy choices, the aim is to gather as much information as possible, and at the same time keep perspective and overview of the problem at hand. Life-cycle assessment is a well-established method of assessing the environmental effects related to alternative strategies. A “cradle-to-grave” approach is assumed, which includes all environmental effects on a global scale. That is, irrespective of where the effect is, it is taken into account. This results in a very large data set. All the data are presented in categories of environmental damage and the alternative solutions are ranked according to how much damage they will entail. A common unit of measurement is not assigned to the damage, so it is not possible to assess how much worse or good one solution is relative to another. This makes it difficult to relate the environmental impacts to other environmental problems that policy makers also wish to solve.

Social cost assessments, including cost-benefit analysis, account for all relevant environmental effects, but the analysis may differ in what is determined to be “all relevant environmental effects”. Most often a cost-benefit analysis will use a life-cycle database as the basis for assessing the benefits (i.e. the environmental damage avoided) in choosing a more environmentally friendly solution. New environmental impact assessments may also be carried out, in order to collect data. A social cost assessment can delimit the analysis to national or regional borders or assume the global focus of life-cycle assessments. This delimitation depends on the aim of the policy and must be mirrored by the same delimitation of the assessment of the social cost. In either case, the social cost assessment will conduct a “social impact assessment” of all relevant social effects, such as the difference in original investment and running costs of the alternatives. Distributional effects may also be assessed to see which groups in society gain and loose by choosing one alternative over another.

The promotion of the hierarchy by both the European Commission and the environmental protection agencies in EU Member States influences the way we think about waste disposal, especially about recycling. Recycling is a good strategy for society. But should we recycle 100 percent?

The answer to this question depends on the scale against which the waste disposal options are measured. If the scale is environmental desirability, the answer is, yes – recycle 100 percent, assuming that the amount of waste has already been minimised as much as possible. But if the scale is social desirability of all the social costs and benefits to society as a whole[1] – the answer is, no. It is not a question of recycle or not, but of how much to recycle. Likewise it is not merely a question of whether to landfill or not, but how much to landfill. To understand why this is an important distinction, it is helpful to think of waste disposal as involving various ways of minimising environmental damage.

From an environmental point of view, recycling is preferred to incineration because the environmental effects are less for some materials when they are recycled rather than incinerated. So the more of such a material (e.g. aluminium) we recycle, the less environmental damage we do. But for most environmental problems it is a fact that abatement costs become marginally higher, the more we increase the level of abatement.

In socio-economic terms this corresponds to the diminishing benefits of cost of abatement. Gradually the environmental benefit from "cleaning up further" becomes smaller compared to the benefits of the first improvements when the starting point was quite bad. Usually, the reasonable level of abatement - or what is sometimes referred to as "allowed pollution" - is neither 0 nor 100 per cent, but somewhere in between. Similarly, there is a difference between the cost of preventing the bulk of the damage and the cost of preventing 100 per cent of the environmental damage.

Social cost-benefit analysis would only support that we recycle until the net social marginal costs of recycling equal the net social benefits. It might be argued that there is not much harm in "overshooting the mark somewhat" when it comes to environmental improvements. This argument is sometimes justified given the uncertainties in determining the true environmental benefits of a given policy. However, given that some guidelines for environmental assessments take some account of the possibility of scientific uncertainties, this is a problematic approach. Too much recycling can be just as wasteful as too little recycling. In environmental policy there is always a very real issue of scarcity of funds and a multitude of competing applications. Our resources are limited and we have to choose carefully how to spend them.

1.3 Summary of the results

The general problem with the waste hierarchy, and the use of it in waste management policies, is that it makes no reference to the costs of disposal. As explained above, waste disposal methods are traditionally ranked according to environmental desirability. In the EU waste hierarchy this has correspondingly been expressed as a ranking of what is most socially desirable. However, in determining what is socially desirable – as opposed to environmentally desirable – one must also consider social costs. In this report the primary focus is on both social benefits and costs. In section 1.3.1-1.3.3, we focus on three central social cost issues concerning the waste hierarchy.

- First, we point out that social cost-benefit studies cast doubts on the validity of the waste hierarchy as the sole ranking principle in waste management strategies.

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[1] This is no small 'if' and the issue of social economic monetisation - or valuation – is debated in box 2.

- Secondly, we draw attention to the inefficiencies of fixed recycling targets in the European Union.
- Finally, we recommend that European legislation on waste move towards more economic regulation, such as green taxes or tradable quotas.

These problems are important reasons for policy-makers and decision-makers to rethink the use of the principles in the waste hierarchy.

1.3.1 Social cost-benefit studies question the hierarchy

Social cost-benefit studies do not always recommend a ranking of waste disposal options that is identical to the ranking in the waste hierarchy. The discussion in this report of recycling versus incineration and incineration versus landfilling demonstrates that the waste hierarchy needs to be treated as a very flexible principle for policy recommendations. Social cost-benefit analysis can help debates on whether a specific policy is worthwhile. Such assessments can therefore assist decision-makers in developing more subtle strategies for waste disposal.

In the case of recycling, some economic appraisals show that this option is preferable to incineration or landfilling. Conversely, some economic studies suggest that the net costs of recycling are far above the benefits. In a literature review of socio-economic studies of waste disposal options (see below), Dijkgraaf and Vollebergh illustrate that other options than recycling are preferred in three studies, while in three studies recycling is the best option. This indicates that recycling is not necessarily always the best solution as recommended by the waste hierarchy. However, Dijkgraaf and Vollebergh warn that care is needed when interpreting the results of these studies, since they differ in methodology.

A second question regarding the waste hierarchy, is whether waste incineration is always preferable to landfilling. Some economic studies support the view that waste incineration should be the preferred strategy, as the net social cost of incineration is lower than the net social cost of landfilling. However, other economic appraisals show the opposite. Dijkgraaf and Vollebergh identify two studies that prefer incineration as the best option, while three appraisals have landfilling is the best option. This indicates that incineration is not always the best solution as suggested by the waste hierarchy.

To illustrate the consequences of indiscriminately following the principles of the waste hierarchy regarding incineration and landfilling Dijkgraaf and Vollebergh have calculated the social costs of either incinerating or landfilling waste in the European Union countries. The social costs are estimated to be Euro 6.1 billion higher for incineration compared to landfilling; a large excess cost.

In chapter 4, Pearce presents another empirical example regarding the question of incineration versus landfilling - the EU Landfill Directive. The Landfill Directive seeks significant reductions in the amount of biodegradable municipal waste being sent to landfill, and also bans some materials, e.g. tyres. Thus, it mirrors the principles of the waste hierarchy. Pearce presents an economic appraisal showing that the Landfill Directive fails a cost-benefit test.

These results are a strong argument for considering whether the waste hierarchy should continue to determine EU policy on waste disposal. However, all the authors of this report agree that the result of a social cost-benefit study of recycling versus other disposal options varies depending on the specific material and location of the programme.

In chapters 2 and 3, Ackerman and Porter identify fundamental differences between the recycling programmes in the US and the EU respectively. In the US, recycling programmes include several materials in the same recycling system, while the Europeans have a tendency to set up a separate recycling system for a specific material. This may be one reason why some European recycling systems are relatively expensive and therefore may partially explain why some systems fail in a social cost-benefit evaluation.

Ackerman also points out that the key to a more cost-effective waste policy is learning from the best models of recycling. Porter, who argues that the future net benefit of cost-effective recycling is one of the major benefits of our investments in recycling today, also supports this perspective.

Relatively low cost-effectiveness in some recycling programmes may be part of the explanation of why some cost-benefit analyses question whether the cost is too high relative to the benefits. The principle of fixed recycling targets may be another explanation, as the following section reveals.

1.3.2 Fixed recycling rates do not match wide variation costs and benefits between countries

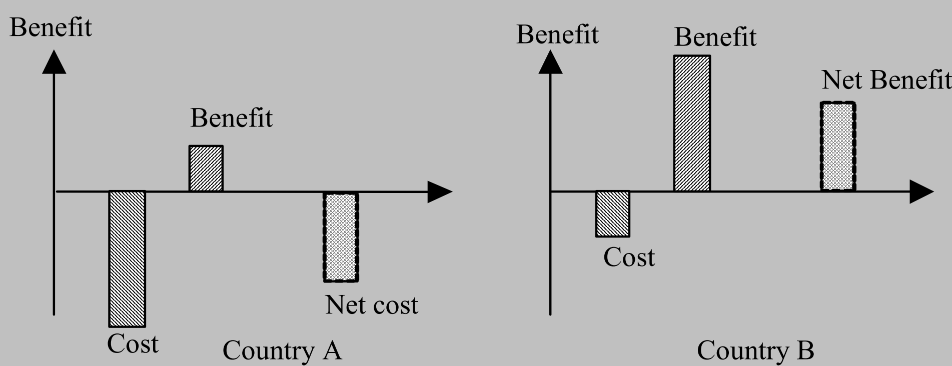
Fixed recycling rates are used as an instrument in the European Union waste policy to promote more recycling. A good example is the 1994 Packaging and Packaging Waste Directive, which requires that Member States recycle and recover specific percentages of packaging waste arisings. The Directive was revised in 2003 and Member States now have to comply with stricter recycling rates. In chapter 4, Pearce concludes that neither the 1994 Packaging and Packaging Waste Directive, nor the revised Directive, pass a cost-benefit test in the UK. In fact, Pearce concludes that both Directives fail by a very large margin.

On the other hand, it may be socially beneficial for other countries in the European Union to attain stricter recycling rates. The problem is that the target recycling rate is the same in each country. Cost and benefit functions vary between the 25 EU Member States. Consequently, some countries may gain net benefits and other countries may pay a large extra cost to achieve the same recycling target. This is not an optimal situation because the EU could achieve the same recycling target at a lower cost. The problem is illustrated in Box 5.

Box 5 - The problem of fixed recycling rates

The figure below illustrates the cost and benefit of recycling one more tonne of packaging waste in two different countries. In country A, the cost exceeds the benefit and in country B the benefit exceeds the cost.

Marginal cost and benefit of recycling waste in two different countries.



Recycling one more tonne in country A result in a net cost. However, in country B the result is a net benefit. The common result for both countries together is approximately zero, i.e. the net cost for A plus the net benefit for B. If country A recycled zero tonnes and country B two tonnes, both countries would be better off.

Also Ackerman, who advocates recycling, writes in chapter 2 that it is hard to understand why all 25 Member States should have the same fixed recycling targets. Ackerman points out that there is a wide variation within the EU in many factors that affect the design of an optimal recycling program. These factors include income levels, size and composition of waste streams, proximity to markets for recycled materials, and costs and convenience of disposal options.

The next section will consider more ambitious ways of rethinking the waste hierarchy, namely by moving away from fixed recycling targets, and towards more price-based policies.

1.3.3 Advantages of price-based regulation

A third relevant issue is how to regulate the waste market. Economists often support the use of price-based incentive systems to achieve an optimal level of externalities, and this is also the case with regard to waste disposal. At EU level, however, the preferred regulation of the waste disposal is fixed target setting, i.e. mandatory, fixed recycling targets.

In chapter 3, Porter emphasises that the best policy to encourage recycling is to use incentive-based systems. These systems attempt to correct the price distortions in the market for waste management. The aim of incentive-based regulation systems is to regulate the price of waste disposal to a level that reflects the net social costs of the different methods. Such prices will provide the right incentives to achieve the optimal level of both recycling and externalities in waste management. Ackerman also supports this conclusion, especially if they help in the process of achieving more cost-effective recycling systems.

Porter explains that the problem with fixed target setting, what he calls "quantitative-based regulations", is that consumers and firms react to input prices that are distorted by hidden subsidies and externalities. Quantitative-based systems encourage people to behave against their economic interests; e.g. recycling targets do not make people prefer recycling to landfilling. It is highly unlikely that such forced behaviour of consumers and will result in an optimal level of externalities. Thus, these regulations can be socially inefficient.

Box 6 – Price-based regulation

The idea behind price-based regulation systems is to raise the costs of waste disposal to a level that also includes the negative impacts on environment and human health, while they also allow the waste generators the flexibility to choose the methods with the lowest cost.

Since the 1970s, economists have promoted the use of economic incentive-based systems in environmental policy. However, so far there is a limited number real applications of these mechanisms, though the successes with tradable quota systems for air pollution in both the US and Europe have gained increased recognition. Nevertheless, governments still often prefer fixed target setting rather than incentive-based systems. The EU is advocating more price-based policies in other areas of environmental policy. One example is the strategies for harmonisation of transport taxes towards “fair pricing” that reflects the environmental externalities.

Three examples of incentive-based policies are proposed in chapter 3:

a) **Advanced disposal fees** paid by firms will influence their decision on choice of material for packaging. The fee should depend on the cost of recycling the individual material.

b) Households should pay a **marginal waste charge** that reflects the extra costs of landfilling or incinerating the waste instead of recycling it.

c) Finally, it is recommended that the external costs of landfilling or incineration are “internalised” through **taxes on landfill or incineration**. A number of the EU Member States already have such taxes in place. A landfill or incineration tax causes a higher waste charge on households and businesses, and this will presumably induce them to seek ways to reduce and recycle in order to avoid the higher landfill cost.

This system is an example of what Porter terms “good policies” – as opposed to “bad policies” such as fixed recycling targets and landfill bans that encourage recycling at any cost.

According to Porter, however, there are a number of institutional factors and reasons that can explain why fixed target setting is often preferred to price-based regulation. He identifies four main reasons why price-based policies generally are used so little. The first is that a fixed target policy is easier for policymakers to apply; it is relatively straightforward to ban something rather than carry out studies on socially optimal prices and taxes. The second reason is that fixed target setting policies act directly on policy goals – the target is the policy. A third reason is that policy-makers can have doubts about whether changing prices would have much effect on behaviour and waste decisions in firms and households. Finally, the costs of waste disposal are hidden in target setting policies. In this way policy-makers avoid imposing a visible tax.

1.4 Recommendations for future EU waste policies

A number of specific recommendations for achieving cost-effective waste policies can be made based on both the US experience presented by Ackerman and Porter and on the analysis of European waste management by Pearce and Dijkgraaf & Vollebergh. The main results of this project relate to both the target setting and the regulatory implementation of waste policy in the EU.

The main recommendations for future waste policies in the EU and Member States are:

- **The waste hierarchy should only be regarded as a general guideline or envelope for policy making.** It is important to bear in mind that what is environmentally preferable is not always a preferred solution, when considered from a relative perspective – the social economic cost-benefit perspective. The reason is that some environmental benefits may come at a comparably socially unacceptable high cost. The marginal costs and benefits will vary depending on material and locality. It is recommended that social costs of new recycling schemes should be analysed and that a critical assessment be made on to determine if further steps are in fact socially desirable.

- Regulation by fixed target setting in waste policy is currently the preferred method of regulation at the EU level. Examples are the packaging waste Directive with fixed target rates for recycling. Instead of current strategies, **price-based policies are recommended, since** they have the ability to change the demand for disposal methods and potentially change the material used towards more sustainable choices. There are two reasons for this recommendation:
    - **There is no upper boundary to the acceptable costs** related to reaching the fixed targets.
    - Fixed targets are **bad at encouraging recycling**, since they have insignificant effects on demand for the environmentally desirable solutions and materials.
- Institutional factors and non-economic reasons can be very important in policymaking and may explain why the European Union prefers fixed target setting such as mandatory recycling targets. However, the empirical analyses in this report suggest a **risk of welfare losses due to fixed target setting policies in EU waste policies.**
- Since waste management is one of the single largest environmental expenses in EU Member States, the potential losses from inefficient policies are very substantial. Equally**, optimal waste policies can potentially free funds for environmental improvements.**

A policy towards fairer pricing in waste policies would be highly recommendable and follow the tendency of the EU strategies for other environmental issues such as transport and energy.

2 Cost-effective Recycling

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2.1 Introduction

People everywhere seem to like recycling. Large numbers of people want their communities to offer recycling programs that can recover a significant fraction of waste. Does economic analysis show that they are wrong? And if not, what does economics have to contribute to our understanding of this popular environmental activity?

