



Addressing the Economics of Waste



ORGANISATION FOR ECONOMIC CO-OPERATION AND DEVELOPMENT

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FOREWORD

Waste generation and waste management receive increasing attention among policy makers. On the one hand, waste causes environmental problems, both if it is handled properly and - especially - if it is mishandled, e.g. through illegal dumping. On the other hand, both waste recovery and final disposal can involve significant financial costs for public authorities, waste handlers and for households. Unfortunately, under existing policy frameworks, both producers and households often lack proper incentives to reduce the amounts of waste being generated, and to ensure that the waste which is generated cause limited adverse environmental impacts. Some countries have adopted ambitious targets in the waste area, but the costs of achieving them through the instruments chosen is sometimes high. It is therefore important to carefully assess the costs and benefits of waste-related policies.

The OECD has for many years been working on waste-related issues, in later years primarily through its Working Group on Waste Prevention and Recycling (WGWPR). The work has inter alia focused on transboundary movements of waste, and on defining environmentally sound management of waste.

In the elaboration of the 2003-2004 work program of WGWPR on waste issues, it was felt that it could be useful for the OECD to place more emphasis on issues related to the “economics of waste”. In order to take stock of recent research findings on such issues, and to help the selection of topics on which the OECD could usefully do additional work, a workshop on the economic of waste was held 14-15 October 2003 in Paris. The workshop brought together leading academics, civil servants working on waste-related topics and representatives of the business community (selected by the Business and Industry Advisory Committee to the OECD, BIAC) and environmental NGOs.

This book contains eight papers that were presented at the workshop - revised to take into account relevant comments made and questions raised in the subsequent discussions. In addition to these papers, two more papers were presented at the workshop: David Fitzsimons of the consultancy firm Oakdene Hollins, Aylesbury Bucks, United Kingdom, had written a paper on “Improving Markets for Used Lubricating Oils”, while Nick Johnstone of OECD, Environment Directorate, had addressed “Market Failures and Barriers in Secondary Material Markets”. While this book would have benefited greatly from the inclusion of these two papers, it has instead been decided to issue them separately, along with two other papers, all presenting findings of a recent OECD project on the functioning of secondary material markets.

The OECD wishes to thank Denmark and France for financial contributions that allowed the workshop to take place.

These proceedings are published on the responsibility of the Secretary-General of the OECD.

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Chapter 1

ADDRESSING THE ECONOMICS OF WASTE • AN INTRODUCTION

By Nils Axel Braathen¹

1. Introduction

OECD has for many years been working on many waste-related issues, in later years primarily through its Working Group on Waste Prevention and Recycling (WGWPR). The work has primarily addressed technical, scientific and health aspects of waste issues and *inter alia* focused on waste management, transboundary movements of waste, waste minimisation, extended producer responsibility and • most recently • on promoting environmentally sound management of waste.

It has now been felt useful to shift the emphasis somewhat, and to focus more on the “economics of waste”. There are several reasons for this: Better understanding of the economic issues involved can facilitate policy measures to limit waste generation, and to promote environmentally benign ways of handling the waste. Increased ambition levels in waste-related policy targets set over the last decades have increased costs to public authorities, industry, waste handlers and/or private households. More focus on “the economics of waste” can help assessing the balance between benefits and costs of existing targets, and making sure that a given target is approached at lowest possible costs to society as a whole.

In order to take stock of recent research findings on such issues, and to help selecting topics on which OECD usefully could do additional work, a workshop on the economic of waste was held 14-15 October 2003 in Paris. The purpose of this chapter is to highlight a few of the issues raised, and the “answers” given, both in the subsequent papers² and in the discussions held during the workshop. Some additional observations are also made.

2. Trends in Waste Generation and Waste Policies

Soizick de Tilly’s paper served as an introduction to the workshop discussions. It describes trends in waste generation and waste policies over the last decade or so, focusing in particular of municipal waste, i.e. waste from households and most smaller businesses. According to OECD statistics, municipal waste generation increased by 14% between 1990 and 2000, from 530 to 605 million tonnes. Measured per capita, municipal waste generation increased from 509 to 540 kg, a rise of 6%, while total population in the area increased 8% over this period.

1. National Policies Division, OECD Environment Directorate. The opinions expressed in this chapter are those of the author and do not necessarily reflect the views of the OECD.

2. All the papers have been revised after the workshop, to reflect comments made and questions asked during the workshop.

The paper explains that the increase in the amounts of municipal waste is the net impact of several, sometimes conflicting “drivers”, like economic growth; a growing number of households; smaller average household size; growing urbanisation (with better waste collection services in urban than rural areas³); changing consumption patterns and changing socio-cultural habits.⁴

De Tilly illustrates that although most municipal waste is still put in landfills, this method of waste management is less and less dominant: Municipal waste landfilling increased by 2% between 1995 and 2000, while municipal waste generation increased by 10%. Incineration of municipal waste with energy recovery, and composting of moist organic waste, is becoming increasingly common. However, major differences between different countries and regions are described in the paper.

There is also a broad trend towards increased recycling. Recycling rates differ according to the type of material, surpassing 80% for metals, 35-40% for glass, 40-55% for paper and cardboard. Recycling rates differ also considerably from one country to another: in Ireland, for example, 10% of paper and cardboard is recycled whereas the figure for Germany is 70%.

Municipal waste constitutes only a minor share of total waste amounts. According to EEA (2001b), manufacturing waste constituted 26% of the total waste amount in EEA countries in the period 1992-1997, while mining and quarrying waste constituted 29%, construction and demolition waste 22% and municipal waste 14%. It should, however, be emphasised that these numbers are uncertain.

De Tilly finds that broadly speaking, environmental impacts of waste management in the OECD countries have diminished over the last ten years, due to extensive regulation, especially concerning landfills and standards for incinerator emissions and the development of highly efficient technologies, such as for controlling dioxin emissions from incinerators.

However, in many cases, current disposal capacities are seen as insufficient. The paper also states that emission regulations and standards are often not complied with, and that poor waste management in the past e.g. have led to long-term contamination of soil and groundwater. Local authorities set waste management charges that do not reflect environmental externalities and fail to provide a coherent basis for the use of the different potential methods of waste management.

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3. This point is also reflected in an observation made at the workshop by Mr. Remy Risser of the French Ecology and Sustainable Development Ministry. He indicated that the amount of household waste in France is mainly known through surveys of treatment facilities. Thus, each policy increasing the supply of treatment facilities tends also to raise the apparent quantity of waste generated. It has been estimated that the amounts of waste eliminated by household through domestic incineration, domestic composting and illicit dumping have decreased from 1.56 Mt to 0.89 Mt between 1993 and 2000, representing now less than 3% of the 32,5 Mt collected and treated yearly. During the same period, available data show a decreasing growth rate of waste generation. This is partly due to better statistics linked to the modernisation of treatment facilities, but also - apparently - to a change in the consumption structure, with a shift from less durable goods to more durable goods.
 4. EEA (2001) states that “(t)here is no correlation between quantities generated (of waste from daily household and commercial activities) and private consumption, suggesting that basic needs that produce waste are being satisfied in all countries and that higher incomes will not result in the generation of more waste of this type. Differences between countries are due to other factors, such as difference in consumption patterns and lifestyle. Single households produce more waste per capita than families, ready-made food produces more packaging waste than the traditional family-prepared meals, while traditional preparation results in more organic kitchen waste. Increases in income are likely to be used for long-lived goods and services, which could result in increasing amounts of other types of waste such as bulky waste and wastes from construction and demolition.”

3. Economic Analysis of Solid Waste Management Policies

The paper by Don Fullerton and Amy Raub addresses the economics of a broad spectre of policy options for the management of household waste. Central to their discussion is a distinction between situations where households can avoid the tax on proper collection of garbage by instead burning or dumping their garbage, and situations where such behaviour is precluded. In a simple model reflecting the latter case, a social optimum can be reached by simply taxing garbage per unit generated, according to the marginal external damages it causes.

In cases where illegal dumping of waste is a possibility, a combination of instruments is necessary to achieve a social optimum. Fullerton and Raub argue for a deposit-refund system, consisting of an advanced disposal fee at the time of purchase, in combination with subsidies to proper disposal. The tax on consumption should reflect the marginal external damages *from dumping*. Recycling ends up with no net tax. Proper disposal of garbage receives back the initial tax upon purchase, but garbage gets an additional tax that reflects its own externality.

If the subsidy is nearly equal to the cost of waste collection, a city can save administrative costs by just collecting garbage for free. Fullerton and Raub find that this logic is already widely applied, as many cities intentionally collect garbage and recycling for free, in order to avert dumping.

Fullerton and Raub find that a virgin materials tax cannot optimally be used to correct for the marginal environmental damages of garbage disposal if a tax is available on garbage disposal. Another policy option discussed in the paper is government mandates, e.g. requirements on minimum recycled-content of certain products. Such mandates *can* be used to reduce solid waste amounts, but an optimal implementation would require the policy maker to have information on the production technologies and costs of all different firms. Thus, even if mandates achieve waste-reduction targets, they may have high social costs.

The paper presents several studies of the impacts of unit-based pricing systems for garbage collection, where households, for example, pay a certain amount for each garbage bag they put out for collection. Central to the discussion is a study by Fullerton and Kinnaman from 1996, where the weight and volume of the garbage and recycling of 75 households were measured by hand over four weeks prior to, and following, the implementation of a price-per-bag program. This study found a slight drop in the weight of garbage (elasticity of -0.076) – indicating that a price per bag is not very effective in reducing waste generation. The 1996 study also points to increased illegal dumping as one of the explanations of the (limited) reductions in measured waste amounts.

The likelihood of increased problems related to illegal dumping following the introduction of a unit-based charging system caused some debate at the workshop.⁵ Participants from several countries • for example Switzerland and Korea⁶, where such systems are widely used • indicated that this – at

5. See also European Commission (2003), where it is stated (on pages 34-35) that “most communities that have introduced PAYT [pay-as-you-throw] schemes have not experienced large and sustained increases in illegal dumping”.

6. According to KEI (2003), there were about 1.1 million “cases of illegal activities nationwide” relating to their newly introduced volume-based waste fee (VBWF) in 1995. The number of such cases had dropped to about 365,000 in 2000, after totalling about 550,000, 640,000, 550,000 and 580,000 in 1996, 1997, 1998 and 1999 respectively. However, “Illegal Dumping” is mentioned first among “Future Challenges” in the report, with the explanation that “[o]ne of the biggest challenges to the successful implementation of VBWF is finding how to effectively monitor and prevent illegal dumping and other unlawful activities. Illegal dumping by rural residents and low-income population take diverse forms and monitoring these activities are difficult and expensive. In rural areas, illegal dumping is more problematic than in urban areas.”

most – was a relatively short-term problem.⁷ In any case, the impacts of unit-based charging systems on • e.g. • household behaviour was singled out at the workshop as one of the issues that usefully could be studied further by OECD.

The paper concludes that the optimal charge at the curb is the full social cost per bag of garbage, including both the direct costs of collection per bag plus the external cost per bag. If dumping is not a problem, then this charge can be imposed directly to control waste quantities. If dumping is a potential problem, then the paper finds that it can be optimal to use a deposit-refund system. This set of taxes and subsidies has the same net revenue as a direct excise tax, but it cannot be evaded by illegal dumping. It therefore has lower enforcement costs, but may • on the other hand • have higher administrative costs.

4. The Development and Implementation of a Landfill Tax in the UK

The next two papers presented case studies of taxes that have been introduced to address the final disposal of waste.

In their paper Bob Davies and Michael Doble describe the preparation and the workings of the landfill tax that was introduced in United Kingdom in 1996. The tax applies to commercial and industrial waste and municipal waste, and there are separate tax rates for active waste (i.e. waste containing putrescible material) and inactive (or inert) waste. The tax was originally designed to internalise the negative externalities of landfilling • but the tax rates have subsequently been increased, in order to strengthen its behavioural impacts, and additional, significant, rate increases are planned for future years.

Depending upon the type of landfill, whether it was urban or rural and whether there was energy recovery, a 1993 study calculated the external costs from landfill to be in the range £1 to £9 per tonne of waste (1.4 to 12.8 per tonne waste, using exchange rates of December 2003). Climate change impacts of methane emissions was found to be the most significant externality, varying between £0.57 (0.8) and £6.27 (9) per tonne of waste. It was assumed that new landfills internalised any externalities related to leachate and that the operator is responsible for cleanup and monitoring costs, while for existing landfills and externality of £0.45 (0.65) was estimated.

On top of these estimates comes disamenity costs, that is the nuisance from noise, odour, visual intrusion, etc. Based on US data, an estimate of approximately £2 (3) per tonne of waste was used.⁸ This gave a total monetised cost for landfill of about £5 per tonne, approximately equivalent to £7 (10) per tonne for active and £2 per tonne for inactive (inert) waste • and these values were at the outset used for the respective tax rates. The revenue was recycled back through a 0.2 per cent reduction in business's national insurance contributions. It is also possible for landfill operators to obtain tax rebates of up to 90 per cent when making contributions to environmental trusts.

A 1998 review of the landfill tax concluded that the landfill tax had influenced business's waste management decisions with almost a third of companies having begun, or considering, waste recycling, re-use or minimisation as a result of the tax in combination with the Packaging Regulations. This finding was supported by an observed 30 per cent reduction in landfilled wastes being taxed at

7. The case Fullerton and Kinnaman studied was the introduction of a 0.80\$ charge per 32-gallon bag or can of residential garbage collected at the curb in Charlottesville, Virginia, USA, as from 1 July 1992. The behaviour of households *ex post* was observed over a four-weeks period in September of the same year, i.e. 2-3 months after the introduction of the charge.

8. This estimate was later underpinned by a large UK study of impacts on landfills on disamenity costs, cf. DEFRA (2003).

the lower rate. However, for active waste there was much less of a reduction in landfilled amounts.⁹ As a follow-up of this review, the tax rate for active waste was increased to £10 per tonne. In addition, an escalator that will increase the tax rate £1 per tonne per year was introduced for the period up to 2004, when the rate will reach £15 (21.4) per tonne.

The paper illustrates that it will be quite challenging for United Kingdom to fulfil its obligations under the EU Landfill Directive (Council Directive 1999/31/EC), e.g. as regards bio-degradable municipal waste. Following additional studies on how best to fulfil these obligations, a decision has now been made to increase the tax rate for active waste by £3 per tonne in 2005 and by at least £3 per tonne in the years thereafter, on the way to a medium to long-term tax rate of £35 (50) per tonne. The purpose of these rate increases is to achieve behavioural change and send a long-term signal to municipalities and business that the relative costs of disposal are going to shift in the future. The additional revenues will be re-distributed back to business.

The paper concludes by pointing out that that the current tax rate increases go beyond the level which would internalise the externalities caused by the waste. Subsequent estimates have confirmed the credibility of the original externalities estimates. There has also been a significant impact on the quantity of inactive waste sent to landfill, in the main due to the re-use of construction and demolition waste. Active waste going to landfill has remained stable and it is clear that if this is to be reduced, the landfill tax will have to increase further.¹⁰

As an a propos to the paper by Davies and Doble, the fact that tax rates significantly higher than the estimated externalities caused by the landfilling of waste has so far proven insufficient to achieve the targets set under EU legislation can indicate that it would be useful to reconsider the foundation of some of these targets. In this connexion it is interesting to note that European Commission (2003) states that “current directives foresee that all Member States should achieve the same recycling target. However, the question is legitimate whether this uniformity in targets is most effective from both an environmental and economic point of view”.

5. Waste Tax in Norway

As a second case study, the paper by Torhild H. Martinsen and Erik Vassnes presents the Norwegian tax on final waste treatment that was introduced in 1999. The objective of the tax is to price the environmental damage caused by final waste treatment. The tax was expected to contribute to an increase in source separation and recycling and thus reduce the amount of residual waste.

When the tax was introduced, it was levied per tonne of waste delivered to landfills and incineration plants. In order to make the tax better reflect the environmental harm done, the tax on landfills has been differentiated, with a tax rate of NOK 327 (about 40) per tonne waste delivered to a landfill with a high environmental standard, and NOK 427 (about 52.5) per tonne waste delivered to a landfill with a low environmental standard. The tax rate on waste delivered to incinerators varies at present on the degree of utilisation of energy produced during incineration. If none of the energy is used, the tax rate equals that of waste delivered to landfills with a high environmental standard.

9. The fact that hardly any British households pay a waste collection charge that varies with the amount of waste deposited obviously contributes to explaining this difference in impact.

10. As long as households only face flat waste collection charges, any increase in the tax rates of the landfill tax will have modest direct impact on household waste generation. The tax rate increases will, however, provide increased incentives for municipalities to stimulate recycling programs, etc. They could also make municipalities more interested in applying unit-based charging systems, but at present there are legal limitations to their use.

Since the introduction of the tax, significant changes in the handling of waste have taken place. In 1998, 43% of household waste was landfilled, while 33% was recycled and 23% incinerated. In 2002, the share of landfilling had dropped to 24%, while those of recycling and incineration had increased to 45% and 31% respectively.

To obtain a correct pricing of the environmental costs of waste treatment, an emissions tax would be preferable to a tax on the amounts delivered to landfills or incinerators. However, an emissions tax requires measuring of the actual emissions.¹¹ It is not yet possible to measure emissions from landfills, but a separate tax on emissions from incineration of waste has now been designed, with tax rates based on estimates of the economic value of environmental damages caused by emissions from incinerators. This emissions tax will give incentives to reduce emissions from waste, i.a. by cleaning emissions.

According to the paper, more than half (approximately 21 out of a total of 40) of the total estimated environmental cost of incinerating an average tonne of waste stems from emissions of hazardous chemicals. Of particular importance are emissions of chrome and manganese, with estimated values of 7.9 and 6.6 per tonne waste respectively. Despite a very high cost per unit emitted (almost 283,000 per gram) used in the estimation, dioxins “only” contribute 2.9 per tonne waste. Emissions of non-greenhouse gases (e.g. NO_x, SO₂, VOC) contribute about 8 all together to the total estimated economic value of the environmental damages from incineration, dust adds about 5.3 and greenhouse gas emissions about 4.8 .¹²

The taxation will be based on continuous measurements for the non-greenhouse gases and for dust, and two measurements per year for heavy metals and dioxins.

For greenhouse gases, the situation is more complicated: While incineration of waste fractions that contain plastics or carbon from other fossil matter cause net emissions of CO₂, incineration of biological waste does not cause *net* climate gas emissions. Even if one *can* measure *total* CO₂ emissions directly, it would only be relevant to tax the *net* CO₂ emissions. As it anyway is not possible to measure the net emissions directly, it has been decided to base the CO₂ component of the tax on the *weight of the waste* incinerated. Plants that can prove that they do not burn any fossil waste will be exempted from the CO₂ component.

The measurement obligations described here are based on requirements already contained in the directive 2000/76/EC of the European Parliament and of the Council on the incineration of waste, and thus represents few additional burdens on the plants.

The Norwegian Parliament has decided to replace the differentiation of the tax on incineration plants according to the degree of utilisation of energy with a subsidy dependent on the amount of energy produced from waste. This subsidy will also cover production of energy from landfills.

11. Despite frequent references to emission taxes in economic text books, very few taxes on measured or estimated emissions are in fact levied in OECD member countries. Among the approximately 1500 tax-bases detailed in the OECD/EEA database on environmentally related taxes (cf. www.oecd.org/env/tax-database), only about 30 tax-bases refer to measured or estimated emissions to air or to water.

12. The estimated externalities of incineration quoted here can seem rather high compared to the estimated externalities of landfilling presented in the context of the UK landfill tax • especially as both papers can seem to agree that the externalities related to landfilling tend to be higher than the externalities related to incineration.

6. Costs and Benefits of Alternative Treatments of PVC Waste in Denmark

The next case study, by Niels Buus Kristensen, looks at the costs and benefits of alternative treatments of PVC waste in Denmark, building on a new study undertaken for Danish authorities. It compares the costs and benefits of potential future implementation of two chemical treatment processes for PVC waste as alternatives to the conventional disposal via incineration or landfilling. One of the processes is based on pyrolysis, the other on hydrolysis.

The current Danish waste strategy states that PVC waste should be separated out and that PVC waste which cannot be recycled *should* be landfilled • and not incinerated • because of the environmental hazards related to the flue gas resulting from incineration of the chloride and heavy metal content in PVC. However, at present, the largest part of the PVC waste *is* incinerated.

In addition to a user fee to cover the waste treatment costs, waste producers also have to pay a waste tax amounting to 50 per tonne for landfilled waste and 44 per tonne for incinerated waste. Denmark also levy product taxes on commodities made of PVC and on phthalates, the plasticizer in flexible PVC.

Treatment costs of landfilling PVC waste were estimated to be about 76 per tonne *mixed* PVC waste. Treatment costs for incineration were found to be 205 per tonne *mixed* PVC. The high costs are primarily attributable to the need for additional lime and lye for flue gas cleaning and the costs of disposal of the flue gas cleaning residue. Impacts of two different scenarios for PVC sorting and collection were studied for chemical treatment processes. For the pyrolysis-based approach, treatment costs were found to be in the range 130-190 per tonne *mixed* PVC waste, while for the hydrolysis-based technology, treatment costs were estimated to be in the range between 85 and 160 per tonne *mixed* PVC waste. The ranges for the treatment costs relates to whether small or large volumes are treated by the processes. The unit costs for chemical treatment as well as incineration and landfilling are also sensitive to other assumptions as well as uncertainties regarding input figures.

The treatment costs are, however, only on part of the total private costs. In general it was found that the two chemical treatment processes would have significantly higher collection and transport costs than both landfilling and incineration. Regarding external costs, these were found to be highest for incineration, and lowest for landfilling.

Per tonne *pure* PVC, the four scenarios with chemical treatment have higher costs than a Reference Scenario, even though the hydrolysis process appeared to be less costly than incineration. The reason is that in all four alternative scenarios about 80% of the volumes for chemical treatment comes from landfilling because only small volumes can be transferred from incineration to chemical treatment. The additional costs varies between 90 and 210 per tonne PVC waste transferred to chemical treatment. Net present value for the total costs over the period 2000-2020 is 18-30 mill higher than for continued incineration and landfilling.

The included environmental costs are also in most cases somewhat higher for the chemical treatment scenarios. However, this should be assessed against the environmental effects which are not included in the cost-benefit analysis.

To conclude, the paper indicates that the current treatment system in total is less costly than either of the scenarios with chemical treatment. It should also be taken into account that if additional PVC waste is sorted out from incinerated volumes, this could also be landfilled with lower costs than chemical treatment. A decision about utilisation of one of the two chemical treatment processes should therefore depend on the political willingness to pay for avoiding those environmental risks that are *not* included in the analysis. These are possible leachate of phthalates and heavy metals from potential

uncontrolled discharges from landfills and possible leachate of heavy metals from landfilled flue gas cleaning residues and from the recycling of incineration slag in road construction, etc.

7. Efficient Targeting of Waste Policies in the Product Chain

The paper by Richard Porter discusses a broad range of waste policy options, from the perspective of how best to target them in the product chain. This means making sure that the actors at each phase of a product's life – from its birth to its death – face prices that reflect the marginal social costs of their actions. He points out that actors may face incorrect prices for two main reasons: waste handling often generates external cost, which means that part of the social cost is passed on to third parties, and waste handling is often subsidised, which means that part of the marginal private cost is paid for by government funds.

Porter explains that the marginal private cost of waste disposal consists of the extra costs of the equipment, wages and the opportunity cost of the land that are needed for the collection and disposal or recycling of one extra unit of trash. The marginal external costs consist of the noise, litter, dust, unsightliness, and potential air or groundwater pollution that are generated by one extra unit of trash collection and disposal or recycling. The marginal social cost is simply the sum of these two kinds of cost, the marginal private cost plus the marginal external cost.

In general the cost of collection is covered either from general revenues or from a time-based charge on residents, i.e. an amount per month or per year that is not related to the amount of trash being put out for pickup and disposal. Whether the general fund or a special time-based trash charge is used – the marginal private cost of putting out an additional unit of waste is zero. And, if a time-based charge is used, it makes no difference whether the total revenue collected from households covers the total collection cost – the marginal private cost of putting out an additional unit of waste is zero.¹³ The “dead-weight loss” created by the lacking marginal price incentives is roughly estimated to 2.4 billion USD for United States alone.

Porter points out that businesses often *do* cover the marginal private cost of waste disposal through payments to private waste haulers and the “tipping fee” they pay to a landfill or incinerator. However, most manufacturers escape responsibility for some of the waste they generate – namely the waste created by the packaging of their products and the waste created by the products themselves when their useful life has ended.

Porter discusses in detail different policy options in three different situations:

- A case where there is no possibility for recycling and littering (illegal dumping);
- A case where there is recycling, but still no littering;
- A case where both recycling and littering might take place.

His preference is for an advance disposal fee on products and packages equal to their net recycling cost,¹⁴ combined with a marginal waste charge on households equal to the excess of the collection-and-landfill disposal cost over the average net recycling cost of household solid waste. The

13. This observation underlines that the so-called “User Pays Principle”, or an obligation on municipalities to recover the full costs of given services, does *not* in itself lead to an efficient pricing of the services provided.

14. The cost of collecting and sorting a recyclable material minus the revenue earned on its sale.

advance disposal fee encourages manufacturers to make products and packages more cheaply recyclable. The marginal waste charge encourages households to reduce, reuse¹⁵ and recycle.

In a number of circumstances the application of a marginal waste charge could be less beneficial:

- a) When illegal disposal is, or is expected to become, a serious problem;
- b) When administrative and monitoring costs are high;
- c) When it is impossible to organize a charge system that does not seriously increase the tax burden of the poor;
- d) When multi-family dwellings dominate the municipal landscape.

If net recycling costs are high as compared to the waste collection and disposal costs, the appropriate marginal waste charge would be small and hence little efficiency gain would be achieved.

Porter stresses that there is no easy formula for deciding what mix of taxes and subsidies is appropriate for waste handling. It could therefore be tempting to rely entirely on non-price policies. But all non-price policies try to get people to do things that are not in their personal economic interest to do, and while there is no limit to the inefficiency that a badly chosen non-price regulation can cause, price policies self-limit their damages, no matter how badly chosen.

The paper ends by suggesting why non-price policies are so often preferred. First, they are easy for policy makers to apply – one can ban or require something without knowing marginal benefits and marginal costs and optimal prices or taxes. Second, they make more immediate sense for non-economists than price-based policies. Third, many waste professionals and policy makers do not believe that changing prices would change behaviour. And fourth, every non-price policy hides the cost of the policy. As a result, non-price policies become acts of faith, and they lead the waste-policy focus to the poles of nothing-discarded-everything-recycled and nothing-recycled-everything-discarded. The optimum is somewhere in-between, and only a greater emphasis on price policies can lead us toward it.

8. Targeting Lead in Solid Waste

Hilary Sigman's paper is a materials-specific case study on optimal targeting of waste policies. Many countries have made significant efforts to reduce exposure to lead, most importantly by phasing out lead additives in gasoline. Other policies target exposures from lead in paint, in food containers, and in drinking water from old lead-bearing pipes. Lead in waste may give rise to human and environmental exposure after wastes are incinerated or disposed in landfills. Sigman's paper discusses policy options to reduce lead discards and quantifies the effects and costs of several policies for battery recycling in the U.S.

The paper indicates that there was a dramatic decline in lead in municipal solid waste between 1985 and 2000 in United States • largely due to an increase in the recovery of lead from car batteries.

15. Whether increased reuse as such • or an extended lifespan of products • necessarily leads to lower waste amounts is questionable. An extended lifespan of a given product would represent a saving of expenses for the households concerned, and the households will normally use the money saved to purchase other products or services. Hence, it is an empirical question whether the amount of waste generated by this increase in consumption possibilities would outweigh the direct impact of an extended lifespan on waste amounts or not. The answer will vary from household to household, depending on how they spend the "extra money" available to them.

At the same time, there was an increase in lead disposed in consumer electronics, in particular the lead in cathode ray tubes (CRTs) from televisions and computer monitors. CRTs now far surpass batteries as a source of lead. However, flat panel display monitors and televisions contain dramatically less lead than their predecessors. Hence, current lead disposal in monitors may turn out to be a temporary phenomenon.

The environmental harms from lead in waste depend on whether the waste is sent to a landfill or an incinerator. In a landfill, the concern is that contaminated leachate may reach groundwater. Given leachate containment and inexpensive alternatives to the use of any contaminated groundwater, Sigman refers to Macauley et al. (2001), who found that land disposal of CRTs imposes few health costs in the U.S. • but she also states that the long-term fate of lead in landfills and the success of post-closure care assurances are unknown.

Combustion poses a bigger immediate risk, as it is likely that a substantial share of lead discarded goes into the incinerator along with other waste. This increases the toxicity of residual ash and may be emitted to the air. Nonetheless, Macauley et al. (2001) find fairly low costs from combustion of computer monitor CRTs in the United States. They conclude that the health costs amounts to \$2.67 million annually. Sigman states that this may be an overestimate because it gives no role to sorting at the facility.

The paper discusses costs and impacts of five policy approaches to limit lead battery disposals:

- Deposit-refund systems
- Taxes on lead
- Subsidies to recycled lead
- Recycled content standard
- Producer responsibility requirements

While the most common approach for a deposit-refund is to place both the deposit and refund on the consumer, Sigman finds that better approach would be to impose the deposit-refund at the producer level, with a charge for lead use in production and a subsidy for recovered lead. That could lower the administrative costs and provide greater incentives to assure that lead collected from consumers is recovered because the refund is not payable otherwise.

A tax on all lead would raise the cost of lead, and thus discourage its use, but would not have a direct effect on recycling. A virgin material tax applied only to primary lead raises the price of primary lead to its users. Because primary and secondary lead are substitutes, users will purchase recycled lead at nearly the same price. Thus, the tax will also raise the price of recycled lead. As a result, the virgin material tax is similar to a deposit-refund in its effect.

A recycling subsidy would lower the costs of recycled lead relative to virgin lead. This should reduce virgin lead production. However, the price of lead declines because recycled lead has become cheaper. Thus, the recycling subsidy creates an incentive for *increased* consumption of the lead.

A recycled content standard could e.g. be implemented at the level of individual products, of individual firms or across all users of lead. Firms could meet the latter, most flexible, requirement by trading recycled content. Then the standard's effect would be as if the government collected a virgin material tax equal to the permit price and used all the money it collected to give a recycling subsidy. The recycled content standard should increase the recycling rate and make lead more costly, as firms

must spend resources either to buy permits for the use of primary lead or to use secondary lead when primary lead might be cheaper.

Governments might also set rules requiring that producers take back lead-containing products. The principal example of this sort of policy for lead is the EU's WEEE Directive.¹⁶ Sigman points out that such requirements resemble a tax and subsidy combination. They create large effective subsidies to recycling because of the funds the producers must spend on collection of scrap products and their recovery. To cover the costs of these subsidies, producers will increase the prices of their products to reflect the additional costs. In most markets, prices will increase by most or all of the costs of the expected collection and recovery costs. Thus, there is also an effective tax on the purchase of the product.

The paper concludes that price-based recycling policies can effectively increase lead recycling, but the policies differ substantially in the costs of accomplishing a given reduction of lead in waste. A recycling subsidy entails nearly twice the private costs of a deposit-refund, with a recycled content standard intermediate in costs. Sigman finds that this ranking also applies to policies aimed at other sources of lead in municipal solid waste, including consumer electronics.

Despite the effectiveness of price-based policies, Sigman refers to earlier studies that suggest the need for caution in pursuing policies that reduce lead in solid waste. For countries with already high recovery rates of lead from batteries, such as the U.S., it may be that the environmental gains of reduced lead disposal are not high enough to merit the cost and environmental consequences.

9. Changing Product Characteristics to Reduce Waste Generation

Matthieu Glachant's paper discusses policies that may efficiently encourage innovation reducing waste at source through changes in product characteristics. Municipal waste is a by-product of the consumption of goods designed upstream by producers. Waste policies should thus seek to influence the behaviour of consumers, retailers, and producers. In this respect, as waste production is tightly correlated with product characteristics, a key goal is to foster product changes. Product change is primarily an economic process in which firms introduce new products with design characteristics reducing waste at the post-consumption stage. A first challenge for waste management policies is thus to provide producers with the appropriate incentives to innovate. Furthermore, less waste-intensive products also need to be bought by consumers. In the end, the question for waste management policies is how to create market conditions favourable to the production and consumption of these goods.

Glachant lists three possible • and possibly contradicting • waste policy objectives related to product change:

- To reduce the quantity of waste generated by consumption;
- To reduce the toxicity of the waste generated; and
- To facilitate recycling or re-use.

The paper distinguishes between two broad product categories differing in waste-related design patterns. A first group includes packaging and non-durable goods, where the challenge mainly is lightening, reducing the development of small containers, and introducing lighter and/or more easily recyclable material. The second group covers durable products such as electronic equipments,

16. Cf. European Union (2003).

household appliances and cars. A significant proportion of their metal contents is already recycled. In this context, complete product redesign might be necessary. This requires radical innovation as opposed to incremental changes that are at stake for non-durables and packaging. A further difference is that durable products are much more complex and embody a higher number of materials. These differences imply differentiated policy consequences depending on product category.

Glachant emphasises that “Business-as-Usual” product changes can have dramatic impacts on waste generation. Therefore, the goal of waste policies is clearly not to *initiate* product change. Instead, the challenge is to *modify the pattern* of Business-as-Usual product change in order to position goods on less waste-intensive innovation trajectories.

He also points out that innovation is risky, and that it represents an investment where the innovator bear innovation costs now in order to obtain future benefits. And innovation costs are *sunk*, in the sense that they cannot be recovered should the innovation project be withdrawn. In order to be effective, policies should reduce as much as possible the level of risk surrounding product innovation. It primarily requires a long-term stability of the policy signals, or at least a predictability of their changes.

The paper stresses that innovation outcomes may benefit others, in particular competitors, through imitation. Imitation is particularly of concern for product innovation as opposed to process innovation. The concern is that the resulting innovation is embodied in the product which is sold in the market. It is then fairly easy for any imitators to exploit the innovation by analysing the product – a process called reverse engineering. Innovators cannot appropriate all the benefits of their innovation. Therefore, they have reduced incentives to innovate. However, imitation also leads to the diffusion of the innovation, thus creating benefits in the economic system. There is thus a trade-off for policy makers here. On the one hand, one should protect innovation incentives. On the other, one should promote innovation diffusion.

There are three generic strategies to solve the dilemma. First, the regulator can grant intellectual property rights such as patents and copyrights. The second solution is public research or publicly funded research. Third, firms can form research consortia, thus mitigating the imitation problem.

In the waste area, co-operation among firms is observed and usually takes the form of so-called Producer Responsibility Organisations (PROs). However, to date, PROs have not engaged much in research co-operation in order to minimise waste generation.

The paper goes on to discuss the impacts various features of PROs might have on firms’ incentives to innovate to limit waste amounts, facilitate recycling, etc. A crucial point is how each producer’s financial contributions to the PRO are calculated. Under an individual regime, each producer contributes on the basis on its *own* products’ collection and processing costs. Here the innovation incentives are strong. Alternatively contributions can be based on e.g. market shares, with no direct relationships with individual producers’ product characteristics. Such schemes fail to provide producers with incentives to alter their products.

Individual financing schemes is common in PROs dealing with packaging waste. Here the unit weight by material of a specific branded product is relatively easy to monitor; furthermore, it provides incentives for lightening and material substitution which are a key part of packaging waste prevention. In the case of more complex products, designing incentive fees is far less feasible. In particular, many more materials are embodied in durable goods like cars or computers and the dismantling ability is a key factor of waste prevention and recycling. As a result, waste prevention generally requires a complete redesign of the products and it is difficult to imagine a product fee providing the adequate incentive. It is thus not surprising that “collective” financing schemes are more widespread for these products.

The paper assesses a number of instruments which can be used to meet waste policy goals, with an emphasis on their impact on innovation incentives – and finds that taxes and charges are more likely to induce innovation than standards. This is so because taxes or charges always provides firms with incentives to innovate. By contrast, such incentives vanish once regulatory thresholds are met.

A distinction is made between “upstream” and “downstream” instruments. It is pointed out that ex post evidence is very scarce on whether upstream taxes • like advance disposal fees paid on each unit of product sold in the market reflecting the disposal cost of the product • induce product changes in practice. However, a comparison between the amounts of packaging used in 2000 with a hypothetical trend in the absence of the German sales packaging program Dual System indicates an 18% reduction.

Downstream instruments, like taxes on final waste disposal, might also be useful tools influencing product innovation. In market economies, the ultimate impact of product design on waste streams depends on the commercial success of re-designed products. In this way, consumers play a crucial role and downstream instruments may better target them. A necessary condition for downstream policies to influence upstream design decisions is to provide consumers with incentives to modify their purchasing behaviour.

Glachant finds that if one seeks significant impacts, the only possibility is to implement unit-based waste collection charges to households. At present, the influence of households subject to unit pricing on producers is *de facto* limited by their dilution among the vast majority of consumers facing flat rate waste collection charges.

10. Issues for Further Work – Conclusions

Based on the papers prepared and the discussions that took place during the workshop, a number of areas where OECD could usefully do additional work have been singled out. These include:

- a) Analysis of what are the main “drivers” for the generation of different categories of waste.
- b) Quantification of the marginal social costs on different waste handling options for different types of waste • in different circumstances.
- c) Evaluations of the impacts of economic instruments, like unit pricing schemes for household waste collection and advanced disposal fees.
- d) Evaluations of specific policy options, like extended producer responsibility schemes.

Subject to the availability of resources, these issues will be addressed further in the coming years. The intention is to give the work a practical • rather than theoretical • focus, and to address several different types of waste.

To conclude, the workshop clearly demonstrated that more economic analysis is warranted in the waste policy area. It was also clear that despite a considerable amount of work on waste economics undertaken so far, additional work is needed. It would be useful to know more about what “drives” the amounts of waste being generated. The environmental externalities caused by different types of waste also needs further clarification, both as regards the physical impacts caused by different

handling options¹⁷ • in different geographical settings • and regarding the economic value that should be attributed to the different physical impacts. While most economists would agree that unit pricing for waste collection would cause some, but relatively limited, reduction in the amounts of waste generated, there is a need to analyse the issue further • and to get a clearer picture of the impact such pricing will have on illegal dumping of waste.

Some waste-related targets and policies have been accused of causing significant net costs to the society.¹⁸ OECD can play a useful role by addressing the validity of some such accusations • and by providing suggestions on how policies might be modified, so that given policy targets can be reached at lower overall costs.

17. It is worth noting that some of the quantifications of physical impacts underlying e.g. evaluation studies referred to in the papers discussing the Landfill tax in the United Kingdom and the Waste Tax in Norway are more than 10 years old.

18. See for example Radetzki (2000).

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Chapter 2

WASTE GENERATION AND RELATED POLICIES: BROAD TRENDS OVER THE LAST TEN YEARS

By Soizick de Tilly¹

1. Introduction

The primary purpose of this paper is to give an overview of the waste generation and management situation in the OECD countries during the past decade. It describes the broad trends and their impact on the environment, and the policies put into practice. Most of the findings are based on OECD Environmental Performance Reviews and Economic Surveys. We have chosen the most recent studies, giving preference to those which cover both the environment and the economy. The list of countries reviewed is contained in an annex.

However, the information contained in these reviews does not at the present time supply all the necessary data for an in-depth analysis and assessment of policy. Other studies carried out either by the OECD or by outside researchers or organisations have therefore also been used.

We then put forward some thoughts on the effectiveness of waste reduction policies and the conclusions that may be drawn.

2. The findings

2.1 *Municipal waste generation*

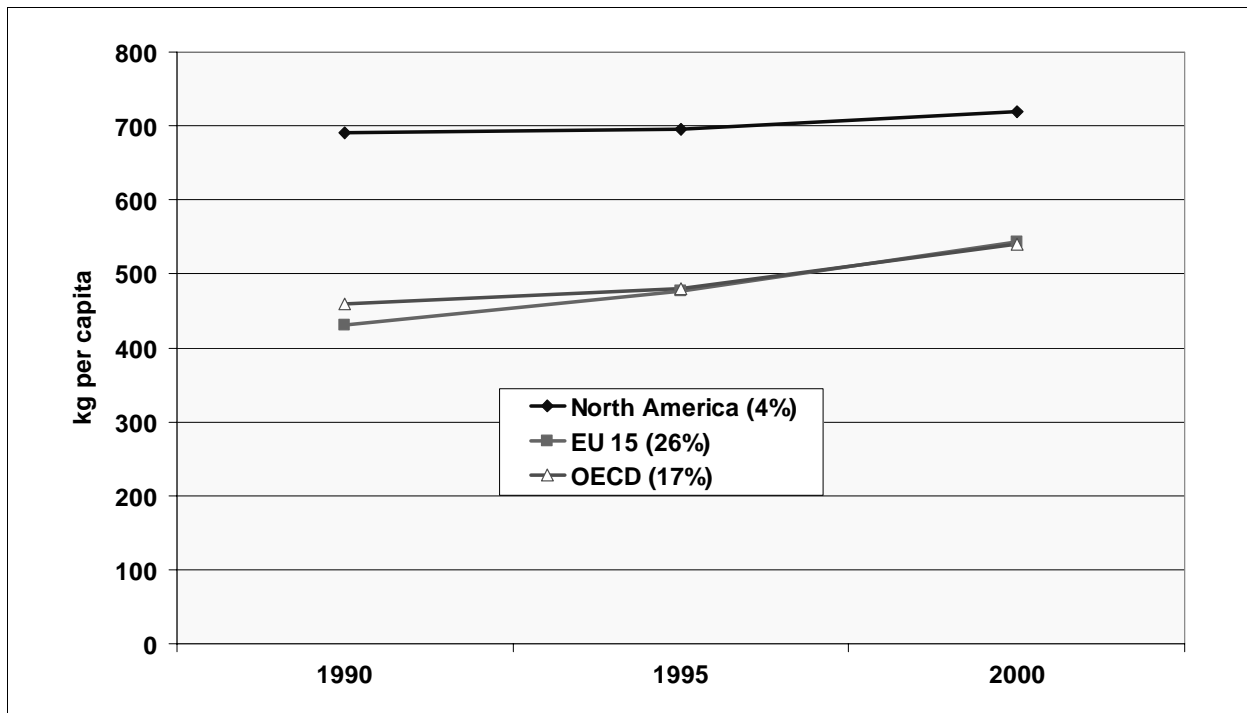
Municipal waste generation continued to rise in OECD countries between 1990 and 2000, not only in absolute terms but also on a per capita basis. This means that population growth is not the only cause of increased waste.

- Municipal waste generation increased by 14% over the period, from 530 to 605 million tonnes.
- Per capita, it increased from 509 to 540 kg on average, a rise of 6%.

The population of the OECD countries increased by 8% over the same period.

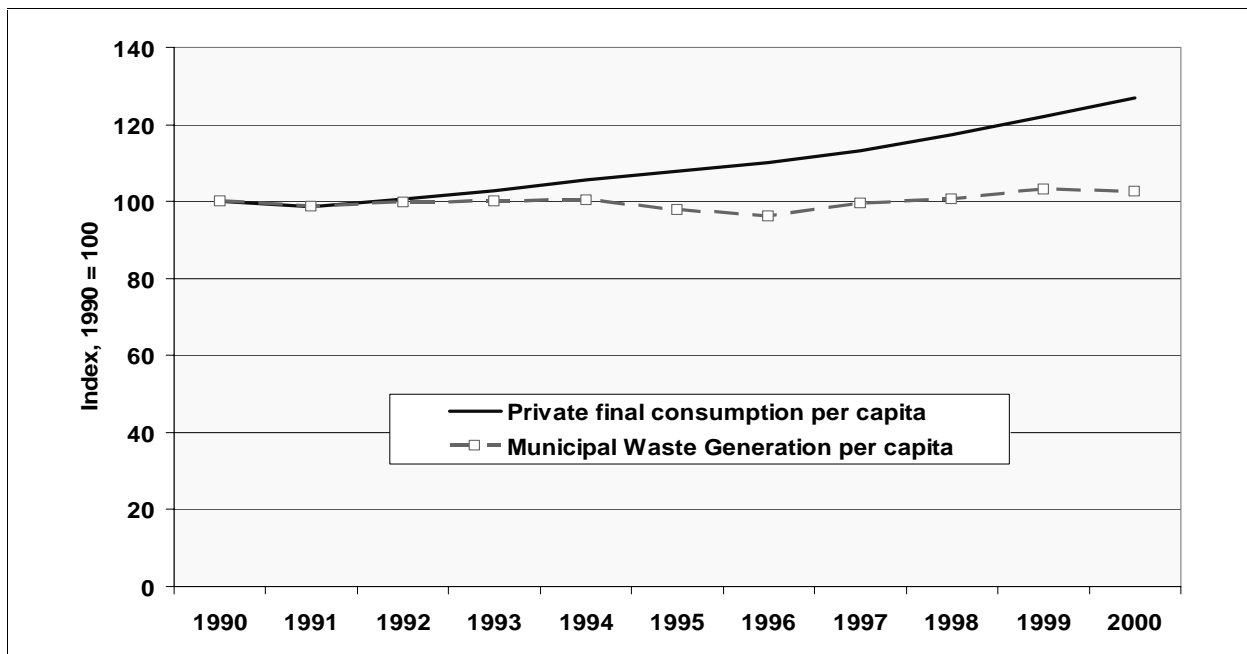
1. National Policies Division, OECD Environment Directorate. The opinions expressed in this chapter are those of the author and do not necessarily reflect the views of the OECD.

Figure 1. Municipal Waste per Capita
Unweighted averages



Source: OECD.

Figure 2. Municipal Waste Generation and Private Final Consumption
United States



Source: OECD.

This is the broad trend, but there have been some rare cases of a slowdown (USA, Japan) or a reduction in municipal waste generation (Korea, Germany), though we cannot be certain whether this

is due to a change in definition or a different way of calculating waste, as was the case with Germany in 1994.

Many factors influence the generation and type of waste, either positively or negatively, but in all events the outcome is increased waste generation, as we have seen. Some of these factors are as follows.

- Economic growth, coupled with enrichment of the population: higher incomes generally lead to higher consumption, for example of domestic appliances and electronic devices such as computers and mobile phones. In Italy, the regions with the highest standard of living produce more municipal waste per inhabitant:
 - Southern Italy (Molise region): 347 kg/inh., (average income per inhabitant: 12,000)
 - Northern Italy (Bologna region): 606 kg/inh., (average income per inhabitant: 20,400)
- Population growth and structure: studies (Van Houtven and Morris, 1999) have shown that households comprising young children and persons aged 25 to 64 produce more waste. However, ageing populations in OECD countries are likely to mean a corresponding reduction in waste generation.
- Growth in the number of households, linked both to growing populations and smaller households: the number of households in Finland increased by 71% between 1970 and 2000, but their size fell by 35% to a current level of 2.15 persons per household (Statistics Finland, 2002). One consequence is an increase in the number of small-unit food products and hence in the amount of packaging, thus increasing per capita waste generation.
- Growing urbanisation of the population: in general, urban dwellers have higher incomes than rural populations, generating greater consumption of goods and services in highly urbanised areas. However, waste collection and recycling are easier to organise in such areas. This may wrongly suggest higher levels of waste generation, whereas it actually reflects better waste management.
- The structure of consumption: the growth of the services and leisure sectors leads to dematerialisation. This in turn reduces the generation of certain types of waste but increases the generation of other types, such as computers; it also leads to the displacement of waste generation, as in the case of waste related to tourism.
- Socio-cultural habits: a more individualistic lifestyle leads to the multiplication of goods, such as cars and individual portions of food products. Less time spent on domestic tasks at home leads to greater consumption of ready-to-eat meals or home deliveries and hence to more food packaging. Socio-cultural habits, the sense of civic solidarity and “environmental awareness” vary considerably from one country to another. Scandinavians, more sensitive to environmental quality and protection, choose greener products which generate less waste and sort waste for recycling in a very disciplined way.

The variable and sometimes contradictory influence of these many factors on waste generation underlines how difficult it is to assess the determining driver(s) and to make forecasts. For example, OECD Economic Surveys suggest that a sharp rise in waste generation, as in Ireland or Korea, is due to high GDP growth and rising private final consumption (Ireland’s GDP increased by 61% between 1995 and 2000, and municipal waste generation by 42 %.) Yet the correlation coefficient between the annual growth of private final consumption and municipal waste generation in OECD countries between 1990 and 2000 is only 0.18. Other studies carried out in the United States and Korea (Kinnaman and Fullerton, 1997; Hong, 1999, etc.) also show that municipal waste generation is relatively inelastic in relation to income.

The question remains open. We shall not linger here on the respective influence of the drivers for waste generation, which is debatable and could be the subject of a study in itself, but focus rather on the circumstances that have led to the introduction of different waste management policies.

Table 1. Growth of Municipal waste generation compared to that of some waste generation drivers (in %) between 1990 and 2000

	Municipal waste generation	Population	Gross Domestic Product	Private final consumption (1)
North America (2)	13	10	37	39
Asia-Pacific (3)	-11	5	25	25
OECD Europe	23	5	23	25
EU (15 countries)	26	3	23	23
OECD Total	14	8	29	31

(1) At 1995 price levels and purchasing power parities; estimates for East German, Hungary and the Slovak Republic in 1990.

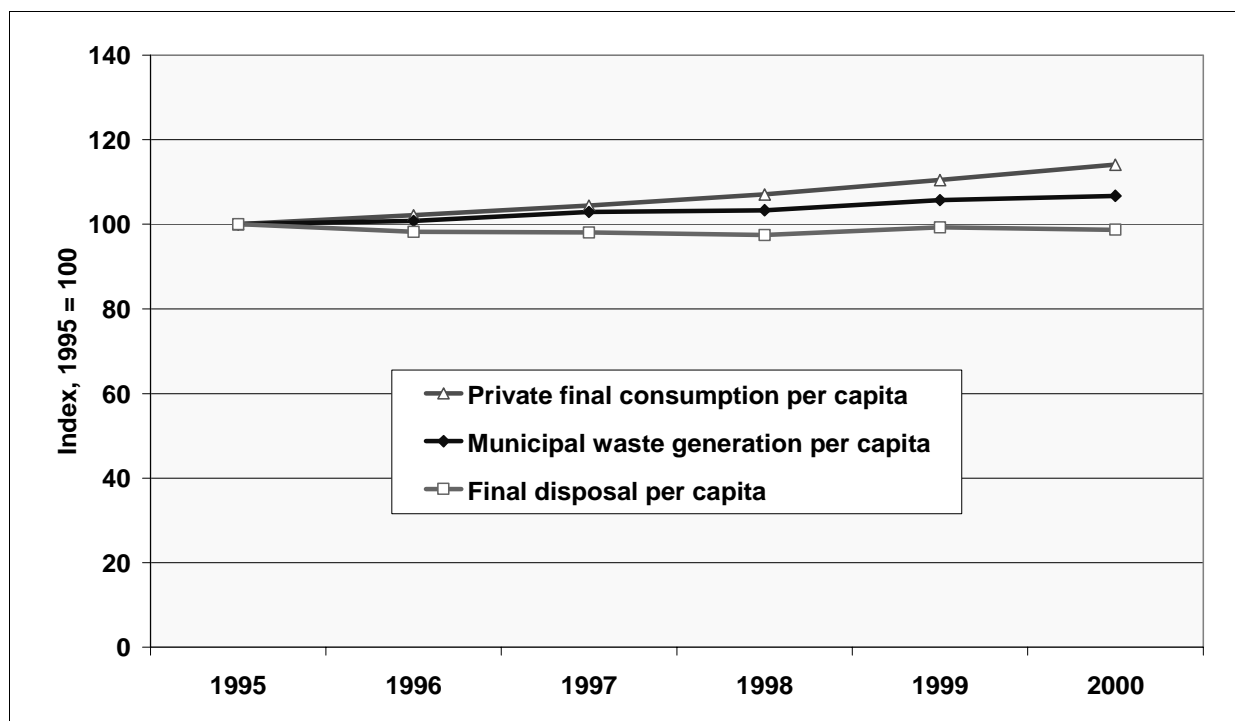
(2) Does not include Mexico.

(3) Includes Japan, Korea, Australia and New Zealand; the negative rate is due to municipal waste generation in Korea which decreased by 45%.

Source: OECD.

However, there is a slight decoupling between GDP growth or rising private final consumption and waste going to final disposal (landfill and incineration), which confirms the growing importance of the role of recycling and waste prevention.

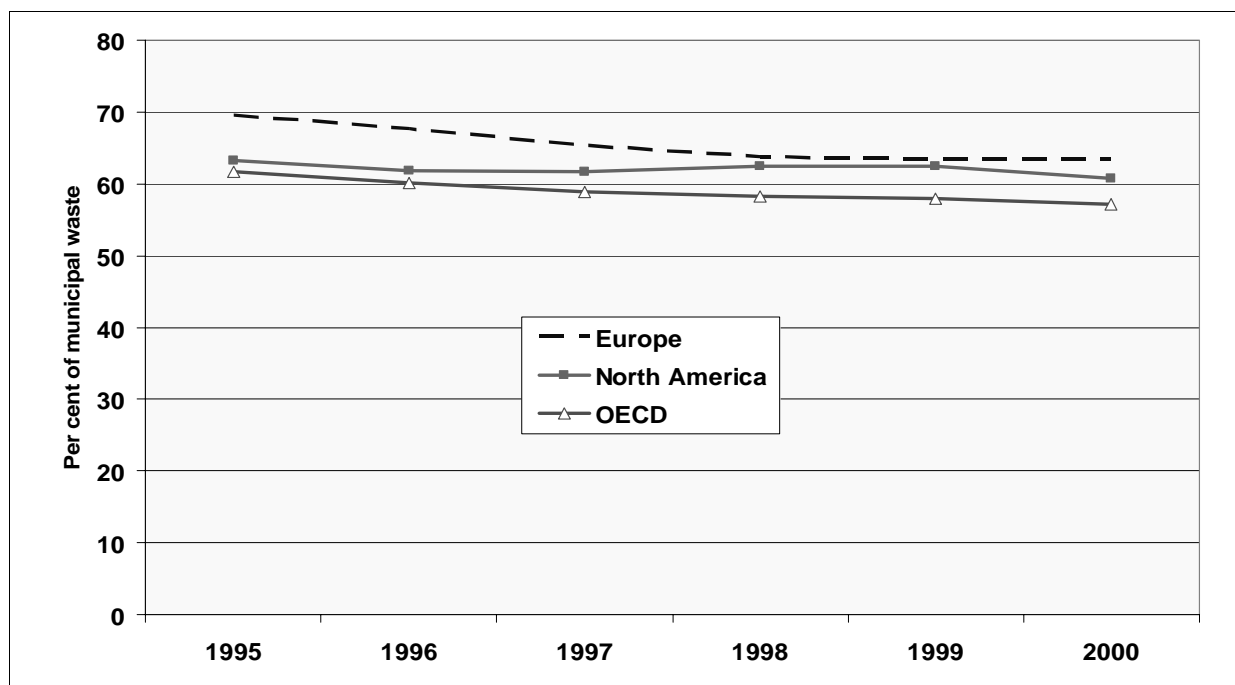
Figure 3. Private Final Consumption, Municipal Waste Generation and Final Waste Disposal OECD



Final waste disposal = Landfilling + Incineration without energy recovery

Source: OECD.

Figure 4. Share of Municipal Waste Destined for Disposal* in OECD Regions



* Final waste disposal = Landfilling + Incineration without energy recovery

Source: OECD.

2.2 *Municipal waste management*

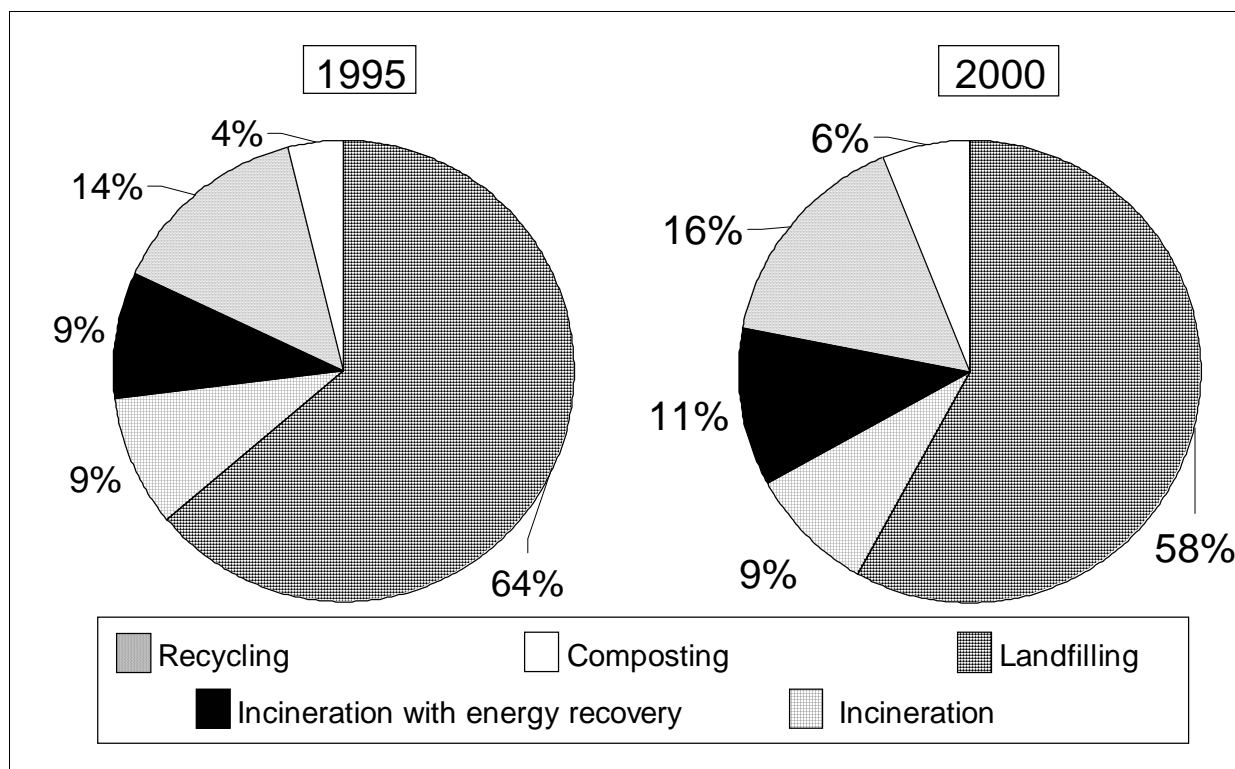
Although most waste is still put in landfills, this method of waste management is less and less prevalent and waste recovery² is increasing: municipal waste landfilling increased by only 2% between 1995 and 2000, while municipal waste generation increased by 10%.

There is also a broad and growing trend towards the incineration of municipal waste with energy recovery and the composting of moist organic waste.

Although these broad trends are valid for the OECD as a whole, each country has one or two predominant waste management methods, determined by its physical, economic and social characteristics and by its regulations. In the United States, for example, landfilling is the predominant waste disposal method, probably because land is plentiful and hence the cost is low. In densely populated countries like Japan, Denmark and the Netherlands, at least 50% of all waste is incinerated (78% in Japan) because these countries' incinerators benefit from economies of scale and incineration reduces the volume of household waste by 90%.

2. Recovery, according to the OECD definition [see all Council Acts relating to waste, especially Decision C(2001)107/FINAL], includes recycling, incineration with energy recovery and composting.

Figure 5. Municipal Waste Management in OECD Member Countries



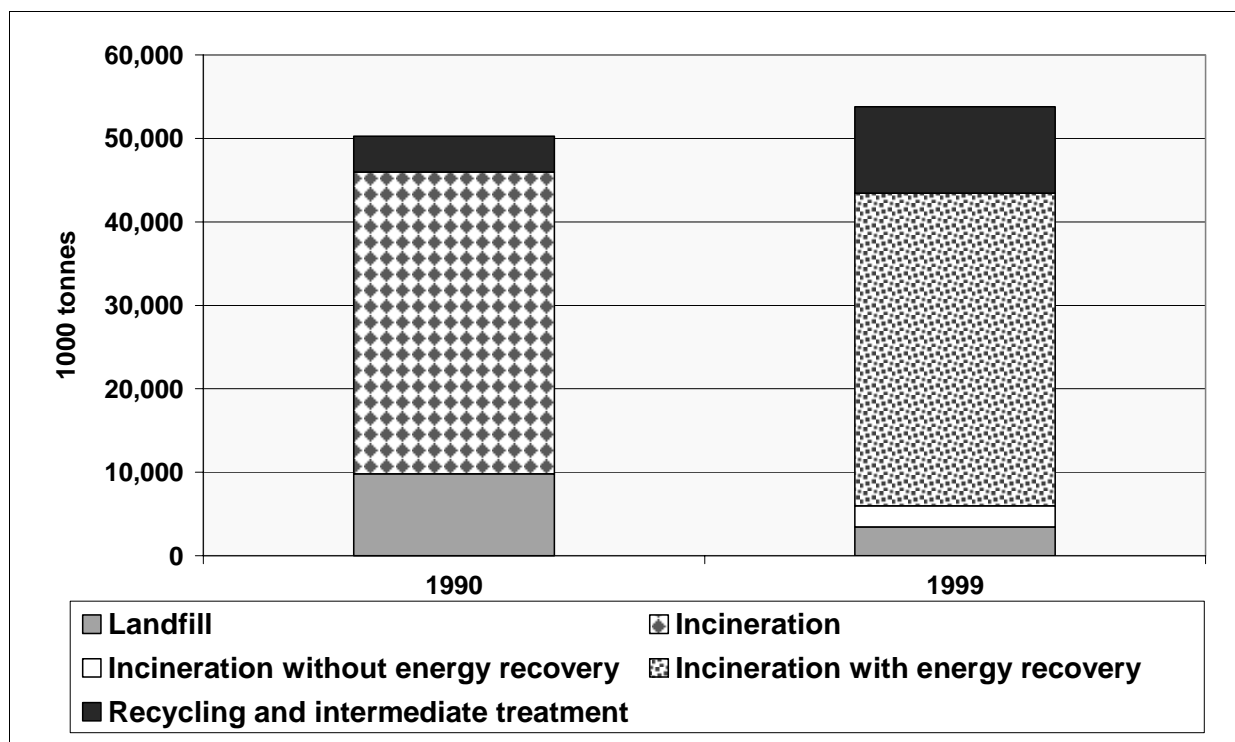
Source: OECD Environmental Data Compendium 2002, OECD Environmental Performance Reviews, OECD Economic Surveys, plus OECD Secretariat estimates.

There is also a broad trend towards improved recycling rates. Recycling rates differ according to the type of material: > 80% for metals, 35-40% for glass, 40-55% for paper and cardboard. Recycling rates differ considerably from one country to another: in Ireland, for example, 10% of paper and cardboard is recycled whereas the figure for Germany is 70%. All the Scandinavian countries have very high recycling rates, as do Belgium, Switzerland, Austria and Germany. The Mediterranean OECD countries (Greece, Portugal, Spain and Turkey) still have the lowest recycling rates, though they are rising fast.

It seems that once a certain level of recycling has been reached, the rate flattens out or even falls back slightly. This would mean that each country has a maximum recycling rate, beyond which technical and economic efficiency would no longer be achieved.