Economic calculations can tell us many interesting and important things about waste management and recycling choices, but they cannot provide a precise monetary valuation of the full range of environmental benefits of recycling. As with other environmental policies, the benefits include avoided human deaths, illnesses, and impacts on other species and ecosystems, benefits that are priceless – not infinite in value, but unpriced and unpriceable.[2]

In the case of recycling, the benefits include reduced air and water pollution, and reduced impact on surrounding communities and ecosystems, due to the reduced demand for virgin raw materials. Other benefits include movement toward sustainable resource use, allowing a richer bequest of natural resources to future generations, and a sense of visible, public participation in environmental improvement. (In areas where land is scarce and expensive, the reduced demand for landfill space is another benefit of recycling; however, it is a mistake to view this as the

2 This point is explained in detail in Frank Ackerman and Lisa Heinzerling, Priceless: On Knowing the Price of Everything and the Value of Nothing (New York: The New Press, 2004).

only, or even the most important, benefit.[3]) Although many of the benefits defy monetization, an ambitious and detailed national study of recycling in Australia found monetized benefits clearly exceeded net economic costs for every recycling scenario in the study.[4]

While recycling has valuable, partially unpriced benefits, it also entails economic costs, which vary widely by material and location. Some recycling has always been profitable enough to occur in the private sector. Scrap metals are efficiently collected and recycled by private firms; without such recycling, the volume of scrapped automobiles in a developed country would require vast, unpopular increases in landfill capacity. Large quantities of some paper and paperboard products, such as the cardboard cartons used in retail trade, are also recycled by private initiative. The support for municipal recycling, however, is for something more than this: it is a call for a visible program that reduces municipal waste disposal, beyond the particular activities that are already profitably occurring in the private sector. Thus it is not surprising to find recycling of municipal waste causes a modest increase in the net costs of waste management. In the mid-1990s, a well-run program under typical American conditions increased the net costs of waste management by an estimated $21 per household per year.[5]

2.2 Valuing Recycling

How much should a community pay for recycling? As an alternative to the impossible task of pricing all the separate environmental benefits of recycling, it is possible to ask people what they are willing to pay – and then see if a recycling program can be run for that amount or less. “Stated preference” approaches, asking for an aggregate valuation of a public policy such as recycling, are methodologically

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[3] See Frank Ackerman, *Why Do We Recycle? Markets, Values, and Public Policy* (Washington DC: Island Press, 1997), especially chapters 1 and 10.

[4] National Packaging Covenant Council, “Independent Assessment of Kerbside Recycling in Australia,” 2001, available at http://www.packcoun.com.au/NPC-FINAL-01.PDF

[5] Ackerman 1997, chapter 4.

much simpler than the disaggregated valuation of the numerous individual benefits that result from the policy. In addition, there is a fundamental political argument in favor of stated preferences: in a democracy, public policy *should* be based on citizens' stated preferences.

Economists have criticized stated preferences on two principal grounds: preferences may be based on misinformation about the actual benefits of a policy; and they may reflect the "warm glow" of altruism rather than a valuation of the specific benefits of the policy in question. The first objection is, in theory, easily testable: if more information is provided, do preferences change? While people may be lacking information about the detailed environmental implications of recycling, they are more familiar with recycling programs than with many of the hypothetical options that economists ask about in contingent valuation studies.

The second objection is potentially meaningful for some academic analyses, but not for public policy. The fact that a warm glow of altruism attaches to environmental protection and humanitarian assistance, for example, can and should influence government spending in those areas. Altruism is a policy-relevant attitude that affects political debate; many religious and other perspectives view it as one of the nobler sentiments in the public sphere. Meanwhile, it is politically troubling for experts to claim that their superior understanding allows them to overrule citizens' stated preferences. Experts who believe that the public is wrong have only one recourse in a democratic system: they can try to persuade the public to change its views.

In a number of cases, researchers have asked people how much they are willing to pay for the existence of municipal recycling programs. Answers to this question in the US are generally above the estimated $21 per household per year cost of a typical program: studies in Utah and Tennessee, far from the most environmentally oriented states in the country, find a mean willingness to pay for municipal recycling of $2 - $7 per household per *month.*[6] This suggests that communities with recycling programs are getting something they are willing to pay for, consistent with the observed popularity of recycling. Cutbacks in recycling, motivated by municipal budget crises, have provoked grassroots opposition: both New York City and Washington DC have attempted such cuts, and both cities have ended up restoring recycling in response to popular demand.[7]

In the European context, a study by Pieter van Beukering cites four estimates by other researchers of the willingness to pay for participation in recycling programs, ranging from Euro 20 to Euro 290 per household per year; the mean of the four values is Euro 134.[8] A policy based on this sentiment might first select a specific value as a budget constraint, such as Euro 20 or Euro 134 or Euro 290 per household per year, and then ask, what is the most environmentally beneficial municipal recycling program that can be created within that budget? The lowest of these figures, Euro 20, is broadly comparable with the estimate of US $21 for the net costs of a typical American recycling program, suggesting that it should be possible to run a recycling program at a price that Europeans are willing to pay.

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[6] David Aadland and Arthur J. Caplan, "Willingness to Pay for Curbside Recycling with Detection and Mitigation of Hypothetical Bias," *American Journal of Agricultural Economics*, vol. 85, no. 2, May 2003, pp. 492-502; Arthur J. Caplan et al., "Waste Not or Want Not? A Contingent Ranking Analysis of Curbside Waste Disposal Options," *Ecological Economics*, vol. 43, no. 2-3, December 2002, pp. 185-97; David Aadland and Arthur J. Caplan, "Household Valuation of Curbside Recycling," *Journal of Environmental Planning and Management*, vol. 42, no. 6, November 1999, pp. 781-99; Kelly H. Tiller et al., "Household Willingness to Pay for Dropoff Recycling," *Journal of Agricultural and Resource Economics*, vol. 22, no. 2, December 1997, pp. 310-20.

[7] The suspension of recycling in Washington DC occurred during a crisis of municipal mismanagement that led to severe budget cuts in 1997. In New York, the recycling program was cut back in 2002, and restored in early 2004 – in part because the cutbacks saved much less money than the city had anticipated. See the Natural Resources Defense Council's account, *Recycling Returns: Ten Reforms for Making New York City's Recycling Program More Cost-Effiective* (New York: NRDC, 2004, available at http://www.nrdc.org/cities/recycling/returns/returns.pdf).

[8] Pieter van Beukering, *Recycling, International Trade, and the Environment: An Empirical Analysis* (Dordrecht, Netherlands: Kluwer Academic Publishers, 2001), Table 4.9, p.74.

2.3 Inside the Hierarchy

The much-debated hierarchy of waste management methods, often evoked with the phrase "reduce, reuse, recycle," offers one approach to choices about recycling and other options. Reducing and reusing materials is said to be preferable to recycling; likewise, recycling is preferable to incineration, and even incineration is better than the worst option, land disposal. This hierarchy of judgments should be understood, not as a mantra or a mandate, but as a series of generalizations about relative environmental impacts. Those generalizations, first articulated many years ago, can be tested against contemporary data. The hierarchy can be divided into three statements, of which the second is the most important and controversial:

> 1) Reduction of material use and reuse of existing products are preferable to recycling or waste disposal;
> 2) Recycling (and composting) are preferable to waste disposal; and
> 3) Among disposal options, incineration is preferable to landfilling.

The first statement is normally true whenever it is applicable: using less material, or reusing existing products, generally will have lower environmental impacts than producing either new or recycled materials. However, it is rarely applicable to the practical problems of municipal waste management systems. There are few opportunities to achieve significant reductions in municipal waste volumes through reducing and reusing. Extensive discussion and experimentation in the US has led to only one large-scale success: reduction of yard and garden waste volumes can be achieved through simple, inexpensive backyard composting techniques.[9] In many parts of the US, yard waste is a surprisingly large fraction of municipal waste; it is less significant in most of Europe. Beyond backyard composting, the attempt at organized waste reduction has led only to isolated product-specific or

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[9] The difficulties of measuring waste reduction, and data suggesting that half or more of all US waste reduction consists of yard waste, are discussed in US EPA's *National Source Reduction Characterization Report for Municipal Solid Waste in the United States* (EPA530-R-99-034, available at http://www.epa.gov/epaoswer/non-hw/reduce/r99034.pdf).

technology-specific innovations, such as the reduction in paper use achieved by two-sided office copying and printing, or schemes for the reuse of formerly discarded shipping cartons and containers. Reduction in the quantity of material used to make many products is already occurring in the marketplace, as companies seek to economize on expensive inputs; even more may need to be done in the long run to achieve sustainable patterns of material use. But these trends and objectives are of limited relevance to waste management practices today.

The third statement, the preference for incineration over landfilling, is normally valid for combustible wastes, such as wood, paper, and plastic products. If these products are headed for disposal, it is environmentally preferable to recover energy from burning them, likely substituting for fossil fuel combustion, as well as reducing the need for landfill capacity. The one important exception concerns materials that give rise to hazardous by-products when burned, such as polyvinyl chloride; such materials are better off in a landfill where they can remain inert. For non-combustible wastes such as metals and glass, on the other hand, nothing is gained by incineration. These materials do not yield energy, and are not reduced in volume by incineration. Worse yet, they may be contaminated by contact with other wastes in the combustion process.

Most of the passion and controversy about the hierarchy surrounds the middle statement. Is recycling generally preferable to disposal? The answer is not the same for all materials in all contexts. Imagine a line-up of all common waste materials in order, from greatest to least environmental benefits from recycling. At the end with greatest benefits are most metals, where recycling involves much lower energy use, air emissions, and other environmental impacts than extraction and refining of new materials. Composting of yard and garden waste also provides substantial benefits, except in those cases where excessive use of pesticides and fertilizers makes the resulting compost too hazardous for land application (a problem that is best dealt with by reducing the use of chemicals, rather than changing waste management practices). Recycling of glass leads to modest energy and emissions savings relative to production with virgin materials, although markets for recycled glass are limited.

At the other end of the line, opportunities for real environmental savings from recycling are limited or prohibitively expensive for many complex multi-material products and multi-resin plastic products, newly engineered composite materials, and the less common plastics. Often an effective recycling program could be devised if a homogeneous stream of materials was available in sufficient quantity – but current waste volumes are too small and heterogeneous. For example, it is technologically possible to recycle aseptic packaging, a multi-layer combination of paper, plastic, and aluminum, but the required equipment is expensive. At least in the US (where aseptic packaging is used much less than in Europe) the volume is too small to justify the equipment unless it can be used to recycle other materials as well.[10] The dilemma of plastic recycling is that on the one hand, recycling of mixtures of different plastic resins produces a low-grade, low-value product that has limited uses, but on the other hand, separation and recovery of small quantities of rare plastics is usually not worthwhile.

In the middle are the controversial cases: there are conflicting judgments about the merits of recycling paper products and the most common plastics (PET and HDPE). Paper and cardboard constitute a huge fraction of municipal waste, and the fraction grows as incomes rise. In the highest-income countries, one-third or more of solid waste consists of paper products.[11] Numerous studies have been done of the relative environmental impacts of recycling vs. incineration of paper, with

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[10] See Ackerman 1997, chapter 5.

[11] For US data see "Municipal Solid Waste in the United States: 2001 Facts and Figures" (Washington: US EPA, 2003, EPA530-R-03-11, available at http://www.epa.gov/epaoswer/non-hw/muncpl/pubs/msw2001.pdf). Among other international comparisons, note the relationship between income level and the paper fraction of solid waste in "What a Waste: Solid Waste Management in Asia," p.6 (Washington: World Bank, 1999), available at http://www1.worldbank.org/wbiep/decentralization/eaplib/waste.pdf.

varying results.[12] In terms of energy use, producing paper from wood uses more *total* energy than recycling used paper, but much of the energy in virgin paper production comes from burning wood waste (parts of the tree that are not made into pulp, and process by-products); paper recycling uses as much or more *purchased* energy, and hence may entail more fossil fuel consumption. In terms of climate change impacts, studies that ignore forestry impacts often find roughly equal life-cycle greenhouse gas emissions from paper recycling vs. incineration. However, if carbon sequestration in forests due to recycling is included in the analysis, recycling is a clear winner.[13]

The decision made about paper is crucial to the viability of a recycling program. Send the paper waste to an incinerator, and a community is left with only scattered, niche recycling activities, involving metals, glass, and composting. Send the paper waste to the recycling facility, and the community has a sizeable, multi-material recycling program. In most US recycling programs, paper accounts for well over half of both the tonnage and the market value of the recovered materials. Since the environmental evidence remains contested, and therefore presumably close to break-even one way or the other, why not just recycle the paper? It seems to be what people want to do.

2.4 Cost-Effective Recycling

As a recycling advocate looking at Europe from outside, it is hard to understand why all 25 members of the European Union should have the same recycling targets. There is wide variation within the EU in many factors that affect the design of

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[12] For a review of this literature and summaries of some of the leading studies, see the *Journal of Industrial Ecology* vol. 1 no. 3 (Summer 1997), a special issue on the industrial ecology of paper and wood.

[13] When recycling increases and demand for virgin pulp decreases, forest owners cannot adjust their standing stocks of timber immediately, due to the long lags involved in growing trees; thus there is more sequestration in forests. The climate change implications of recycling are explored in Frank Ackerman, "Waste Management and Climate Change," *Local Environment* vol. 5 no. 2 (2000), 223-229, and Frank Ackerman, "Waste, Recycling, and Climate Change: US Perspec-

an optimal recycling program, including income levels, size and composition of waste streams, proximity to markets for recycled materials, and costs and convenience of disposal options. A comparative study of paper recycling in 25 developed countries (including 18 EU members) found strong positive correlations of recycling rates with population density, and with average incomes.[14] That is, high-income and high-density areas recycle more than low-income and low-density ones. The willingness to pay for recycling, discussed above, presumably is correlated with income as well. As incomes rise in the future, recycling rates should also be expected to rise.

Suppose, then, that nationally appropriate recycling targets were set, and communities were told to spend no more than a fixed sum per household (perhaps based on national estimates of willingness to pay for recycling), to run a multi-material recycling program, including paper, metal, glass, yard and garden waste, and the most common plastics. Can it be done? Some critics of existing European recycling programs suggest that the programs are enormously expensive, and impose huge time requirements on households. This is likely to be a sign of poorly designed programs, and should be addressed by reorganizing and streamlining recycling efforts, not giving up on the whole enterprise. Both advocates and critics of recycling should favor efforts to improve efficiency; the widespread desire to participate in recycling does not imply an urge to spend any more money and time on recycling than is necessary.

Organized municipal recycling is a relatively new activity, and there is a steep initial learning curve in the development of efficient programs. Here the different institutional structure of American recycling programs may have given the US an advantage. Rather than being driven by national mandates, recycling has been implemented in the US on an extremely decentralized basis. In the years when

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tive," in Velma I. Grover et al., eds., *Recovering Energy from Waste* (Enfield, NH, USA: Science Publishers, 2002), 261-269.

municipal recycling efforts took off, from the late 1980s onward, the national government was cutting back on many forms of aid to localities, and provided only modest support and technical advice for recycling efforts. State governments varied in their involvement in recycling, but typically had limited resources available.

As a result, the introduction of recycling occurred through numerous local initiatives. Roughly 9,000 communities now have curbside recycling programs, serving about half the US population.[15] Programs are often staffed by recycling enthusiasts, and have always operated in a budget-constrained environment. The result has been a remarkable process of experimentation and innovation in methods, as recyclers have tried many different program designs, copied each other's successes, and constantly worked on increasing recycling rates and revenues. When New York City's recycling program was restored in early 2004, environmental advocates welcomed it back with a proposal of ten reforms to make recycling more cost-effective.[16] Economic incentives, such as volume-based charges for waste disposal, have played only a small part in the success of American recycling to date.[17] Most of it has occurred without any new market incentives, based only on the interaction of grassroots enthusiasm and tight municipal budgets.