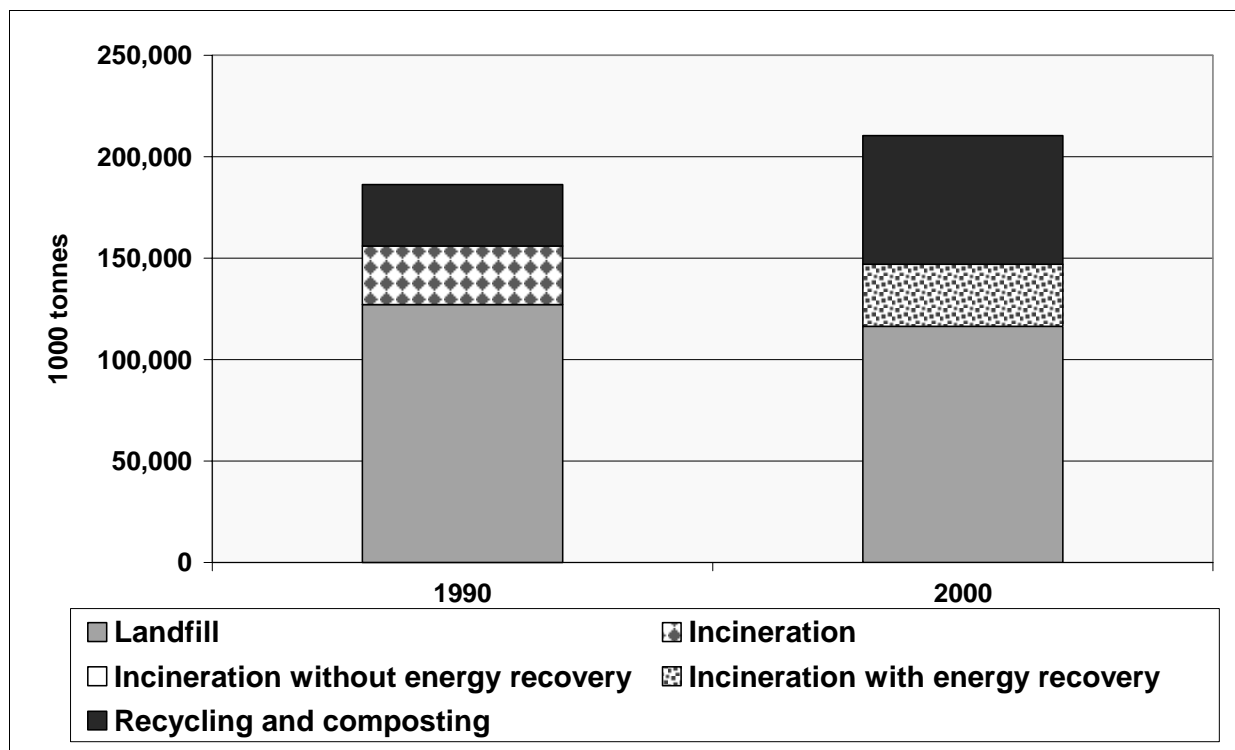
Broadly speaking, municipal waste generation is rising steadily (it is forecast to reach 640 kg/inh. for the OECD as a whole by 2020), despite all efforts to prevent waste generation and increase waste recovery in relation to disposal.

Figure 6. Municipal Waste Management - Japan



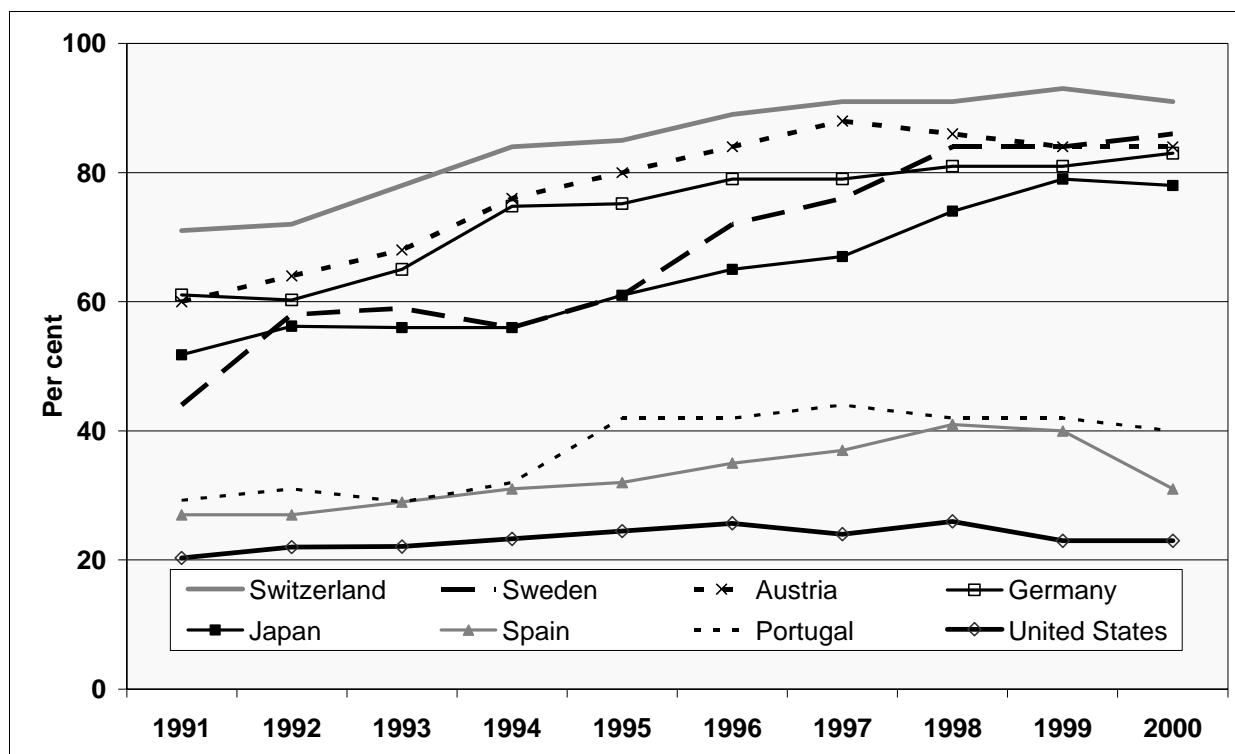
Source: OECD.

Figure 7. Municipal Waste Management - USA



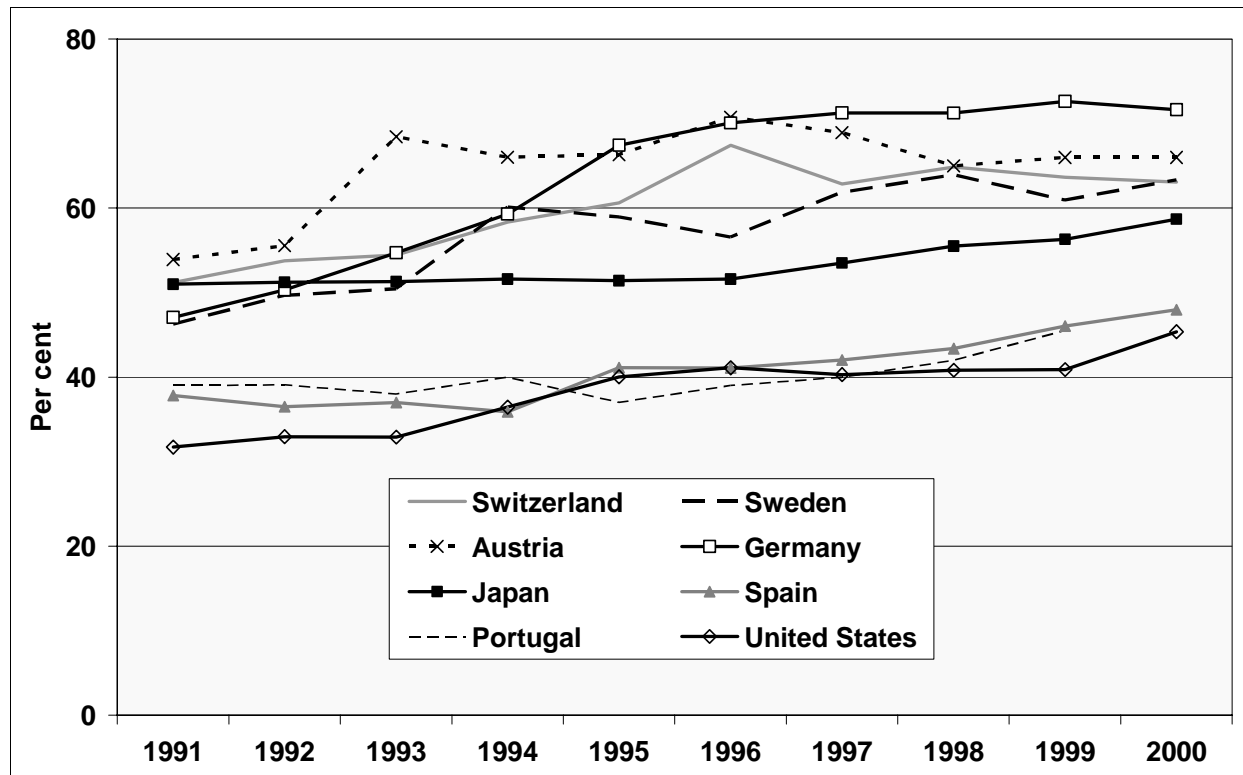
Source: OECD.

Figure 8. Glass Waste Recycling Rates



Source: OECD Environmental Data – Compendium 2002.

Figure 9. Paper and Cardboard Waste Recycling Rates



Source: OECD Environmental Data – Compendium 2002.

2.3 *Industrial waste generation*

Industrial waste generation is linked to economic growth and hence to GDP. For example, Korea, which has experienced rapid growth over the last decade, has seen a corresponding rise in industrial waste generation. In Ireland, the volume of ordinary industrial waste increased by 56% between 1995 and 1998. Industrial waste as a proportion of total waste naturally varies according to each country's industrial structure, the industries which produce the most waste being the chemical, pulp and paper and iron and steel industries.

Unfortunately, as industrial waste is generally managed at source by the industry itself and as information on the subject is regarded as confidential, data are difficult to obtain.

2.4 *Hazardous waste generation*

Hazardous waste is estimated to account for 3-4% of total waste on average. The characteristics of hazardous waste, produced mainly by industry, are the same as for industrial waste: generation increases in line with GDP growth, and the amount of hazardous waste produced varies enormously from one country to another according to their industrial structure. The lack of data on the subject and differences in definitions make it difficult to give any assessment for all OECD countries as a whole.

3. *The environmental impacts of waste management*

Waste has many impacts on the environment. They occur throughout the product life-cycle, from production through consumption to disposal; they are caused by a number of different players (humans, animals) and activities (eating, travelling, transporting, working, etc.) that are extremely varied in both type and location; and they affect many areas, such as water, air, vegetation, landscape, human health, etc. The range of impacts resulting from waste generation is therefore extremely wide in both space and time and this greatly complicates government action, as we shall see.

Broadly speaking, environmental impacts in the OECD countries have diminished over the last ten years, for various reasons:

- extensive regulation, especially concerning landfills and standards for incinerator emissions. This is particular true of Europe as a result of EU directives on landfill, incineration, the management of certain waste flows such as electrical and electronic waste, end-of-life vehicles, the content in certain materials of dangerous substances such as lead, mercury and cadmium, etc.
- the development of highly efficient technologies in areas such as the control of incinerator emissions, especially dioxins (for example, dioxin emissions in the United Kingdom fell by 90% between 1990 and 1997), recycling, the health and safety characteristics of landfill sites (methane emissions from landfills are falling in most countries since most sites have gas recovery systems).

However, problems still remain.

- In many cases, disposal capacity is insufficient, as with incinerators in the UK and Germany, and landfill sites in Japan, Ireland and Germany (see OECD Environmental Performance Reviews and Economic Surveys). The problem is all the more difficult to solve insofar as many plants and landfill sites prefer to close rather than meet new standards. It is

compounded by increasingly prevalent NIMBY³ attitudes, preventing any further expansion of disposal capacity. This leads in turn to higher prices for waste disposal and exports of waste to neighbouring countries.

- In other countries, in contrast, the availability of disposal capacity (such as incinerators in the Netherlands) encourages maximum use in order to amortise the investment, increasing movements of waste and the environmental impacts of transporting it. This situation may run counter to recycling policies.
- Emission regulations and standards are often not complied with. Unauthorised dumps and sites that do not meet the technical requirements may spring up, as in Italy and Ireland (OECD Environmental Performance Reviews of Italy in 2002 and Ireland in 2000, and OECD Economic Survey of Ireland in 2001). Incinerators may exceed air pollution limits, as with dioxins and furans in France (see “Aménagement du territoire et environnement – Politiques et indicateurs”, IFEN – DATAR, France 2000). Groundwater pollution may also exceed authorised levels, often due to leachate from landfills. Non-compliance may partly be explained by the lack or inadequacy of environmental monitoring and penalties and by higher taxes on landfill and final disposal or the introduction of “pay-as-you-throw” taxes, based on the weight of waste produced (Kinnaman and Fullerton, Horton).
- Poor waste management in the past can create an inherited burden. In several countries, unsupervised landfills have led to the long-term contamination of soil and groundwater. The costs of restoring the environment are very high and for that reason sometimes have to be assumed by central government, as is the case in Spain. This report merely touches on the matter without going into any more detail, since soil contamination is a related issue which deserves to be treated separately.
- The cost of managing waste, whether locally or nationally, is generally hard to evaluate. Consequently, local authorities set waste management prices that do not reflect the environmental externalities and fail to provide a coherent basis for the use of the different potential methods of waste management.

4. Measures and policies

The governments of OECD countries have introduced a certain number of measures and policies in an attempt to reduce the environmental impacts and costs connected with rising waste generation.

- Waste management planning is on the increase, generally at national, regional and local level. This includes forecasting the quantities of waste produced, collected and treated, setting objectives such as recycling rates for certain materials, and financing waste management. These plans all give priority to preventing and reducing waste generation through recovery; disposal by incineration or landfilling is a least desirable solution of last resort from a public health and environmental standpoint.
- Almost all the OECD countries now apply the principle of extended producer responsibility (EPR) for a certain number of products that pose end-of-life problems either because of their volume or because they contain dangerous substances. Producers, importers and distributors are generally required to organise the collection and recycling of these end-of-life products. Increasingly, they are getting together to sub-contract their obligation to specialist private organisations that cooperate with local authorities in organising the collection, sorting and

3. “Not In My Back Yard”, meaning that local populations refuse to allow polluting facilities to be located nearby.

recovery of materials. Collection and recovery rates everywhere have since improved considerably. The types of waste to which EPR applies are practically the same in all OECD countries: packaging, electrical and electronic waste, used tyres, used oils, batteries, end-of-life vehicles and batteries.

- Economic instruments are increasingly being used to internalise the costs of waste management and the environmental impacts of waste. These costs are passed on to economic agents, especially consumers, for example through taxes or charges on purchase (deposit refund systems) designed to finance the disposal or recovery and recycling of end-of-life products. End users who return the used product (end-of-life vehicle, used oil, electrical appliances, batteries, etc.) to approved centres can reclaim the tax or charge. The instruments most widely used in waste management are charges for the collection of household waste calculated either by weight or at a flat rate (per person, per household or according to the habitable surface area) and taxes on tipping or incineration. The level of these taxes, and hence their deterrent effect, can be adjusted to direct waste flows towards other, more environmentally desirable methods of disposal.
- The control and monitoring of hazardous waste, from generation through transport to final disposal, have been stepped up both within countries and outside their borders. In the Slovak Republic, for example, an information system tracks all hazardous waste by origin, quantity and method of disposal. In Japan, there is a tracking system for used oils, residual acids and bases, hospital waste and asbestos, and in Germany for all hazardous waste.
- Internationally, there has been a realisation of the need to cooperate in order to ensure better management of international movements of waste, reflected in:
 - the introduction of international systems to control exports and imports of waste (OECD decisions, European regulations, Basel Convention) and greater efforts to combat the illegal movement and disposal of waste. This can sometimes represent very substantial amounts of money, as in Italy, where in 1999 the value of the illegal traffic in waste was estimated at 6 billion euros (Environmental Performance Review, Italy – OECD 2002);
 - agreements seeking to ensure environment-friendly waste management, like the London Dumping Convention and the 1996 Protocol to the London Convention which prohibits the dumping and incineration of waste at sea.
- The infrastructure needed to treat waste is improving. The capacity to treat certain types of hazardous waste, such as PCBs, is being commissioned or adjusted at national level with the aim of achieving self-sufficiency in order to comply with certain principles of a political nature, especially at European level. From the standpoint of economic efficiency, however, such policies are open to question since they imply repeat large-scale investment and run counter to the achievement of economies of scale. Systems are being set up to collect environmentally harmful waste produced in isolated and limited quantities, such as hazardous household waste, lead batteries, used oils, etc.
- Administrative structures are being reorganised to cut costs and ensure more efficient management. In Japan, France and Denmark, for example, local authorities are getting together for the provision of waste removal and treatment services, generating economies of scale.
- More ecological waste management methods are being developed. In Ireland, for example, a ban was introduced in 1999 on discharging sewage sludge into the sea and better slurry spreading practices have been introduced. Increasingly, landfills may accept only inert or pre-treated waste; other forms of waste, like organic waste, inflammable materials and old tyres, are refused. New and cleaner waste treatment technologies are being developed, such as the automatic sorting of metals, batteries and plastics. New incinerators, whether for household

or hazardous waste, must comply with very strict limits on atmospheric emissions, especially of dust, hydrogen chloride and fluoride, sulphur dioxide, heavy metals, dioxins and furans, etc.

5. Conclusions

It is very difficult to draw an accurate and precise picture of the situation and trends in the generation of different types of waste because of the lack of reliable, comparable data.

- Definitions have changed: the terms “waste” and “hazardous” vary over time as regulations have changed, creating series breaks (in Germany, Ireland, the Slovak Republic, etc.) and suggesting enormous increases or the decoupling of waste generation and economic growth. This factor makes forecasting all the more difficult.
- The most detailed and reliable data concern municipal waste, which accounts for only a small proportion of total waste generation – approx. 14% – and is generally not hazardous by nature.
- There is very little information on which to base an assessment of the economic efficiency of the various instruments and types of approach, according to whether they are voluntary or coercive and the level at which they are implemented (national/federal, regional, local).

Economic efficiency takes second place to environmental effectiveness as the principal concern in framing environmental policy. In addition, populations strongly oppose waste management solutions like incineration and landfill that they regard as a local and environmental nuisance even though they cost less. Under pressure from public opinion, governments therefore introduce recovery policies that prove to be much more expensive. That is the case with the EPR systems for packaging introduced in Germany and Sweden: their economic efficiency is far from optimal, but their results in terms of meeting recycling objectives are highly satisfactory. Preference is given to waste management methods which are not necessarily rational from an economic point of view. For example, the marginal cost of recycling a ton of packaging waste in Sweden is reckoned to be 250, compared with 160 for incineration and 130 for landfilling (Economic Survey, Sweden – OECD 2001). A recent cost-benefit analysis by the Danish Environmental Protection Agency (“Should household food waste be burned or recovered?”, 2003) showed that it costs more to compost organic waste than to incinerate it; moreover, the environmental benefit in comparison with incineration is very small. Questions are increasingly being asked about the recycling of certain materials, like plastics, and studies advance “optimum” recycling rates that are much lower than those currently being practised. Inefficient management in economic terms may well pose problems of long-term viability. The question of a better allocation of resources may also be raised, and it is all the more relevant in that these so-called “environmental” waste management methods benefit from subsidies.

How much does waste management cost and who should pay? The major problem at present is that the evaluation is still very vague. Where waste is concerned, given the number of players involved throughout the product life-cycle, it is particularly difficult to identify polluters in accordance with the polluter-pays principle (PPP). Few cost-benefit analyses have been made of the real and total costs (including externalities) of waste management, especially recycling, because such studies are both complex and expensive. Broadly speaking, and at first sight, recycling seems to be “greener” but also more expensive than incineration and landfilling. Consequently, recycling receives support from governments that favour it as a solution. However, internalising the environmental and social costs could call this preconception into question. In addition, the way in which public authorities set more or less arbitrary and uniform charges for municipal waste collection is unsatisfactory in several respects:

- first, the charges are not generally high enough to cover local authorities' waste management costs and do not include the external costs, even though they are borne by the local community;
- second, they do not create any incentive for citizens to reduce their own waste generation or to recycle, since those who create the waste are not aware of and have no responsibility for the costs they generate for society. Most citizens have no idea of the charge they have to pay for household waste collection: in Vienna, only 8% of inhabitants were aware of it.

That is why PPP – i.e., taxation according to the weight of household weight produced – would seem to be the most effective method from several points of view, because it places responsibility with the producer of the waste.

- In environmental terms, it generally goes hand in hand with a 15-30% increase in recycling and a sharp fall in landfilling (Miranda and Aldy, 1996, Kinnaman and Fullerton, 1996).
- In economic terms, collection and treatment costs are adjusted according to the weight treated. The result is a drop in the volume of waste to be collected and treated, generating lower costs for local authorities. However, this fall is partly or even entirely wiped out (Fullerton and Kinnaman, 1996) by the higher costs of administering the system.
- It is also the fairest solution, since the cost of the service is individual and billed according to use.

The drawback of this method of taxation, called “pay as you throw” (PAYT) in the United States, is that it often leads to the illegal disposal of waste by unauthorised tipping and incineration (Kinnaman and Fullerton, 1996 and 2000, Horton, 1996). Although the generator of the waste disposes of it at least cost, the disposal has a significant environmental impact and implies social and rehabilitation costs borne by the community as a whole. In order to limit such side-effects, regulations backed up by penalties and waste tracking systems may prove necessary, as has been the case in Korea, Japan and Germany.

What are the most ecologically and environmentally efficient solutions, bearing in mind the extreme variety of factors that, in all countries, influence waste management, whether physical conditions (climate, surface area, population density, extent of urbanisation, etc.), demographics (age pyramid), social and cultural features (lifestyles, consumption, behaviour) or economic factors (standard of living, structure of economic activity)? Is the arbitrary definition of a hierarchy of waste management methods, which all waste policies recommend, valid in all cases? Waste management costs vary enormously from one country to another, and even from one region to another within the same country, depending on local conditions. Consequently, shouldn't the arbitrary and uniform definition of recycling quotas by international legislation be adapted according to the circumstances (regional, local, social and cultural differences)? Some countries try to link charges to related externalities: in Austria, for example, the landfill tax depends on the risk potential of the waste and the facilities of the landfill site.

In a nutshell:

- a very complex situation,
- full of contradictions,
- full of shadowy areas,
- in which a uniform approach may well be prejudicial to environmental and economic efficiency.

6. Avenues for further research:

1. Better knowledge of the facts (quantities of waste produced, recycled, composted, incinerated with or without energy recovery, put in landfills, exported, imported, by type, by origin, treatment capacity, treatment costs, etc.) is essential in order to monitor how the situation develops and frame effective waste management policies.
2. Detailed cost-benefit analysis would certainly provide better information about externalities and make it possible to adjust waste management policies in such a way as to optimise their environmental effectiveness and economic efficiency. However, given the difficulties of carrying out such studies, mentioned above, we should not take a wait-and-see attitude or refrain from acting on the grounds that we do not know enough or that it costs too much, especially as waste generation is continuing to rise.
3. We must find the lever or levers which can be used to reduce waste generation, especially municipal waste generation. As we have seen, it is difficult to identify these levers and assess their importance, and it is not easy to see how governments could go against broad demographic, social and cultural trends which are not susceptible to policy intervention. The aim must be rather to control their effects on waste generation and waste management and find the most effective instruments, i.e. those most liable to influence the behaviour not only of consumers but also of product designers and producers.

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Chapter 3

ECONOMIC ANALYSIS OF SOLID WASTE MANAGEMENT POLICIES

By Don Fullerton and Amy Raub¹

Worldwide quantities of household solid waste have been rising. By the year 2000, the United States processed an estimated 544 million tons of solid waste – about 4.4 pounds per person per day. Of this total, landfills were used to dispose of approximately 370 million tons (68%).² Federal legislation has made the siting of new landfills increasingly difficult and costly, while state and local governments continue to pay for landfill costs. Europe faces similarly high municipal solid waste levels. On average, Europe produces 306 million tons of solid waste per year – about 2.5 pounds per person per day. Fifty-seven percent of that waste is put into landfills.³

Moreover, most households think garbage collection is free. Traditionally, residents pay for garbage collection services through property taxes or a monthly fee that does not depend on the number of bags or cans placed out at the curb for collection by the city. This pricing practice provides no incentives for households to reduce quantities of waste generated.

Many nations and cities have begun to look at a wide range of environmental policies to alleviate solid waste related problems. Whether it involves a tax on consumers or extended responsibility for producers, an optimal policy must reflect the full social marginal cost at the optimum.⁴ That is, the price per bag of garbage would need to reflect marginal environmental damages (MED) along with the internal or direct costs of collection and disposal.

This paper sets out to examine the choice among policies. Section 1 of this paper looks at data on the quantities of solid waste disposal in landfills and incinerators, and it discusses explanations for the increasing fraction of waste that is recycled. Section 2 then reviews the basic theory of Pigou (1932), as

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1. Fullerton's email address is dfullert@eco.utexas.edu, and Raub's address is raub@eco.utexas.edu. We are grateful for financial support from the OECD, and for comments from Richard Porter and Nils Axel Braathen.
 2. Both of these figures are taken from the website at Chartwell (www.wasteinfo.com). Note that "municipal solid waste" (MSW) includes not only household garbage and recycling but also some institutional, commercial and industrial waste, greenwaste (often composted), some construction and demolition debris, and certain special wastes (like batteries, tires, solvents, and small quantities of hazardous wastes). This paper will only discuss policies that might apply toward different components of household solid waste.
 3. These figures are taken from the European Environmental Agency (EEA, 2003).
 4. The theory of Pigou (1932) for optimal environmental policy is described in Baumol and Oates (1988). This social marginal cost (SMC) is not a narrow concept, but includes all the present and future costs associated with one more bag of garbage: internal costs like labor, trucks, and space in the landfill, but also external costs like odor, litter, noise, aesthetic costs, leachate, methane, and air pollution from incineration.

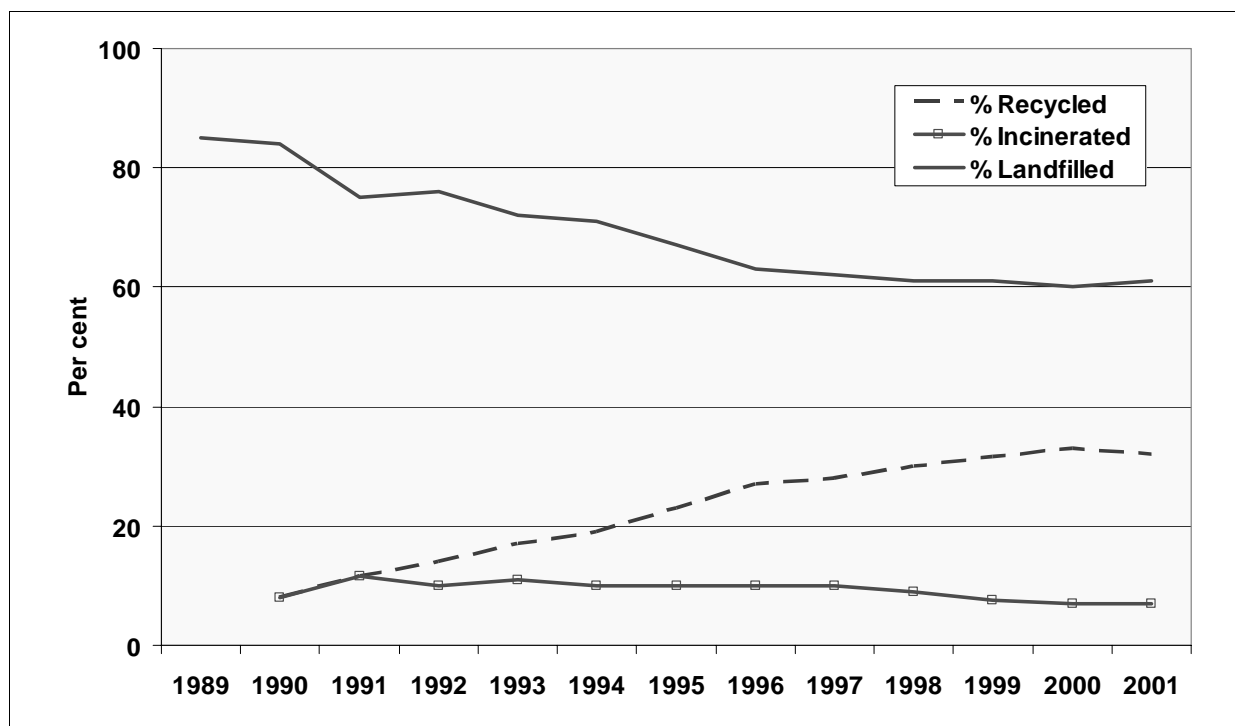
applied to the problem of household disposal options such as garbage, recycling, and potentially illegal burning or dumping. A tax on garbage might encourage recycling, but it might also increase dumping. To be “optimal”, a tax must apply to *each* form of disposal. A tax on dumping cannot be collected, but this section shows that a deposit-refund system (DRS) can match all of the effects of having both the tax on garbage and the unavailable “tax on dumping”. This DRS optimally reduces dumping, and it collects the same revenue as the optimal taxes on garbage and dumping.

Section 3 then addresses how to make choices among various policy options available to policymakers. Although policies may be similar in theory, they are likely to differ in terms of practical considerations like enforcement, distribution effects, and administrative costs. Section 4 reviews results of an empirical literature studying unit pricing systems and discusses some of the shortcomings of those systems. Section 5 describes the benefits and costs of a DRS. Section 6 assesses the relative merits of mandates. Section 7 looks at manufacturer take back programs, and particularly, the successes and failures of the German Green Dot program. Section 8 concludes.

1. Solid Waste Around the World

Of total household solid waste in the United States, the portion incinerated has remained near 10% over the last decade, but the portion placed in a landfill has decreased from roughly 85% in 1989 to just over 60% in 2001.⁵ This decrease in use of landfill disposal was associated primarily with the simultaneous increase in recycling. Figure 1 shows that the portion recycled in the U.S. has increased threefold, from just 10% in 1989 to roughly 32% of household waste in 2001. In the European Union, the recycling of municipal waste has increased similarly: from 11% in 1985-1990 to 29% in 2000.

Figure 1. Disposal Rates in United States



Source : Biocycle Magazine.

5. The information in this section is taken primarily from successive issues of *Biocycle* Magazine, which in 1989 began annual surveys of the 50 states (Glenn 1998, *Biocycle* 2001). Also, see Kinnaman and Fullerton (2000b).

This dramatic increase in the recycling rate can be attributed to a number of possible interrelated factors. First, the number of curbside recycling collection programs increased monotonically from just 1,000 programs in 1989 to over 9,700 programs in 2000. This trend clearly facilitates household recycling, but then the question is what has induced cities to provide this curbside recycling collection.

Second, some have pointed to increased disposal costs, including the “tipping fee” charged by a landfill per ton of garbage. This cost varies by region, but overall U.S. tipping fees have not increased by much (Glenn, 1998, *Biocycle* 2001, and Kinnaman and Fullerton, 2000b). For example, tipping fees in the more-densely-populated northeast are greater than in other regions of the U.S. where land is cheaper. From 1988 to 1997, the average tipping fee in New Jersey has increased from about \$50/ton to about \$55/ton (in nominal dollars). Accounting for increases in the general price level, the real tipping fee has not changed much at all over the past decade. Thus the national rise in curbside recycling is unlikely to be attributable to increases in the tipping fee. However, the newly-available technology of curbside recycling has become most popular in the northeast where the tipping fee is higher. Thus, while the change in recycling cannot be explained by a change in disposal cost, some other factor may have increased recycling more in the region where disposal costs already were relatively high.

Third, recycling might have increased due to changes in the market price of recycled materials. When accounting for increases in the general price level, however, the prices of recycled materials have remained rather constant (Ackerman, 1997). Interestingly, prices of recycled materials are highly variable over time. For old newspaper, six spikes have appeared over the past 30 years, most recently in 1995 when the price for many materials hit all-time highs. This latest spike has been attributed to new recycled-content laws passed by several state governments (Ackerman, 1997). But overall, these trends do not appear to support the argument that economic forces are responsible for the growth in curbside recycling.

Fourth, the dramatic increase in the number of curbside recycling programs in operation in the United States could be related to changes in voter tastes for the environment and other political concerns. Misinformation may have contributed to the public’s perception of a shortage of landfill space. This perception may have emerged in 1987, when the barge “Mobro”, loaded with Long Island garbage, was unable to unload its cargo after repeated attempts (see Bailey, 1995, for a discussion of the incident). A wave of state and local legislation encouraging or mandating recycling was passed soon after this incident. In addition, voters read newspaper stories that landfills were closing faster than they were opening. Indeed, the number of landfills in operation in the U.S. fell by nearly 75%, during 1988-2000, from about 8,000 to only 2,100 (Glenn, 1998 and *Biocycle*, 2001).

Yet the United States is *not* running out of landfill *space*. The estimated capacity of remaining landfills has been steadily rising. Based on state-reported estimates, available landfill space doubled from 1988-97, from roughly 10 years of remaining capacity to 20 years of remaining capacity.

The explanation for the apparent contradiction is the replacement of many small town dumps with fewer large regional sanitary landfills. This trend is due mostly to Subtitle D of the Resource Conservation and Recovery Act (RCRA) of 1976, designed to reduce the negative externalities associated with garbage disposal. This law imposed technology-based standards on the construction, operation, and closure of solid waste landfills. Each landfill now is required to install thick plastic linings along the base, to collect and treat leachate, to monitor groundwater, and to cover garbage within hours of disposal. Because the fixed costs of constructing and operating a landfill have increased, the cost-minimizing landfill size has increased, and fewer landfills are being built. The trend toward larger landfills is also related to the increased political difficulty in siting them: once decision-makers get past the problem of “Not in My Back Yard” (or NIMBY), they choose to build one large enough to last longer.

The increase in the percentage of waste that is recycled in the United States seems to have come entirely from a reduction in the percentage put in landfills. The portion incinerated has been fairly stable – around 7-11% of total household solid waste. It reached something of a peak in 1991 when 170 incinerators operated nationally, but then the number of incinerators in operation began gradually to decline. In 2001, incineration accounted for only 7% of total household solid waste. Incineration was once considered a dual solution to the solid waste and energy crises, but that assessment changed with some complicated technological considerations. Fixed costs are high, and so average costs can be reduced by greater garbage throughput. Yet incinerators could not lower their tipping fees to levels necessary to attract more business without incurring financial losses. For this reason, and because incineration was thought to be a good environmental solution to the dual problem of waste and energy, many local governments passed laws *requiring* that all local garbage come to the incinerator, effectively giving the incinerator monopoly power over local garbage.

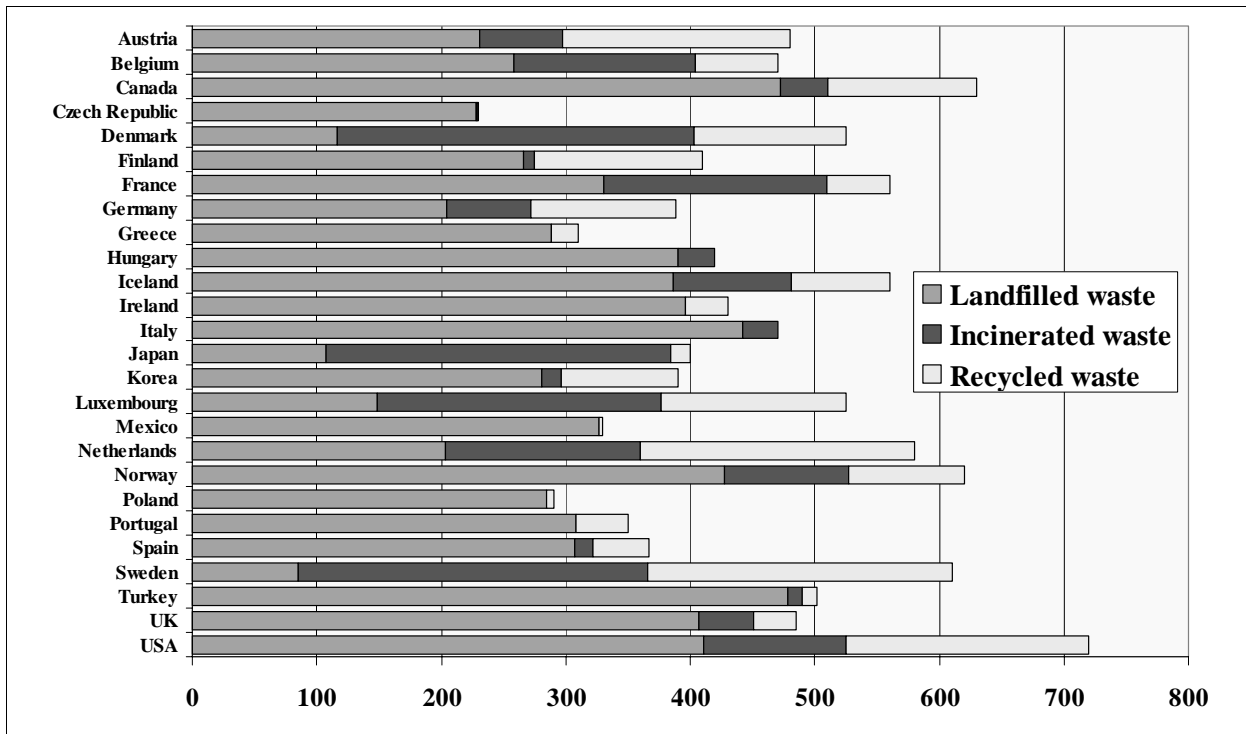
The U.S. Supreme Court struck down these laws, exposing the incineration industry to competition from cheaper landfills. Then the Supreme Court dealt a second blow to the incineration industry when it ruled that incinerator ash must be tested and if toxic must be placed in an expensive toxic waste landfill. A third decision disallowed local control over waste imports. As a consequence of these three decisions, the U.S. private sector built many large regional “megafills” (Bartone, 2002). These new facilities have state-of-the-art leachate and methane gas management systems, and tipping fees are required to include financial provision for 30-year environmental aftercare. Thus, external damages are falling.

The increased use of recycling in the early 1990’s further reduced the quantity of garbage available to incinerators, adding to their financial problems. Then the public began to oppose the resulting air pollution emitted by incinerators, and policymakers are no longer eager to rescue the industry.

The choice of method depends on land scarcity. In the more-densely populated northeastern U.S., incineration accounts for 36% of waste. Incineration is also popular in Japan and several European countries where population densities and land values are high. Landfills are used almost exclusively in the U.K., Ireland, and Greece, but incineration accounts for most garbage in Sweden and Denmark.⁶ As Figure 2 shows, incineration accounts for nearly 70% of Japan’s municipal waste disposal versus less than 10% of the U.K.’s total waste. Facing less competition from land-intensive landfills, incinerators in densely populated areas can capture the economies of scale necessary to keep down the average cost of incineration. But even though many countries rely heavily on incineration, Brisson (1997) finds that the private and external costs of incineration exceed those of landfill disposal in most European countries.

6. See Jenkins (1993) and Kinnaman and Fullerton (2000b).

Figure 2. Per Capita Municipal Waste
Mid-1990's

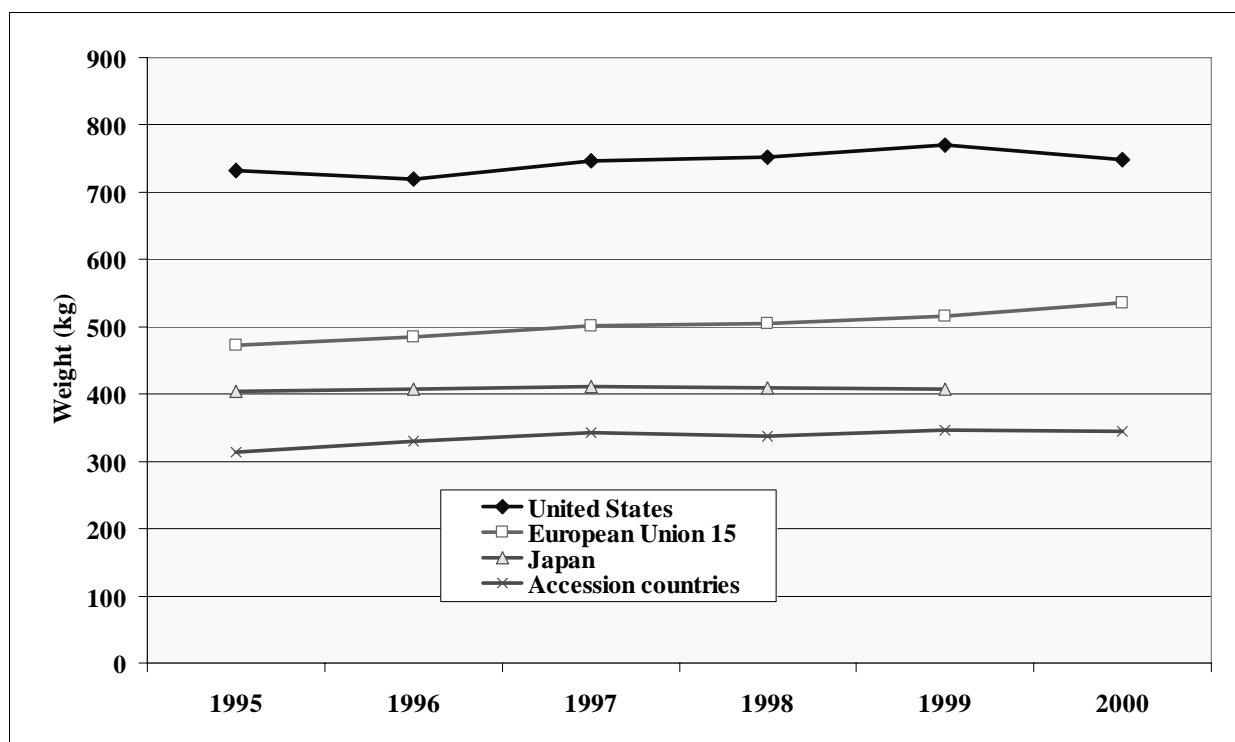


Source: OECD.

Figure 3 shows that between 1995 and 2000, total waste generation increased by 11% within the European Union (E.U.) and the European Free Trade Area. European efforts to decouple economic growth from waste generation have been relatively unsuccessful. As a whole, Europe generates more than 306 million tons of municipal waste every year, an average of 415 kg per capita in 2000.⁷ From 1995 to 1999, landfilling decreased from 67% to 57% within the E.U. Recycling efforts have had some success. From 1985-1990, recycling accounted for 11% of total municipal waste. That figure increased to 21% in 1995 and by 2000, recycling made up 29% of the municipal waste stream.

7. The data in this section are taken primarily from Environmental Assessment Reports of the European Environmental Agency. See <http://reports.eea.eu.int>.

Figure 3. Per Capita Municipal Waste Generated
1995-2000



Source : EEA.

The E.U. has mandated a waste management policy based on a “hierarchy” of options that give top priority to waste minimization, followed by recycling, and then incineration, with landfill disposal last. Despite this policy, landfilling remains the predominant waste disposal method in countries throughout Europe. Incineration accounted for only 18% of municipal waste in 1999. Public opinion is hesitant to accept incineration as a safe disposal option, and local conditions may prohibit the long-term sustainability of operating some incinerators. Indeed, Dijkgraaf and Vollebergh (1998) find that this hierarchy is not justified on economic grounds: incineration has lower external costs than landfills, but sufficiently higher internal private costs – such that incineration has higher total social costs than landfills. The lowest net cost option seems to be landfill disposal with energy recovery through capture and flaring of methane.

Circumstances in developing countries are almost completely different from those considered here. For example, Medina (1997) suggests that municipal solid waste (MSW) can hardly be reduced from levels that may already be only 0.2 to 0.5 kg per person per day. Markets may not work well enough to charge a price or tax on waste. Instead, efforts may need to concentrate just on collection, transportation, and upgrading of the present open dumping sites into controlled landfills.⁸

2. A Simple Conceptual Model

A simple skeletal model is developed here to frame the discussion of optimal policy design, but it is a fully general equilibrium model that captures all of the essential elements. We avoid the problems of second best by assuming that lump sum taxes are available. Instead we focus on the technology of waste disposal and include substitution between different methods of disposal. In particular,

8. For more on circumstances in developing countries, see Bartone (2000) and the references therein.

consumption generates waste that can either go into garbage collection or into recycling that can be re-used in production.⁹

Assume that n identical consumers each maximize utility subject to a budget constraint and a mass-balance equation given by $c = c(g, r)$, where c is consumption, g is garbage, and r is recycling generated. The general form $c(g, r)$ represents the various combinations of g and r that are consistent with any particular level of consumption, possibly with a varying rate of tradeoff (but strict mass balance would require $c = g + r$). All lower case letters represent per capita amounts, while upper case letters are aggregate, so $G = ng$ is total garbage. Utility is $u = u[c(g,r), G]$, where each individual can choose g and r (and indirectly c) but cannot affect G . Consumption has a positive effect on utility ($\partial u / \partial c \equiv u_c > 0$), but a negative externality from all others' garbage means that $\partial u / \partial G \equiv u_G < 0$.

The household budget constraint is $y = (p_c + t_c)c(g, r) + (p_g + t_g)g + (p_r + t_r)r$, where y is income, each p is a price, and each t is a tax rate. The price p_r may be negative if consumers are paid by a private firm for recycled material, and any tax rate may be positive or negative. The production function is $c = f(k_c, r)$, where k_c is the amount of labor or other resources used in production of c . General equilibrium conditions require that the amount of recycling generated by households, r , must be the same amount that re-enters production of c . Garbage collection and disposal also uses resources through the production function $g = \gamma k_g$, and the overall resource constraint is $k = k_c + k_g$. To get conditions of an optimum, a social planner is assumed to maximize utility subject to this resources constraint, recognizing that choices about individual r and g affect aggregate garbage in utility ($G = ng$). To get private market conditions, consumers view G as fixed but choose g and r to maximize utility subject to their income $y \equiv kp_k$. Private firms maximize profits ($cp_c + rp_r - k_p p_k$) under perfect competition and constant returns to scale, so they set the price of each input equal to its marginal product. Similar conditions for garbage disposal imply that $p_g = p_k / \gamma$. Substitution of these producers' conditions into the consumers' first order conditions yield a set of conditions for private markets that can be compared to the conditions for social optimality.

Every extra bag of garbage in the landfill might emit more foul odor, pollute more groundwater, worsen the eyesore, and contribute to climate change.¹⁰ If all tax rates are zero, with no government action, then households fail to internalize the full social costs of their disposal decisions. Too much garbage and too little recycling are produced by a decentralized economy. In a similar model, Fullerton and Kinnaman (1995) show that several different tax and subsidy combinations can achieve the efficient allocation of resources in the presence of the external costs from garbage disposal. In the model above, however, the simplest way to make all of the private market conditions match all of the social optimality conditions is to set all tax rates to zero except:

$$(1) \quad t_g = -n u_G / \lambda,$$

where λ is the marginal utility of income at the optimum. Since u_G is negative, this tax rate is positive. It is merely an example of the general principle of Pigou (1932): for an activity that causes a negative externality, the optimal corrective tax is "marginal external damages" (MED). The expression in (1) reflects the negative effect on utility (u_G) for all n individuals, converted into dollars when divided by the marginal utility of income.

9. The theoretical framework in this section builds on the model of Fullerton and Kinnaman (1995).

10. Decomposition of material in a landfill creates methane gas, which is a greenhouse gas. An estimated 6% of the world's emissions of methane are released from landfills (Beede and Bloom, 1995).

So far, our theory supports not only a price per bag, as would be charged by a private firm (p_g), but also a tax that raises additional revenue (t_g). As intended, the price per bag induces consumers to substitute out of garbage and into recycling.

This optimality disappears, however, if consumers can avoid the tax on proper collection of garbage by instead burning or dumping their garbage. Suppose that the technology of disposal is represented by $c = c(g, r, b)$, where b is “burning or dumping”. Suppose further that utility is given by $u = u(c, G, B)$, where $B = nb$, and where $\partial u / \partial B \equiv u_b < 0$. A reasonable assumption is that social costs are higher for waste that is dumped than for waste in the landfill ($u_b < u_c$).

In this case, if consumers pay little or nothing to dump their trash, then any positive price ($p_g + t_g$) can induce some households to substitute out of garbage and into the more damaging activity of dumping their waste.¹¹ The normal Pigouvian solution in this case would require a tax on *each* activity equal to marginal external damages:

$$(2) \quad \begin{aligned} t_g &= -nu_c / \lambda & \text{and} \\ t_b &= -nu_b / \lambda \end{aligned} ,$$

while other tax rates are zero. Yet a simple Pigovian tax on dumping is generally considered to be infeasible because evasion is easy. Indeed, in this case, just collecting the price per bag ($p_g + t_g$) is a problem, as consumers can avoid that tax by dumping. The revenues from the system in equations (2) may be very low indeed.

For one type of alternative, the city could make dumping illegal, impose a stiff fine, and devote police resources toward establishing some positive probability of discovery.¹² Using a model where consumers maximize expected utility, the fine can be set such that the *expected* fine per unit of dumping is marginal damages ($-nu_b / \lambda$). The fine revenue bears no particular relationship to the cost of the effort to catch those who are dumping, however, so the net revenue may still be small, or negative.

Another set of alternatives, though imperfect, is that government could impose an array of command and control policies such as mandatory recycling for households and minimum recycled-content standards on producers (Palmer and Walls, 1997). A version of the model above can be used to show the quantity restrictions that achieve efficient outcomes, at least in theory, but the information required to achieve those efficient outcomes is not likely to be available to policymakers. This point is just a variant of the usual economic efficiency case for incentive instruments rather than mandates. The general case for mandates is discussed more below.

11. Demand for each type of disposal in this model depends explicitly on price and income, and implicitly on demographic characteristics and intrinsic motivation. For discussions of these latter variables, see Miranda and Bynum (2002) and Frey (2002). It is difficult for policy to modify the latter variables, however, so we focus here on the effect of changing each price. Some individuals might litter or dump readily, but others who believe it wrong would not do so at any price. This simple model has only one type of individual, but it can be said to represent the average or overall response to a change in any price.

12. Again the model simplifies by aggregation, since different kinds of dumping have costs that would require different prices or enforcement policies. Putting household garbage in a commercial dumpster may not have the same environmental cost as throwing it along the roadside, and locks on dumpsters might be cheap and effective enforcement, but it does impose social costs by requiring the locks and by transferring the cost of disposal in a way that is economically inefficient. See Miranda and Bynum (2002).

As a final type of alternative, households could be required to pay an advanced disposal fee at the time of purchase (t_c). This fee by itself cannot achieve the efficient allocation (Palmer, Sigman, and Walls, 1997). In combination with the right subsidies to proper disposal, however, the result is an optimal deposit-refund system (DRS). With the strict mass balance condition ($c = g+r+b$), Fullerton and Kinnaman (1995) show that the optimum in the above model is achieved by:

$$(3) \quad \begin{aligned} t_c &= -nu_b/\lambda \\ t_r &= nu_b/\lambda \\ t_g &= n(u_b - u_c)/\lambda . \end{aligned}$$

Note that the tax on purchase of consumption c is positive, and it reflects the marginal external damages *from dumping*. The tax on recycling is negative, a subsidy that exactly returns the tax collected upon the purchase of the item. Recycling has no “external” effect, and ends up with no net tax. The proper disposal of garbage also receives back the initial tax upon purchase (since t_g includes nu_b/λ), but garbage gets an additional tax that reflects its own externality, $-nu_c/\lambda$. The basic logic is that any item purchased must eventually become disposal in one of three forms. If the consumer does not get the tax back by recycling the item or by proper garbage disposal, then the item must have been dumped. But the marginal damages from dumping were already collected upon purchase, so even that activity has been properly priced.

A disaggregate version of this model would require many different tax rates and refunds that reflect the toxicity or other social cost of dumping each good. As discussed below, the administrative cost of such a policy might be high indeed, but these costs can be reduced by using few “categories” of goods, by employing existing sales taxes, and by bulk subsidies to recycling (per ton of glass or aluminum rather than per bottle or can).

Because of the assumption above that dumping must be more socially damaging than proper disposal ($u_b < u_c$), we know that the sign of t_g must be negative. Proper disposal of garbage is subsidized, to avert illicit dumping. If the subsidy is nearly equal to the positive cost of collection (p_g), then the city can save administrative cost just by collecting garbage for free. This logic has already been widely applied, as many cities intentionally collect garbage and recycling for free, *in order to avert dumping*. In other words, most cities *already* have a deposit-refund system, in the form of a sales tax on consumption goods ($t_c > 0$) and subsidized collection of curbside garbage ($t_g < 0$) and recycling ($t_r < 0$).

The deposit-refund system in (3) is designed so that all private first order conditions exactly match the socially optimal conditions, just exactly as would the Pigouvian taxes in (2). With all the same outcomes, then, the DRS must generate the same *net* revenue as the Pigouvian taxes. But while the tax on dumping is unenforceable, because dumping is unobservable, the deposit-refund system applies a tax or provides a subsidy only to observable market transactions: purchase of c , sale of recycled materials r , and collection of garbage at the curb for disposal at a sanitary landfill. While this paper will not analyze the relative administrative costs of collecting various excise taxes, it would seem reasonable to believe that the administrative cost of taxing these market transactions are the same as for any other market transactions. Costs of collection of the positive net revenue from the DRS in equations (3) must certainly be less than the costs of collection of the Pigouvian taxes in equations (2).¹³

13. If either the Pigouvian tax or DRS were to encourage recycling of some types of materials more than others, then the change in composition of the remaining waste stream may change the MED used to calculate the optimal rate of tax (Linderhof et al, 2001). For example, composting or anaerobic digestion of yard waste significantly reduces methane emissions relative to disposal in a landfill.

3. Various Considerations in the Choice Among Policies

These policies have the same economic outcomes in a theoretical framework, but they may differ when the simplifying assumptions of that model are relaxed. Policymakers must consider not only the welfare effects of a particular policy, but also enforcement and monitoring issues, distributional effects, information requirements, and administrative costs. In this section, we start with measures of the welfare effect and then discuss these other considerations.¹⁴

In general, the net welfare effect depends on estimates of the social marginal cost (SMC) and the social marginal benefits (SMB) of disposal. SMC include all costs associated with garbage disposal. Some of these costs are easy to observe, such as costs of labor and space at the landfill, while other costs are more difficult to quantify, such as external costs of leachate and methane emissions. The SMC curve depends on the waste disposal method. Landfills impose aesthetic costs on individuals through noise pollution from collection trucks and less scenic views. They may also have negative health effects from toxins in the leachate that seeps into the groundwater.¹⁵ As the organic material in a landfill degrades, methane gas is produced. Landfills are the source of 35% of methane emissions in the U.S. and 28% of methane emissions in the E.U., or 4% of *all* greenhouse gas emissions.¹⁶ If the methane is flared, then the combustion converts the methane into carbon dioxide, a less potent greenhouse gas. Flaring thus reduces the greenhouse effect, but it also generates other local air pollutants.

Incinerators have higher private costs of operation, but they have lower external costs than landfills. Incineration reduces municipal waste to about 30% of its original weight, substantially reducing the quantity of waste that must be landfilled. The remaining slag is much more stable than untreated waste and can be reused in road construction projects, embankments, and noise barriers. Many incinerators also utilize the energy obtained. However, the combustion process releases acid gases, polycyclic aromatic hydrocarbons, dioxins and furans, dust, and heavy metals into the air. In addition, the advanced cleaning systems create flue-gas residues that are highly contaminated and are usually classified as hazardous waste.

Thus, the choice of disposal method will determine the height and shape of the SMC curve. For a particular estimate, Jenkins (1993) and Repetto et al. (1992) find that the full private plus external social marginal cost in the United States is \$1.43-\$1.83 per 32-gallon bag, depending on local conditions.¹⁷ For illustrative purposes, Figure 4 show this flat SMC curve in a simple partial equilibrium model of garbage.

14. For a good and recent discussion of disposal methods and policies, see Porter (2002).

15. These external costs could be estimated from effects on property values (Hite et al, 2001). Housing values are estimated to rise by 6.2% for each mile (up to two miles) away from a landfill (Nelson et al., 1992, as cited in Beede and Bloom, 1995). Ten studies reviewed by CSERGE (1993) found that prices are 21-30% lower for houses within half mile of a landfill, and they increase 5-7% for each mile further away (up to four miles). From interviews, Roberts et al. (1991) find that households are willing to pay \$227 per year to avoid having a landfill nearby. Reported amounts increase with income, education, and dependency on well water. Since these effects pertain to the existence of a landfill, they might not seem to affect the SMC per bag at the margin. In the long run, however, any small but permanent increase in garbage per person will eventually necessitate another landfill with negative effects on another neighborhood.

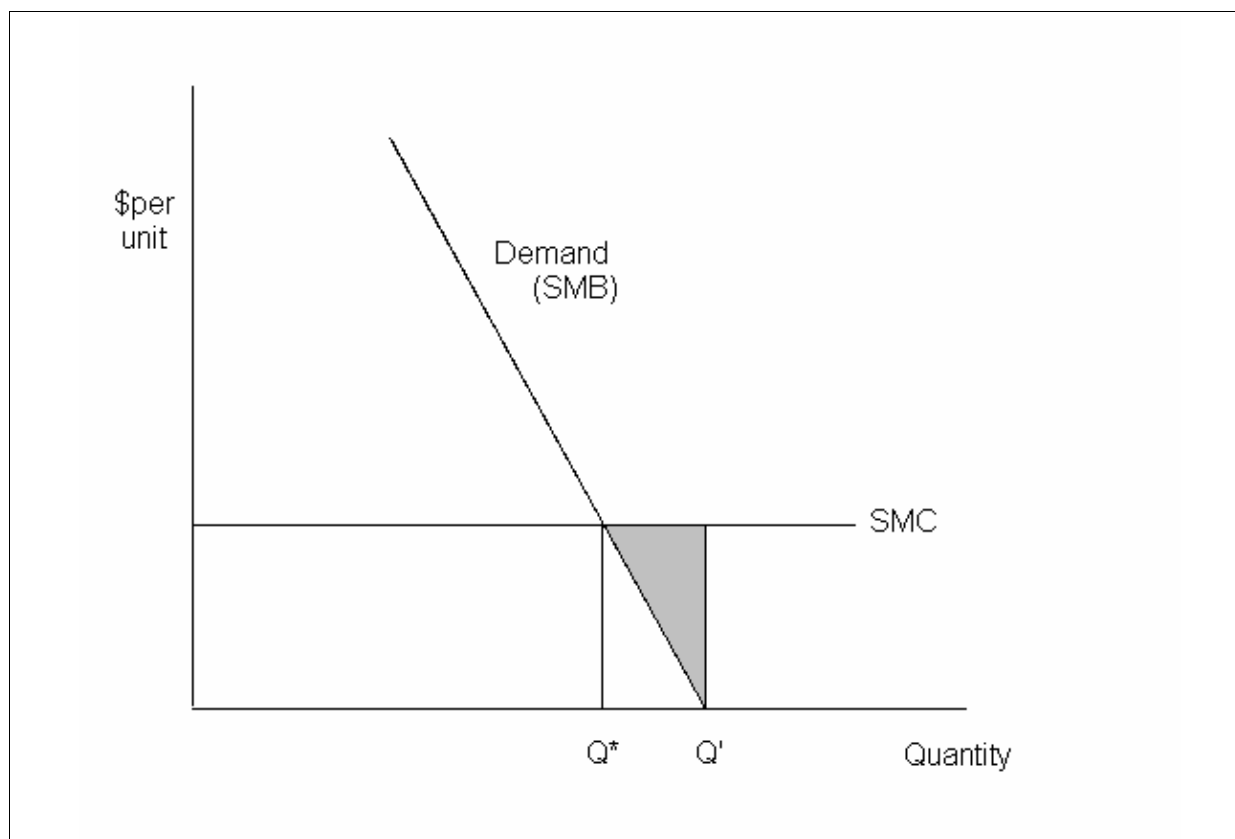
16. See U.S. EPA (2001). However, landfills are also considered to be carbon “sinks” because they keep the carbon in material such as wood products from escaping into the atmosphere.

17. This estimate includes private and external collection and disposal costs (with a depletion allowance). The external costs are based on the work of Stone and Ashford (1991) and the Tellus Institute (1991).

The second major piece of information is the social marginal benefit of disposal, that is, the amount that consumers are willing to pay for one more unit of disposal. Empirical estimates are reviewed below, but the general result is that this demand is fairly steep. Thus, a steep SMB curve is shown in Figure 4. The optimal quantity in this diagram is Q^* , found where $SMC=SMB$.

Yet most cities and towns in the U.S. still finance garbage collection through property taxes or monthly fees, with no price at the margin. This price of zero leads consumers down their demand curve to Q' in Figure 4. The welfare cost of the excess garbage is defined by Jenkins (1993) and Repetto et al. (1992) as the extent to which SMC exceeds SMB for each of those extra units, the shaded area in the figure. They use their estimate of demand to reflect social marginal benefits, and they calculate the welfare cost arising from the current under-pricing of garbage to be as much as \$650 million per year in the U.S., roughly \$3 per person per year. Fullerton and Kinnaman (1996) use household data and also estimate the potential benefits of marginal cost pricing to be in the neighborhood of \$3 per person per year. Podolsky and Spiegel (1998) study a cross-section of towns in New Jersey and estimate the economic benefits of charging per unit of garbage to be as great as \$12.80 per person per year. Still, the estimated demand in Figure 4 is steep, which makes the triangle relatively small.

Figure 4. The Welfare Cost of Excess Garbage



This partial equilibrium model considers only the externality from garbage, however, so the price-per-bag can reduce garbage and avoid this welfare cost. The alternative must be some clean activity like recycling. In a more general model, households may have multiple alternatives, each with its own negative externality. If households are able to avoid paying the garbage fee through illegal

For further information about the SMC of waste, see U.S. EPA (2001), EEA (1999, 2003), CSERGE (1993), Miranda and Hale (1997), and European Commission (2000).

burning or dumping, then Figure 4 is not an accurate depiction of welfare gains. Waste policies need to account for methods of avoidance or evasion. If households are able to dump their garbage, the social cost of dumping may far exceed the social cost of additional waste disposal in a landfill. In this case, it would be better to offer free garbage collection than to implement a pricing policy that leads to widespread dumping. Policies must also reflect monitoring capabilities. Thus the goal of monitoring and enforcement might be met more easily by some kind of deposit-refund system (DRS), or if that DRS has high administrative cost, then perhaps by a simple mandate.

Even if policies have the same aggregate net welfare gain, they may differ in terms of who bears the costs and who receives the benefits. Figure 4 refers to a case where households pay a fee per bag of garbage. All households must pay for their waste, but only some may receive social benefits of reduced waste in landfills. Also, the government receives additional revenue that can be used to lessen other distorting taxes on particular individuals. If households are subsidized for recycling, instead of paying a tax on garbage, then households receive the money from their recycling along with the welfare gains from reduced garbage. In this case, the government would also have to finance the subsidy by raising taxes on someone else. Similarly, firms' profits and government revenues depend on the choice between taxing virgin materials or subsidizing the use of recycled materials in production. In either case, customers may be forced to pay higher prices, reducing their consumer surplus.

In theory, waste levels can be restrained by traditional mandates, by market-based incentives (MBI) like a price-per-bag, or by manufacturer take-back requirements. More generally, however, economic efficiency also requires minimizing the cost of achieving that reduction in waste. A mandate is only able to match the efficiency of a market-based policy if the regulator has perfect information. A MBI policy is likely to impose lower economic costs than a mandate, because it induces the firm or household to *find* the lowest cost production or waste disposal choice.

Other policy goals are to minimize administrative costs to government and compliance costs to firms and households. A policy that imposes different disposal taxes for newspapers and batteries may better reflect true social damage, but that policy may be too complex to administer in a cost-effective manner. Similarly, the cost to households and government of printing, distributing, and buying stickers for a unit pricing system may exceed the social benefits of reduced waste. Thus the administrative and compliance costs may be lower for mandates than for MBI policies. With these considerations in mind, we now study each type of policy in more detail.

4. Unit Pricing Systems

A unit pricing system (UPS) requires households to purchase a sticker or special bag for every unit of garbage they generate. Instead of viewing garbage collection as free, households face a positive price for every bag. Theoretically, this policy can induce households to recycle more of their waste.

Wertz (1976) is the first to derive the impact of a user fee on garbage quantities. By simply comparing the average quantity of garbage collected in San Francisco, a town with a user fee, with the average "urban" town in the United States, Wertz calculates a price elasticity of demand equal to -0.15. In the initial econometric study, Jenkins (1993) gathers monthly data from 14 towns (10 with unit-pricing) over several years. Jenkins also finds inelastic demand for garbage collection services; a 1% increase in the user fee is estimated to lead to a 0.12% decrease in the quantity of garbage.