The process of experimentation is continuing, and different models are being used in different communities. Nonetheless, some patterns are emerging; at this point one can describe a semi-standard American recycling program. Most communities have found that curbside collection of recyclables is more effective than asking households to bring their materials to a collection site; the exceptions are often small, rural communities where volumes are too small for cost-effective curbside collection. Recycling collection works best on the same day as waste collection, so

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14 van Beukering 2001, chapter 5.

15 Scott Kaufman et al., "The State of Garbage in America," *BioCycle*, January 2004, vol. 45 no. 1, available at http://www.jgpress.com/archives/_free/000089.html.

16 NRDC, Recycling Returns.

17 This point is argued in Ackerman 1997, chapter 2.

that residents can set everything out for collection at once. The degree of sorting required by households is one of the key program parameters: more household sorting yields cleaner materials that are easier and cheaper to process, but also discourages participation. Many (not all) American recycling programs now ask residents to make only one or two basic separations. In my community, like many others, we separate only three categories: yard waste, recyclable paper, and all other recyclable materials. Multi-compartment trucks pick up these materials and dump the paper in one place, and the other recyclables in another, in a large industrial sorting and processing facility. Yard waste goes to a separate, much simpler composting facility.

Useful foreign models of recycling efficiency are not limited to the US. The Canadian province of Ontario, an early leader in the development of curbside recycling, matches or outdoes the best US programs – and now offers the interesting innovation of negotiated financial contributions from industry, in lieu of more expensive producer responsibility schemes.[18] Other Canadian provinces, including Alberta and British Columbia, may have the world's lowest-cost and most popular beverage container deposit-refund programs, another area where improvement in program design can make all the difference in costs.[19] Australia has moved faster than the US toward mechanizing the collection and processing of recycled materials, and as a result has noticeably lower costs for curbside recycling.[20]

To place this discussion in a broader economic context, municipal recycling is much newer than its competitors in the extractive industries and traditional waste disposal. As often happens in a new, rapidly growing industry, recycling has experienced a steep learning curve and rapid reductions in unit costs. The process of

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[18] See the Recycling Council of Ontario website, http://www.rco.on.ca/, and many sources cited there.

[19] Clarissa Morawski, "Who Pays What: An Analysis of Beverage Container Recovery and Costs in Canada 2001-2002," (Toronto: CM Consulting, 2003), available at http://www.bottlebill.org/geography/canada.htm.

innovation and improvement has not been completed; there is more reason to expect big future cost reductions in recycling than in mature industries. A static cost-benefit analysis of recycling would miss the fact that the costs are dynamic and changing (declining) at a significant but unpredictable rate.

In the end, we are left with a mix of economics, politics, and environmental values. People want to recycle; they perceive it as providing environmental benefits, and are willing to pay for those benefits. A democratic society should respect and implement this desire. Europeans may not be willing to pay the high costs of some of the least efficient existing programs, but the limited available data (more is surely needed) suggests that they are willing to pay the modest costs of a typical American-style curbside recycling program. If new market-based policy instruments can help the process along, more power to them – but the successes of recycling to date have not depended on such policies. It is more important to learn from the best existing models of recycling, and to join in the ongoing adventure of developing more affordable and sustainable technologies for future material use.

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3 Benefit-Cost Analysis and the Waste Hierarchy – US experiences

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3.1 Introduction

The U.S. Environmental Protection Agency (USEPA) has - like the EU - declared a hierarchy of waste strategies: "source reduction (including reuse) is the most preferred method, followed by recycling and composting, and, lastly, disposal in combustion facilities and landfills"(http://www.epa.gov/epaoswer/non-hw/muncpl/facts.htm, accessed 18 May 2004). Since reduction and reuse are never likely to be quantitatively significant options, this hierarchy boils down to the question, how much should we recycle? If waste were like broccoli, we could safely let "the market" decide what, where, and when to recycle. But waste is not like broccoli.

The solid waste market is filled with hidden subsidies and external costs. Subsidies occur whenever a waste disposer receives a waste service for less than its marginal private cost, with the excess cost being covered somewhere else through some other (usually public) source of funds. And external costs – the gap between social cost and private cost – occur throughout the waste market. Waste collection generates noise and traffic congestion; landfilling can generate methane and groundwater pollution; incineration can release mercury and other air pollutants; and more recycling means less virgin material extraction, with concomitant damage to the air, water, health, and esthetics from production in the mining, agriculture, and forestry sectors.[21]

Can benefit-cost (B/C) analysis help us decide what, where, and when to recycle? I think, as do most economists, that it can – and I will outline such a B/C analysis of curbside recycling in the next section of this paper. But not everybody agrees. Dissenters correctly raise many problems with B/C analysis. While hidden subsidies are readily tracked down, external costs are sometimes hard to identify, and even harder to quantify. The expected illnesses avoided and lives lengthened must be estimated and given dollar values and then present-valued, each of which is a philosophically troublesome step. B/C analysis does not easily handle issues of environmental justice or the "precautionary principle" or distant and uncertain events.

B/C dissenters have a point.[22] B/C analysis is not everything, but it is a long way from nothing. Even badly done B/C provides an agenda and a starting point for a discussion of a project or policy. To fully appreciate the value of a B/C analysis to a policy or project discussion, consider a world without it. Decisions would have to be made solely on the basis of emotional allure, media hype, and lobbyist pressure. Compared to these other criteria, B/C analysis has the great virtue of transparency – it is easy to dissect and critique. Moreover, the process of dissection and criticism illuminates -- rather than obfuscates -- the decision process. Charles Schultze put it nicely many years ago:

> *Where complex issues are involved, we must rely on analysis to help. Intuition and goodwill alone will not suffice. It is not really important that the analysis will be accepted by all the participants*

[21] Usually, virgin material extraction uses more energy than does recovering recyclables, so external costs in the energy sector need also to be considered.

[22] Ackerman and Heinzerling, 2004, should be required reading for everyone who does benefit-cost analysis of environmental or health issues.

in the bargaining process. We can hardly expect that information systems will be so complete, necessary assumptions so obviously true, or constraints so universally accepted, that a good analysis can be equated with a generally accepted one. But analysis can help focus debate upon matters about which there are real differences of value, where political judgments are necessary. It can suggest superior alternatives, eliminating, or at least minimizing, the number of inferior solutions. Thus, by sharpening the debate, systematic analysis can enormously improve it.

(Schultze, 1968)

3.2 Benefit-Cost Analysis of Curbside Recycling

It would be nice if we could do one definitive B/C analysis of all the recycling in the industrialized countries and thereby decide, from an economic viewpoint at least, whether it is a good or bad thing.[23] But, as we shall shortly see, the benefits and costs of recycling vary greatly in different parts of countries and in different kinds of cities. So all we can do in a general way is to outline the process of doing a B/C analysis.

Every B/C analysis must start with a list of the relevant benefits and costs and a plan for measuring them in dollar terms. Recycling is no exception.

The major readily measured benefits (+) and costs (–) of recycling are:

1) + the recovery and reuse of recyclable materials;

2) + the reduced use of landfills (or incinerators);

3) + the reduced need for solid waste collection;

4) – the costs of collecting recyclable materials;

[23] Recycling in developing countries is an entirely different matter. There, with low wages and high underemployment, extensive recycling takes place without any government subsidy or encouragement. Indeed, the government policy is often quite the opposite – for example, for many years the government of Indonesia classified scavenger-recyclers with prostitutes, beg-

5) – the costs of processing the materials for reuse; and

6) – the time and space costs to households of separating and storing the recyclable materials. We next look at each of these in turn.

3.2.1 Recovering Materials

Recovering and reusing previously discarded materials are major benefits – after all, this is the main reason why we recycle. The problem is putting dollar figures on the tons of this and the tons of that. The obvious, and the usual, way of valuing recovered recyclables is the revenue gained from their sale. The recovered materials are worth what users of these materials are willing to pay for them.[24]

Market prices of recyclables are, however, not always a full measure of the social benefit of recovered and reused materials. Market prices reflect only the private benefit of the materials to their buyers. For almost all the products we recycle, the recycled material replaces a substitute virgin material, and production of that virgin material requires greater activity in sectors that receive heavy government subsidies or generate great external cost – mining, agriculture, forestry, and energy. Increased recycling means fewer government subsidies (which burden the taxpayers) and fewer external costs (which burden the neighbors) in virgin material production.[25]

Traditionally, economists have argued that market failure should be corrected <u>directly</u> by means of policies in the market where it occurs, not by artificially encouraging a substitute activity. This is fine "in theory" but the market failures in

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gars, and thieves as "urban undesirables" and forcibly relocated them in distant rural areas (Porter, 1996).

24 The price of recyclable materials may vary greatly in different parts of a country since transport costs of recyclables are usually high relative to their value.

25 More precisely, what must be added here are the uncorrected external costs of virgin material production that are avoided by recycling. The correction of external costs becomes a cost to the producers, and that cost is presumably embedded in the price of virgin materials, which in turn raises the price of substitute recycled materials. But the price of recycled materials has already been counted as a benefit of recycling, so adding the external costs of virgin material production would be double-counting the benefit of recycling. Only when the virgin material external costs have been sub-optimally corrected should the benefit of recycling be augmented on this account. I thank the editors for reminding me of this.

mining, agriculture, forestry, and energy are legion, and the U.S. federal and state governments have made very few efforts over the last century to remove the subsidies or correct the external costs in these sectors. If they can be corrected through greater recycling, those benefits should be attributed to recycling. Thus, the reuse benefit of recycling depends partly on the avoided subsidies and external costs of the virgin product it replaces.

3.2.2 Reduced Solid Waste Disposal

Materials that are recycled do not have to be disposed of, and all the social costs of that disposal are avoided. The private cost of that avoided waste disposal is fairly easy to estimate – it is just the volume of waste avoided by recycling multiplied by the price of "tipping" at the landfill (or incinerator).[26] Unfortunately, what we need is the social cost, which exceeds private cost to the extent of the external cost of landfills, and the money "value" of polluted groundwater, odor, litter, greenhouse gases, etc. is not easy to estimate and will be affected by the geography and population of the landfill's neighborhood.[27]

26 In very few industrialized countries is incineration as important as landfilling as a means of solid waste disposal – Belgium, Japan, Sweden, Switzerland (OECD, 2002). Both the private costs and the external costs of incinerators differ from those of landfills.

27 For a fuller discussion of landfill externalities, and a review of efforts to quantify them, see Fullerton, 2002.

"Jobs" Are Not a Benefit of Recycling

Several things are wrong with seeing "jobs" as a benefit. One, the fact that lots of people are needed to carry out recycling programs is basically evidence that recycling is expensive, requiring lots of labor (as well as capital) that could have been used to fulfill other goals of public policy. Two, any jobs created by recycling programs do not reduce unemployment but simply replace jobs elsewhere in the economy. Where these jobs come from we cannot be sure – it depends upon where government spending is decreased or taxation increased when spending on recycling is increased.[28] And three, even if we were sure that jobs were created, and that the national unemployment rate actually went down as a result of a recycling program, we would still have to be sure that recycling was the best way of achieving this outcome.

3.2.3 Reduced Solid Waste Collection

What is collected for recycling does not need to be collected by the usual municipal solid waste (MSW) collection system. There is clearly a saving and hence a benefit here – but how much? The average cost of collection is readily estimated, but the marginal cost is certainly lower than the average cost. The MSW trucks must still drive by and stop at each house even though they pick up less when some is being separately recycled. Even advocates of recycling sometimes admit that there will be little or no saving here due to recycling (Ackerman, 1997, p. 70).

The cost of MSW collection varies widely from city to city, depending on such things as household density, pickup frequency, and wage level. Of course, these very factors also affect the cost of collecting recyclables, so what is interesting is how the sum of the costs of the two kinds of collection change with increased recycling.

3.2.4 Increased Recyclables Collection

It is tempting to throw this category of costs together with the preceding category of benefits and think that they cancel each other out. Alas, they usually do not. First of all, there are all the overhead costs of two collection systems where there used to be only one. More importantly, the variable cost of collecting recyclables is much higher than the variable cost of collecting traditional solid waste. Why the difference? Why is a ton not a ton when it comes to collection?

People are often shocked to discover that collecting recyclables costs some two to three times as much as collecting trash – usually well over $100 per ton for recyclables versus something like $50 per ton for trash.[29] They shouldn't be shocked. Consider the following very simplified story. A city sends out two trucks, similar in capacity (in cubic meters), cost, and crew. One truck picks up 3/4 of the tonnage (the trash) and compacts it to 1/3 of its original size; the other truck picks up the other 1/4 of the tonnage (the recyclables) and doesn't compact it at all. The two trucks fill up at the same time – the recyclables collector has less to load but also does some sorting and quality control. The two trucks then go to the relevant unloading site (assumed to be the same distance for each truck), empty themselves, and proceed again on the pickup route. Their total costs are identical (ignoring time differences at each stop and differences in gasoline usage owing to the differ-

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[28] If new recycling expenditure comes from increased taxation, then it depends upon what consumers would have spent their now-taxed income on.

[29] Throughout, past U.S. dollar figures have been converted to current values using the U.S. GDP deflator, and all Euro figures have been converted to U.S. dollars at the rate of €1 = $1.17.

ent loaded weights of the two trucks), but the trash collection truck is collecting three times as much material. The difference in the cost per ton basically resides in the fact that you cannot compact the recyclable materials (and still hope to sort them later). As a result, while the average cost of collecting recyclables declines as more is collected at each stop, the overall average cost of collecting trash-plus-recyclables rises as ever larger fractions of household trash are recycled – the compaction disadvantage of recycling can never be overcome.[30] Quintupling the recycling rate -- from about 5% to about 25% -- raises the total MSW-and-recyclables collection costs by something like one third.

Not only are recyclables collection costs high, they also vary a great deal across cities, depending on crew size, truck type, volume collected per household, number of different materials collected, average distances between stops, whether public or private collection, etc. To a great extent, the high and varying costs per ton of collecting recyclables is due to the newness of the activity, especially compared to trash compaction, where cost-minimizing collection techniques have been evolving for over a century. As cities and recycling collection truck manufacturers learn, from their own experience and from the experiences of other cities and manufacturers, the cost per ton should come down – but never to the level of trash collection until we find a way to compact recyclables and still sort them.

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30 In one careful study (Stevens, 1994), 60 randomly selected U.S. cities were surveyed in 1993, with the following average costs per ton collected for trash and recycling, given below according to what percentage of the MSW was recycled:

	Cost per Ton of Collection		
Percent Recycled	MSW	Recycling	Weighted Ave.
0-9%	$51	$343	$65
10-19%	64	123	73
20% and over	79	112	88

Stevens' estimates have been converted to current prices, and the weighted average has been taken at recycling percentages of 5%, 15%, and 25%, respectively.

3.2.5 Processing Recyclables

Recyclables are processed at a materials recovery facility (MRF, pronounced *murf*), where the heterogeneous collection is turned into homogeneous and hence marketable bales of recyclables. The MRF has both capital and operating costs.[31]

MRFs are expensive. Inside these unprepossessing buildings, many things are happening: 1) shredding, where mechanical force is used to break the collected materials into small, uniform sizes; 2) screening, where particles of different sizes are separated; 3) air or water flotation, where particles of different densities are separated; 4) magnetism to isolate iron materials; and 5) just plain hand-sorting. The first four operations require costly equipment, and the final operation is labor-intensive. The good news is that, since MRF technology is still quite young, cost-saving (and especially labor-saving) innovations will probably bring the costs of MRFs down a lot in the near future.

How expensive are MRFs? Since MRFs are relatively new, and different MRFs do different things in different ways, cost estimates vary greatly. A sampling of studies turned up cost estimates (converted to current dollars) ranging from just over $30 per ton of processed recyclables to nearly $90 per ton (Francis, 1991; Miller, 1992; Scarlet, 1993; Ackerman, 1997). Almost all such studies, if they look also at MRF revenues, find that the processing cost at the MRF exceeds the revenue gained from the sale of the recovered materials. The typical MRF, seen in isolation (*i.e.*, ignoring the costs of collecting the recyclables), is not <u>currently</u> a commercially viable entity unless it charges tipping fees.