Two studies rely on self-reported garbage quantities from individual households (rather than as reported by municipal governments). Hong et al. (1993) utilize data based on 4,306 surveys. Households indicate whether they recycle and how much they pay for garbage collection. Results indicate that a UPS increases the probability that a household recycles, but does not appreciably affect the quantity of garbage produced at the curb. In a later study, Hong (1999) shows that as households

engage in more recycling, they reduce their source reduction efforts. Thus, these households may offset increased recycling by producing more total waste. Reschovsky and Stone (1994) mailed questionnaires to 3040 households and received 1422 replies. Each household reported its recycling behavior, income, and demographic information. The price of garbage alone is estimated to have no significant impact on the probability that a household recycles. When it is combined with a curbside recycling program, however, recycling rates increase by 27 to 58%, depending on type of material.

Miranda et al. (1994) gather data from 21 towns with UPS programs and compare the quantity of garbage and recycling over the year before implementation of unit-pricing with the year following it. Results indicate that these towns reduce garbage by between 17% and 74% and increase recycling by 128%. These large estimates cannot be attributed directly to pricing garbage: in every case, curbside recycling programs were implemented during the same year as the unit-pricing program. Callan and Thomas (1997) also predict a large increase in the portion of waste recycled, especially when the user fee is accompanied by a curbside recycling program.

Only Fullerton and Kinnaman (1996) use household data that are not based on self-reported surveys. The weight and volume of the garbage and recycling of 75 households were measured by hand over four weeks prior to, and following, the implementation of a price-per-bag program in Charlottesville, Virginia. A curbside recycling program had already been in operation for over one year. Results indicate a slight drop in the weight of garbage (elasticity of -0.076) but a greater drop in the volume of garbage (elasticity of -0.23). Indeed, the density of garbage increased from 15 pounds per bag to just over 20 pounds per bag.

Since collectors and landfills compact the garbage anyway, the compacting by households does not help reduce the actual costs of disposal. We want to know the change in space used in the landfill, and that is not well measured by the change in the number of bags at the curb. It is better measured by the change in the weight at the curb. Unfortunately, with an elasticity of only -0.076, a price per bag is not very effective at reducing that measure of the space used in the landfill.

Van Houtven and Morris (1999) look at two policy experiments in Marietta, GA. The traditional bag or tag program requires households to pay for each additional bag of garbage presented at the curb for collection. The second program type requires households to pre-commit or “subscribe” to the collection of a specific number of containers each week. The household pays for the subscribed number whether these containers are filled with garbage or not. Many towns in California and Oregon have used subscription programs since early in the century. One advantage of subscription programs is that their direct billing systems may reduce administrative costs. Yet most economists believe the first type of user fee more truly represents marginal cost pricing. The subscription program does not effectively put a positive price on every unit of garbage, since the can may be partially empty most weeks. Indeed, Van Houtven and Morris (1999) find that the bag program reduces garbage by 36%, while the subscription can program reduces it by only 14%.¹⁸

Two studies expand on the work of Jenkins (1993) by increasing the number of towns in the sample. Podolsky and Spiegel (1998) employ a 1992 cross-section of 159 towns clustered in New Jersey, twelve with unit-based pricing programs. They estimate the largest price elasticity of demand in the literature (-0.39). The authors attribute this estimate to the fact that all towns in their sample had mature recycling programs in place, and no towns in their sample had implemented subscription programs (as was the case for Wertz and Jenkins). Kinnaman and Fullerton (2000a) use a 1991 national cross-section of 959 towns, 114 that implemented user fees. They find that accounting for endogeneity of the policy variables raises the demand elasticity to -0.28, but that is still not very high. They also

18. Miranda and Aldy (1998) find that subscription programs can be effective if pricing applies to smaller trash containers. Nestor and Podolsky (1998) employ self-reported household data to estimate that subscription programs are about as effective as bag/tag programs.

estimate that subscription programs have less of an impact than bag/tag programs on garbage and recycling quantities.¹⁹

In 1995, South Korea became the first country to implement a UPS nationwide. Households must purchase garbage bags authorized by local governments and stickers for the disposal of larger items. These prices vary between cities. Hong (1999) finds that, after the adoption of this UPS, waste dropped from 58,118 tons per day to 47,774 tons per day (a decline of 17.8% that year), and recyclables increased from 8,927 tons per day to 11,322 tons per day (an increase of 26.8%). He calculates a price elasticity of -0.154 . Despite an increase in the proportion of waste disposal costs supported by service charges, households' monthly service charges did not change.

Recall that Figure 4 depicts the welfare gain from a UPS. Yet even this small welfare gain is not necessarily available merely by charging for garbage, because of three big problems. First, Figure 4 is a partial equilibrium model that looks only at garbage, not other disposal methods. It does not convey why demand slopes down, that is, what substitutes are available. That welfare gain calculation is correct if recycling is the only alternative, but not if dumping is possible, as that can be *more* costly than garbage.²⁰ Second, the administrative costs of implementing the garbage-pricing program may exceed the social benefits (the shaded area in Figure 4). Fullerton and Kinnaman (1996) estimate that the administrative costs of printing, distributing, and accounting for garbage stickers in Charlottesville, Virginia, could exceed the \$3 per person per year benefits mentioned above. Third, a uniform tax on all types of garbage may be inefficient if materials within the waste stream produce different social costs (Dinan, 1993). If the social cost of disposal of flashlight batteries is greater than that of old newspapers, for example, then the disposal tax on flashlight batteries should exceed that on old newspapers.

Available data rarely allow for direct comparisons between illegal dumping before and after the implementation of unit pricing. Many economists have asked town officials whether they believe illegal dumping has increased, and many have stated that it has, but many more have stated otherwise. Reschovsky and Stone (1994) and Fullerton and Kinnaman (1996) asked individual households whether they observed any change. In the former study, 51% of respondents reported an increase in dumping. The most popular method was household use of commercial dumpsters. For the 20% who admitted to burning trash, the authors were unable to confirm whether these burners did so in response to the program. Roughly 40% of the respondents to the Fullerton and Kinnaman (1996) survey said that they thought illegal dumping had increased in response to the UPS. Many of these lived in the more densely populated urban areas of the city. Those authors also use survey responses with direct household garbage observations to estimate that 28% of the reduction of garbage observed

19. Other important studies include Hong and Adams (1999) who look at the effect of unit pricing on aggregate disposal and recycling behavior, and Jenkins et al (2003) who use household level data to look at recycling behavior by material. They find that unit pricing has no effect on recycling but that curbside collection has a big effect on recycling of all materials.

20. Of the reduction in garbage resulting from pricing garbage at the curb, Fullerton and Kinnaman (1996) estimate that 28% may have been dumped. Evidence of increased dumping was also found by Blume (1991), Jenkins (1993), Rechovsky and Stone (1994), Miranda and Aldy (1998), and Hong (1999). A number of other studies find minimal changes in dumping, including Podolsky and Spiegel (1998), Strathman et al. (1995), Miranda et al. (1994), Miranda and Bauer (1996), and Nestor and Podolsky (1998). Nobody has estimated both short run and long run effects, but we suspect that some individuals might "protest" the imposition of a price on garbage by dumping initially – but then return to compliant behavior.

at the curb was redirected to illicit forms of disposal. Nonetheless, Miranda and Bynum (1999) estimate that more than 4000 communities use some form of unit pricing in the United States.²¹

To avoid illegal dumping, communities may choose to provide free garbage collection for the first bag of garbage, in a system where a fee must be paid for every additional bag. This pricing system leaves some distortion in economic incentives, however, in that households have no incentive to reduce their garbage generation below one bag per week.

5. Deposit-Refund Systems

The dumping problem might also be fixed by implementation of a deposit-refund system (DRS), but such systems entail their own administrative costs. Those administrative costs might be quite low if the DRS is implemented implicitly by the use of a sales tax on all purchased commodities at the same rate, together with a subsidy to all recycling and proper garbage disposal. That practice is currently followed in the U.S., at least implicitly, since local governments do impose local sales taxes and they do provide free collection of curbside recycling and garbage.²² In order for the local sales tax to approximate the deposit portion of a DRS, it should reflect the SMC of dumping garbage. A sales tax set lower than the SMC of dumping will not encourage an efficient level of proper garbage disposal. If the recycling subsidy needs to be larger, administrative costs can be reduced by providing a subsidy per ton, paid to recyclers, rather than providing an amount for each bottle recycled by each household. But then optimality may require a different tax and subsidy amount for each type of material – a plan that might be very costly to administer.

The oldest DRS implemented in the U.S. at the state level is for empty beverage containers. The state of Oregon was the first to pass this form of legislation in 1971, and nine other states followed in the 1970's and early 80's, but then no state implemented a new DRS until Hawaii in 2002.²³ Worldwide, these programs have been successful at reducing waste and recovering recyclable materials.²⁴

Several economic studies have favored the use of deposit-refund systems to correct for the external costs associated with garbage disposal, including Dinan (1993), Dobbs (1991), Fullerton and Kinnaman (1995), Palmer and Walls (1994), Palmer et al. (1997), Fullerton and Wu (1998), and Atri and Schellberg (1995). To achieve the efficient allocation, the deposit for each good should be set equal to the social marginal cost of dumping the post-consumer waste, and the refund on return is that deposit minus the marginal external cost of recycling. If the external cost of recycling is zero, then the refund matches the deposit. The deposit could be levied either on the production or the sale of goods. As long as transaction costs are low, the refund can be given either to the households that recycle the materials or to the producers that use the recycled materials in production. If the refund is given to the households, then the supply increase is expected to drive down the price of recycled materials paid by

21. ISWA (2002) reports that a recent study for Denmark recommended against weight-based charges after finding that municipalities with such charges had more illegal disposal and less recycling than other municipalities. Bartone (2002) reports low participation rates and widespread littering where households contract directly with private firms for collection services in two Latin American cities (Merida, Mexico and Guatemala City, Guatemala). Some municipalities may themselves engage in dumping.

22. Since money is fungible, it does not matter if the subsidized collection of garbage and recycling (the "refund") is financed from sales taxes (the "deposit") or from some other source like property taxes.

23. According to www.bottlebill.org, the eleven states with current bottle bills are: California, Connecticut, Delaware, Hawaii, Iowa, Maine, Massachusetts, Michigan, New York, Oregon, Vermont. In Europe, Austria, Belgium, Denmark, Finland, Germany, the Netherlands, Norway, Sweden, and Switzerland are all listed as having beverage container DRS. Canada has also had success with their program.

24. See Porter (1983), OECD (1998), and Naughton et al. (1990).

firms. If the refund is given to firms, then firms increase demand for recycled materials and drive up the price received by households (Atri and Schellberg, 1995). In addition, Fullerton and Wu (1998) find that the refund given under a DRS encourages firms optimally to engineer products that are easier to recycle. Households demand such products in order to recycle and thereby to receive the refund. This result is important, since directly encouraging the recyclability of product design can be administratively difficult.²⁵

If the administrative cost of operating the DRS is high, then Dinan (1993) suggests that policymakers could single out products that comprise a large segment of the waste stream (newspaper) or that involve very high social marginal disposal costs (batteries). Palmer and Walls (1999) argue that a tax on produced intermediate goods combined with a subsidy paid to collectors of recycling would preserve the efficiency effects of a DRS but would be less costly to administer.

Although a “virgin materials tax” might be used most directly to internalize the MEDs of material extraction (e.g. cutting timber or strip mining), some researchers have suggested the use of these taxes to encourage recycling. When taxing or pricing garbage directly is problematic, a virgin material tax has been suggested as a way to increase manufacturers’ demand for recycled materials, driving up the price of recycled materials and thus increasing the economic benefits to households that recycle. Miedema (1983) finds that a tax on virgin materials set equal to the social marginal cost of disposing of any resulting waste materials produces welfare gains greater than those resulting from other policies. A tax on virgin materials would discourage use of virgin materials, while simultaneously encouraging the development of the market for recycled materials.

On the other hand, Dinan (1993) finds that, although a tax on virgin materials encourages the use of recycled materials in industries where the recycled input is a substitute for the taxed virgin input, other industries that do not use the taxed virgin input will not increase demand for recycled materials. For example, farmers could use old newspapers for animal bedding, but a tax on paper manufacturers’ use of virgin wood pulp will not encourage this form of recycling. Additionally, a tax on virgin materials does not encourage exporters to use recycled materials in their production.

Palmer and Walls (1994) suggest that a tax on virgin materials, while producing an efficient mix of inputs, may discourage production and consumption in the overall economy, resulting in an inefficiently low quantity of garbage. They posit that the tax is only optimal when combined with a subsidy on final goods. Additionally, both Fullerton and Kinnaman (1995) and Palmer and Walls (1997) find that as long as other policy options are available, then a tax on virgin materials is only necessary to correct for external costs associated with extracting the virgin material. The virgin materials tax is not optimally used to correct for the marginal environmental damages of garbage disposal if a tax is available on garbage disposal.

Furthermore, taxes on virgin materials may be more difficult to implement than a deposit-refund system. Whereas firms can organize a strong defense against virgin material taxes, households often lack political organization. Additionally, households with strong preferences for a clean environment are likely to support a subsidy for recycling. Efficient implementation of a DRS also requires less information. Virgin material taxes require information on each firm’s rate of technical substitution between virgin and recycled materials, while a DRS requires only knowledge of the social marginal cost of waste disposal (Palmer and Walls, 1994).

25. On the other hand, this result depends on the assumption that recycling markets are complete. Calcott and Walls (2000a, 2000b) argue that imperfections in recycling markets prevent attainment of the first-best. It is costly to collect and transport recyclables, and it is difficult for recyclers to sort products according to their recyclability and pay consumers a price based on that degree of recyclability. With these transaction costs, price signals may not be transmitted from consumers and recyclers back upstream to producers.

6. Mandates

Instead of administering a detailed set of tax and subsidy rates of a tailored DRS, many states simply regulate household solid waste in particular ways. One item with the lowest SMC of disposal is yard-waste, and many states prohibit it from landfills because composting facilities can accommodate it more cheaply. The European Union landfill directive imposes a target reduction of the amount of biodegradable waste in landfills by 35% from 1995 to 2016. Several other states have passed laws prohibiting from landfills other materials such as automobile tires, batteries, motor oil, and old appliances. Similarly, the E.U. has passed a directive on end-of-life vehicles that emphasizes recovery, reuse, and recycling.

Mandatory source separation is another command and control policy that is meant to increase household recycling. Households must pay a fine for not separating recyclables from other waste. Effects are limited, however, unless the program is strictly enforced (Goldin, 1987).

Another common policy is a minimum recycled-content standard. This command and control policy requires firms to employ a certain minimum portion of recycled materials in their product. Palmer and Walls (1997) point out the problems associated with such rules. First, they only achieve efficiency if carefully implemented with other policies. If recyclable materials are highly productive at the margin but are not used because of their high price, then a recycled-content standard could increase production and thus increase solid waste. Therefore, a simultaneous tax on consumption is also necessary. If, however, recycled materials are not very productive at the margin, then standards can decrease output and therefore decrease solid waste. In this case, a subsidy to consumption is necessary to achieve efficiency. Most importantly, efficient implementation requires information not ordinarily available to policymakers about the production technologies and costs of all different firms.

7. Manufacturer Take-Back Programs

Other policies to reduce solid waste have focused on producers. One such proposal is a manufacturer take-back requirement. In this case, firms are required to accept their own packaging and products back from consumers, after use, and then dispose of it. Intuitively, this policy should give firms the proper incentives to reduce packaging and to design more recyclable products. However, this policy alone may not do enough. If the firm is only responsible for the private cost of garbage disposal, it still does not internalize the full social marginal cost. Therefore, firms may still need to be made to pay the full marginal external damages of disposal. If firms internalize this cost, then they have the right incentives without requiring taxes on packaging, disposal-content charges, recycled-content standards, or subsidies for “green design”.²⁶

In Korea, a national waste management law requires that manufacturers and importers pay deposits on various items, including beverage containers, pesticide containers, batteries, tires, engine oils, TV's, washing machines, and air conditioners. These deposits are refunded if manufacturers can show proof of proper disposal. The deposit-refund system gives firms incentives to collect goods to recycle, resulting in less landfilling of wastes and more recyclable products. However, the recapture rate has been extremely low. Disposal costs are higher than the deposit in Korea, so firms do not provide consumers with enough monetary incentives to return their goods to the manufacturers for proper disposal (Hong, 1999).

In 1991, Germany adopted a new policy called the ‘Law on Waste Management,’ which requires manufacturers to pay to recycle their post-consumer packaging. The goal of this program is to internalize costs of packaging choices. Originally, firms were required to recycle 80% of all packaging they produce, but amendments lowered the standard to 50% in 1996 and to 60% in 1998 (OECD, 1998).

26. For a more detailed analysis, see Fullerton and Wu (1998).

In order to capture economies of scale and to reduce administrative costs, over 400 retail and packaging firms combined with waste hauling firms to create the Duales System of Deutschland (DSD). Instead of returning each bottle to its original manufacturer, local waste management firms collect *all* recyclable bottles of member organizations in exchange for payment from the DSD. A green dot on their packaging identifies DSD members. Waste management firms are reimbursed by the DSD for all collection, sorting, and marketing costs incurred. The DSD then charges manufacturers according to the quantity and type of packaging used, reflecting marginal cost pricing.

The success of the Green Dot program in achieving the efficient quantities of garbage and recycling hinges on two critical issues (Fenton and Hanley, 1995). First, households must be willing to recycle materials. Mandatory deposits are only on beverage containers, and so consumers have no incentives to recycle optimal quantities of other materials. Second, private waste collectors must recycle the materials. Unless they are regulated or taxed, private collectors face private rather than social costs for disposing of materials. Hence, some collectors may find it cheaper to export waste instead of recycling it locally. To correct this problem, the DSD is no longer allowed to export packaging material for recycling.

In the first year following the implementation of the Green Dot Program, the volume of packaging material in circulation fell by 500,000 tons.²⁷ By 1998, packaging had fallen by an additional 500,000 tons. From 1991-1998, the amount of packaging waste sent to landfills or incinerators fell by 66%. Additionally, producers began using more paper-based materials that are easier to recycle. New technologies were also developed, particularly for plastics, to recycle additional materials. The development of new sorting technologies has also further automated the recycling process.

Despite these promising signs, the program had not met its quotas as of 1998. Additionally, problems are associated with free riders. Because the program relies on households to separate Green Dot program members' waste from the rest of household trash, some packaging from non-member firms is picked up by the DSD, costing an estimated 400 million DM a year. New legislation was passed requiring non-members to prove they are collecting their packaging. If they are unable to do so, stiff fines are imposed. This new policy is expected to help alleviate the free ride problem. Other problems include possible waste management monopolies and cartels, consumer misidentification of the Green Dot symbol, the importance of public participation, and duplicated waste disposal costs.²⁸

Palmer and Walls (1999) argue that these problems can be alleviated while preserving the proper incentives by replacing a program like the Green Dot with a combined tax on intermediate goods and a subsidy paid to collectors of recycled materials. Similarly, Runkel (2003) finds that under perfect competition, extended producer responsibility (EPR) misses a first-best solution but still leads to welfare increases relative to free garbage collection. The EPR may be a good substitute for unit pricing of garbage when households dump or are poorly informed. When these conditions do not hold, however, it is unclear whether EPR is the best option. Even when assuming households do not illicitly dump and are rational, an EPR program under imperfect competition may result in welfare losses.

8. Conclusion

Solid waste quantities have been rising for the past several decades, so waste reduction has become an important item on the agenda of nations and municipalities. Under the most common existing pricing arrangement for local garbage collection, the marginal cost to an individual household for disposal of another bag of garbage is essentially zero, even though collection and disposal costs

27. Information on the Green Dot Program in this section is taken from OECD (1998).

28. Communities must pay for general municipal waste collection. The Green Dot packaging is generally collected separately. Significant cost savings may arise with only one pickup per household.

increase with the amount of garbage. Several communities have started a “pay-by-the-bag” program, and most of these programs have been viewed as successful. This paper shows that the optimal charge at the curb is the full social cost per bag of garbage, including both the direct costs of collection per bag plus the external cost per bag. If dumping is not a problem, then this charge can be imposed directly to control waste quantities.

On the other hand, if dumping *is* a potential problem, then this paper shows that the optimal charge can be collected using a deposit-refund system (DRS). This set of taxes and subsidies has the same net revenue as the direct excise tax, but it cannot be evaded by dumping. It therefore has lower enforcement costs. On the other hand, it may have higher administrative costs.

Government mandates can also be used to reduce solid waste. These policies may be cheaper to enforce, but they require the policymaker to have more complete information. Thus, even if mandates achieve waste-reduction targets, they may have higher social costs.

Manufacturer take-back systems may encourage companies to reduce their packaging. These programs require firms to pay the costs of their packaging disposal. This change alone is not enough. Firms must be charged the full marginal environmental damages of disposal to achieve an optimal outcome.

Overall, this paper has examined the available menu of solid waste environmental policies. By looking at the costs and benefits associated with various policies, the paper provides a framework for thinking about policy choice.

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THE DEVELOPMENT AND IMPLEMENTATION OF A LANDFILL TAX IN THE UK

By Bob Davies and Michael Doble¹

1. Introduction

This paper describes the development of the UK tax on landfilled waste, its implementation and subsequent history highlighting some key areas of more general interest for environmental tax design.

The landfill tax was first introduced in the UK in 1996. It applies equally to commercial and industrial waste and municipal waste, and there were separate rates for active and inactive waste. The tax was designed as an environmental tax to internalize the negative externalities of landfill. The Government published a review of the tax in 1998, and an increase in the active rate was announced for 1999/2000. This one off increase was followed by a landfill tax escalator which was announced for 2000/01 to 2004/05. Budget 2003 confirmed that there will be subsequent increases in the active rate of the landfill tax from 2005/06 when the current escalator finishes, towards a set medium-long term target.

2. Background to the introduction of the landfill tax

The impetus for the introduction of a tax on landfilling waste came from a number of initiatives through which attitudes to waste management in the UK were changing. The UK was also amongst those EU countries with the highest level of landfill, as Figure 1 shows. In the early 1990s, over 90% of municipal waste went to landfill.

In 1990, the Environmental Protection Act (EPA) and publication of the Environmental White Paper² set the UK Government's agenda as follows: encouraging the minimisation of waste, tightening waste disposal standards, and promoting recycling of as much waste as possible, including the recovery of energy. A target was also set under the EPA to assist in attaining this objective of recycling 25% of household waste by the year 2000 (50% of the estimated potentially recyclable content of household waste).

In 1991 the Advisory Committee on Business and the Environment (ACBE) recommended that the price of landfill be increased significantly to levels reached elsewhere in the EU. As a result the Government agreed to further investigate landfill pricing and the possible use of economic instruments. A number of issues had to be resolved before the case for a landfill tax could be made

1. The authors work in the Department for Environment, Food and Rural Affairs (DEFRA), United Kingdom. However, the views expressed in the paper do not necessarily represent the views of DEFRA.

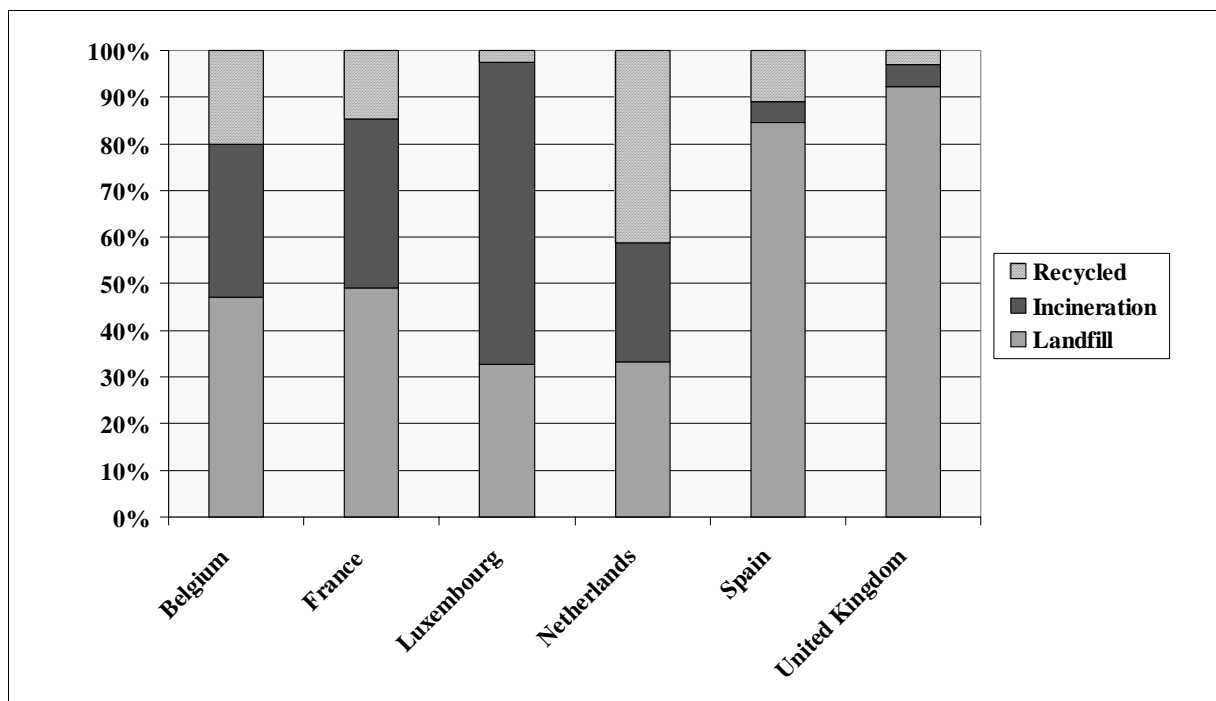
2. DoE (1990).

and implementation could take place and a number of studies were undertaken which are discussed further in Section 3.

A further White Paper in 1992 stressed the usefulness of economic instruments in achieving environmental goals:

'Economic instruments are an inherently more flexible and cost effective way of achieving environmental goals. The Government believes that the time has now come to deploy them more fully to achieve environmental objectives'.³

Figure 1. EU Waste Disposal Methods for Municipal Waste
1993



Source : DEFRA.

3. Development work on a Landfill Tax

A number of issues had to be resolved before the case for a landfill tax could be made and implementation could take place. To resolve these, three studies were undertaken and published on the rationale for using economic instruments in the context of waste; valuing the externalities from waste disposal to landfill; and evaluating the sectoral impact on business.

3.1 Economic instruments and waste

A report was commissioned from Environmental Resources Management consultants⁴ which examined a range of economic instruments for environmental protection and concluded that the internalisation of externalities (from waste disposal) would reduce the amount of waste going to final

3. DoE (1990).

4. ERM (1992).

disposal either through recycling or waste minimisation. It was deemed that this could be achieved most simply via collection or disposal charges.

3.2 *Impact on landfill costs and behaviour of a landfill levy*

The implication of a landfill levy were examined in a report commissioned from Coopers and Lybrand (1993).⁵ This report prepared estimates of the current and likely future costs of waste disposal and evaluated options for a levy on landfilled controlled waste, the implications of implementing the options and their impact on behaviour.

It was envisaged that landfill operators would pass almost all of the levy onto their customers in the form of higher landfill prices. There would then be little change in the short term in the quantity of waste being landfilled because there were few competitive alternative disposal routes. In the longer term as underlying costs rose it was then expected that the effect of a levy would be to increase the incentive to incinerate waste. This would be even more marked in the major conurbations where it was expected that rising costs alone would make incineration more attractive. A level of £20 per tonne was estimated to provide an incentive to undertake incineration throughout England and Wales. Whether or not this would actually take place was seen to depend upon land use planning controls and other regulatory restrictions to cost effective incineration.

A levy at £20 per tonne might also incentivise recycling by promoting the introduction of bring systems and central composting. However, by itself a levy at this level was only expected to increase recycling from 2 to 12%.

3.3 *Valuing the externalities of waste disposal*

In response to ERM's conclusions work was commissioned from the Centre for Social and Economic Research of the Global Environment (CSERGE), Warren Spring Laboratory and EFTEC (1993) which produced the first major estimate of the externalities from landfill and incineration with energy recovery for the UK (see Tables 1 and 2). Most of the work undertaken thus far had looked at single waste management options, single materials, or compared the environmental costs of incineration and landfill. CSERGE (1993) compared the respective contributions of landfill and incineration of waste across most of the main environmental impacts. Depending upon the type of landfill, whether it was urban or rural and whether there was energy recovery, the external costs from landfill were calculated to be in the range £1 to £9 per tonne. Waste incineration with energy recovery was estimated to result in an external benefit of £2 to £4 per tonne of waste. However, this does not reflect potential disamenity impacts and it assumed that the electricity generated displaced that of a coal-fired power station.

As can be seen from Table 1, the climate change impacts of methane emissions is the most significant externality from landfill, with other impacts being of a roughly similar order to each other. The global damage estimates for the impact of CO₂ and methane released were evaluated and translated into a cost per tonne of emission. It was concluded that estimates by Fankhauser (1992) were the most reliable and using these as a basis, the parameters used to calculate the externality ranges from global pollution in Table 1 are based on £ per tonne externality from CO₂ (as C) of £4.1 to £31.0 and the impact of methane is valued at £31.9 to £138.5 per tonne.⁶ Multiplying these by the carbon and methane content of emissions from a tonne of waste gives an external cost range (depending on the type of landfill) for CO₂, of between £0.08 and £1.27 per tonne, and for methane of between £0.86 and £5.40 per tonne.

5. Coopers and Lybrand (1993).

6. Fankhauser (1992).

The extent to which leachate is an externality depends on whether the landfill operator is responsible for cleanup and monitoring costs. It was assumed that for new landfills the externality was internalised and that the operator is responsible. A Departmental working group estimated the costs of leachate accidents over 30 years would be £2.6m per annum and that monitoring costs would be £91.3m per annum. Existing landfill sites would therefore have a maximum externality value of $((£2.6m + £91.3m) / 102m \text{ tonnes of landfilled waste})$ £0.9 per tonne. In addition, the externalities from transport impacts are evaluated, in terms of pollution and accidents associated with the transportation of waste to landfill. The mean values of this impact are generally, depending upon the type of landfill, less than £1 per tonne.

The mid estimate total externality across the whole waste stream, averaged for the four scenarios in Table 1 was about £3 per tonne.

**Table 1. Values of landfill externalities (£ per tonne) from CSERGE
Warren Spring Laboratory and EFTEC (1993)**

			Existing urban landfill without energy recovery	New urban landfill with energy recovery	Rural landfill without energy recovery	New rural landfill with energy recovery
+	Global Pollution	CO ₂ as C	0.32 (0.08-0.87)	0.46 (0.12-1.27)	0.32 (0.08-0.87)	0.46 (0.12-1.27)
		CH ₄	2.36 (0.86-5.40)	1.36 (0.45-3.32)	2.36 (0.86-5.40)	1.36 (0.45-3.32)
+	Air pollution		N/A	N/A	N/A	N/A
+	Transport impacts	<i>Pollution - Conventional (UK only)</i>	0.09 (0.05-0.16)	0.09 (0.05-0.16)	0.38 (0.10-1.06)	0.38 (0.10-1.06)
		<i>Pollution - Conventional (UK and ECE)</i>	0.10 (0.06-0.17)	0.10 (0.06-0.17)	0.46 (0.14-1.19)	0.46 (0.14-1.19)
		<i>Accidents</i>	0.23 (0.13-0.33)	0.23 (0.13-0.33)	0.55 (0.31-0.79)	0.55 (0.13-0.79)
+	Leachate		0.45 (0-0.9)	0	0.45 (0-0.9)	0
-	Pollution displacement	<i>Conventional (UK only)</i>	0	0.81 (1.54-0.45)	0	0.81 (1.54-0.45)
	Pollution displacement	<i>Conventional (UK and ECE)</i>	0	1.12 (1.92-0.69)	0	1.12 (1.92-0.69)
=	Total	<i>Conventional (UK only)</i>	1.12 to 7.66 (3.45)	-0.80 to 4.63 (1.33)	1.35 to 9.02 (4.06)	-0.57 to 6.00 (1.94)
=	Total	<i>Conventional (UK and ECE)</i>	1.13 to 7.66 (3.45)	-1.17 to 4.91 (1.03)	-1.58 to 9.15 (4.14)	-0.91 to 5.89 (1.72)
The mean value shown for the total, and for pollution displacement, transport and global impacts reflects specific statistical techniques used to capture the uncertainty in CH ₄ and CO ₂ estimates. The mean does not therefore equal the midpoint of the range values.						