Are larger MRFs cheaper (per ton) than smaller MRFs? We must be careful in answering this question. Doubling the size and throughput of a MRF, while processing the same materials, much reduces the cost per ton of the operation. But increasing the tonnage processed by increasing the number of different materials does not reduce cost per ton. Indeed, there is a hierarchy of recyclables – some are relatively low-cost to process, others relatively high-cost.[32] In short, MRFs display economies of scale but diseconomies of scope.

3.2.6 Household Costs

So far, all the benefits and costs have started with the trash already at the curbside. We have ignored both the costs to the household of preparing the recyclable materials for separate collection and the benefits to the household of participating in an activity that may lead to a more sustainable economy. Some attempt to estimate these costs and benefits.[33] I suspect that these benefits and costs roughly cancel each other and that just omitting them does not much alter the B/C results, but in any case they are not easy numbers to estimate.

3.2.7 Summary

While we have not been able to put firm money figures on most of the benefits and costs discussed in the previous section, it does not look good for recycling's B/C grade. The collection of recyclables is two to three times as expensive per ton as the collection of ordinary trash, so the net there is a big negative for the B/C analysis. The cost of the MRF itself is usually larger than the revenues the MRF earns on its materials sales, so there is another net negative. Avoided landfill costs are not

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31 When the nearest MRF is much further away than the nearest landfill, the MRF also has differential transport costs that must be factored in. In the sparsely populated U.S. Rocky Mountain states, this factor alone accounts for the near absence of recycling there.

32 Studies generally find that paper, cardboard, and metals are low in cost, net of revenues, and plastics are high in cost (Huhtala, 1997; Brisson 1997).

33 Recycling detractors can get some big numbers on this score – costs up to nearly $3,000 per ton of recyclables collected (Tierney, 1996). On the other hand, recycling boosters often decry the effort to put dollar values on the cost of civic duty – "life is not a business, and participation in society is not a reimbursable business expense" sounds great but is no excuse for omitting some of the real costs of recycling when assessing the activity (Ackerman, 1997, p. 13). Others find households so eager to recycle that their willingness-to-pay for it exceeds $40 per capita per year (Kinnaman, 2000). While I think one should count household costs in princi-

large -- in most parts of the United States landfill costs (including their external costs) are not (as yet) very high. A huge estimate for the uncorrected external costs of virgin materials – a benefit for recycling – is necessary if recycling is to carry its B/C day.

In short, recycling probably does not now pay off in a social B/C sense for the typical municipality in the United States. Although the methodologies vary a lot, many, though definitely not all, empirical studies agree that the bottom line on the average city's recycling has been negative in the 1990s (Deyle and Schade, 1991; Specter, 1992; Scarlett, 1993; Curlee *et al.*, 1994; Franklin, 1994; Kinnaman, 1996; Shore, 1997). Not only does the net benefit of recycling vary a lot across cities, but there is evidence that cities somehow recognize this in making their decisions whether to adopt curbside recycling. One study examined 80 towns in (the U.S. state of) Massachusetts, 31 of which had curbside recycling and 49 did not, and found that the towns that did recycle saved on average about $80,000 per year because of the decision to recycle while the towns that did not recycle saved on average about $100,000 per year because of the decision not to recycle (Tawil, 1996).

ple, I find it hard in fact to conceive of a net cost greater than a few dollars per ton of recyclables for a typical American family.

We Should Not Try To Recycle Everything

In the United States, there has been a long and steady trend toward plastic containers and away from glass containers. As a result, roughly half the U.S. glass container factories have closed in the last few decades. This means that demand has declined for "cullet" – *i.e.* used glass that can be cheaply recycled into new glass production – and that cullet has to travel further on average to find a user. Accordingly, the price of recyclable glass has fallen dramatically – often to below zero, which means that MRFs have to pay to have it taken away. Some cities have stopped recycling glass, even though it is technically one of the easiest materials to recycle.

Ann Arbor (Michigan), which is passionately determined not to "backslide" into recycling fewer materials, has agreed to pay a higher tipping fee for its trash at the landfill on the condition that the landfill also recycles our glass.[34] The landfill "achieves" this by using the cullet as landfill cover, which counts (in Michigan and many other states) as recycling. Glass goes onto the landfill but not into the landfill!

Trying to recycle everything regardless of cost raises the average cost of recycling and ultimately will risk a backlash from citizens and cities facing budgetary stringency.

So, the answer to the question, does recycling pass its B/C test is yes and no – it depends on what kind of municipality we are considering. In many places in the

[34] The only material that Ann Arbor has ever stopped recycling is textiles.

United States, recycling already passes a B/C test.[35] Refuse and recycling collection costs, as well as tipping fees, vary greatly across U.S. municipalities, and these costs are a critical ingredient in the B/C analysis of recycling (Apothekar, 1993; Tawil, 1996).

Recycling will pass in more and more cities and towns as time goes on. Recycling is still in its "shakedown" phase. The average national costs of collecting and sorting recyclables will come down over time as cities learn how to operate their recycling program in less costly fashion and as cities discard recycling techniques that they learn are dominated by other, better techniques. Moreover, markets for recyclable materials and for products using recyclable materials are also in their infancy, but they are growing, partly in response to the growth in the supply of recyclable materials. Furthermore, the time when recycling passes the test will be nearer because we are already recycling now. In a sense, we should count the future net benefits of cost-effective recycling as one of the major benefits of our investments in recycling today.

3.3 Good Policies That Encourage More Optimal Recycling

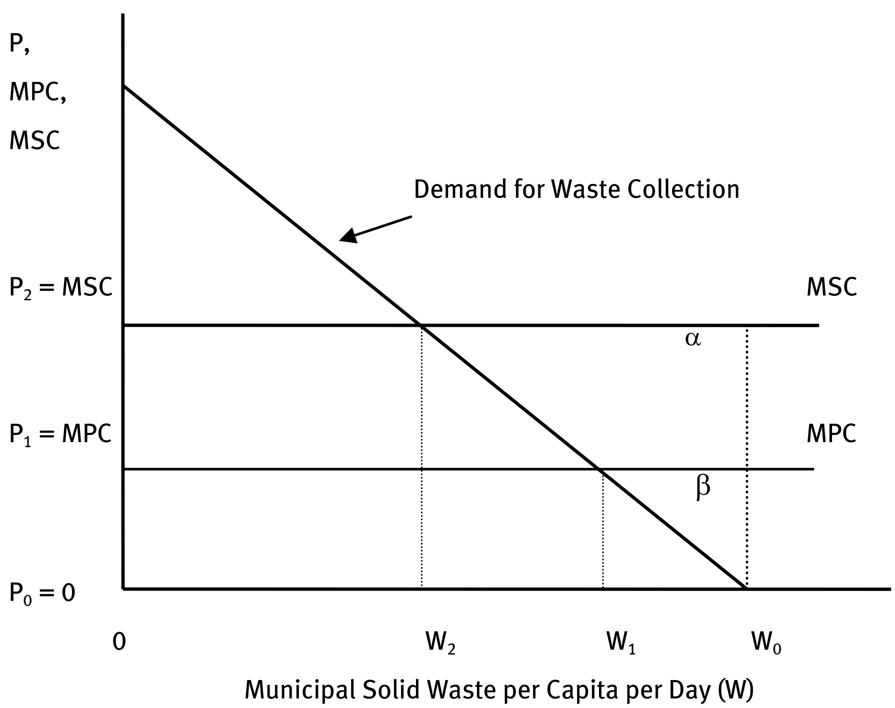
Market failures abound in waste handling. Too many of the actors in the waste market are reacting to prices that are distorted by hidden subsidies and external costs. A simple diagram lets us add precision to this general point. Figure 1 shows three lines, the demand curve for municipal solid waste collection and the marginal private cost (MPC) and marginal social cost (MSC) of collecting it. (Here, for simplicity, the downward-sloped demand curve is assumed to be a straight line, and each cost is assumed constant.) Often, the price of such collection (P_0) is zero, and households react to that price by creating a large volume of waste (W_0). If the price of waste collection were raised to P_1 (equal to MPC), less waste would be

35 In many municipalities where recycling fails a benefit-cost test, it could pass if fewer materials were recycled. When a material is costly to collect and sort, and its market price is low, it should not be recycled just because "it is there" and sufficient municipal subsidization is available.

produced ($W_1 < W_0$). And if the price were raised still further to P_2 (equal to MSC), even less waste would be produced ($W_2 < W_1$). Indeed, W_2 is the optimal amount of waste -- for any waste in excess of W_2, households are not willing to pay as much as the marginal social cost of collecting and disposing of the waste.

Pricing waste at zero instead of the optimal price (P_2) creates a deadweight loss (DWL) equal to the excess of all the social costs over the WTP of households – in Figure 1, the DWL is measured by the sum of the two areas marked α and β, which show the total amount by which MSC exceeds WTP in the range of prices between P_2 and zero.

Figure 1 Deadweight Loss of Underpricing Waste



Good policies to encourage recycling are those policies that remove the price distortions. Three such policies are worth a quick look:

1. Advanced Disposal Fee (ADF). Manufacturers often have no reason to consider the cost of disposal or recycling of their packaging. An ADF on

manufacturer's packaging would make them aware of the disposal/recycling problem. Many different ADFs are suggested by theory, but the one I prefer is a fee on packaging waste that depends upon the cost of recycling the packaging (Porter, 2002 and 2004). Packaging that is costly to collect and sort or earns little revenue as recycled material would carry a high ADF; packaging that is easy to collect and sort or is highly valued by recyclers would carry a low ADF. Table 1 gives some examples of ADFs that are being applied in practice.[36]

Table 1: ADFs in France and Germany, 2002 (in U.S. dollars per kilogram)

Material	France	Germany
Glass	0.0039	0.0889
Steel	0.0241	0.3346
Aluminum	0.0482	0.8962
Paper/Cardboard	0.1299	0.2387
Plastic	0.1892	1.7644

Source: Porter, 2004, p. 128.
Note: These are the weight-based ADFs. There are also small per-unit fees and some rebates.

2. Marginal Trash Charge (MTC). Households also have no reason to worry about their trash volumes as long as the marginal cost of disposing of it is zero. An MTC makes households aware that MSW disposal is not free. MTCs take many forms, the marginal charge may be per bag or per barrel, and the charges may be by volume or by weight.[37] And there is no single correct MTC. I prefer a MTC that reflects the excess cost of landfilling (or incinerating) the waste over the net cost of recycling it – because that is

36 The French fees are not supposed to reflect net recycling cost, but rather the amount by which the cost of developing packaging recycling exceeds the cost of traditional waste management. The German ADFs reflect the total waste management cost – *i.e.* net recycling cost if the material is 100% recycled or collection/landfill cost if the material is 100% landfilled. As a result, the German ADFs are 2-20 times higher than the French ADFs. As examples of the burden of these ADFs in Germany, a (75 centiliter) glass wine bottle would pay about $0.04 and a (mostly) plastic pail would pay about $0.50.

37 MTCs also go by other names – frequently, pay-as-you-throw (PAYT) and unit-pricing.

the extra cost the household imposes on the waste system when it puts waste out as trash rather than for recycling.[38]

3. Landfill Tax. The external costs of landfills still exist, though they are much smaller with liners, monitors, methane flaring, and post-closure vigilance. Nevertheless, the remaining external costs should be "internalized" through taxes on landfill use. These taxes will be passed on into higher trash charges on households, businesses, and cities, and this will presumably induce them to seek ways to reduce, reuse, and recycle in order to avoid the higher landfill cost.

3.4 Bad Policies That Encourage More Recycling at Any Cost

Price-based policies, such as those suggested in the previous section are good because they place a ceiling on the cost of recycling. If you have a choice between recycling something or paying a tax of say ten cents, then you will recycle only if the cost of that activity is less than ten cents. With quantity-based policies, there is no ceiling on the cost of compliance. Consider three quantity-based policies that are commonly implemented:

1. Producer Take-Back Responsibility. With this scheme, producers are required to physically re-collect the packaging on their products (and usually also required to recycle it, though incineration is sometimes considered acceptable). Where this take-back responsibility has been demanded, the affected firms usually band together to create a single firm that will pick up and recycle all their packaging. This greatly reduces the

[38] MTCs, however, may not be appropriate under many circumstances for a variety of reasons: 1) if illegal disposal is expected to become a serious problem; 2) if administrative and monitoring costs of an MTC system are high; 3) if income distribution considerations are important, and it is impossible to organize an MTC system that does not seriously increase the tax burden of the poor; and/or 4) if multi-family dwellings dominate the municipal landscape, so that the dumpsters/skips behind these apartment houses essentially become a "commons" where each resident has almost no personal incentive to reduce his or her trash (Hardin, 1968).

cost, but it does not limit it. Moreover, since the take-back scheme adds a new waste collection system to the already existing municipal waste and recycling systems, it is inevitably expensive. Estimates of the cost of the German "Green Dot" take-back law always run to at least several hundred dollars per ton – one OECD study concluded that the Green Dot costs "effectively approach the costs of handling a tonne of hazardous waste" (OECD, 1998, p. 33). Is recycling a little more paper or plastic really worth as much to us as the careful handling of hazardous waste?

2. Landfill Ban. When the use of landfills for waste disposal is forbidden for certain items, waste generators are forced to find an alternative means of disposal of these items – export, incineration, reuse, reduction, maybe even recycling. The generators will presumably choose the least expensive alternative, but the ban stands regardless of how expensive is this least expensive alternative.[39] Forcing an expensive disposal on items in order to encourage more recycling is a roundabout and wasteful policy.

39 When the construction of new landfills is banned, there is little effect until the remaining landfill capacity nears zero, when in effect the ban on landfill construction becomes a ban on putting anything in landfills.

Mandatory Deposits on Beverage Containers.

Throughout North America and Western Europe, many governments have attempted to return to refillable, reusable beverage containers by requiring bottlers and brewers to demand a deposit from their customers that is redeemed when the customer returns the empty container. This deposit system has everywhere failed to reverse the trend away from potentially refillable glass containers toward one-way, non-refillable aluminum and plastic containers. Mandatory deposits have, however, provided other benefits -- reduced litter, reduced landfilling, and increased recycling of beverage containers (and hence reduced extraction of virgin materials). The question is, at what cost? Most studies suggest that the cost may well be thousands of dollars per ton of containers, far too high for the benefits achieved.

Again, we see that a quantity-based regulation provides no ceiling on the cost.[40] All this could easily be remedied if beverage producers were given a choice of two systems: 1) the standard mandatory deposit system; or 2) a tax of say five cents per container, which frees the producer of any obligation to handle deposits or re-collect the empty containers. Producers who find the tax cheaper will choose it.[41] But that means that neither system will cost more than $2,500 per ton to operate.[42] Adding a price-based alternative to mandatory deposits limits the possible cost incurred.

40 Mandatory deposits are price-based to consumers, who are given monetary incentive to return (recycle) containers, but they are quantity-based to producers, who must re-collect and recycle all returned containers.
41 More precisely, the tax must be compared to the cost of re-collecting the empty container, minus the revenue earned from the sale of the recyclable material, plus the inconvenience cost experienced by the customers who must return empty containers.

3. Recycling Target. In the United States, a popular way to encourage recycling is for the states to mandate a target recycling rate, which municipalities are expected to meet by a certain date. Most U.S. states have set such targets, apparently with little concern for cost or benefit. For example, we should expect that the more urban, densely populated states would be sensibly doing (or planning to do) greater recycling, but as Figure 2 shows, there is little relationship between state recycling targets and population densities.[43] Moreover, as Figure 3 shows, there is little relationship between currently achieved state recycling rates and targeted recycling rates, which suggests – correctly – that the targets are weakly enforced.[44] The point, however, is that not all cities in a state are identical, and that almost any target chosen will be too high for some and too low for others. When a city is forced to meet a target that is too high, deadweight loss occurs as the marginal cost of recycling exceeds the marginal benefit. And when a city fails to exceed a target that is too low, deadweight loss also occurs as the marginal benefit of foregone recycling exceeds the foregone marginal cost. Are the cheerleading values of recycling targets worth possibly large deadweight losses?

42 There are approximately 50,000 beverage containers to the ton, and $0.05 * 50,000 = $2,500.

43 Source for state recycling targets: www.afandpa.org/Content/NavigationMenu/Environment_and_Recycling/Recycling/State_Recycling_Goals/State_Recycling_Goals.htm (accessed 18 May 2004). Some states target waste reduction rates, others recycling rates (which are not quite the same thing -- see Porter, 2002, p. 158)

44 Source for actual 2002 state recycling rates: Kaufman et al., 2004 (2000 rates used for Alabama, Alaska, and Montana, for which 2002 data are not available.

Figure 2 Relationship of State Recycling Target to Population Density

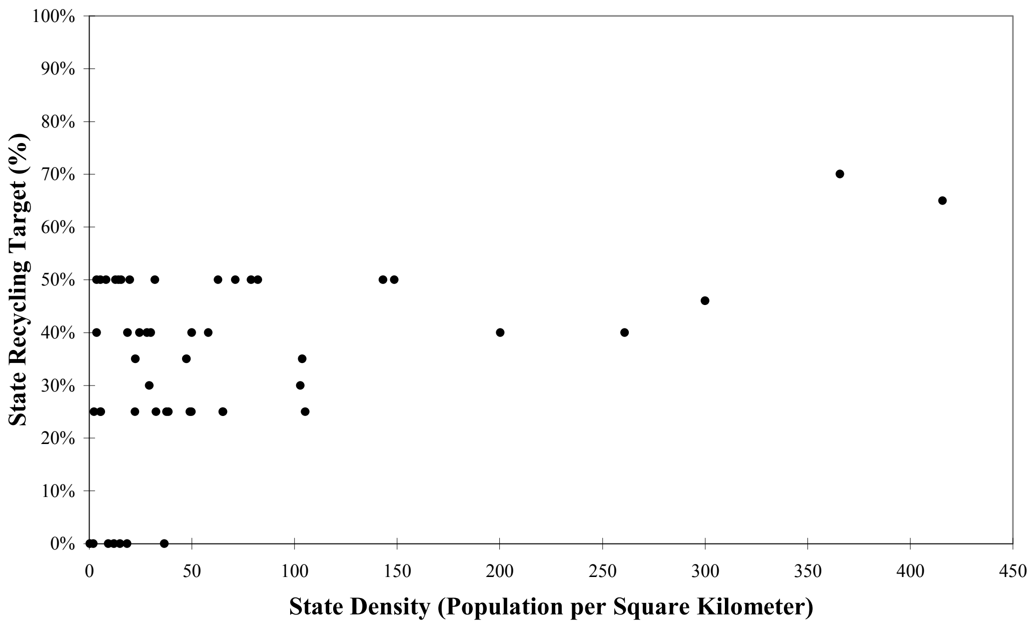
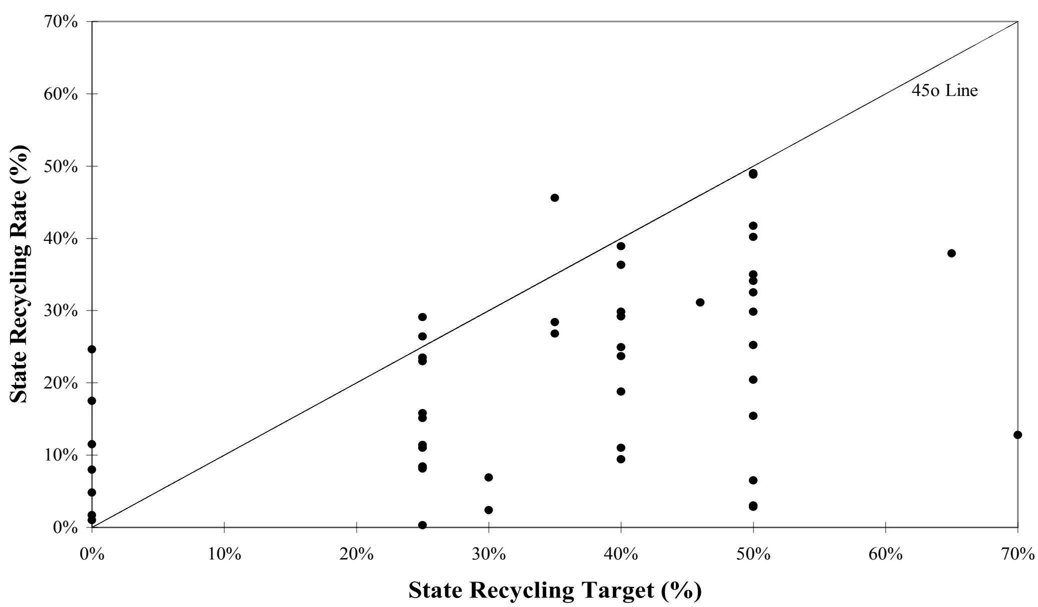


Figure 3 Relationship of Actual State Recycling Rate to Target Rate



3.5 Conclusions

Recycling is a good thing, but not at any cost. And quantity-based regulations to spur recycling – such as producer take-back, landfill bans, mandatory recycling targets, and mandatory deposit systems – can be very expensive. Price-based systems are better because they avoid both of the two serious defects of quantity-based systems.

The first defect arises from the fact that all non-price policies try to get people to do things that are not in their personal economic interest to do. Producer take-back requirements do not make producers want to take back their packaging. Landfill bans do not make people prefer recycling to landfilling. Price-based policies essentially say, you do what you want, x or y, but if you do x there is a fine or tax – many people will suddenly want to do y.

The second defect is that there is no limit to the inefficiency that a badly chosen quantity-based intervention can cause, while price-based policies self-limit their damages, no matter how badly chosen. An example may illustrate this. Suppose cardboard is the packaging material that is cheapest to recycle. With a price-based policy, they would receive a relatively low ADF, and manufacturers would be greatly encouraged to use cardboard for their packaging. Manufacturers who could easily switch to cardboard would quickly make the switch in order to reduce their ADF costs. But those manufacturers who needed to package with plastic (for the security, safety, or sanitation of their product) would not switch and would pay the higher ADF on plastic. A quantity-based regulation, such as a ban on packaging other than cardboard, does not make this distinction between the different packaging needs of producers – or if it tried to, would have to make the distinction by means of a long series of cumbersome, bureaucratic, case-by-case procedures. This example, though simple, captures the essence of the difference between price-based and quantity-based policies.

If price-based policies are superior to quantity-based policies for correcting waste externalities, it is curious that they are so little used. I have four suggestions as to why quantity-based policies are so often preferred in real-world policymakers: 1) they are easier for policymakers to apply – one can ban or require something without the need for a difficult empirical investigation into things like marginal benefits or marginal costs or optimal prices or proper taxes; 2) non-economists can more easily comprehend quantity-based policies because they act directly on the policy goal – mandating that recycling increase by a percent makes more immediate sense than, say, raising the price of trash disposal by b percent; 3) many waste professionals and policymakers do not believe there is price elasticity in the waste decisions of manufacturers or households, which would mean that changing prices would not much affect behavior; and 4) every quantity-based policy hides the cost (as well as the benefit) of the policy – no explicit tax or higher price is imposed on anyone.

Can economists overcome the easy virtues of quantity-based environmental policies? The cynic accepts Paul Krugman's words: "So now you know why economists are useless: when they actually do understand something, people don't want to hear about it" (Krugman, 2000). The optimist continues to preach.

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4 DOES EUROPEAN UNION WASTE POLICY PASS A COST-BENEFIT TEST?

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4.1 The European Union and cost-benefit analysis

The European Commission is committed to applying *some* form of cost-benefit test to its Directives. In the context of environmental legislation, Article 130r of the Treaty on European Union (1992) requires that:

> 'in preparing its policy on the environment, the Community shall take account of – available scientific and technical data, environmental conditions in the various regions of the Community, *the potential benefits and costs of action or lack of action*, and the economic and social development of the Community as a whole and the balanced development of its regions' (italics added).

There is no implication in Article 130r that Directives need pass a cost-benefit test for each and every Member State affected by the Directive, nor that the comparison of costs and benefits should take the form that economists would regard as conforming to cost-benefit analysis, CBA. In the former context, unless natural resource endowments and human preferences are identical across all Member States, what is economically efficient in one country need not be economically efficient in another country. In the latter respect, the Commission has conducted, and continues to conduct, cost-benefit analyses of some Directives, but not others. In other contexts, the Commission relies on individual Member States to conduct a CBA and then 'borrows' or cites the analysis if it supports the Directive.

In an earlier paper, Pearce (1998) surveyed the extent to which environmental Directives were subject to any form of formal appraisal – whether CBA, cost-effectiveness, multi-criteria or some form of environmental impact assessment. The general finding was that up to around 1990 very few formal appraisals were conducted. In the 1990s formal appraisals increased primarily in the sphere of water pollution and air pollution, but they vary significantly in quality and in the extent to which they provide a clear comparison of costs and benefits. It seems clear that the presence of Article 130r in the Treaty of Union has acted as some kind of spur to conduct formal analysis. It remains the case, however, that procedures are far from systematic, as the review of more recent Directives and regulations below shows.

4.2 CBA and the waste hierarchy

The waste hierarchy refers to a ranking of waste management options according to their desirability. Desirability is interpreted to mean environmental desirability, i.e. the waste hierarchy makes no reference to financial costs of disposal. This immediately limits its validity as a waste management tool - financial costs are important and both environmental and financial costs need to be included in any economically rational waste policy. The waste hierarchy as it is widely interpreted within the European Commission is follows:

1 Source reduction
2 Re-use
3 Recycling
4 Composting
5 Incineration with energy recovery
6 Landfill with energy recovery
7 Incineration without energy recovery
8 Landfill without energy recovery

However, the basis for the ranking has never been clear and it appears to have emerged as a 'consensus' ranking, although it is also not clear across which parties the consensus is supposed to apply. Yet it is possible to derive a hierarchy based on economic analysis of the baseline mixes of management options. The ranking can be derived by measuring the (marginal) net social cost of each management option. *Net* social cost here refers to the financial plus environmental costs of the option, minus any associated benefits. The management option with the lowest net social cost will be the most preferred, and so on. It is important to note that the ranking may well change as the quantity-mix of waste going to the various options changes. This is not widely understood. Essentially, as more waste goes to, say, recycling, so the financial costs of costs may rise at the margin. As recycling costs rise, so other options may then have lower marginal net social costs. Hence the ranking will not be invariant with the mix of waste going to each option. The idea of a waste hierarchy that is fixed and immutable is therefore erroneous. As noted above, an additional observation is that the hierarchy will not necessarily be the same for every location: both financial and environmental costs will vary according to where waste is sent.

One attempt to estimate the waste hierarchy based on these social cost principles, which are simply CBA expressed slightly differently, is Coopers and Lybrand et al. (1996). While broadly supporting the waste hierarchy, there are some important differences between the social cost ranking and the ranking based on the conventional wisdom.

First, landfill is not ranked the least desirable option once *both* financial and environmental costs are included, as they should be. Much depends on what incineration does by way of displacing other energy sources. A modern incineration plant generates energy (heat and/or power) which can displace low merit order electricity plant which is itself polluting. Hence incineration may have significant environmental benefits as well as some environmental costs. If the plant displaces ‘average’ generation plant elsewhere, the environmental benefits of incineration are significantly less. Overall, the Coopers and Lybrand et al. study suggests incineration could be more expensive in net social costs terms than landfill, reversing the waste hierarchy ranking as far as these two management options are concerned.

Second, composting, highly ranked in the conventional hierarchy, is unattractive on the social cost approach. By and large this is because it is financially more expensive than other options.

Third, re-use is limited to only some specific materials and is not applicable to others. This is reasonably consistent with the conventional waste hierarchy because re-use has always been interpreted as ‘re-use where possible’.

Fourth, the hierarchy needs to be differentiated by type of waste. Whilst the Coopers Lybrand report finds that recycling is indeed the second best option (after source reduction which effectively removes the entire environmental life cycle of the other management options), the environment benefits of recycling some plastics can be negative, i.e. there is an environmental cost to such recycling.

Overall, whilst the conventional hierarchy is shown not to be wildly inaccurate, the general message of social cost analysis is that the hierarchy will vary by waste type and perhaps by country as well. Where there is geographical variation in overall net social costs, policy should be location-specific which, as is well known, runs counter to the requirement of environmental harmonisation with in the EU.

4.3 Are waste targets economically rational?

Whatever the rationality or otherwise of the waste hierarchy, a separate but linked issue relates to the targets set within the European Union for waste recycling, reduction in landfill, etc. Again, an economic approach would require that such targets be set according to the balance of costs and benefits. Formally, the requirement is that the target should be set where marginal social benefit equals marginal social cost. Thus, a recycling target should be relaxed if it implies that the extra cost of recycling exceeds the extra benefit, and strengthened where marginal benefits exceed marginal costs. There is no evidence to suggest that any of the waste targets in the various EU Directives have been determined on this basis, although there is one example where the Commission may have believed it was basing its targets broadly on such an analysis. Unfortunately, the analysis in question has no rational basis, as we see shortly. In what follows we discuss several examples of waste policies and subject them to a cost-benefit test. Such tests are self-evidently limited in scope since full appraisals would require significant resources. The analysis focuses on UK experience simply because Regulatory Impact Analysis of environmental policy is required in the UK, and this has helped to generate some data and information.

4.4 The Packaging and Packaging Waste Recovery Directives

The 1994 Packaging and Packaging Waste Directive requires that Member States recycle and recover specific percentages of packaging waste arisings. Recycling refers to reuse of the collected material in making further packaging, whilst recovery includes both recycling and the incineration of packaging waste with energy recovery. The Directive has been extended, with further, stricter targets being proposed for achievement by 2008. Table 1 summarises the various targets. The 2008 targets were originally envisioned for 2006 but there were delays in revising the Directive. The new targets were effectively agreed in 2003.

Table 1 European Union recovery and recycling targets for packaging waste

	1994 Directive targets for June 2001 % waste stream	2003 Revised Directive targets for 2008 % waste stream
Overall recovery target	50-65	60
Overall recycling target	25-45	55
Materials specific recycling targets:		
Paper	15	60
Glass	15	60
Metal	15	50
Plastics	15	22.5

In May 2001 the European Commission released a cost-benefit study that appeared to lend support to the Commission's proposals for revised targets for 2006 under the Directive (PIRA- RDC, 2001). This report used a combined life-cycle analysis and cost-benefit analysis procedure to estimate 'optimal' recycling rates. The procedure is one of estimating the life-cycle impacts of various recycling options, including options relating to different collection schemes. The various impacts are then given monetary values based on the economic valuation literature. The suggested range for optimal recycling of all household plus industrial packaging waste in the whole of the EU was 50-68%. The ranges vary by country, as one would expect, since the optimum is determined by the financial costs of collection and reprocessing and the external (environmental) costs. For the UK, for example, the overall range is 49-69%. The ranges for individual materials are given only generally for the EU and are shown below.

PIRA-RDC	
Plastics	28-38%
Steel	60-75%
Aluminium	25-31%
Wood	47-65%
Paper	60-74%
Glass	53-87%

By and large, it can be seen that the ranges in the PIRA-RDC report support the suggested targets that the European Commission proposed and which were subsequently adopted with some revisions. On the face of it, the European Commission actually set targets that were at least informed by the CBA. But how valid is this cost-benefit study?