These estimates exclude disamenity costs, that is the nuisance value from landfill sites from noise, odour, visual intrusion etc. The CSERGE report reviews the methods by which disamenity values can be estimated, principally by analysing the variation in property prices which can be attributed to the proximity of a facility; or by surveys of willingness to pay for (or to avoid) a facility being located at a specific site. All of the recent evidence was North American: although clearly a second best approach the US 'willingness to pay' estimates were transferred to the UK landfill context. The use of US figures, suggested willingness to pay estimates of £160 per household (located within 4 miles of a site) per year for landfill sites. This gives an estimate of disamenity value of approximately £2 per tonne of waste.

**Table 2. Values of incineration externalities (£ per tonne) from CSERGE
Warren Spring Laboratory and EFTEC (1993)**

			New urban incinerator with energy recovery	New regional incinerator with energy recovery
+	Global Pollution			
		CO ₂ as C	2.55 (0.69-6.70)	2.55 (0.69-6.70)
		CH ₄	not applicable	not applicable
+	Air pollution			
	or	Conventional ¹	1.62 (1.16-2.07)	1.51 (1.03-2.02)
		Conventional ²	2.01 (1.47-2.55)	1.14 (0.80-1.49)
		Toxics	not estimated	not estimated
+	Transport Impacts			
		Pollution ¹	0.23 (0.11-0.52)	0.36 (0.11-0.93)
		Pollution ²	0.26 (0.12-0.57)	0.42 (0.15-1.03)
		Accidents	0.20 (0.11-0.29)	0.33 (0.18-0.48)
-	Pollution Displacement ¹	6.87 (11.93-4.30)	6.87 (11.93-4.30)	
	Pollution Displacement ²	9.40 (14.81-6.49)	9.40 (14.81-6.49)	
=	Total ¹	Range	-9.86 to 5.28	-9.78 to 5.88
		Mean ³	(-2.26)	(-2.01)
=	Total ²	Range	-12.41 to 3.61	-12.32 to 4.27
		Mean ³	(-4.38)	(-4.09)

¹⁾ Conventional air pollution including damage to the UK only.

²⁾ Conventional air pollution including damage to the UK and the rest of the ECR region.

The mean value shown for the total, and for pollution displacement, transport and global impacts reflects specific statistical techniques used to capture the uncertainty in CH4 and Co2 estimates. The mean does not therefore equal the midpoint of the range values.

Adding the disamenity and non disamenity externalities gives a total monetised cost for landfill of about £5 per tonne, approximately equivalent to £7 per tonne for active⁷ and £2 per tonne for inactive waste.

3.3 Evaluating the sectoral impact of a landfill tax on business

The Department of the Environment also commissioned a report from MEL Research to assess the impact on business of a landfill tax.⁸ To estimate the additional costs to industry of a proposed tax, the waste arisings figures for each industry sector were multiplied by the cost increase of disposal expressed in £ per tonne and converted into millions of pounds. Total cost to industry was estimated at around £366m using survey data which was collected as part of the report and at £416m using previously collected Waste Disposal Plan data.

In order to determine which industries would be the most heavily affected by the tax, the cost was also evaluated in terms of proportion of gross output. Information on gross output was obtained from the Census of Production. This Census only covers manufacturing industry, however, no

7. Waste containing putrescible material.

8. DoE (1995).

comparable data could be found for construction or commerce. These costs generally ranged between 0.01% and 1% of annual gross output, the average being around 0.1%.

4. Implementation of the Landfill tax proposals

4.1 *Rationale for the tax*

The intention to introduce a landfill tax was announced in the 1995 Budget. The Government issued a consultation paper on details of the tax proposal along with the announcement. The stated objectives of the tax were:

- To ensure that landfill waste disposal is properly priced, which will promote greater efficiency in the waste management market and in the economy as a whole; and
- To apply the polluter pays principle and promote a more sustainable approach to waste management in which we produce less waste, and reuse or recover value from more waste.⁹

There was much criticism of the initial proposal that the tax was to be on an *ad valorem* basis, that is, a fixed percentage of the cost, rather than being weight-based. An *ad valorem* tax was recommended on the basis that it would provide a straightforward proxy for the environmental impacts of landfill disposal. The rationale for this was that charging in proportion to the costs of landfill disposal, it would result in a higher tax for more 'difficult' wastes, which are more expensive to dispose of and a lower tax for inactive waste, which is cheaper to dispose of. The responses to this approach were that the initial intention of internalising externalities had been abandoned in favour of a revenue-raising exercise and an *ad valorem* tax would increase the existing price differential between sites. This would also penalise more costly sites with higher environmental standards and contribute both to an increase in environmental pollution and the transport of wastes over long distances.

A number of responses came out in favour of a weight-based landfill tax as it was argued that this would result in lower price differentials, that the revenue would be more predictable and that with calibrated weigh-bridges there would be less scope for fraud and evasion.

A consultation exercise on a lower tax band of waste was completed in February 1996. It was proposed that the lower tax band should be linked to inactive category waste listed in the Department's national waste classification scheme which was introduced in April 1996. This category is limited to those wastes which do not chemically or physically react, biodegrade, or adversely affect other matter with which they come into contact in a way that is likely to give rise to environmental pollution.

4.2 *Introduction of the tax*

The landfill tax was introduced on 1 October 1996. It was to be a weight-based tax, which was set at two levels, reflecting the CSERGE estimated externalities associated with landfill. Inactive waste, such as construction waste, which does not release greenhouse gases was to be taxed at £2 per tonne. Active waste, on the other hand, which includes all other types of waste including biodegradable waste was to be taxed at a higher, standard, level of £7 per tonne, taking into account the higher potential environmental impacts. A key feature of the scheme in its final form was the return of revenues. The revenue was hypothecated back to those paying tax through reductions in business's national insurance contributions; and it was also possible for landfill operators to obtain tax rebates by making contributions to environmental trusts.

9. HM Customs and Excise (1995).

4.3 *Recycling of revenue – National Insurance Contributions reduction*

The Chancellor had indicated that the landfill tax would not impose extra costs on business overall. He announced in the March 1996 Budget a reduction of 0.2% in the main rate of employers' national insurance contributions from April 1997. (This change brought the main rate of employers' national income contributions to the same rate as employee national income contributions, exactly 10%).

4.4 *The environmental bodies credit scheme*

When the landfill tax was introduced, a landfill operator making payments to a registered environmental trust for expenditure by the trust on approved environmental purposes, received a rebate from their landfill tax of up to 90% of the payments made to trusts. Trusts would need to be non-profit distributing bodies within the private sector. There was an upper limit to the payments to trusts qualifying for rebates of 20% of the landfill tax payments made by landfill operators.

The aims of the scheme were to help reduce reliance on landfill and to compensate those who lived in the vicinity of landfill sites through 'environmental improvements'. The scheme was regulated by ENTRUST¹⁰, which managed the schemes to which landfill operators contributed if they wished to claim a tax credit. The objectives which a project should meet were: the reclamation of land, reduction of land pollution, protection of the environment, public amenities within the vicinity of a landfill site and restoration or maintenance of environment or buildings within the vicinity of a landfill site. A wider variety of environmentally based organisations can enroll as Environmental Bodies, given that they are non-profit making and that they are not controlled in any way by a local authority or landfill operator.

A number of practical flaws began to emerge with the scheme and at the policy level there was a certain amount of disquiet that not enough of the funds were being directed to areas that would reduce reliance on landfill. The 1999 Budget statement emphasized the intention to extend the protection of the environment objective to include recycling. A Select Committee on Environment, Transport and Regional Affairs (1999) report expressed concern that a disproportionate amount of funding was going into the 'information' categories within the objectives.¹¹ The scheme is due for revision with a view to concentrating revenues on sustainable waste management activities and a consultation on the options took place in 2002.¹²

5. *Changes to the landfill tax since its introduction and recent new proposals*

5.1 *The 1998 landfill tax review - changes to the rate and proposed landfill tax escalator*

A review of the landfill tax was begun by HMCE in 1997. It considered the impact of the tax in relation to its environmental objectives and whether the structure could be simplified or improved in line with the Government Statement of Intent on Environmental Taxation.¹³ It examined whether the tax encouraged companies to reduce waste production, dispose of less waste to landfill and recover more value from waste that is produced.

10. The Environmental Trust Scheme Regulatory Body Limited, a private company appointed by HM Customs and Excise.

11. Cf. <http://www.publications.parliament.uk/pa/cm199899/cmselect/cmenvtra/150/15002.htm>.

12. DEFRA (2002).

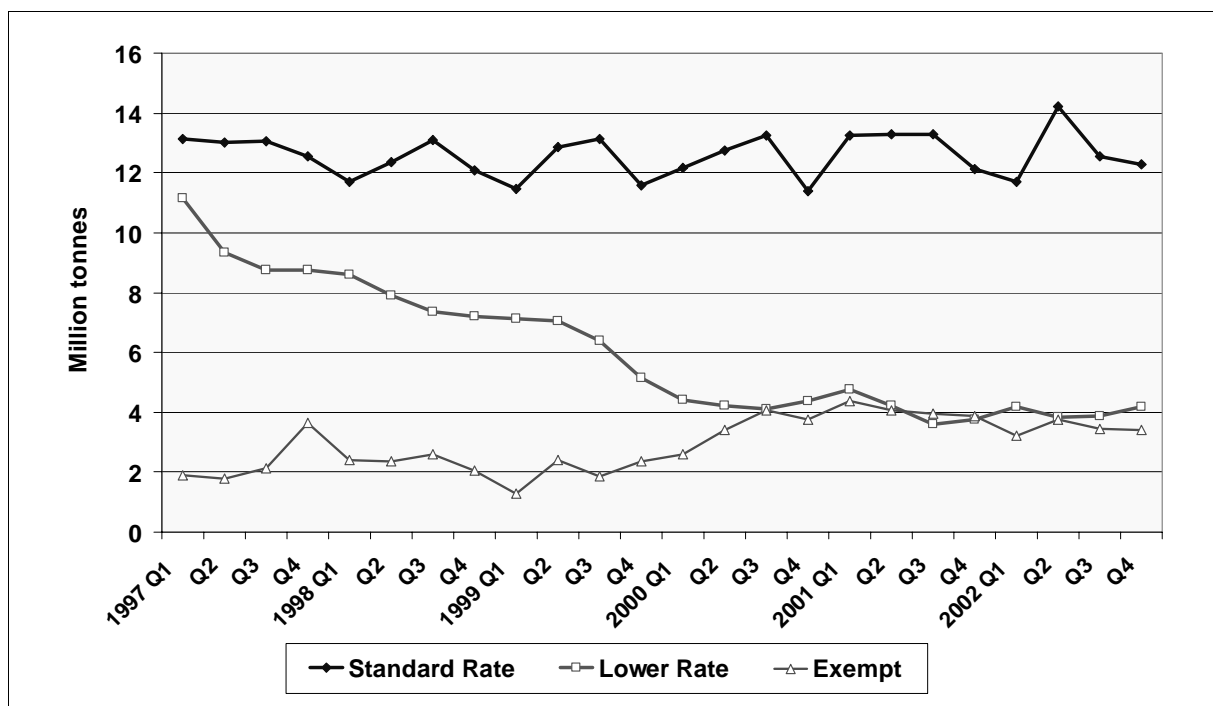
13. Cf. http://www.hm-treasury.gov.uk/topics/environment/topics_environment_policy.cfm.

The review was published in March 1998 on the basis of responses to a consultation exercise.¹⁴ It concluded that there was evidence that the landfill tax has influenced business's waste management decisions with almost a third of companies having begun, or considering, waste recycling, re-use or minimisation as a result of the tax in combination with the Packaging Regulations.¹⁵ A large section of respondents to the consultation believed that there had been an increase in the re-use and recycling of wastes as a result of the tax. It also noted that there appeared to have been a substantial reduction in the quantity of inactive wastes going to landfill. This impression was confirmed by figures for three month moving averages over the period November 1996 to October 1998, which show a reduction over the period of over 30% in lower rate wastes. There was, however, less evidence of any significant impact on volumes of active waste going to landfill (see Figure 2, which graphs all the available data, for this period of time).

The review considered two alternative approaches to setting landfill tax, firstly setting the landfill tax on an assessment of externalities; and secondly setting the tax on the basis of what would be required to deliver environmental targets. It is explained that at its introduction the externalities approach was preferred and regarded as providing a more acceptable basis for taxation. At that point in time, predicting the impact of the tax on behaviour was regarded as problematic because of the lack of information regarding the waste industry and the state of knowledge with regard to the elasticity of demand for landfill.

As experience of the tax had increased HMCE was now in a better position to set rates in a way designed to affect behaviour. The existing approach whilst internalising externalities had little actual impact on reducing the volume of active waste and this was becoming a policy imperative. Landfill tax was described as being a potentially cost effective means of meeting various targets such as those included in the Packaging Directive and the then, draft EU Landfill Directive.

Figure 2. UK Quarterly Landfill Tonnages



14. HM Customs and Excise (1998).

15. Ecotec Research and Consulting (1997).

Thus, a key recommendation of the review was that the basis of calculation of the tax rates should shift towards setting rates to achieve environmental targets:

'We...recommend that consideration be given to further increases in the standard rate of tax, as a potentially cost-effective means of meeting targets arising from the proposed EU Landfill Directive and the revised national waste strategy; and that consideration be given to setting out a programme of such increases over a number of years'.

In the March 1998 Budget, following the recommendations of the Landfill Tax Review, there were a number of changes made to the structure of the tax.

- The standard rate of landfill tax for active waste was increased from £7 to £10 per tonne from 1 March 1999; the standard rate would be further increased “*should research indicate that tax is a cost-effective means of achieving expected new waste strategy targets*”.
- The lower rate of landfill tax for inactive wastes remained frozen at £2 per tonne because of concerns about the availability of inactive construction and demolition waste for constructing, operating, and restoring licensed landfill sites; and
- There was an exemption from landfill tax for inactive wastes used in the restoration of landfill sites from 1/10/99.

Discussions were then held on whether to introduce a 5% real increase on the standard rate of tax (without a pre-announced end date but one that is subject to a five yearly review); or alternatively introduce an escalator involving a £1 per tonne per annum increase in the standard rate of tax for the next five years.

In the 1999 budget, the government announced a landfill tax escalator of an additional £1 per tonne each year. This escalator to apply for at least five years until 2004 when the rate will be £15 per tonne.

5.2 Changes in the policy context 1999-2001

The Landfill Tax review anticipated the introduction of the EU Landfill Directive in 1999 (Council Directive 1999/31/EC) and the need to provide the incentives to divert waste from landfill. Landfill was seen as being at the bottom of the 'waste hierarchy', which ranks waste management options by desirability: waste reduction at source is the most desirable, followed by re-use of materials, recycling, incineration and landfill. The focus of the Landfill Directive is bio-degradable municipal waste and the requirements of the Directive are as follows:

- To reduce the volume of biodegradable municipal waste sent to landfill to 75% of that produced in 1995 by 2010, 50% of that produced in 1995 by 2013 and 35% of that produced in 1995 by 2020 (these targets take into account a 4 year derogation offered by the EU to countries like the UK which are heavily reliant on landfill).

In addition, there are tighter controls on the materials that can be landfilled. Other measures to be put into place are:

- An end to the co-disposal of hazardous and non-hazardous wastes from 2004 when separate landfills are required for hazardous, non-hazardous and inactive wastes;
- Landfill of whole tyres is banned in 2003 and shredded tyres in 2006;

- A ban is already in place for liquid wastes, certain clinical wastes and certain types of hazardous waste;
- To accompany these measures there are provisions for controlling, monitoring, reporting and closure of sites.

The 25% recycling/composting target set in the UK's Environmental Protection Act was largely aspirational and was set in the absence of any clear policy or delivery instruments. In 1999 a Government document, 'A Way with Waste' recognised that this target would not be met by 2000 (the recycling rate for 1997/98 was 8%) and formed part of a consultation for a new waste strategy.¹⁶ This was realised in the publication of the UK's Waste Strategy 2000,¹⁷ which identified the need to minimise waste and recycle and compost or re-use waste that was produced, it also put instruments into place to do this, in particular, some of the measures introduced were:

- Setting local authorities statutory targets for recycling and composting of household waste for the first time, these equate in aggregate to a national target of 25% in 2005/06 and 30% in 2010/11;
- Introducing legislation for a landfill allowance scheme setting limits for the amount of biodegradable municipal waste which councils may send to landfill;¹⁸
- Setting up of Waste and Resources Action Programme (WRAP), which aims to help strengthen the market for recyclables;
- Increased resources through the Environmental Protection and Cultural Services (EPCS) Standard Spending Assessment, establishing a Waste Minimisation and Recycling Fund and funding through the Private Finance Initiative to provide for projects with very high recycling rates.

5.3 Background to increases above the landfill tax escalator

November 2002 to May 2003 saw five key publications which have implications for the future of the landfill tax:

1. The Strategy Unit report 'Waste not, Want not' (November 2002);
2. Government guidance 'Tax and the environment: using economic instruments' (November 2002);
3. The Pre-Budget Report 2002 (November 2002);
4. Budget 2003 (April 2003); and
5. The Governments response to the Strategy Unit Report (May 2003).

Following the waste summit that the Government held in 2001, the Prime Minister commissioned his Strategy Unit to look at how improvements could be made in order to make greater progress in

16. DETR (1999).

17. DETR (2000).

18. Waste and Emissions Trading Bill, November 2002.

this area. They were asked to examine the economic framework, targets, regulations and other instruments that will help us to meet our international obligations in the most cost effective and environmentally sustainable way.

The Strategy Unit also recommended a rise in the landfill tax to £35 a tonne for active waste in the medium term.¹⁹ On the basis of their analysis, they stated that this would reduce reliance on landfill by making it economic to develop alternatives, but stressed that the revenue needs to be redistributed to ensure alternatives are developed without significant impact on business competitiveness and Local Authorities.

Taking into account the Strategy Unit recommendation, Governments guidance on the use of economic instruments and the environment and analysis undertaken by HMCE, the Pre-Budget Report 2002 stated that:

'The current landfill tax escalator, introduced in 1999, commits the Government to raise the standard rate of tax for active waste by £1 per tonne each year until 2004-05, by which time it will have reached a rate of £15 per tonne. As announced in Budget 2002, there is a strong case for increasing the tax rate significantly in future years to provide incentives for the development of alternatives to landfill and to reduce the volume of waste disposed in this way. The Government will therefore consult on a revenue neutral proposal to increase the landfill tax escalator to £3 per tonne in 2005-06 and to increase the rate of tax by at least £3 per tonne in future years, on the way to a medium to long-term rate of £35 per tonne. The Government's intention is that increases will be introduced in a way that is revenue neutral to business as a whole. The Government will consult with stakeholders on options for the package, including the recycling of revenue, before making its decisions'.

Budget 2003 confirmed the increases announced in Pre-Budget Report 2002. Work is ongoing to determine how best to re-distribute the additional landfill tax revenues (i.e. above £15 p/t) back to business and local government.

The Government published its response to the Strategy Unit report in May 2003,²⁰ but it had already acted in the following key areas:

- Landfill tax will be increased by £3 per tonne in 2005/06 and by at least £3 per tonne in the years thereafter, on the way to a medium to long term rate of £35 per tonne. This will be the foundation for the economic framework the Strategy Unit recommended;
- The Landfill tax Credit Scheme has been reformed and a proportion of the funding will be re-directed to a new Sustainable Waste Management Programme in England in 2004/04, 2004/05 and 2005/06;
- A new Sustainable Waste Management Programme managed by Defra's Environment Department, will concentrate on improving waste minimisation, recycling and composting, and researching new technologies for dealing with those wastes which are not readily reduced, reused or recycled;
- A new Delivery Team and Steering Group is being established in Defra to drive forward implementation of the Government's response to the Strategy Unit report and new programmes of work in Defra and WRAP;

19. Strategy Unit (2002).

20. DEFRA (2003b).

- Local authority funding of £90m each year for 2004/05 and 2005/06 has been provided for the Waste Minimisation and Recycling Fund or its successor the Performance Reward Fund.

5.4 Interaction of the landfill tax and other new policy instruments which aim to achieve municipal landfill diversion

The original landfill tax was designed as an environmental tax to internalise the negative externalities of disposal to landfill. The purpose of the announced landfill tax increases (and the current landfill tax escalator) is to achieve behavioural change and send a long-term signal to municipalities and business that the relative costs of disposal are going to shift in the future. The resulting change in the price differential between landfill and the other waste management options will create an incentive to divert waste from landfill, and help move the UK up the waste hierarchy and towards its EU Landfill Directive targets.

As well as the announced landfill tax increases, the Government is also introducing a Landfill Allowance Trading Scheme (LATS) for England for biodegradable municipal waste in 2004/05.²¹ This should provide greater certainty of achieving the EU Landfill Directive targets in the most cost effective way. A new sustainable waste management programme is being introduced, the Waste Implementation Programme (WIP), which will concentrate on improving waste minimisation, recycling and composting, and researching new technologies for dealing with those wastes which are not readily reduced, reused or recycled.

The WIP programmes, the landfill tax and LATS should be viewed as complementary initiatives. The WIP programmes provide the resources and framework to divert biodegradable municipal waste from landfill, the landfill tax provides the incentive and the Landfill Allowance Trading Scheme provides the certainty of hitting Landfill Directive targets (something the WIP programmes and landfill tax cannot guarantee).

The case for using both taxes and a tradable permit system in tandem are set out in HMT guidance on economic instruments for the environment:²²

'In some circumstances there may be a case for combining taxes and trading schemes where they achieve greater net benefits, or benefits can be achieved with more acceptable impacts. For example, a tax may help to reduce the regulatory burdens of a tradable permit scheme and provide incentives for behavioural changes, which might help to ensure that the objectives are met at lower overall cost'.

The simultaneous application of a tax (landfill tax) and a tradable permit system (Landfill Allowance Trading Scheme) does not change the total resource costs, but does affect the distributional outcomes (which are dependent on how the revenue is used and how the permits are allocated).

6. Evaluation of the success of the UK landfill tax

There are a number of criteria by which success could be assessed, but based on the aims for the landfill tax set out in the original 1995 HMCE consultation, two questions are key:

21. Under the Landfill Allowance Trading Scheme, landfill allowances will be allocated to each Waste Disposal Authority (WDA) at a level that will enable England to meet its EU Landfill Directive targets. These allowances will be tradable, allowing WDAs with low costs of diversion from landfill to divert more than they need to, and to sell their surplus allowances to those WDAs with high costs of diversion.

22. HM Treasury (2002).

1. Was landfill waste disposal 'properly priced', were the calculations of externalities on which the tax were based reasonable?
2. Has the tax promoted more sustainable waste management through diversion from landfill?

6.1 Were the calculations of externalities on which the tax were based reasonable?

i) A new evaluation of the values of the externality estimates

Was landfill waste disposal 'properly priced', were the calculations of externalities on which the tax were based reasonable? The original estimates were criticised but current estimates are broadly consistent with these.

Coopers & Lybrand *et al.* (1996)²³ undertook a cost-benefit analysis of the different municipal solid waste management systems. The analysis considered municipal solid wastes only across the Member States of the European Union. The results and analysis undertaken broadly support the methods used and externality estimates derived by CSERGE *et al.* (1993).

Coopers & Lybrand *et al.* (1996) used a life cycle model to predict the environmental impacts of the waste management systems, including landfill. The report adopted a waste composition from Warren Spring Laboratory (1991) for the UK. The use of this waste composition, which it quotes as being "still relevant data", re-confirms the relevance of the waste composition used by CSERGE *et al.* (1993).

Table 3. External cost estimates per tonne of waste (1999 prices)

Options (1999£/tonne of waste)	UK
Present – mixed refuse collection, bring system for recyclable and organic materials	
Landfill – no gas recovery	4
Landfill – gas flared	4
Landfill – energy generation (marginal displacement)	3
Landfill – energy generation (average-mix displacement)	4
Landfill – no transfer	3
Incineration – electricity generation (marginal displacement)	-17
Incineration – electricity generation (average-mix displacement)	10
Recycling	-161
Composting	NP
Present – co-collection of mixed refuse and recyclable and organic materials (blue box)	
Landfill	3
Incineration – electricity generation (marginal displacement)	-17
Incineration – electricity generation (average-mix displacement)	10
Recycling	-167
Composting	NP
Present – separate collection of mixed refuse and recyclable and organic materials (wheelie bins)	
Landfill	3
Incineration – electricity generation (marginal displacement)	-17
Incineration – electricity generation (average-mix displacement)	10
Recycling	-161
Composting	NP

Note: All externality estimates are quoted as external costs. A minus sign therefore precedes those values which are external benefits. NP indicates an option which is not practiced. It should also be noted when comparing these figures that the data taken from the 1996 paper has been presented in 1999 prices

Economic values were then applied to each of the environmental impacts of the waste management system, to calculate the net external cost of each option. As with the CSERGE report the damage value for greenhouse gas emissions were all based on the assumptions made by Fankhauser

23. Coopers & Lybrand, CSERGE and EFTEC (1996).

(1992), with the exception of CO. This implies that the assumptions made in the CSERGE report were considered to be well founded and were still appropriate 3 years on. The CSERGE figures take account of the costs of global pollution, air pollution, transport impacts, leachate and pollution displacement. The Coopers & Lybrand report breaks total damage down into transport pollution, energy use, operation, displaced pollution, bags/bins and transport accidents. Table 2 details the external cost estimates per tonne of waste for the UK.

Examining the figures for present – mixed refuse collection, the external costs of landfill vary from £3 per tonne of waste for landfill with energy generation (marginal displacement) and landfill with no transfer, to £4 per tonne of waste for all other landfill options.

ii) A recent study to estimate disamenity cost of landfill – confirmation of the original estimate

In 1999 the Government commissioned a study to identify and estimate the disamenity costs of landfill in Great Britain, that is those local nuisance costs experienced by households living close to landfill that are associated with it such as odour, dust, litter, noise, vermin, and visual intrusion.²⁴ The study uses landfill data made available by the Environment Agency and the Scottish Environmental Protection Agency together with individual house price and ward-based socio-economic data drawn from Cambridge Econometrics' AHPD database. The study covers Great Britain, except for small parts of the North West and East Midlands where data are not currently available. Data for Northern Ireland are not yet available on a comparable UK basis. The combined database identifies a core data set of 11,300 GB landfill sites (some 6,100 licensed as operational in 1993/94) and the study has associated these sites with 592,000 housing transactions from 1991–2000 inclusive.

Controlling for both physical and socio-economic factors there remained a statistically significant stock disamenity effect for houses located closer than 0.5 miles to a GB landfill site. This gave an average reduction of about £5,500 in the value of houses lying within the zone of 0.25 miles from operational GB landfill sites and about £1,600 for those between 0.25 and 0.5 miles of such sites. Taking house prices at their 1995 values, but updating for consumer price inflation, this generated a mean estimated total present value of fixed disamenity of £2,483m at current prices, within a 95% confidence interval of £2,041m–£2,925m. The equivalent 95% confidence interval estimate of the present value of fixed disamenity effects of landfill is between £334,350 and £478,990 per landfill site in GB, and this corresponds to a nominal measure of fixed disamenity cost of between £1.52 and £2.18 per tonne of landfill at current prices for an assumed average flow of 100 million tonnes pa for 28 years at a 6% discount rate.

6.2 *Has the tax promoted more sustainable waste management?*

i) Diversion from landfill

It was observed in the 1998 HMCE landfill tax review that the tax had incentivised a reduction in inactive waste but that the quantity of active waste to landfill had remained unchanged. As is demonstrated in Figure 3 this trend has continued, active landfill has remained relatively stable whilst there has been significant reductions in inactive waste to landfill due in part at least to significant re-use of construction and demolition waste. From the first full year of data, 1997/98, inactive waste landfilled has fallen from 35.4m to 15.7m tonnes in 2001/02 (see Table 3).

24. DEFRA (2003a).

Table 4. Quantities of waste and landfill tax receipts

Year	Standard Rate Tonnage Waste (a) 000's	Lower Rate Tonnage Waste (a) 000's	Exempt Tonnage Waste (a) 000's	Total Receipts(b) £million	Standard rate of tax £ per tonne
1997-98	50,363	35,440	9,961	360.7	7
1998-99	49,006	29,606	8,302	333.1	7
1999-00	49,743	23,012	9,215	429.7	10
2000-01	50,182	17,262	15,836	462.1	11
2001-02	50,349	15,685	15,275	502.1	12

ii) UK business's evaluation of the landfill tax

The Advisory Committee on Business and the Environment (ACBE) produced a report on landfill tax and resource productivity in 2001 which recommended an increase in landfill tax to achieve environmental objectives. The report concluded that the UK landfill tax at its current levels has 'not stimulated a reduction in the amount of waste taxed at the standard rate'.²⁵ The report was produced prior to announcements about the increase in the escalator but contains an assessment of the landfill tax on the current escalator and what direction it should take.

The message contained in their report was that Government needs to send strong and consistent signals to business regarding the need for waste minimisation and greater resource productivity. The landfill tax regime as it stood at that time had not stimulated a reduction in the amount of waste taxed at the standard rate. The costs of landfill remain much lower than in other European countries. They conclude that higher landfill tax would not be harmful to business as they are a very minor cost and that there was widespread support throughout the waste reuse and recycling industry for change. On the basis of this evidence they conclude that what is needed is:

'A significant increase in landfill costs from 2004 and that increases in tax revenues are used to provide increased levels of financial support and incentives to business to improve resource productivity'.

Apart from the need for the tax to be higher, earlier, this view concurs strongly with current policy in using the landfill tax to create behavioural change. In addition, tax revenues are going to be used, at least in part, to provide targeted support for businesses to minimise waste and the provision of incentives to invest in alternative waste management options.

7. Conclusion

The UK landfill tax, its level and the purpose for which it was implemented has evolved and developed since 1996. From a starting point of seeking to internalise externalities and incentivise sustainable waste management, policy considerations have changed the focus. There have been three changes to the £7 per tonne standard rate of tax since its introduction, an increase to £10, an escalator to £15 and more recently the announcement that from 2005/06 an annual increase of £3 per annum to a level of £35 per tonne in the medium term. These changes have been driven by an acceptance that landfill tax must be increased to achieve behavioural change, through closing the cost gap on methods of diversion from landfill and ultimately to contribute to the incentive to achieve diversion to meet EU Landfill Directive targets on municipal waste.

25. ACBE (2001).

This is not to say that the landfill tax at its inception was set incorrectly. It is clear that the current increases go beyond the level which would internalise the externalities. Within the remit the original HMCE consultation set for the tax it could be said that the UK landfill tax has been a success. Subsequent estimates have confirmed the credibility of the original externalities estimates. There has also been a significant impact on the quantity of inactive waste sent to landfill, in the main due to the re-use of construction and demolition waste. Active waste going to landfill has remained stable and it is clear that if this is to be reduced landfill tax will have to increase further.

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WASTE TAX IN NORWAY

By Torhild H. Martinsen and Erik Vassnes¹

1. Background

1.1 Green Tax Commission

In Norway, the question of taxation of waste was first raised by a Green Tax Commission in 1996². The Commission recommended a partial shift in the tax system from taxation of labour to taxing the use of natural resources and harmful emissions. The Commission pointed out that the most important environmental problems in the area of waste treatment, were associated with polluted leachate and emissions to air. The costs of environmental problems were not fully reflected in municipal charges. In order to confront those in possession of waste with more correct relative prices between waste supplied to landfills and waste incinerators, and to stimulate delivery of waste for recycling and waste reduction, the Commission proposed that a tax on final waste disposal should be considered. The Commission recommended to give such considerations high priority. Further, the Commission recommended that the tax as far as possible should cover all external costs associated with disposal and incineration of waste.

1.2 Introduction of a tax on final waste treatment

The Government followed-up with proposals to the Storting (the Norwegian parliament) in 1998 [St. prp. nr 54 (1997-98) Grønne skatter]. The waste tax was implemented 1 January 1999. The objective of the tax was to price the environmental damage connected to end treatment of waste. The tax was expected to contribute to an increase in source separation and recycling and thus reduce the amount of residual waste. It was also a political aim to stimulate the utilisation of energy from incineration of waste. The tax rate of waste to incineration plants was therefore designed with a basic tax and an additional tax which was reduced according to the degree of utilisation of energy from waste in the incineration plants. The tax is estimated to generate about 500 million NOK³ in governmental revenue in 2003.