To appreciate why the RDC-PIRA study cannot be used to determine optimal recycling rates, it is necessary to state the conditions that should prevail for any amount of recycling to be declared 'optimal' in a cost-benefit sense. Taking a very simple model in which there are just three options, recycling, landfill and incineration, the optimality condition is that the marginal net social cost of landfill must be equal to the marginal social cost of incineration which must be equal to the marginal net social cost of recycling. Only if this condition is met are social costs minimised for the system as a whole. These conditions are implied, though not clearly stated, in the PIRA-RDC report. But both financial and external costs vary with the amount of waste. Generally, as more and more collection takes place so the marginal (i.e. extra) cost of collection will rise. The same may be true of reprocessing costs, although it is possible that there are some economies of scale in the recycling industry. If the marginal social cost of recycling was always below the marginal net social cost of landfill, then all waste should be recycled and none landfilled. The reason that an overall waste management system will contain a mix of recycling and other options is because the marginal costs vary with waste throughput. An optimal recycling rate therefore can only be identified if we have knowledge not just of unit costs, but of the *cost function*. Unfortunately, information on such functions is not available in the PIRA-RDC report, nor could it be easily obtained. What the report does is to 'optimise' across mixes of waste management options. That is valuable and goes some of the way to determining optima, but it does not resolve the problem of what an optimal recycling rate is. Overall, then, the cited optimal recycling rates in the PIRA-RDC report are artefacts of a methodology that is far from clearly stated, but which certainly does not embrace the requirement that cost *functions* be estimated.

The UK government (DEFRA, 2001) produced a cost-benefit appraisal in its Regulatory Impact Assessment of the UK Producer Responsibility scheme, which is designed to achieve the EU targets. But the appraisal is not satisfactory. What should be estimated is the monetised benefit of diverting waste from landfill and incineration to recycling. The Regulatory Impact Assessment (DEFRA 2001, Annex 6) had some curious logic. For example, it argued that a strategy has benefits because it induces the infrastructure necessary to achieve the targets. This is clearly not a benefit, but a cost. On benefits, DEFRA remarks that 'It is difficult to value the environmental benefits of a reduction of the environmental impact of packaging waste' (Para 23, Annex VI, DEFRA 2001). While it is indeed difficult, it is not impossible, as we show below.

DEFRA (2002) estimates the direct costs of compliance with the *original* Directive as follows:

£ 78 million
£ 35 million
£ 36 million
£ 70 million

giving a total of £219 million (undiscounted). Taking the base year to be 2001, the rate of inflation to be 2% p.a. and the social discount rate to be 3.5%, the £219million converts to £233.7 million at 2001 prices and with 2001 as a base year. DEFRA (2002, p.160) estimates that compliance with the Directive has resulted in the recovery of a cumulative additional tonnage of material of 3.16 m. tonnes of material. Hence, the per tonne cost of this recovery has amounted to some £74 per tonne for the UK.

There are various estimates of the externalities from landfill and incineration in the UK. The relevant costs are (a) the environmental costs linked to throughput, and (b) disamenity costs[45]. The externalities will clearly vary with the environmental standards associated with the disposal method. Table 2 brings together some estimates of externalities associated with these options. The recycling cost of £74 per tonne of waste was derived above. Disamenity costs are not shown separately, but there is evidence from the UK that Table 2 may well exaggerate the implied disamenity costs of £8-10 tonne waste: Cambridge Econometrics (2003) places UK landfill fixed disamenity costs at some £2 per tonne waste.

[45] We ignore collection costs, making the broad assumption that they will be similar whether the materials are recycled or sent to landfill and incineration. Again, a more careful study would need to test this assumption.

Table 2. External costs from landfill and incineration: European Union average. £ per tonne, 2001 prices

Option/Scenario	Study	'Best'	'Medium'	'Worst'
Incineration, *including* disamenity Incineration, *excluding* disamenity	COWI (2000) Brisson and Powell, 1995	-28.7 -17.5	24.7	51.3
Landfill, *including* disamenity Landfill, *excluding* disamenity	COWI (2000) Brisson and Powell, 1995	7.3 - 0.5	n.a	13.3 4.0 - 5.0
Recycling private costs (see text), excluding disamenity			74.0	

Notes: 'best', 'medium', 'worst' refers to the environmental restrictions on disposal. Thus, a 'best' incinerator complies with the latest EU Directive on incineration practice. The negative values (i.e. negative costs = positive benefits) arise when allowance is made for energy recovery which displaces marginal electricity production. Disamenity costs are taken from US studies and greatly exceed those found in Cambridge Econometrics (2003) for the UK – see text. The values shown in this table are also generally consistent with those in Coopers and Lybrand et al. (1996).

Taking the relevant mix of landfill and incineration disposal in the UK to be 95% landfill and 5% incineration, and using the 'best' estimates in Table 2, the avoided externalities would range from £6 - 7 per tonne. In addition to the avoided landfill externalities, recycling will save the costs of landfill 'beyond the gate', i.e. the actual operational costs of landfill excluding any collection costs. In the UK such costs are put at £30-50 per tonne waste (DEFRA, 2003).

Recycling would thus cost £74 per tonne of waste, but would generate environmental benefits of £6-7 per tonne, plus perhaps £40 per tonne saved landfill costs. This indicates that the 1994 Directive fails a cost-benefit test by a significant margin: the benefit-cost ratio is, say, 47/74 = 0.6.

In 2003 DEFRA changed its position on the comparison of costs and benefits, arguing that they could now be compared in quantitative terms (DEFRA 2003). Using the same procedure as that outlined above, net benefits for various options within the overall 2008 targets were evaluated. These options primarily relate to the 'loading' of the reductions in time, i.e. when the reductions occur, and to different wood and metals targets. This Regulatory Impact Assessment concluded that benefit-cost ratios were of the order of 0.62 to 3.25, i.e. 'on average' there would be a positive cost-benefit ratio and the revised targets would be justified. However, the critical issue in this cost-benefit comparison is the 'social cost of carbon', i.e. an estimate of the marginal (global) damage done by one extra tonne of carbon (equivalent) released now. The comparison with 'low' costs of carbon produces the below unity benefit-cost ratios, and the 'high' costs of carbon produce the upper ends of the range of the benefit-cost ratios. The problem lies in the range adopted by DEFRA for the social cost of carbon, some £35 to £140 per tonne carbon. In extensive reviews Tol (2004) and Pearce (2004) find that the *lower* end of this carbon damage cost range is above that indicated in the damage cost literature. That is, even a figure like £35 (say $60 per tonne carbon) cannot be justified. If this is correct, then the results in DEFRA (2003) show that the revised Packaging Directive targets also fail a cost-benefit test, contrary to the conclusion reached in that document.

The argument here is that both the original and revised Packaging Waste Directives fail an economic efficiency test. The original Directive was not accompanied by a cost-benefit analysis. The cost-benefit analysis produced for the European Commission in 2001 clearly gave support for the revised recovery and recycling targets, but, unfortunately, the study is deeply flawed and fails a test of competent CBA. Thus, while the European Commission honoured its obligation under Article 130r of the Treaty of Union to compare costs and benefits, the study they argued supported its new targets should have been peer-reviewed. Perhaps more curiously, the UK government, with its tradition of RIA, systematically avoided estimating the benefits of the original Directive, only to make at least an honest attempt at it for the 2008 targets. However, this later attempt founders on non-credible estimates of the marginal social cost of carbon.

4.5 The Landfill Directive

The Landfill Directive was under discussion for nearly ten years before its introduction in 1999 (Council Directive 1999/31/EC). The Directive seeks significant reductions in the amount of biodegradable municipal waste being sent to landfill, and also bans some materials altogether - e.g. tyres. For each Member State, by 2006, the biodegradable fraction of municipal solid waste must be less than 75% of the 1995 level. By 2009 the fraction must fall to 50% and by 2016 it must fall to 35%. The UK has availed itself of a four-year derogation under the Directive, so that the relevant dates are 2010, 2013 and 2020. The assumption must be that the financial costs of disposing of this waste are currently minimised by sending the waste to landfill. Hence the financial costs associated with the Directive are equal to the quantities of the individual waste types multiplied by the difference between the cost of their new disposal method and landfill. The benefits are the environmental benefits that will ensue and these will be equal to the avoided environmental damage from landfill and the environmental damage from incineration. In short, the comparison is between:

(a) the total financial cost of diverting landfilled waste to achieve the Directive's targets:

$$TFC_{DIV} = Q_{DIV}.(C_i - C_l)$$

and

(b) the environmental benefits:

$$TEB_{DIV} = Q_{DIV}.(E_l - E_i)$$

where TFC is total financial cost, TEB is total environmental benefit, Q is the quantity of waste diverted from landfill, C is unit financial cost per tonne waste, E is unit environmental damage (in money terms) per tonne waste, subscript l is landfill and subscript i is incineration. Since Q_{DIV} is common to both benefits and costs it can be ignored.

No CBA appears to have been performed on the Directive at EU level. The UK has undertaken several Regulatory Impact Assessments for different aspects of the Directive. DEFRA (2000) contains a Regulatory Impact Assessment of the UK's *Waste Strategy* which encompasses a wider range of actions than complying with the Landfill Directive alone. Fortunately, some effort was made to separate out the costs of meeting the Directive. The prime assumption in the modelling used in the Regulatory Impact Assessment is that the Directive's targets are met primarily by diverting landfilled waste to incineration (Mixes 1 and 2 in the DEFRA model). The basic finding is that C_i in the above equations is £71 tW, and C_l is £45 tW (inclusive of collection and transportation costs). Hence the relevant financial cost of meeting the Directive is £71-45 = £26 tW.

CSERGE et al. (1993) estimated the externalities from UK landfill as shown in Table 3. As well as estimating landfill externalities, the study also estimated externalities from incineration. The rationale at that time was that the information might be needed for an incineration tax. In the event, the incineration externalities were negative because of the effect of establishing new incinerators on the displacement of inefficient coal-fired power stations in the UK electricity system. In other words, electricity from incineration displaces electricity elsewhere in the system and hence 'saves' the pollution associated with the displaced electricity. A similar assumption was made for new landfill sites at which energy recovery was practised. We return to this negative externality assumption shortly.

Table 3 CSERGE et al. estimates of externalities from UK landfill (£1993 per tonne waste)

Externality	Urban site, no energy recovery	Urban site with energy recovery	Rural site, no energy recovery	Rural site, energy recovery
CO_2	0.32	0.46	0.32	0.46
CH_4	2.36	1.36	2.36	1.36
Transport pollution	0.10	0.10	0.46	0.46
Transport accidents	0.23	0.23	0.55	0.55
Leachate	0.45	0	0.45	0
Pollution displacement	0	-1.12	0	-1.12
Total	3.46	1.03	4.14	1.71

Source: CSERGE et al. (1993) with minor corrections.
Notes: energy recovery refers to the capture of methane for energy supply; pollution displacement refers to the reduced pollution from power station output displaced by methane recovery; rural sites have longer transportation distances, hence the higher accident and transport pollution estimates; leachate is assumed to be a problem for existing sites only - new sites are designed to preclude leachate.

Table 3 shows the importance of methane as a major externality, with other externalities being roughly equally important as each other[46]. A notable omission from the landfill externality estimates was the value of landfill site disamenity which was, unfortunately, not included in the terms of reference for the CSERGE et al. study. Since then, Cambridge Econometrics (2003) has estimated these externalities to be around £2 per tonne.

The original landfill externality estimates require revision for several reasons.

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[46] Fairly uniquely in the history of environmental taxation, the original UK landfill tax was set approximately equal to the externality per tonne estimated in CSERGE et al. (1993).

- The first is the potential change in the marginal fuel displaced by energy recovery in landfill – from coal to perhaps gas. This will raise the net external cost from landfill *with* energy recovery by reducing the credit received for displacing more polluting fuels. But it will leave most landfill unaffected as energy recovery tends not to be not practised.
- The second is the lost value of disamenity from landfill sites. This is allowed for in the Cambridge Econometrics (2003) study which estimates the effects of disamenity on house prices as being the equivalent of £1.5 to £2.2 per tonne of waste. This estimate requires modification because of the use of an outdated discount rate in the Cambridge Econometrics study[47]. We propose to use the upper end of the range, i.e. £2.2 per tonne.
- The third factor is the shadow price of carbon which, in the CSERGE study was put at £4-31 tC, and which can be compared to £2-16 supported by Pearce (2003, 2004). We retain the original CSERGE range.
- The fourth factor is the 'user cost' of landfill space. Since landfill space is, by and large, akin to an exhaustible resource, its shadow price should be equal to the sum of the marginal private costs of disposal to landfill, any pollution externalities, and the user cost or 'depletion premium' that reflects the scarcity of land. This element of the shadow price, if it is relevant, was not estimated in the CSERGE study. However, the relevance of user cost is debatable. Where landfill sites are sold on the market, mar-

[47] The Cambridge Econometrics study estimates a disamenity effect on house prices of proximity to landfill sites. Because house prices are capitalised values, i.e. they reflect the discounted future flow of services from houses, a discount rate is implicit in the capital value. However, the study did not attempt to 'unbundle' this discount rate. The study then takes this capital value (present value) and divides by the *present value* of annual waste streams over a period of 28 years, using a discount rate of 6 %. 6% was the Treasury recommended discount rate until 2003 when it was changed to 3.5%. If the present value of waste streams is discounted at 3.5%, the effect is to *raise* the present value of waste and hence *lower* the house price effect per tonne waste, which is not what is intended. If discount rates fall, the value of disamenity should rise. What is happening is that there are two discount rates: an unknown one implicit in the housing market, and the 6% (now 3.5%) which is only allowed to have an effect on the present value of the waste stream. We have therefore made a crude adjustment by selecting the upper end of the Cambridge Econometrics range.

ket prices should reflect user cost, just as any exhaustible resource price reflects future scarcity if that resource is traded on the open market.

- The CSERGE values are in 1993 prices and need to be updated to 2001/2 prices. But there is also the likelihood that the *relative* price of environmental goods has risen as well since environmental preferences tend to increase in relative terms with income.

Taking landfill without energy recovery, the adjustments are roughly as follows:

- Original value £3.46 tW
- Revised to 2001 prices at 2.5% p.a. = £4.20
- Adjusted for a relative price effect, with an income elasticity of willingness to pay of 0.3 and an income growth of 2.5% p.a.= £4.48
- Add disamenity effect at £2.2 tW = £6.68 tW

It is unclear if any adjustment should be made for user cost. The theory of exhaustible resource pricing would calculate the (marginal) user cost as the price of the 'backstop' technology that replaces landfill as landfill rises in price due to its scarcity. However, it appears to be very difficult to estimate this scarcity factor since it depends on the availability of new landfill space as opposed to the remaining capacity of existing sites. Taking the £74 tonne cost for recycling packaging waste noted earlier, and assuming recycling represents the backstop technology, and that landfill sites become extremely scarce in 20 years' time, then the (marginal) user cost would be (at a 3.5% discount rate) some £37 tW. With a 40-year time horizon it would be around £19 tW, and a 10-year horizon would raise it to £52 tW. However, if incineration is seen as the backstop technology, then the backstop price would be around £30 tW (Coopers and Lybrand et al, 1996) and the user cost element for landfill would be £21 tW for the 10-year horizon, £15 tW for the 20-year horizon and just under £8 tW for the 40-year horizon[48].

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[48] The relevant formula is simply $P_B/(1+s)^T$ where P_B is the price of the backstop technology, T is the time this technology comes in, and s is the social discount rate

Overall, then, a case can be made for an externality value of £6.7 tW, with the caveat that the inclusion of user cost, if judged to be relevant, would substantially increase this figure.

Comparing the incremental financial costs of £26 tW from using incineration to meet the Landfill Directive targets for biodegradable municipal waste, with the £7 or so externalities reduced by diverting waste from landfill implies a substantial net social loss from complying with the Directive in the UK. Two factors may improve the picture. The first is the extent to which landfill user costs are relevant. Given that incineration is the assumed means of meeting the Directive in the UK, a figure of £15 tW would perhaps be relevant, i.e. £26 tW cost now plays £22 tW benefit. As noted however, the extent to which user costs can be thought of as being relevant is open to serious question. Moreover, an analogous argument applies to incinerator 'space' since there has been and still is serious public opposition to the siting of new incinerators in England and Wales. A second element of benefit is the displaced pollution benefit arising from expanding incineration. In the original CSERGE et al. (1993) study, new incinerators were assumed to be associated with energy generation which displaced low merit order coal fired power stations. The overall external cost from incineration was put at about *minus* £4 tW, i.e. there was a benefit of £4 tW. At 2001 prices and allowing for the relative price effect, this would be about £5 tW. Ignoring the user cost issue, adding in displacement benefits would mean that £26 tW cost then plays £12 tW benefit. The user cost argument now appears to be crucial if it is to be argued that the benefits of the Landfill Directive outweigh costs, at least in the UK. But the balance of argument is against benefits being greater than cost for several reasons: (a) the user cost argument is not persuasive even in terms of landfill space; (b) there is a parallel user cost argument for 'incinerator space'; (c) while site disamenity has been calculated for landfill, it has not been estimated for incineration, and there is strong public opposition to new incinerators; and (d) the pollution displacement credit to incineration assumes displacement of marginal coal plant, whereas today the displaced plant could well be gas, in which case there would be a much lower benefit to incineration.