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1. Ministry of Finance, Norway. This paper is to a large extent based on a report from a working group which considered a change in the waste tax in Norway. Their report was submitted to the Ministry of Finance 15 June 2001. The authors of this paper bear the sole responsibility for its contents, and our views do not necessarily represent the views of the Ministry of Finance.
 2. "Green Taxes – Policies for a better Environment and High Employment", NOU 1996: 9. English summary available in the Ministry of Finance.
 3. 1 EURO equals approximately 8 NOK.

Table 1. Tax rates on waste in 2003

	Tax rates in NOK per tonne
Landfills,	
High environmental standard	327
Low environmental standard	427
Incineration plants,	
Basic tax	82
Additional tax dependent of energy utilisation	0-245 ¹⁾

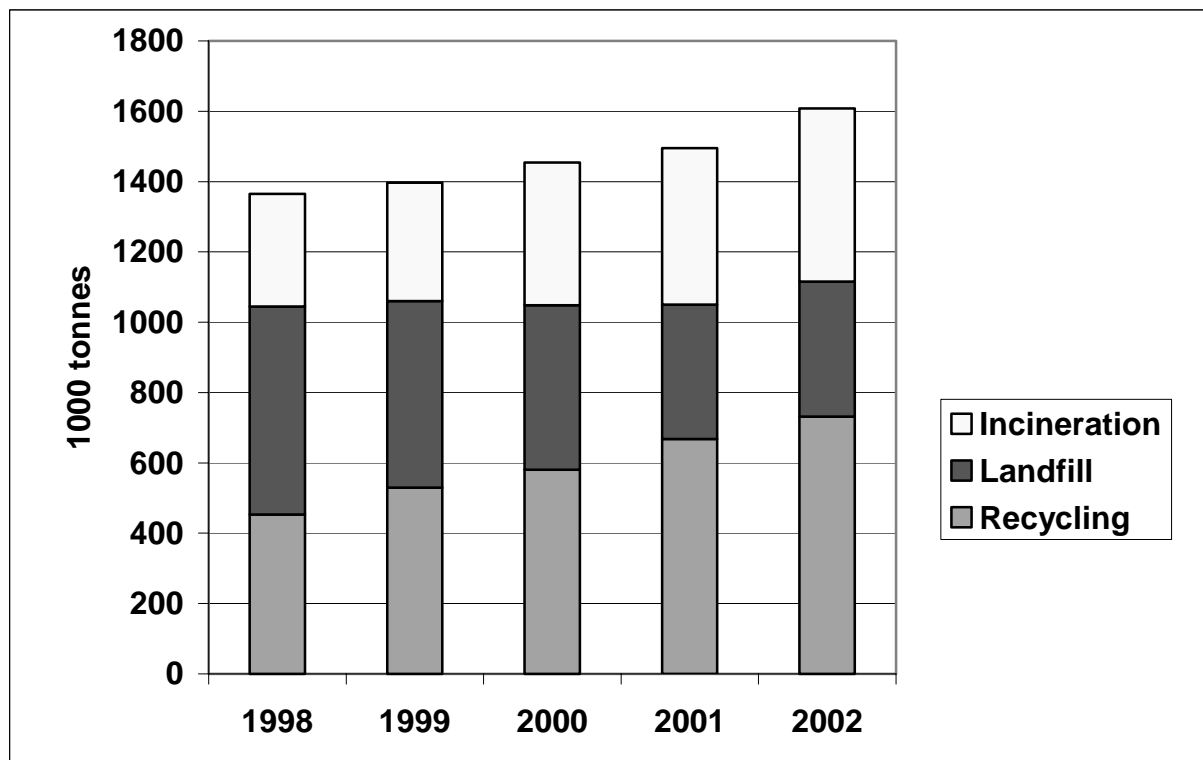
1) Depending on the degree of energy utilisation in the plant

1.3 *Effects of the tax on generation of waste, recycling and emissions*

It is difficult to evaluate the effects of the tax on generation of waste, recycling and emissions due to simultaneous effects of other factors. According to Statistics Norway, the amount of household waste entering municipal waste collection systems has gradually increased between 1998 and 2002 cf. Figure 1.

The percentage of recycled waste has increased considerably during the same period. The relative increase in recycling is assumed to be, at least partially, an effect of the waste tax. In the same period there has been a decrease in the amount of waste going to landfills and an increase in the amount of waste going to incineration. This development can be due to the fact that the average tax per tonne waste is higher for waste delivered to landfills than for waste delivered to incineration plants.

Figure 1. Treatment of household waste 1998-2002
1000 tonnes



To make households face the real environmental costs of waste, the total social costs have to be included in the municipal waste charges. According to the Norwegian Pollution Act all costs connected to the treatment of waste must be included in the municipal waste charges. The municipal charges on waste have increased in recent years. This can indicate that households and others taking part in the municipal waste collection systems to a larger extent than before face the real environmental costs of waste.

Introduction of differentiated charges will increase the economic motives for households to increase recycling and stimulate waste reduction. In practise the differentiation of charges can be related to environmental costs according to weight or waste fractions. Even though most municipalities have some sort of differentiated waste charges, there is considerable potential for further differentiation of these charges. According to the Norwegian Pollution Act, the municipalities are only *requested* to introduce differentiated waste charges. A recent statistical analysis by Statistics Norway indicates that municipalities with relatively low waste charges have the highest percentage of recycling (Hivju and Smith (2003)). This might indicate that a high rate of recycling and thus relatively less waste delivered to landfills and incineration can result in lower waste charges.

The tax differentiation according to energy utilisation leads to reduced tax on incineration plants that produce energy from waste. This implies that waste delivered to such plants does not face the full environmental costs of final waste treatment. There have also been indications that the existing differentiation has led to increased incineration, but not to a similar increase in the production of energy from waste. This has raised the question if the tax differentiation could be replaced by a more targeted measure to stimulate energy production from incineration plants.

1.4 *Changes in the waste tax*

The Storting has asked the Government to consider how the tax on waste treatment could be changed in such a way that it to a larger extent would stimulate utilisation of energy from waste. The Storting also asked the Government to consider levying a tax directly on emissions, and that the tax to a larger extent should correspond to the environmental costs of waste.

To meet the requests of the Storting, a working group was appointed by the Ministry of Finance in order to consider changes to the waste tax. The working group submitted a report to the Ministry of Finance 15 June 2001. In the budget proposal for 2002, the Government proposed that the tax on waste delivered to incineration plants be changed to a tax on emissions. A tax on emissions would to a larger extent correspond to the environmental costs of waste. It was also proposed to replace the differentiation of the tax on incineration of waste according to the degree of utilisation of energy with a direct subsidy dependent on the amount of energy produced from waste. The subsidy would also apply to production of energy from landfills. Compared to a tax reduction dependent on the degree of energy utilisation, a direct subsidy dependent on the actual energy produced will be a more precise measure and can to a larger extent stimulate production of energy from waste.

The Government proposed a differentiation of the tax on waste to landfills dependent on the environmental standard of the landfill, see Table 1. This tax differentiation implies that the tax to a larger extent corresponds to the environmental costs of depositing waste.

The proposals were all adopted by the Storting and were meant to be implemented 1 July 2003. However, only the differentiation on waste to landfills was implemented from 1 July. Since the Government considered the tax on emissions from incineration plants to be connected to the subsidy scheme for energy production, the tax was also postponed until 1 January 2004, pending the acceptance of the subsidy scheme by the EFTA Surveillance Authority (ESA). The subsidy scheme is not yet accepted by ESA.

2. Environmental costs of final waste treatment

2.1 Theoretical background

The purpose of environmental taxes is to introduce a more correct pricing of environmental damaging activities and thereby lead to a more effective use of common resources. Simultaneously such taxes create governmental revenue and thus reduce the need for other distorting taxes.

Environmental taxes give polluters incentives to reduce their emissions. To ensure that the reductions are made at the lowest possible cost, environmental taxes should be levied directly on the harmful emissions. For a given input of waste, emissions can be reduced either by reducing the amount of harmful components in the production process or by cleaning the emissions. The social costs of reducing emissions are connected to cleaning of emissions, decreased production and to the fact that more resources are put into the production processes.

According to economic theory, a tax on emissions should equal the marginal environmental cost of the emissions. When all polluters faced the same tax, emissions are generally reduced where the reduction costs are lowest. As long as it is more expensive for the polluter to pay the tax than to reduce emissions, emissions will be reduced. Theoretically the polluters will reduce emissions to the level where the marginal cost of reducing emissions equals the tax rate. Emission reductions will thereby be distributed among the polluters in a cost-efficient way.

Exemptions from environmental taxes do not represent a cost-effective policy, and are not in line with the polluter pays principle.

To introduce an emissions tax, it must be possible to measure the actual emissions. In practice this might be very complicated and also expensive. As described in section 2.3, this is the case for landfills. An approximation to an emissions tax might be to tax input factors that cause emissions or products resulting from polluting production processes. When the tax on final waste treatment was introduced in Norway, it was designed as a tax per tonne of delivered waste. Taxing input factors or products does not necessarily lead to cost efficiency. Although such taxes give incentives to reduce the use of products causing emissions, in this case the amounts of waste, they give no incentives to reduce emissions.

As described above, an optimal tax rate on emissions would equal the marginal environmental cost of emissions. To be able to determine such a tax rate, one has to estimate the environmental damages in monetary value. During the last ten years, there has been considerable research to develop value estimates for different kinds of harmful emissions to water and air. By translating the external effects of emissions into a monetary value, a foundation is laid for determining the level of environmental tax rates. Estimation of environmental damages is subject to severe uncertainty. When calculating such estimates one has to investigate how the society values the damages caused by emissions. Because these damages are not priced in the market, one has to use other, indirect, methods to evaluate and calculate the social costs.

It is complicated to evaluate environmental costs. Final waste treatment causes harmful emissions to soil, air and water, and is a source to local and global environmental problems. Today the most severe problems are emissions of chemicals that are hazardous to human health or the environment, greenhouse gases, organic matter and nutrients. A further description of environmental damages from final waste treatment is given in sections 2.2 and 2.3.

2.2 *Taxing emissions from incineration*

2.2.1 *Different tax rates for different substances*

The most important factor contributing to harmful emissions from incineration of waste, is the content of substances in the waste. This specifically applies to emissions of carbon dioxide, heavy metals, sulphur and chlorine, conditions concerning incineration technology and cleaning of emissions. Cleaning equipment might reduce emissions to a large extent. Today it is not possible to clean emissions of greenhouse gases at acceptable costs, while it is economically feasible to clean other emissions, i.a. emissions of chemicals.

Emissions to air from incineration of waste can be divided into four main categories:

- *Greenhouse gases*, mainly CO₂, but small amounts of methane due to unfavourable incineration conditions, might also occur.
- *Other gases*, hereby nitrogen oxides, sulphur dioxide, volatile organic compounds (VOC), hydrogen fluoride and hydrogen chlorine.
- *Dust*, containing non-combustible (inert) particles and carbon which is not combusted (soot). Dust contains i.a. heavy metals.
- *Chemicals that are hazardous to human health or the environment*, of which the most important ones are dioxins, mercury, cadmium, lead, chrome, copper, manganese, nickel and arsenic.

When introducing the tax on incineration of waste, the environmental costs of emissions of hazardous substances and chemicals causing local air pollution were estimated against three valuation alternatives and estimated at 160, 330 and 800 NOK per tonne of household waste respectively. Later estimates are 900-1,500 NOK per tonne of household waste. The costs of emissions of dust, NO_x, SO₂ and VOC were estimated at approx. NOK 80-120 per tonne (ECON, 2000). These estimates are of course subject to uncertainty. Using the same basis of estimated values adjusted for actual emissions in Norwegian incineration plants in 1999, an implicit tax on emissions of hazardous substances and chemicals causing local air pollution was estimated at an average of NOK 180 per tonne of waste (NOK 4-1,400 per tonne). For other gases and dust the estimation was respectively NOK 65 per tonne (NOK 24-952 per tonne) and NOK 43 per tonne (NOK 2-93 per tonne) (ECON, 2001). The implicit tax per tonne waste is based on the estimated environmental costs due to emissions from incinerating an average tonne of mixed waste.

Based on these estimates it seems like the existing tax rate does not fully cover the environmental costs of some incineration plants while overpricing the costs of others.

Incineration of waste also causes emissions of climate gases, particularly CO₂. The environmental costs of these emissions are not priced by the existing tax. ECON (2001) estimates an implicit tax based on net CO₂ emissions at approximately NOK 40 per tonne waste for existing plants.

CO₂-emissions from incineration of biological waste do not cause *net* climate gas emissions. Waste fractions that contain plastics or carbon from other fossil matter on the other hand, cause net emissions of CO₂ when incinerated. This must be taken into consideration when designing a tax on emissions. In practice however, it is impossible to separate CO₂ emissions from biological waste and CO₂ emissions from fossil matter. If one wants to take *net* emissions of CO₂ into account when estimating the tax, a share reflecting the amount of fossil waste in mixed waste can be calculated. Waste fractions causing net emissions of CO₂ when incinerated account for approximately 13% of mixed waste from households. A share of 13% of total CO₂ emissions from incineration plants can therefore be used as an estimate of net emissions and be added to the tax base.

As shown in Table 2, ECON (2001) estimated that the joint implicit tax on waste for incineration in 2001 amounted to about NOK 324 per tonne of delivered waste. The calculations were based on emissions in 1999.

The emission tax will vary among plants and over time, dependent on the level of emissions. On average, the tax level on incineration of waste will correspond to a tax rate of NOK 327 per tonne of waste (implicit tax). This level corresponds to the existing tax level when adjusted to prices in 2004, and when the tax is not reduced according to the degree of energy utilisation. The plants will probably adjust to the tax, i.a. by cleaning their emissions, so that the tax actually paid will be somewhat lower.

When determining the tax level on waste, the possible harmful effects to the environment due to abandonment, dumping or uncontrolled disposal of waste should also be taken into account.

Table 2. Implicit tax on incineration of waste, 2001

	Environmental cost, NOK per unit	Implicit tax, NOK per tonne waste	Measurement required by the EU-directive ¹⁾
Climate gases, tonne		39,00	
CO ₂	130	38,40	
Methane	2.730	0,60	
Other gases, kg		65,40	
SO ₂	17	7,10	Continuous
NO _x	15	25,20	Continuous
VOC	4	2,80	-
HF	20.000	25,40	Continuous
HCl	100	4,90	Continuous
Dust, kg	565	43,00	Continuous
Chemicals hazardous to health or the environment, gram		176,50	
Dioxins	2.300.000	23,80	Every 2 years
Mercury (Hg)	27	2,30	Every 2 years
Cadmium (Cd)	52	2,50	Every 2 years
Lead (Pb)	62	14,30	Every 2 years
Chrome (Cr)	559	64,30	Every 2 years
Copper (Cu)	0,3	0	Every 2 years
Manganese (Mn)	93	53,50	Every 2 years
Nickel (Ni)	9,1	14,00	Every 2 years
Arsenic (As)	9,5	1,60	Every 2 years
Sum		323,90	

¹⁾ Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste.

2.2.2 Practical and technical conditions

Introducing an emissions tax requires that the incineration plants measure their emissions. Measuring will increase the plants' costs. As of 2005 the EU directive on incineration of waste (Directive 2000/76/EC of the European Parliament and of the Council of 4 December 2000 on the incineration of waste) requires continuous measuring and cleaning of several emissions. To which extent the costs of measuring should be attributed to the tax will therefore depend on whether the tax is introduced before the directive comes into force and whether the tax requires further measuring than the directive. Regulations corresponding to the EU directive came into force in Norway from 1 January 2003 for new plants and will come into force from 1 January 2006 for existing plants.

Incineration plants that intend to operate after 2005 already have installed, or are in the process of installing, cleaning equipment to fulfil the requirements in the new EU directive. Increased use of

chemicals in the cleaning process to reduce emissions will also increase the plants' costs, see 2.2.3. This is not only because the expenses for chemicals will rise, but also because the amount of special waste will increase. The plants may invest in equipment that recycles the chemicals so that the chemicals can be used again.

The EU directive intends to prevent or restrict the negative environmental effects from waste incineration. The directive establishes stringent requirements according to the technical conditions at the plant, i.e. by determining maximum emission values and requiring measuring of emissions. The requirements are more stringent than the existing Norwegian ones, and implementing the directive will therefore have consequences for the plants, but to a varying extent, depending on the technical conditions at each plant.

Table 3 below shows the measurements made today at four of the Norwegian plants, and the measurements required by the EU directive. The incineration plant in Bergen is one of the most modern and clean plants in Norway, while the plant in Hallingdal is a smaller plant which would profit from a presumptive tax, see below.

Table 3. Measurements in four Norwegian incineration plants today and measurements required by the EU directive

Parameter	Current measurements				EU directive	
	Bergen, cont.	Trondheim, per.	Årdal, cont.	Hallingdal, per.	Continuous	Periodic
Dust	x		x	x	x	
Hg		x		x		x
Cd+Tl		x		x		x
Pb+Cr+Cu+Mn+Sb+As+Co+Ni+V+Sn		x		x		x
CO	x		x	x	x	
HF	x			x	x ¹	
HCl	x		x	x	x ¹	
TOC	x			x	x	
NO _x	x		x		x	
NH ₃	x					
SO ₂	x		x	x	x ¹	
Dioxins		x				x

¹⁾ Periodic measurements are allowed when absolute conditions of maximum emissions and cleaning systems are fulfilled.

The EU directive on incineration of waste requires continuous measurements of CO, HF, HC₁, TOC, NO_x and SO₂. (For HF, HC₁ and SO₂ periodic measuring is accepted as long as the conditions on maximum emissions and cleaning systems are fulfilled.) In addition the directive requires periodical measurements of HG, Cd+Tl, Pb+Cr+Cu+Mn+Sb+As+Co+Ni+V+Sn and dioxins.

Climate gas emissions from waste are not intended to be covered by an early Norwegian quota trading scheme that is being prepared. Climate gas emissions from incineration of waste should therefore be covered by the emission tax. As described above, only a small share of the total climate gas emissions, however, comes from fossil matter that should be priced by a tax on emissions. Measuring CO₂ emissions will cause increased measurement costs for incineration plants that have not yet installed the necessary equipment. The EU directive does not require measuring of CO₂ emissions. Although some of the plants in Norway already have equipment that can be upgraded to perform continuous measurements of CO₂ emissions, such upgrading will still not make it possible to measure the *net* CO₂ emissions that we want to tax. As mentioned above, CO₂ emissions can not be cleaned at an acceptable price, and a tax on these emissions will therefore in practice not lead to cleaning of these emissions. Viewed against this background, it was decided to base the CO₂ element of the tax on the

weight of the delivered waste, i.e. a tax per tonne. The disadvantage is that the tax on waste will be based on both emissions and weight, and this will make the tax more complicated. For plants liable to pay the tax this adjustment will not be of any importance, since the EU directive requires that plants register weight of delivered waste anyway. Plants that can prove that they do not burn any fossil waste will be exempted from the CO₂ component. If there is evidence that the assumed share of 13% fossil waste is no longer a reasonable approximation, it can be adjusted.

In principle it would be preferable to do continuous sampling and monthly analyses for *heavy metals* and samples and analyses every six months for *dioxins*. The fact that equipment for continuous measuring of emissions of heavy metals (except for Hg) and dioxins does not exist is probably the reason why the EU directive only requires two samples per year for these emissions. To simplify the control for the authorities and to prevent further costs for the incineration plants, it was decided to base the tax for heavy metals and dioxins on the requirement of two annual measurements according to the EU directive.

Gases that give negligible contribution to the overall emissions were kept outside the tax base for practical reasons. Today this applies to methane and VOC. New information about the various substances might, however, make it relevant to include or exempt substances from the future tax base. When methane and VOC are not included in the tax base, the implicit incineration tax in 2001 would amount to about NOK 285 per tonne, see Table 2.

In the tax proposal approved by the Storting, only chemicals that are covered by the directive were included in the emission tax.

The emission tax will not require more measuring than what is required under the EU directive. Incineration plants that are not yet fulfilling the requirements of the directive must however, as a consequence of the emission tax, fulfil these requirements before the directive comes into force.

Small incineration plants, i.e. plants receiving less than 20 000 tonnes of waste each year, will be allowed to choose a presumptive tax. The tax will be based on the expected emission coefficient from a tonne of mixed waste. This was done to avoid that small plants be exposed to extensive measuring costs. In 2002 only three of the small plants would benefit from choosing the presumptive tax.

The changes in the tax may cause increased administrative costs for the authorities in a transitional period. The costs are related to obtaining technical knowledge regarding measurement of emissions. The change in the tax will also require assistance from the pollution control authorities as to measuring emissions and to the correct stipulation of the tax rate.

2.2.3 Expected effects of introducing an emission tax on incineration plants

Incineration of waste causes emissions of several components to water and air. The amount of emissions depends on the incineration technology, cleaning technology and what kind of waste that is burned. In principle the emissions from incineration plants can be reduced in four different ways: by optimizing the incineration technology, through cleaning the emissions, by sorting out especially harmful waste fractions or by reducing the amount of waste delivered to the plants.

The current tax on final waste treatment is assumed to lead to increased source separation and recycling, which lead to reduced amounts of waste delivered to incineration. The tax, however, does not lead to changes in the incineration conditions, in the cleaning of emissions or in the sorting-out of certain types of waste. Theoretically, a tax on emissions will also create economic incentives for cleaning in a broad perspective.

A tax on emissions will probably affect the choice of cleaning technology when upgrading existing plants or building new plants. Because a tax on emissions will price the emissions directly, it will be profitable to invest in cleaner technology since the tax due will decrease when emissions decrease. In addition, a tax on emissions will probably increase the sorting-out of unwanted waste components. Improved arrangements for handling special waste and electric and electronic waste and increased ambitions regarding use of chemicals in products that are hazardous to health or the environment, will be of great importance for the emission results. Waste fractions leading to the largest emissions (special waste, electric and electronic products, etc.) are already given alternative treatment, according to prevailing regulations. Further source separation or changing of existing schemes will increase costs related to final waste treatment.

In principle it is desirable that an emissions tax on incineration of waste covers all activities involving incineration of waste or waste-based fuels. As of today there is not sufficient data to analyse all the consequences of broadening the base of the waste tax, i.e. the consequences for industries is so far not possible to evaluate. When the results from the impact analysis of the implementation of the EU directive are available, a broadening of the tax base should be considered further.

In practice there are four main ways to reduce emissions in existing plants:

- Improved company routines;
- Increased use of input factors in cleaning processes (chalk and active coal);
- Technological improvements, using best available technology (BAT);
- Reduce the amount of unwanted components in the waste (salts, heavy metals and dioxins).

ECON (2001) found that company routines in Norwegian plants are good, and that further improvement of the routines would not lead to reduced emissions. They also suggested that some plants may extend their use of chemicals in the cleaning processes. Chemicals, however, generate special waste and use of chemicals will only be profitable up to a certain level. For the time being, installation of bag filters is considered to be the best available technology to fulfil the EU requirements. All incineration plants will most likely install bag filters as a consequence of the EU directive. The introduction of an emission tax will therefore probably not change the choice of technology.

As to the building of new incineration plants, an emission tax will probably have crucial effect on the choice of cleaning technology. As described above, a more correct pricing of the environmental costs of incinerating waste will make higher investment costs more profitable, because the emissions tax burden decreases as emissions are reduced.

2.3 *Taxing emissions from landfills*

The most damaging emissions from landfills are methane emissions. Emissions of methane gas from landfills amount to about 7% of Norway's total emission of climate gases. CO₂ and methane are two of the 6 climate gases included in the Kyoto protocol. The global warming potential (GWP) of methane is considered to be 21 times larger than CO₂ and methane thus amounts to 21 CO₂ equivalents.

Methane emissions depend on the amount of carbon in the waste, technical conditions at the landfills and on the climate. Given today's technology, it is impossible to measure methane emissions from landfills. It is, however, possible to measure the amount of methane collected from the landfills. Collected methane is used either for energy purposes or simply flared.

When introducing the tax, several options to price methane emissions from landfills were considered. It was agreed that it would be practically impossible to levy the tax directly on emissions or to differentiate the tax by the content of carbon in the waste. Because organic waste amounts to over 80% of mixed waste, a tax on municipal waste was considered to be relatively accurate for pricing the environmental cost of methane emissions. The cost of methane emissions from landfills was estimated to about NOK 100-550 per tonne mixed waste in landfills with gas collection and NOK 200-1,100 per tonne for landfills without gas collection. As the large intervals indicate, the estimates are uncertain. In more recent analyses the cost of methane emissions from landfills with gas collection is estimated to respectively about 170 and 230 NOK per tonne of mixed waste (ECON, 2000). Since these estimates indicate a lower environmental cost, the existing tax rate might be considered to be higher than the environmental cost of methane emissions. Since most of the landfills were required to have gas collection when the tax was introduced, a tax differentiation was not proposed. According to the EU directive on the landfill of waste⁴, all landfills should have gas collection.

The existing tax rate does not cover the environmental costs of emissions of hazardous substances from landfills. These costs are estimated to about NOK 360-2,170 per tonne of mixed household waste for existing landfills with 25% gas collection (ECON, 2000). As the large interval indicates, these estimates are uncertain. For landfills with 50% gas collection and cleaning of leachate the estimates are reduced to NOK 12-30 per tonne. It is, however, regarded as unlikely that these landfills can reduce their emissions to this low level.

The emissions depend on the content of hazardous substances in the waste, the type of sealing of the base and sides of the landfill and the type of gas collection. Incentives to reduce emissions of hazardous substances could therefore be created by differentiating the tax either by actual emissions, type of waste or by technical conditions at the landfill. When introducing the waste tax, it was considered that the tax would not be an accurate measure to price the emissions of hazardous substances. Technological conditions still make it unfeasible to do so. In practice it is also considered impossible to differentiate the tax by the type of waste delivered to landfills. From 1 July 2003 the tax was however differentiated, based on whether the landfills are in compliance with EU requirements on sealing of the base and sides of the landfill or not, see below.

As described above, it is impossible to measure methane emissions from landfills, and an emissions tax on landfills is not practically possible. Therefore the tax on waste delivered to landfills is designed as a tax per tonne delivered waste. When introducing the tax, the main cost was considered to be emissions of methane, and the tax level was based on the pre-existing CO₂ tax. The cost of methane emissions from landfills was estimated to about NOK 100-550 per tonne mixed waste in landfills with gas collection. When the waste tax was introduced 1 January 1999, the tax rate was set to NOK 300 per tonne waste delivered to landfills. The tax rate corresponded to the tax rate on incineration of waste.

From 1 July 2003 the tax was differentiated so that landfills fulfilling EU requirements on sealing of the base and sides of the landfill face a tax rate of NOK 327 per tonne, while the remaining landfills pay a tax rate of NOK 427 per tonne. The difference of NOK 100 per tonne is considered to correspond to the environmental cost of leachate.

The change of the tax increased the costs for landfills that do not fulfil the EU requirements on landfill sealing. As for the rest of the landfills, the consequences of the tax change are considered to be minor.

4. Council directive 1999/31/EC of 26 April 1999 on the landfill of waste.

3. Measures to stimulate energy production from waste

When the waste tax was implemented, it was also a political objective to stimulate the utilisation of energy from the incineration of waste. The tax rate on waste delivered to incineration plants was therefore reduced, according to the degree of energy utilisation from waste in the incineration plants. The differentiation implies a reduction of the tax rate between 0 and 75%, depending on the degree of energy utilisation in the incineration plant. Utilisation of energy from waste does, however, not reduce the emissions from incineration of waste, and the environmental costs of final waste treatment are not influenced by the use of energy from waste. As a result of the energy differentiation, the tax rate on waste delivered to incineration plants only covers about 45% of the environmental costs of waste incineration.

In some analyses it is taken into account that use of energy from waste can replace more harmful energy sources, such as oil. Some have therefore claimed that the tax on waste should take such factors into account. The objective of the waste tax is, however, not to increase the energy use from waste, but to give incentives to waste reduction. Impacts from different energy sources should in principle be addressed by taxing energy sources directly, and according to the environmental costs they involve.

According to economic theory, the number of objectives should not be higher than the number of policy measures. When the objective of the waste tax is waste reduction (to increase recycling) the tax should not at the same time be used as a measure to increase the use of energy from waste. When one policy measure is used to achieve more than one objective, a conflict between the different objectives may arise. The energy differentiation of the waste tax contributes to lower the price on environmental costs from waste treatment. When the environmental costs of incineration of waste are not fully implemented in the price facing the households, the economic motives for waste reduction is decreased. The differentiation of the waste tax according to energy utilisation can imply both a weakening of the incentives to waste reduction and a too strong subsidy of waste-based energy generation.

In addition to weakened economic incentives for waste reduction, the differentiation of the waste tax is an inaccurate measure to increase the production of energy from waste. If it is a political objective to increase the energy production from waste, it would be a more accurate measure to introduce a direct subsidy on the amount of energy produced. Such a direct subsidy will to a larger extent stimulate to incineration of waste with high calorific value. This is not the case with the existing differentiated waste tax according to the degree of energy production of the incineration plant.

A direct subsidy on produced energy should be given to all producers of energy from waste, i.e. not only production of energy from incineration but also energy production of gas from landfills.

As described in section 1.4., the Storting has decided to replace the differentiation of the tax on incineration plants according to the degree of utilisation of energy with a subsidy dependent on the amount of energy produced from waste. This subsidy will also cover production of energy from landfills.

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PVC WASTE IN DENMARK - COSTS AND BENEFITS OF ALTERNATIVE TREATMENTS

By Niels Buus Kristensen¹

1. Introduction

This paper summarizes the main findings from an economic analysis of the costs and benefits of potential future implementation of two different chemical treatment processes for PVC waste as alternatives to conventional disposal via incineration or landfilling:

- the “**Watech**” process based on pyrolysis
- the “**Stignæs**” process based on hydrolysis

The two processes have been developed by Danish companies and have so far only been implemented on test-scale. The purpose of the analysis has been to provide background material for political decision on revision of the environmental regulation of the treatment of PVC waste in Denmark.

1.1 *The environmental problem*

PVC (polyvinyl chloride) is an important material with many applications in industry, commerce and households. Its stability and unproblematic blending with a wide range of substances combined with low production costs have resulted in its use as basic material for a very wide range of products. By adding stabilizers, plasticizers, pigments, fillers, and other additives various PVC compounds with very different properties can be obtained. The products are numerous ranging from shoe soles and cables based on PVC in its flexible form to window profiles, roofing and pipes in its rigid form.

The raw materials for vinyl chloride, the basic building block of PVC, are hydrocarbons from oil or natural gas, and sodium chloride from salt deposits. The production process is energy demanding and may result in environmental impacts if not controlled. However, this study focuses on the environmental aspect of alternative ways for the disposal of PVC products. The production and installation of PVC-based products generates a pure form PVC waste, e.g. cut-off, which is typically recycled directly into the production in a closed system and hence not considered to be genuine waste. Apart from that the majority of European PVC waste is either incinerated (15%) or landfilled (82%)².

As described below, landfilling and incineration of PVC give rise to different direct environmental impacts, long-term environmental risks and avoidance costs to minimize these

1. COWI A/S, Denmark. This paper is based on a study undertaken by COWI for the Danish Environmental Protection Agency. The results are final apart from any comments from peer reviewers.

2. AEA Technology (2000) p. 26.

problems. Alternative treatment technologies for PVC waste have been developed either based on “mechanical recycling” through for example grinding of disposed PVC-products or “feedstock recycling” involving a chemical treatment for decomposition of PVC waste.

1.2 The current Danish situation

The Danish Waste strategy “Waste 21” formulates a so-called “waste hierarchy” for measures to reduce environmental problems caused by waste. First of all waste amounts should be reduced, and, secondly, waste should be recycled to the extent that it is technically and economically possible. Finally incineration is given priority over landfilling. But specifically for PVC the strategy states that PVC waste should be sorted out separately and that PVC waste which cannot be recycled should be landfilled and not incinerated because of the environmental hazards related to the flue gas resulting from incineration of chloride and heavy metal content in PVC Miljø- og Energiministeriet(1999).

Apart from a user fee to cover the costs of waste treatment waste producer also have to pay a waste tax amounting to 375 DKK/tonne (50 /tonne) for landfilled waste and 330 DKK/tonne (44 /tonne) for incinerated waste. Since 1 July 2000 Denmark has levied environmental taxes on commodities made of PVC and on phthalates, the plasticizer in flexible PVC. In 2002 certain products of rigid PVC were exempted from the PVC-tax.