The balance of argument suggests that the Landfill Directive also fails a cost-benefit test.

4.6 Conclusions

The philosophy underlying European Union policy on solid waste has two basic components. The first is the waste hierarchy, and the second is the notion of harmonised targets for recycling and recovery. The waste hierarchy, whilst developed more on the basis of environmental beliefs than any rational appraisal, may not be too divorced from a ranking that would be dictated by economic analysis. Nonetheless, what economic appraisal exists suggests that the hierarchy needs to be treated as a very flexible ‘envelope’ for policy: the rankings of landfill and incineration could easily be reversed, and the desirability of recycling varies by material. Far more serious, however, is the European Commission’s continued allegiance to arbitrary target setting. Whilst accepting that harmonised targets are part of the whole philosophy of the Union, and hence unlikely to change, the chances are that a large excess cost is imposed by selecting targets that have little or no economic rationality to them. Two examples are used to illustrate this finding. The first looks at the packaging waste policy and concludes that neither the original Directive, nor its revisions, pass a cost-benefit test. Indeed, they appear to fail it by a very large margin. The second looks at policy on landfill, itself a product of the waste hierarchy which places landfill at the bottom of the ranking, and suggests that this probably fails a cost-benefit test.

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5 Literature review of social costs and benefits of waste disposal and recycling

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5.1 Introduction

The European Union has adopted a hierarchical approach to solid waste management, including final waste disposal options. First of all, waste should be reduced, otherwise recycled, next incinerated and, only if nothing else works, landfilled. The EU confirmed this hierarchy in preparing its directives on landfilling and incineration:

> *"The 1996 Commission Communication on the review of the Community Strategy for Waste Management confirmed the hierarchy of waste principles established by the Communication of 1989. The principle of prevention of waste generation remains the first priority, followed by recovery and finally by the safe disposal of waste i.e. landfilling. In the Community Waste Strategy landfilling represents the option of last resort because it can have substantial negative impacts on the environment. ... Landfilling as a waste management method has no effect on the prevention of waste and does not make use of waste as a resource, which has a higher priority in the Community Waste Strategy."*
>
> (COM(97), p. 3)

Apparently, the hierarchical approach of the European Commission relies on a rather asymmetric judgment in comparing the different waste strategies. For example, it does include environmental costs for landfilling, but only stresses environmental benefits for the other options. Also, its recent directive on landfilling

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prohibits flammable waste being landfilled (see COM(99)31), while it remains unclear whether private costs have to play a role in decisions regarding waste disposal options by Member States.

In this paper we ask whether available social cost-benefit evaluations of waste disposal provide some support for this strict hierarchy. The next section discusses the structure of a social cost-benefit analysis proper in this case of final waste disposal options (landfilling and incineration) and recycling. The third section provides a summary of the available literature, mainly from country-specific research reports. The fourth section analyses welfare losses if treatment options are chosen that are not in accordance with the outcome of these studies. The fifth section evaluates the findings in the literature. The last section concludes.

5.2 Choosing between waste disposal options

Ideally, the strategic choice between waste strategies should be based on proper social cost-benefit estimates of different options. Not only is it important to find individual welfare weights for the different disposal options and their associated externalities, but also disposal costs provide important information. As individual welfare weights are inherently difficult to get, the literature usually restricts its analysis to cost-cost evaluations. The best option in such a case is simply defined as that waste disposal technology that minimises social cost at the margin. Obviously, how much waste should be recycled, incinerated and/or landfilled depends on the overall social cost function, viz. on the marginal cost of all options together. It is only optimal for the government to dispose of its waste through all options if the marginal social costs of the three opportunities are equal for a given amount of waste to be treated. If, for instance, the marginal social cost of landfilling exceeds the marginal social cost of incineration for a given range of waste to be processed, the government should incinerate all waste from a social cost perspective and vice versa.[50]

The social cost of waste disposal methods, first of all, includes the private and direct environmental cost of the disposal technologies. For a given technology, whether this is recycling, landfilling or incineration, private costs include labour and the capital cost of operation and maintenance. Direct environmental costs are related to the set of environmental externalities of a specific technology, in particular emissions to air, water and soil. Indeed, the set of externalities widely differs across different specific technologies. Moreover, there is a clear link between private and environmental cost. In general, with better measures against emissions to air, water and soil, private cost can be expected to be higher whereas environmental cost will be lower.

Apart from these direct costs related to final waste disposal technologies, one has to take appropriate account of the jointness[51] characteristics of the different waste disposal methods. Indeed, due to their jointness characteristics, recycling, landfilling and incineration not only affect the processing of final waste, but also - indirectly - the private and environmental cost of energy and materials production.[52] By choosing a specific technology, the government not only makes a decision regarding final waste reduction and its environmental cost, but also the composition of energy production and the amount of material recycling in the economy. This is particularly important because final waste disposal technologies differ

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[50] As the model used by Brisson (1997) is restricted to Lagrangian optimisation, corner solutions are excluded beforehand. However, it is by no means clear that such solutions would not appear in practice.

[51] Jointness means that incineration and landfilling not only results in the treatment of waste, but also produces electricity (for landfilling) and electricity, heat, iron and aluminium (for incineration). Thus, landfilling and incineration influences not only the waste market but also the market for energy and materials. At least the environmental costs need to be taken into the picture of these joined produced outputs as now less emissions result in the materials and energy production plants.

widely in their potential for these jointly produced useful outputs. Moreover, each different combination of functions produces a different set of environmental externalities. The better a specific technology exploits its recovery function, the more it might save on both private and environmental costs of producing these outputs.

It is important to recognise that all disposal options potentially contribute to these joint outputs, although they may differ in important respects. Accordingly, a wide variety of options are available for the government in choosing between different waste disposal technologies and their joint contribution to waste reduction, together with energy and material recovery. For instance, at one extreme there is a modern, best practice landfill site, which not only generates electricity, but also runs a small risk of leakage. At the other extreme, there is an old-fashioned incineration plant without electricity production and lacking flue-gas abatement technologies to prevent air emissions including dioxins. Note that our approach reflects a much more general notion of how to choose between waste disposal options than is usually perceived in both waste theory and policy. For instance, the claim that waste incineration has become more favourable in recent years due to the growing potential for energy recovery is not necessarily true in the more general framework we address. Such a claim requires a comparison of this technical option with the social cost characteristics of others, like recycling or a modern landfilling site with energy recovery.

5.3 Break-even costs for recycling, incineration and landfilling

This section summarises the available literature on comparing different waste disposal options.

Brisson (1997) published in her PhD thesis a social cost-benefit analysis of municipal solid waste management for the representative or average EU country. She analysed the waste hierarchy of (1) recycling (including composting), (2) incinera-

52 See Vollebergh (1997) for an application of this cost-cost framework in the context of joint production to biofuels and waste-to-energy incineration plants. Dijkgraaf and Vollebergh (2004) have applied the same framework to the choice between waste incineration and landfilling.

tion and (3) landfilling. Her calculated private and external costs of different waste treatment options suggest that recycling is the best treatment option from a social cost-benefit point of view (see Table 1). Composting, however, is not better than other treatment options from a cost-benefit perspective. Landfilling is always better than composting, and also incineration is better than a bring system or separate collection of compostable waste. Finally, the preference of incineration above landfilling cannot be confirmed on the basis of all relevant costs and benefits.

Table 1. Net social costs per waste-treatment option, average for EU-countries (ecu per tonne)

	Bring system	Co-collection kerbside	Separate collection
Recycling	-170	-131	24
Landfill	92	91	96
Incineration (coal)[1]	115	114	119
Incineration (EU-av.)[1]	150	148	155
Composting	170	102	133

1. Incineration (coal) means incineration replacing fossil fuel use in coal fired electricity plants and Incineration (EU-av.) means incineration replacing fuel use of the average EU-plant.
Source: Brisson (1997)

Vollebergh (1997) calculated the social costs for Waste-To-Energy plant in the Netherlands. He explicitly distinguished between private and environmental costs for both the waste and energy function of this technology. Landfilling has been used here as the opportunity option for the waste function and the average Dutch fossil fuel energy reference system as the opportunity option for the energy system. Table 2 summarises his findings. As both gross private costs and environmental costs of waste processing by landfills tend to be rather low, the costs for the electricity function of waste incineration are quite high. His calculations show that the preference of the Dutch government for waste incineration has raised the social cost of waste processing and electricity production by Dfl. cts 18.2/kWh or Dfl. 103,- per tonne waste (total cost of landfilling plus Fossil Fuel Based Reference System generation). Thus the government implicitly subsidises Waste-to-Energy electricity through its waste management policy. The cost of this policy is partly paid for by current consumers of electricity. The electricity consumers pay a price equal to the avoided cost of electricity generation of Dfl. cts 7.3/kWh, while the marginal cost to produce electricity from waste is almost zero. Waste suppliers, such as households and firms, have to pay the other Dfl. cts 10.9/kWh through higher tariffs necessary to finance waste incineration plants.

Table 2 Social Cost Calculation for Waste-to-Energy- and Fossil Fuel Generated Electricity (Dfl. cts/kWh)

PRIVATE	Gross Private Cost	Environmental Cost	Total Cost
Waste incineration:			
- Electricity and Waste Function	41.4	2.4	43.8
- Opportunity Cost of Waste Function	13.1 -	0.5 -	13.6 -
Waste-to-Energy Function	28.3	1.9	30.2
Fossil Fuel Based Reference System (FFBE)	8.5	3.5	12.0

Source: Vollebergh (1997)

Ayalon et al. (2001) evaluate the cost-effectiveness of different waste treatment options from the perspective of greenhouse gas mitigation according to the Kyoto Protocol. They do not present a comprehensive cost-benefit analysis, but focus on the investment costs of landfilling, incineration, aerobic composting and anaerobic digestion. Operating and maintenance costs are not included because they are, according to the authors, site specific, highly variable and a function of a large number of country-specific factors. The authors note, however, that there is a rough correlation between investment costs and operating and maintenance costs. Figures are based on the situation in Israel, where organic waste is a large part of municipal waste. In 2001 nearly all municipal waste in Israel was landfilled without energy recovery. This makes investments in other options from a greenhouse gas perspective potentially interesting.

Table 3. Cost-efficiency greenhouse gas reduction Israel

	Efficiency of CH4 reduction (% total CH4)	Investment costs of reduction (Euro/tonne CO2-eq.)[1]	Annualised costs (Euro/tonne CO2-eq.)[1]
Landfilling with LFG flare	50	16	1.08
Landfilling with energy recovery	50	40	2.70
Incineration	100	174	11.59
Aerobic composting	90	8	0.52
Anaerobic digestion	100	35	2.32

1. Calculated from US dollars using the exchange rate for 2001 (1 euro = 0.8956 dollars).
Source: Ayalon et al. (2001)

Aerobic composting has the lowest annualised costs per unit of CO2-equivalents reduced (see Table 3). In this option all organic waste is treated by the aerobic composting plant, while some materials like paper and plastic are recycled and non recyclable waste that cannot be treated by the aerobic composting plant is landfilled. Incineration is the worst option with far higher costs than all other options. Strikingly, the second-best option is landfilling with flaring of the methane emitted from the landfill.

Nolan-Itu (2001) presents an evaluation of the kerbside recycling system in Australia. Based on data for 200 communities, total social costs and benefits are calculated for recycling compared with landfilling. The authors find a net social benefit for all recycling systems in use (see Table 4). This conclusion applies also for less densely populated areas. Although the costs increase and benefits decrease when population density falls, net social benefits remain positive. However, not much data are available for rural towns and remote communities, as kerbside services are not often present in these regions.

Table 4 Net social costs kerbside collection compared with landfilling (euro[1])

System	Net financial costs ($/hh./year)	Environm. costs ($/hh./year)	Net social costs ($/hh./year)
Existing kerbside systems at current yields	23	-61	-38
Existing kerbside systems at higher yields	22	-86	-64
Kerbside recycling of paper and glass only	16	-45	-29
Energy Recovery from Plastics & Paper	24	-44	-20
Mechanical-Biological Treatment (MBT)	50	-64	-14

1. Calculated from US dollars using the exchange rate for 2001 (1 euro = 0.8956 dollars).
Source: Nolan-Itu (2001)

Döberl et al. (2002) evaluate different waste management scenarios for municipal solid waste and sewage sludge in Austria using a cost-benefit approach.[53] The authors include a large number of emissions of nine different waste treatment scenario's (see Table 5), apply shadow prices and add the private costs of the different treatment options to these external costs. They especially focus on the long-term effects as they account for emissions in the next 10,000 years. Their analysis shows that incineration is the best option, followed by mechanical-biological treatment. Landfilling is the worst option. That incineration performs better than mechanical-biological treatment follows from the fact that the residues of incineration have a better quality. This saves emissions in the landfilling phase of the residues.

[53] The authors use also a modified cost-effectiveness analysis. This last type of analysis allows to include the long-term impacts of the landfilled material. As the results of the cost-benefit analysis corresponds to those of the MCEA we only present results for cost-benefit analysis.

Table 5 Cost-benefit results Austria

Scenario	Rank order[1]
Status-quo continues, no change of waste management measures	8
Landfilling of untreated waste	9
Incineration without after-treatment	1
Incineration with cement stabilisation of the residues	2
High temperature treatment	3
Mechanical-biological treatment with the light fraction from sorting and splitting (LF) processed in a fluidised-bed furnace	7
Mechanical-biological treatment with the light fraction from sorting and splitting (LF) processed in a cement kiln	6
Mechanical-biological treatment with the heavy fraction of high calorific value (HF) processed in an incinerator and the LF in a fluidised-bed furnace	5
Mechanical-biological treatment with the heavy fraction (HF) of high calorific value processed in an incinerator and the LF in a cement kiln	4

1. Rank order based on cost-benefit analysis
Source: Doberl et.al. (2002)

As the time period of the analysis is very long, discounting seems to be important. The authors, however, use a discount rate of zero referring to the precautionary principle and the impossibility to estimate technological progress. They argue in the paper that (p. 32):

> *"It is interesting to note, that the ranking of the scenarios does not depend on the time period investigated (years, centuries, millenniums) but is constant for all three cases.*
>
> *The longer the observation period is, the more distinct becomes the advantage of thermal treatment."*

How distinct the position of thermal treatment is in the various periods is not clear as the paper only presents figures for the long-term effects.

Bruvoll and Nyborg (2002) analyse whether the recycling efforts of households should be integrated in cost-benefit calculations of waste treatment options. They show, based on a theoretical analysis, that this is the case when government policies influence these efforts indirectly. Using data for Norway they calculate that the willingness to pay for others to perform the sorting activities is 87 dollars per tonne of waste. Compared with total treatment costs this is a significant contribution.

EPA (2002) presents a cost-benefit analysis of three recycling options and incineration for plastic bottles. Recycling options differ with respect to collection (only in municipal recycling stations or also local) and the way the bottles are treated after collection (export to Germany or production of granulate in Denmark). Table 6 presents the results. On the basis of social costs incineration is much cheaper for society than recycling. According to the authors this stems primarily from the much higher collection costs for recycling. Sensitivity analysis is performed by the authors on a large number of variables. This showed that the ranking of options is robust with only a few exceptions.

Table 6. Social costs in euro per tonne of waste

Option	Collection	Treatment	Private	Environmental	Social
Incineration	With other waste	Incineration plant	178	119	297
Recycling	Municipal and local stations	Granulate in Denmark	521	52	573
Recycling	Municipal and local stations	Export to Germany	407	85	492
Recycling	Municipal stations	Export to Germany	280	49	329

1. Using the 2002 conversion rate (7.4 DKK/EUR)

EPA (2003) analyses the costs and benefits of increases in the recycling of organic waste. This analysis shows that incineration is cheaper for society than anaerobic digestion or central composting. Although treatment is cheaper for these last two options this is more than compensated by the much higher collection costs. External costs were low for all options and were only responsible for 5 to 10% of net social costs. Sensitivity analysis shows that in general the conclusions are not dependent on the assumptions made.

Vigsoe & Andersen (2002) presents a cost-benefit comparison of the collection and recycling of drink containers with incineration. Using data for Denmark, where a refund system was just put into use, the report concludes that the costs of the deposit system are relatively high compared with the environmental benefits.

Petersen & Andersen (2002) compares the costs and benefits of paper recycling and the use of paper for energy recovery. While the last option is forbidden in Denmark, a social cost-benefit analysis shows that energy recovery saves net social costs[54]. Not only is the market price lower than for coal (compared on the basis of the same energy content), using paper instead of coal saves CO2 emissions.

Table 7. Net social cost (Euro) per tonne liquid board container collected

	Unsorted waste			Sorted at source		
				Collection system		Delivery system
	Landfill	Incineration	WTE[1]	Mixed	Separate	
Private costs						
- Household[2]	0	0	0	188	188	232
- Collection/sorting	92	92	92	272	67	85
- Treatment	73	81	107	438	438	438
- Savings	0	0	-51	-64	-64	-64
- Net	165	173	148	834	629	691
Environmental costs						
- Treatment	139	86	27	5	5	5
- Savings			-45	-34	-34	-34
- Net	139	86	-18	-29	-29	-29
Net social costs	304	259	130	805	600	662

Notes:
1. WTE = Waste to energy (Incineration with energy recovery)
2. Use of time and resources.
Source: Ibenholt and Lindhjem (2003)

Ibenholt and Lindhjem (2003) analyse whether separate collection and recycling of liquid board containers is a better option than incineration or landfilling from a social cost-benefit analysis. They show, using data for Norway, that separate collection and recycling is very expensive as liquid board containers are only a small

54 The result is based on incineration of five percentage of the paper recycled in Denmark and only valid when the price for mixed paper is low.

part of total waste. Incineration with energy recovery is the option with the lowest net-social costs compared with landfilling and recycling (see Table 7).

Rasmussen & Reimann (2004) analyse whether the growth of municipal solid waste should be accommodated by increasing the capacity of waste incineration plants or by using waste as a substitute of fossil fuels in private production plants. Although no gains can be made by transferring waste from existing waste incineration plants to private production plants, this conclusion does not hold for waste not currently incinerated. Net social benefits arise if the capacity of private production plants is used.

Table 8. Net social cost estimates for landfilling and incineration (euro per tonne)

	Landfilling	Incineration
Gross Environmental costs:		
- Emissions to air	5.84	17.26
- Emissions to water	0.00	0.00
- Chemical waste	2.63	28.69
- Land use	17.88	0.00
Total	26.35	45.95
Environmental cost savings:		
- Energy function	-4.21	-22.55
- Materials function	-0.00	-5.76
Net environmental costs	22.14	17.64
Gross private costs	40.00	103.00
Private cost savings		
- energy function	-4.00	-21.00
- materials function	0.00	-3.00
Net private costs	36.00	79.00
Net social costs	58.14	96.64

Source: Dijkgraaf and Vollebergh (2004)

Dijkgraaf and Vollebergh (2004) present a social cost-benefit analysis for landfilling versus incineration in the Netherlands. The data provide support for the widespread policy preference for incineration over landfilling, but only if the analysis is restricted to environmental costs alone and includes savings of both energy and material recovery (see Table 8). Gross private costs, however, are so much higher for incineration, that landfilling is the social cost minimising option at the margin, even in a densely populated country such as the Netherlands. Furthermore, they show that results generalise to other European countries and probably to the US. Implications for waste policy are discussed as well. Proper treatment at, and energy recovery from landfills seem to be the most important targets for waste policy. Finally, this study confirms the earlier estimates of Vollebergh (1997) that Waste-to-Energy plants are a very expensive way to save on climate change emissions.

Table 9 presents an overview of the studies discussed so far. Each row gives the ranking of the treatment options analysed in the papers. A number of conclusions appear from the table:
Recycling, which is the best option according to the EU hierarchy, is not always the best option according to cost-benefit studies. In three studies other options are preferred, while also three studies confirm the EU hierarchy. Thus, evidence exists that recycling is not always the best treatment option. This stems mostly from the sometimes very high collection costs if waste has to be separately collected. This shows the importance to account for waste as a heterogeneous commodity.
Composting, which is included in recycling in the EU hierarchy, is the preferred option only in one study which also restricts its analyses to greenhouse gases. Two other studies find that incineration or even all other options are preferable to composting.
Incineration, which is preferred in the EU hierarchy above landfilling, is preferred to landfilling from a social cost perspective in two studies. However, three other studies find that landfilling is the best option. Accordingly, the EU hierarchy does not seem to be supported by the available cost-benefit studies.

Table 9. Overview studies (rank order)

Study	Type of waste	Recycling	Composting	Incineration	CFEP	Landfilling
Brisson (1997)	General	1	4	3		2
Vollebergh (1997)	Burnable			2	1	
Ayalon (2001)	Compostable		1	3		2
Nolan-Itu (2001)	Recyclables	1				2
Döberl et al. (2002)	General			1		2
EPA (2002)	Plastic bottles	2		1		
EPA (2003)	Compostable		2	1		
Vigsoe & Andersen (2002)	One-way drink containers	2		1		
Petersen & Andersen (2002)	Paper	2			1	
Ibenholt et al. (2003)	Liquid board containers	3		1		2
Rasmussen & Reimann (2004)	General			2	1	
Dijkgraaf & Vollebergh(2004)	General			2		1

Note: CFEP = Coal-fired electricity plant.

5.4 Welfare loss of suboptimal strategies

Welfare losses of second or even third-best treatment options can be substantial. Let us take the findings for landfilling and incineration as reported by Dijkgraaf and Vollebergh (2004) as an example. Table 10 gives in the second column estimated figures for the total residential waste collected in EU countries (see Dijkgraaf et al., 2001, for calculation method). In the third and fourth columns the net private and net environmental cost per tonne of waste (see Table 8) are multiplied by these quantities. Thus, the fifth column gives an estimation of total social costs. The last three columns repeat this exercise for the situation when waste is not landfilled but incinerated. It shows that total social costs are Euro 6.1 billion lower if landfilling is chosen instead of incineration.

Table 10 Social costs households landfilling versus incineration in million euro (waste in Mtonne)

	Total	Costs if all waste is Landfilled			Costs if all waste is incinerated		
	waste	Private	Environmental	Social	Private	Environmental	Social
Belgium	3	112	69	181	245	55	300
Denmark	3	108	66	174	237	53	290
Germany	25	901	554	1455	1977	442	2419
Finland	2	75	46	121	164	37	201
France	24	857	527	1385	1881	420	2301
Greece	5	192	118	309	420	94	514
Ireland	2	67	41	108	146	33	179
Italy	29	1036	637	1672	2273	507	2780
Luxembourg	0	5	3	8	10	2	13
Netherlands	5	173	106	280	380	85	465
Austria	2	89	54	143	194	43	238
Portugal	5	181	111	292	396	88	485
Spain	20	712	438	1150	1563	349	1912
UK	30	1067	656	1724	2342	523	2866
Sweden	4	129	79	208	283	63	346
Total EU	158	5702	3507	9209	12514	2794	15308

5.5 Evaluation

This section briefly discusses some qualifications with respect to the studies discussed so far:

i) Firstly, only few of these papers have been published in referred international scientific journals. Therefore the quality of the papers is likely to vary substantially. However, a meta-analysis, for instance by checking the framework described in section 5.2, is beyond the scope of our contribution.

ii) Second, a number of studies are not very clear about the shadow prices used to monetarise emissions. While some studies explicitly mention the values used (for example EPA (2002), EPA (2003) and Dijkgraaf and Vollebergh (2004), others only mention that emissions are monetarised, but not how.

iii) Third, some studies use a dynamic approach but without discounting like the study of Döberl et al. (2002). Indeed, it is certainly valid to challenge the idea of discounting in view of the precautionary principle, and hyperbolic discounting may also be an issue here. Simply disregarding discounting, however, is a very debatable assumption.

iv) Fourth, not many studies account for local and national circumstances that may prevent generalisation of their results to other areas or countries Exceptions are Brisson (1997) and Dijkgraaf and Vollebergh (2004), who explicitly address the applicability of their calculations to other countries.

v) Fifth, the definition of private costs is often unclear and this may substantially affect outcomes. For instance, it seems likely that a number of studies use figures for inefficient plants. Furthermore, private costs in the status quo are usually the consequence of previous policy choices and make comparisons over time more difficult;

vi) Sixth, the benefits of landfilling are not always properly defined. Dijkgraaf and Vollebergh (2004) showed that using emissions of methane as a source of energy production greatly influences the social costs of landfilling versus incineration (see Fullerton, 2004 for a similar result). The reason is not only savings on energy production, but mainly the much lower emissions to the air if methane is collected and used as an energy source. This illustrates the importance to base cost-benefit analysis for strategic policy choices on the best available techniques.

vii) Finally, the benefits of incineration are sometimes too large. As incineration saves energy production in electricity plants, the saved environmental costs due to lower emissions are subtracted from gross external costs. It is important to know the assumption about the energy reference system of electricity plants. Some studies assume that this is a coal-fired electricity plant. However, using substitutes (like gas-fired power plants or even hydro or wind power) would result in much lower benefits for incineration because fewer emissions would be saved. This illustrates once again how sensitive specific analyses are to local circumstances.

These qualifications illustrate that the comparison in Table 9 should be interpreted with care. Without further research and more knowledge about the underlying assumptions one should be careful in interpreting the results of these studies.

5.6 Conclusions

We have shown that the available social cost-benefit studies raise serious doubts on the current waste hierarchy advocated within the EU. For instance, WTE plants reduce the net social costs of final waste disposal only if waste incineration without energy recovery is applied already, or if infrastructures for the use of heat exist in the status quo. It may also be questioned whether the overall environmental cost savings from the current hierarchy are large enough to compensate for the sometimes substantial larger private cost. Because waste is heterogeneous and local circumstances differ substantially (for instance in terms of local environment or the existing energy system), a one-size-fits-all solution is rather unlikely. Indeed, a lot more should be done in terms of proper cost-benefit evaluations of existing waste policies. In particular, not much has been done yet on the cost and benefit evaluation of recycling in practice.

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6 Personal reflections on December 14 workshop

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The December 14 workshop at EAI in Copenhagen, at which the papers in this report were presented, drew a crowd of more than 40 people and led to a lively discussion. It is difficult to summarize the broad range of points made in discussion; some participants generally agreed with the speakers' perspectives, while others clearly did not.

In retrospect, it seems unfortunate that this report does not include criticisms or responses from workshop participants with a differing point of view. In discussion after the fact, I offered that as the speaker most critical of cost-benefit analysis, I could attempt to summarize the environmentalist response to our panel. Here, therefore, is the perspective of the opponents of the panel, as I interpret it, followed by my own response.

6.1 A Critic's Perspective

Cost-benefit analysis as a means of evaluating European environmental policy is unnecessary and undesirable. Europe's rigorous environmental standards, for waste management and for other problems, meet with broad popular support. These standards express the will of a large majority of citizens, which is the essential requirement for public policy in a democracy. There is no evidence that any great economic losses are imposed by environmental protection; on the contrary, Europe enjoys both a high standard of living *and* high environmental standards.

The use of cost-benefit analysis to identify otherwise popular environmental policies that are "too expensive" is the first step on a slippery slope toward American-style deregulation. In this unattractive and unequal model, the most affluent members of society enjoy greater private consumption (for instance, bigger houses and cars) than in Europe today, but at the expense of numerous and growing threats to human health and the natural environment.

Turning specifically to waste management, the solid waste hierarchy is a widely understood and supported principle, calling on us to "reduce, reuse, recycle" as much as possible before discarding waste, and to incinerate the residual waste before resorting to the worst option, landfilling. This hierarchy has withstood the test of time, and has inspired widespread commitment to recovery and reuse of valuable materials that would otherwise be discarded. It is supported by environmental research, such as life-cycle analysis, identifying the environmental impacts of different waste management options. EU policy based on the hierarchy, such as high and uniform recycling targets, maintains the important principle of transparent, uniform standards for all 25 countries. At present, rates of recycling differ widely across the EU; ambitious recycling standards will lead to "harmonizing upward," pushing all countries to emulate those who are doing best.

Cost-benefit analysis of recycling provides a technical cover for opponents of recycling to attack this popular and environmentally sound policy. Is this workshop an attempt to replace political discussion of popular recycling efforts with highly technical analysis that ends with the "experts" telling us to recycle less? Despite some difference of views on the panel, the predominant message was that economists can straighten out our mistakes and help us save a lot of money by cutting back on recycling.

Finally, the workshop would have us believe that cost-benefit analysis has shown that landfilling is preferable to incineration. This, too, flies in the face of both popular preference and scientific evidence -- and makes the impossible claim that economists have evaluated and monetized all the environmental and social impacts of landfilling. Incineration at least recovers the energy value of material that is being discarded, and greatly reduces its volume. This economizes on scarce landfill space and reduces the well-known, undesirable impacts of landfills on surrounding communities. One economic analysis is not enough to overturn the extensive scientific and political support for the solid waste hierarchy, either on the ranking of disposal options or on the desirability of recycling.

6.2 Response to critics

You are right to worry about cost-benefit analysis as a standard for environmental policy; the recent American experience should serve as a disturbing warning in this respect. In the hands of the Bush administration, a biased application of cost-benefit analysis has become a weapon against popular, desirable environmental regulations. Sensible environmental policies have been adopted in the past, without the use of cost-benefit analysis, and the same can and should be done in the future.[55]

But while cost-benefit analysis should not govern policy debate, it can certainly inform it. From that perspective, I urge you to look again at the analyses presented in this report. Whether or not you agree with the theoretical framework advocated by the other authors, I think there is a great deal to be learned from their interesting work. Note that they are a far cry from the partisan, anti-environmental style of cost-benefit analysis that is deployed in Washington today: David Pearce concludes that most EU waste policies, based on the hierarchy, would pass a cost-benefit test; and Richard Porter is much more sympathetic to recycling and other environmental initiatives than the dominant voices in the American debate.

[55] For a critique of cost-benefit analysis in general, and the recent American experience in particular, see Frank Ackerman and Lisa Heinzerling, *Priceless: On Knowing the Price of Everything and the Value of Nothing* (New York: The New Press, 2004).

How, then, should you interpret the results of this workshop? Calculations that a particular recycling program would fail a cost-benefit test do not necessarily imply that the program should be cancelled, since it is effectively impossible to have a meaningful and complete monetary valuation of the benefits of recycling. However, it could be taken as a warning that the program is expensive, and it is time to look for ways to lower its costs. There is an extraordinary range of program designs and cost levels in recycling programs around the world; learning from the best existing programs can frequently lead to big improvements.

Good program design depends on local conditions, which vary widely across an area as large as Europe or the United States; this is a reason why recycling advocates might question the usefulness of uniform standards throughout the EU.

The most surprising finding of the workshop, and the most direct challenge to the hierarchy, is contained in Herman Vollebergh's study of incineration versus landfilling. He argues that economic costs are far greater for incineration than for landfilling, even under the high-density conditions of the Netherlands, while environmental impacts for the two options are broadly similar -- in large part due to the substantial, easily overlooked impacts of disposal of incinerator ash. Although this is expressed in the language of cost-benefit analysis, the validity of the argument does not depend on that language or framework. Even for those who approach policy in terms of the solid waste hierarchy and life-cycle analysis, Vollebergh's remarkable research should lead to re-examination of the bottom rungs of the hierarchy.