Only a small fraction (1200 tonne) of Danish PVC waste is recycled by using the so-called “Wuppi”-process, which is a mechanical recycling process for rigid PVC. The PVC is collected via the municipalities’ local recycling centres, washed and granulated and subsequently reused by PVC-producers as raw material for new PVC-products. Most of the remaining PVC waste is incinerated and the rest landfilled.

2. Approach

The overall objective of the analysis was to assess the social costs of chemical treatment of PVC waste as an alternative to landfilling and incineration³. The analysis compared four different scenarios with a Reference Scenario over a twenty year period. The Reference Scenario simulating “business-as-usual” assumes that all PVC waste is landfilled or incinerated. In the alternative scenarios a certain proportion of the total amounts of PVC waste is sorted out for chemical treatment instead. The four alternative scenarios are based on combinations of the two chemical treatment processes, “Watech” (1) and “Stignæs” (2), and two levels, “Moderate” (A) and “Maximal” (B) of sorting out PVC waste for either of the two chemical treatment processes.

The quantification of the waste volumes in the scenarios is carried out in three steps:

- Firstly, the total, future volumes of PVC waste are estimated
- Secondly, the potential for sorting out PVC waste for chemical treatment is assessed
- Finally, the chemical composition of the sorted out PVC waste is specified

A short description of each step in setting up of the scenario is summarised in the subsequent sections.

3. It was planned that the existing used mechanical recycling process “Wuppi” should also be analysed in parallel to the two chemical treatment processes, but Wuppi was left out because the sufficient data could not be provided.

2.1 *PVC volumes*

PVC has been used in Denmark since the 1950's. Consumption has steadily increased and is roughly estimated to amount to about 60-80,000 tonnes per year today.

The volumes of PVC waste in Denmark are not well established. The current consumption of PVC-based products is not directly reflected in the waste volumes for PVC because most PVC products are not disposed of the same year as they are bought. The life expectancy of PVC-products ranges from a few years for some consumer goods, such as rubber boots, toys etc., to several decades for building materials, such as window frames, gutters and cables. Some PVC-products, e.g. drain pipes, might not end up as waste at all as they will not necessarily be removed after end usage. Hence, the volumes of PVC waste today and in the years to come depend to a large degree on the volumes of PVC consumption as well as the types of products in the past.

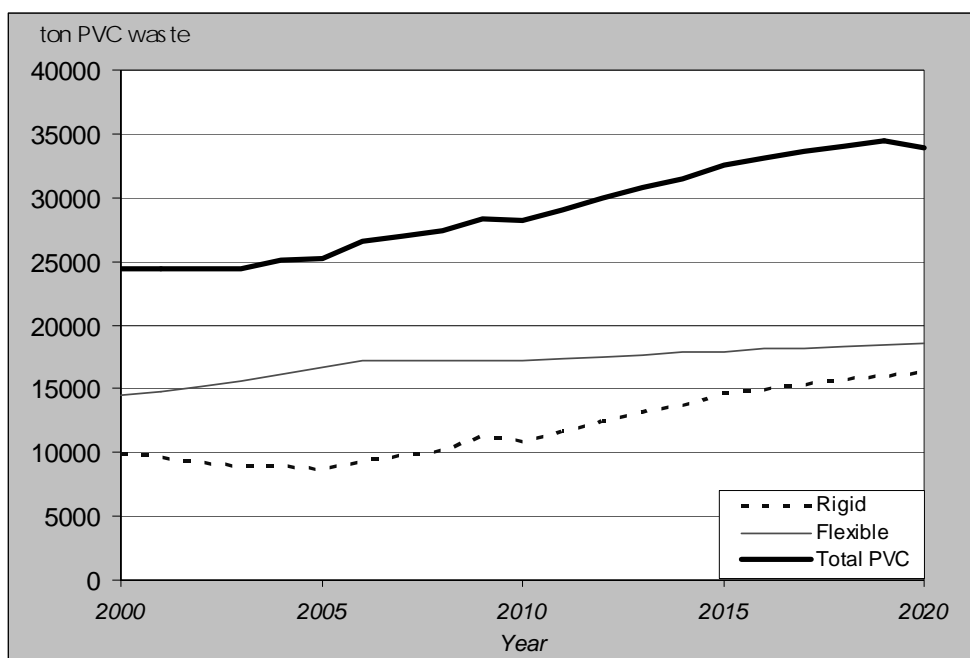
A prognosis for the volumes of PVC over the next twenty year has been established based on the following assumptions⁴:

- Yearly total consumption of PVC since 1950 has been estimated by the Danish Environmental Protection Agency in agreement with the federation Danish Plastic Industry.
- Consumption has been allocated on commodity groups for each year in the period.
- All commodity groups of rigid and flexible PVC-products have been assigned fixed service lives after which they are disposed of.

The resulting estimates of the yearly volumes of Danish PVC waste are presented in Figure 1 below. It appears that the overall volumes are expected to rise from about 25,000 tonnes today to just above 35,000 tonnes and that flexible PVC amounts to slightly more than half of the total amounts throughout the period.

4. RUC/TEKSAM (1986), Miljøstyrelsen (1990), Miljøstyrelsen (1996), Miljøstyrelsen (2003a).

Figure 1. Prognosis for volumes of PVC waste (Compound) 2000-2020



2.2 Sorting out potential for PVC waste for chemical treatment

The majority of PVC waste today is not disposed of in a pure fraction but as mixed municipal solid waste, as part of construction and demolition or as part of the mixed combustible waste material collected at local recycling centres. Only a small fraction is deposited in separate cells in landfills.

This means that a shift to chemical treatment in a significant scale would imply a change in the current flows of PVC waste which would require an effort. Information campaigns, stricter regulations or economic incentives are among the political instruments which could be applied to increase the amounts of sorted out PVC waste. In reality it would never be possible to sort out all PVC waste for chemical treatment. But the bigger the effort the higher would be the rate of PVC which could be shifted away from incineration and landfilling and made available for chemical treatment. This in turn would increase the costs of handling PVC waste even before it enters the gate of the chemical treatment plant.

Estimating realistic levels of sorting out of PVC waste is very difficult. The approach taken is to distinguish between types of waste producers:

- Households and small enterprises
- Construction (“pre-consumer”)
- Demolition (“post-consumer”)
- Other industries

and take into account their shares of the different PVC commodity groups. They produce very different types and amounts of PVC waste per waste producer which will in turn reflect their capability of sorting out significant amounts of PVC waste with a reasonable effort.

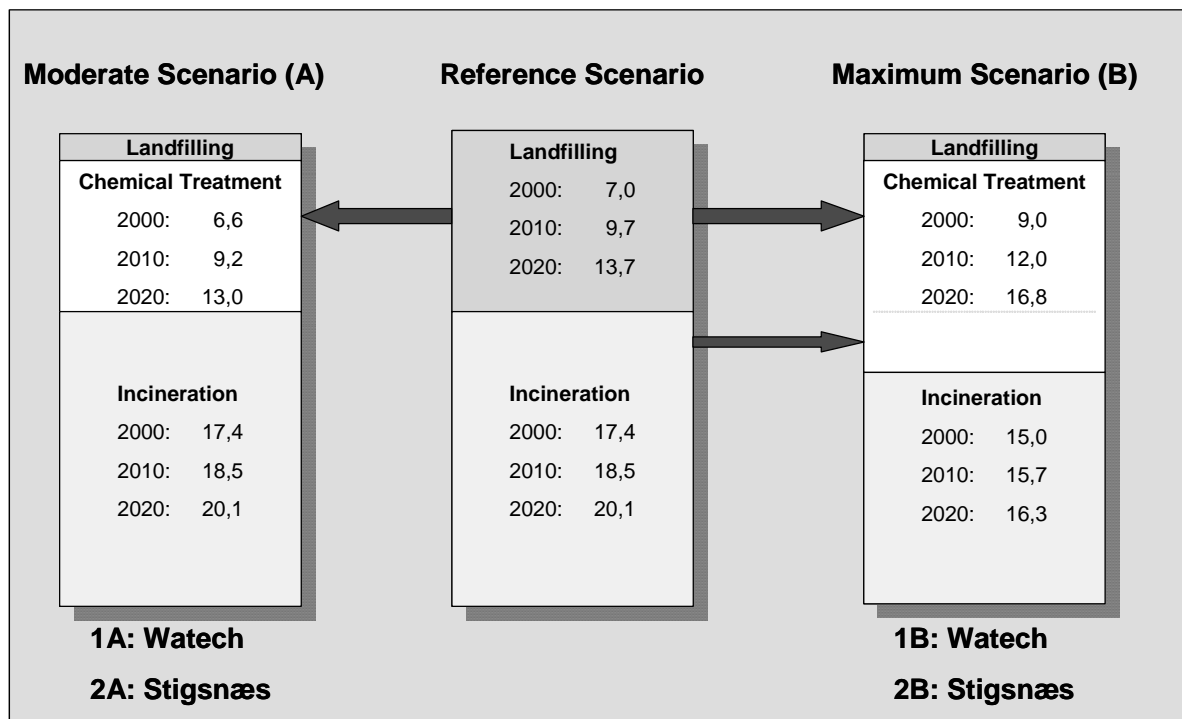
2.3 Scenarios

The level of effort from waste authorities and waste producers put into sorting out PVC waste in separate fractions is what differentiates the “Moderate” and “Maximum” scenarios of the analysis:

- In the *Moderate* Scenarios it is assumed that the effort to increase the sorting out more PVC waste is concentrated on construction and demolition industries which are among the major PVC waste producers, measured in tonnes, and where costs are relatively low. The majority of the waste from these producers would be land filled. No additional initiatives are taken to encourage households and small enterprises to sort out their PVC waste. PVC waste from these two types of waste producers is assumed to be primarily disposed of as part of the municipal solid waste or combustible waste from local recycling centres. In any case will the majority of the waste from these waste producers will end up in incinerators. Hence, in the Moderate Scenario PVC waste shifts predominantly from landfilling to sorting out for chemical treatment.
- In the *Maximum* Scenarios it is assumed that in addition to the sorting out in the Moderate Scenario an effort will be made by minor PVC waste producers by encouraging households and enterprises to hand in their disposed PVC products in separate containers at the local recycling centres. Hence, a shift will also occur from incineration to chemical treatment in these scenarios.

The resulting distribution of the total PVC waste volumes on the alternative treatment processes is illustrated in Figure 2 below.

Figure 2. The allocation of the total volumes of PVC waste on incineration, landfilling and chemical treatment in the four scenarios: 1A, 2A, 1B and 2B. 2000, 2010 and 2020.



In the Moderate scenarios all landfilled PVC waste, except for a few hundred kilograms, are expected to be sorted out for chemical treatment whereas the volumes for incineration remain

unchanged. The shift from landfilling to chemical treatment is maintained in the Maximum scenarios but in addition about 15-20% of the incinerated amounts will be shifted to chemical treatment as well.

2.4 *Composition of sorted out PVC waste*

In order to assess the environmental impacts and the costs of the treatment of the PVC waste it is important to keep track of the composition of the PVC waste. This includes three steps of quantifications:

- The share of sorted out PVC waste which actually consists of PVC
- The distribution on rigid and flexible PVC
- The average composition of the rigid and the flexible PVC

The volumes of PVC presented above equals “pure” PVC waste in the sense that all impurities in terms of other waste materials are not included. For a typical waste producer it can be difficult to determine with certainty whether a given waste product is actually PVC or another type plastic, e.g. PE. This implies that other plastic types will unavoidably be sorted out as PVC by mistake. In addition, many PVC-based products will also contain parts made of other materials such as metals or rubber etc. Finally, dirt will be attached to some PVC products which will also cause that sorted out PVC will contain other materials than PVC.

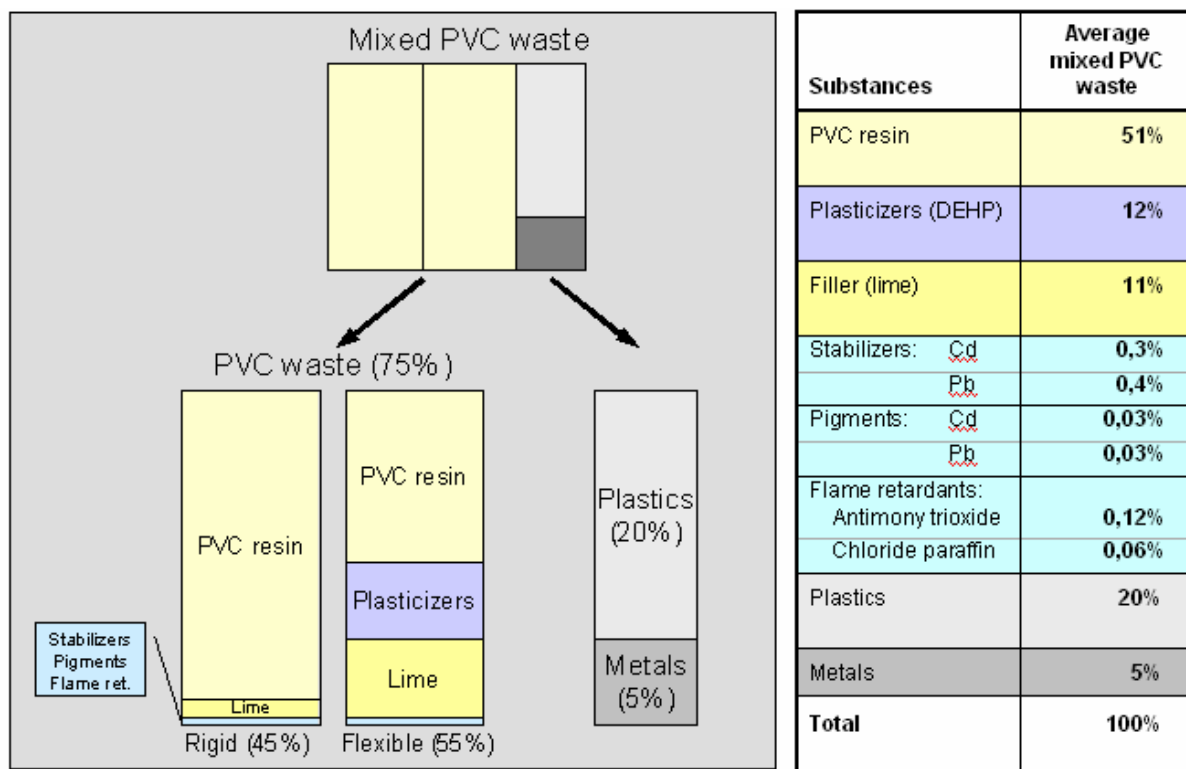
Analysis of samples of sorted out, mixed PVC waste from local recycling centres shows some variations in the share of non-PVC material. It is assessed that if an extended sorting out and collection system is established and backed up with information campaigns it will be possible to maintain PVC shares of about 75% with the rest consisting mainly of other plastics (20%) and metals (5%). This corresponds to the upper end of the observed interval.

Further, the distribution of genuine PVC waste on rigid and flexible PVC will vary over time reflecting historical changes in the composition of consumption of PVC products. The prognosis for PVC waste (see above), which was based on individual commodity groups indicates that the composition will change somewhat from today where rigid PVC amounts to about 40% to a little more than half of the pure PVC waste in 2020. For the sake of simplicity a fixed share of 45% rigid and 55% flexible PVC is used in the subsequent environmental and economic analyses. This corresponds to the forecasted situation in 2010.

The content of PVC resin in the PVC compound is high in rigid PVC, about 95%, whereas flexible PVC often contains 50% or more fillers (e.g. lime) and plasticizers (phthalates). Until recently PVC-products often contained lead, cadmium and other heavy metals as stabilizers and pigments but these substances are being phased out in Denmark. But due to long service life of PVC products heavy metals will still appear in PVC waste in the future.

The estimated composition of the average mixed PVC waste which is sorted out for chemical treatment is illustrated in Figure 3 below.

Figure 3. Composition of mixed PVC waste sorted out for chemical treatment. Average for 2000-2020



3. Treatment technologies

This section briefly describes the four considered treatment technologies (landfilling, incineration, Watech and Stigsnaes) with regard to technology, environmental impacts and costs of the processing of PVC.

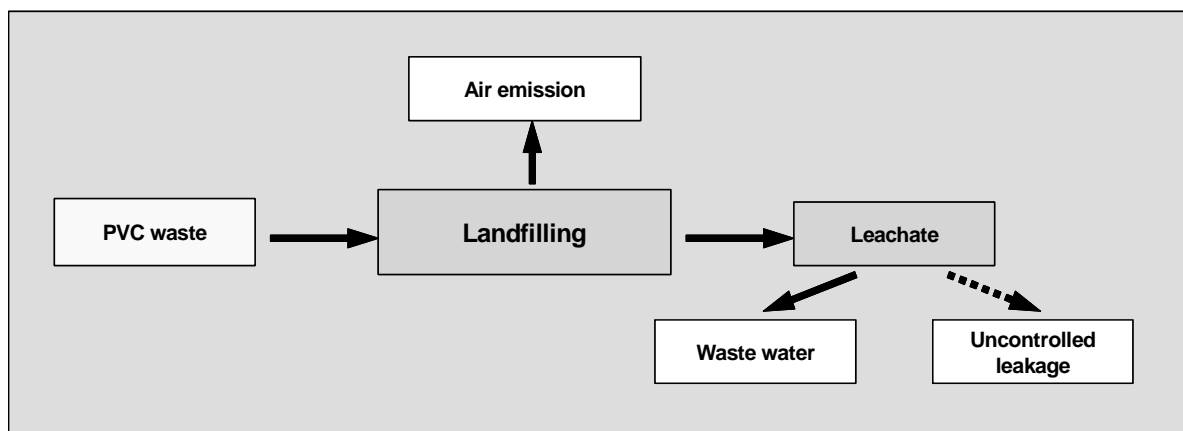
3.1 Landfilling

Landfills in Denmark have been established over a long period concurrently with increases in demand for waste disposal. As a result not all landfills have same level of environmental protection. Stricter environmental regulation imply that all new Danish landfills have a membrane and draining system to secure that leachate is collected and sent to waste water treatment⁵. It is foreseen that over the coming years landfills which do not fulfil the stricter regulation with regard to groundwater protection laid down in the new EU Directive will be closed down.

Here, it is assumed that PVC waste will in the future be disposed of in landfills with effective systems for collection and treatment of leachate so that all waste water discharges fulfil current legislation.

5. Bekendtgørelse nr. 650 af 29. juni 2001.

Figure 4. Flow chart for landfilling



Environmental impacts

The main environmental problems from landfilling are related to air emissions and emissions to soil and water⁶:

- *Air emissions* are primarily CO₂-emissions from degradation of the plasticizers in the PVC compound whereas the PVC resin is considered to be practically inert under landfill conditions. The risk of emission caused by landfill fire is assessed to be negligible as landfilling of organic waste is no longer permitted by the regulations of landfills.
- *Emissions to soil and water* have several aspects. First of all landfilling grounds will of course be contaminated with the harmful substances in the PVC waste, primarily phthalates and heavy metals (Pb and Cd) from stabilisers and colorants whereas chlorine content is assumed to be tied up in the (inert) PVC. Collected leachate will be treated to comply with regulations for waste water emissions. Therefore, environmental damages are considered to be small and the effects not attempted monetarised. Costs of the risk of uncontrolled leachate discharge are roughly estimated based on clean-up costs and probabilities of a uncontrolled discharge. Clean-up costs are allocated equally on all volumes of landfilled waste. The resulting costs per tonne PVC are small as compared to other treatment costs.

There are also other nuisance effects from landfills such as noise, visual intrusion and smell. Smell nuisance is not related to PVC waste. Noise and visual intrusion are not considered because the relevant viewpoint is not the level of nuisance as such but differences between alternative treatment technologies and it is very difficult to assess whether these nuisances are higher or lower than for incineration plants or the chemical treatment plants.

Costs

The costs of landfilling PVC waste have been estimated on the basis of the costs of construction and operation of a new landfill fulfilling adopted environmental EU-requirements. The size of landfill has been determined as the expected average size for future landfills, i.e. a full capacity of 750,000 tonnes and a lifetime of 20 years. On this basis a gate fee of 299 DKK (40) has been calculated for typical landfilled waste. However, PVC waste will in general have a lower density and consequently take up more landfill capacity than typical waste. This has been taken into account by determining which cost components are weight dependent and which are dependent on volume of

6. For further details see: EU DGXI (1999).

the waste. The latter part of costs (about 75%) have been adjusted by a factor 1.6 to take into account that the volume of 1 tonne landfilled PVC waste is 60% higher than for typical landfilled waste. Consequently, the estimated social costs of landfilling PVC waste amounts to 485 DKK (65) per tonne PVC waste.

In Denmark landfills are regulated by the principle of full cost recovery and cross-subsidisation is in principle not allowed. Therefore, the above estimated social costs of 299 DKK for landfilling of typical waste can be used for comparison with gate fees for “mixed non-combustible waste”. These fees differ significantly across landfills depending on local conditions, historical construction costs and regulatory demands at the time of construction. A survey indicated a range from 110 DKK (15) to 750 DKK (100) with an average of 260 DKK (35) per tonne, Miljøstyrelsen (2002a).

3.2 Incineration

Denmark is among the countries which uses incineration most widespread for waste disposal. Danish incineration plants are producing power and heat from the energy content released in the combustion process. All facilities have flue gas cleaning equipment to comply with the Danish regulation of air emissions which is regularly monitored.

The analysis of incineration takes as point of departure a specific incineration plant, “*Vestforbrænding*”, a large, modern plant, which incinerates of about 500,000 tonnes per year. For some specific aspects adjustments of the assumptions are made in order to make the plant more representative for Danish incineration plants. This is for example the case for flue gas cleaning technology where an average allocation on ‘wet’, ‘semi-dry’ and ‘dry’ has been assumed. Costs are also adjusted to compensate for a higher efficiency as compared to the typical plant. Average PVC content of incinerated waste is estimated to be about 0.7%. If this content would be reduced due to sorting out for chemical treatment it would at the margin have an impact on costs and environmental effects from the incineration plant.

Environmental impacts

The main problem with incineration of PVC is related to Chlorine and heavy metal content. Formation of hydrogen chloride in the combustion process places a high demand of alkaline reagents (lime and/or lye) which in turn increases the amount of residue generated and requiring disposal at special landfills.

The flue gas cleaning residues are landfilled as hazardous waste on Langøya in Norway and related environmental costs are assumed to be taken into account in the Norwegian price for receiving the residue. Incineration of PVC will not give rise to a noticeable increase in the amount of slag but will increase the heavy metal content.

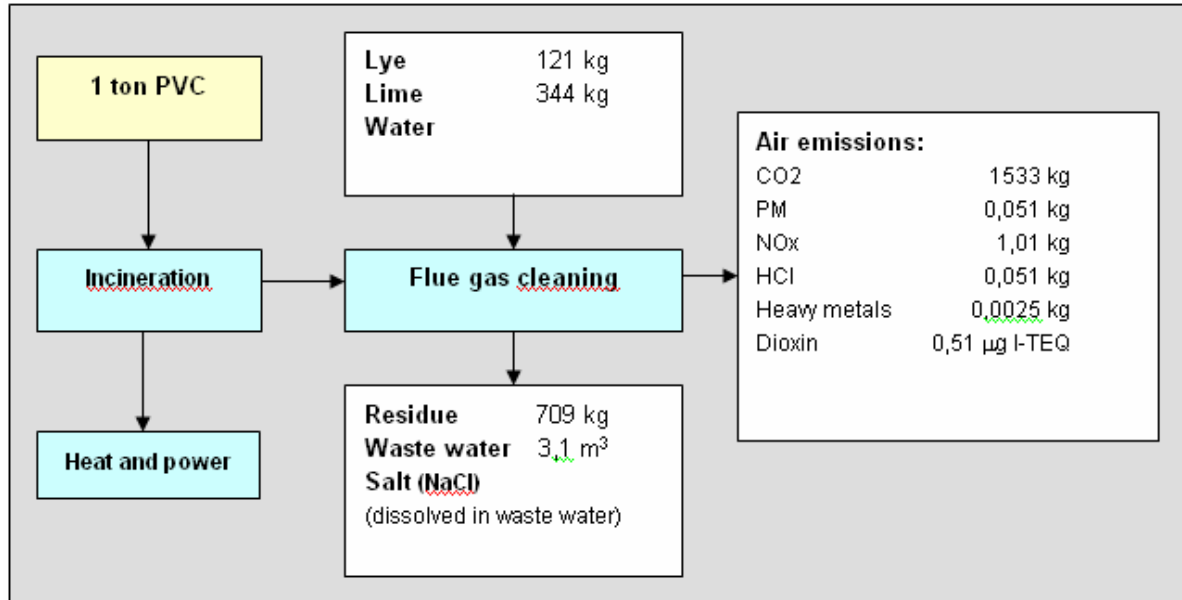
Before cleaning the flue gas from combustion of PVC-containing waste will probably have a higher concentration of certain pollutants, e.g. HCl and perhaps dioxin, whereas SO₂ will not be affected because of the absence of sulphur in PVC. The CO₂ emissions will be directly related to the carbon content of the PVC waste. However, after cleaning the flue gas will still have to comply with the requirements of the regulation which sets standards for the content of the flue gas. This study follows the findings from AEA Technology (2000) and Hjelmar (2002).

In summary, the incineration of PVC gives rise to the following effects as compared to average municipal solid waste:

- Increased use of flue gas cleaning inputs (lime, lye and water)

- Increased volumes of residue from flue gas cleaning
- Changed composition of slag (heavy metal content)
- Increased volumes of waste water (from wet flue gas cleaning)
- Air pollution

Figure 5. Incineration of 1 tonne average PVC waste



Costs

The calculation of incineration costs per tonne PVC waste takes as a starting point average costs per tonne municipal solid waste and makes correction for:

- the higher demand for flue gas cleaning and disposal of residues
- the increased production of waste water
- the increased revenues from the higher energy content
- the higher capital costs due to the higher energy content

All corrections, apart from the increased revenue from the energy production, contribute to increasing the costs per tonne PVC as compared to average municipal waste. Energy content of PVC per tonne is 1.8 times that of municipal solid waste. Therefore, capital costs are also assumed to be 1.8 times higher. This is probably an over-estimate as not all capital costs are related to the energy consumption. On the other hand, variable costs are assumed to be mainly related to the amounts of waste in tonne and therefore assumed to be equal to average costs per tonne municipal solid waste although some of variable costs could also be attributed to the energy production and, hence, allocated according to energy content.

Incineration costs per tonne (pure) PVC waste are estimated to 1.424 DKK (190) per tonne after deduction of revenues from energy production. This is more than twice the costs for municipal solid

waste and is primarily attributable to costs of additional flue gas cleaning input (lime and lye) and disposal of the flue gas cleaning residue costs per tonne.

3.3 *Chemical treatment - the Watech process*

The Watech process for PVC treatment is a pyrolysis based patented technology which is not yet in operation but small scale test runs have been conducted. An overview of the process is given in the process diagram in Figure 6 below.

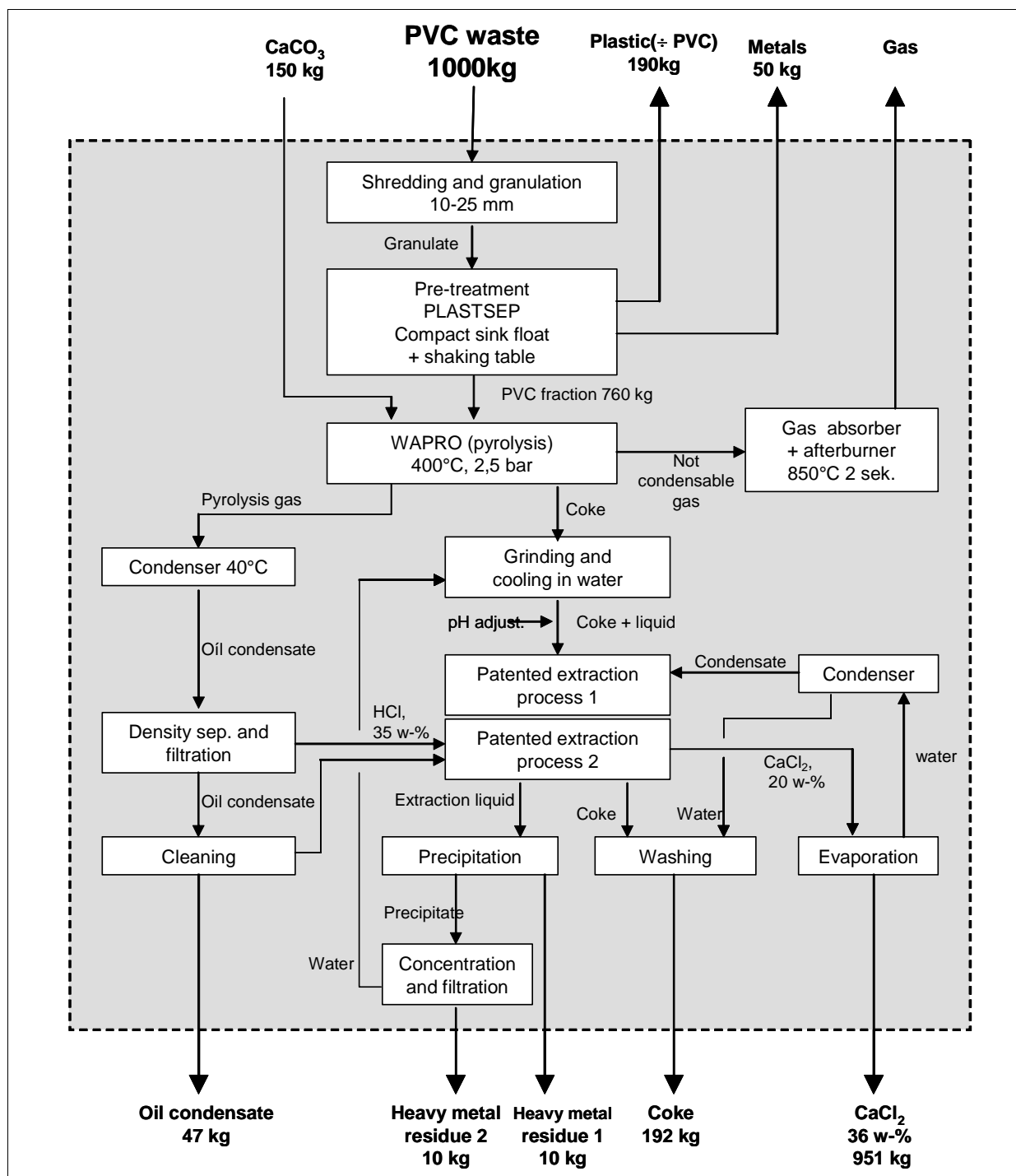
The initial step in the process is to granulate and separate the PVC-, plastic- and metal- fractions of mixed PVC waste. The plastic fraction is sent to an incineration plant and the metal is sold for reprocessing. The PVC-fraction which now contains an acceptable plastic fraction is mixed with calcium carbonate (CaCO_3) and enters the WAPRO-pyrolysis process. At a temperature of 400°C the calcium reacts with the chlorine contents of the PVC forming calcium chloride, water and carbon dioxide. The pyrolysis gas is condensed and the remaining hydrogen chloride is extracted for use in a patented extraction process. The oil condensate is also partly reused for heating purposes in the extraction process. The extraction removes heavy metals from the coke and the CaCl solution is led to evaporation.

The coke, oil condensate and CaCl -solution are expected to be marketable while the two heavy metal residues (40% Pb and 40% Cd) will be landfilled as hazardous waste.

Environmental impacts

The combustion of oil condensate and gas give rise to air emissions of CO_2 , CO, NO_x and dioxin/furan. The coke, which is marketed for energy purposes, will eventually have potential emissions of lead and cadmium when combusted. The disposal of the concentrated heavy metal residues creates a long-termed risk for soil and water pollution.

Figure 6. Process diagram for the Watech process



Costs

Costs per tonne for treatment of PVC waste by the Watech process have been calculated based on information from the process owner. The unit costs depend on the volumes processed because of economics of scale and substantial capital costs. However, the process is designed so that extra modules can be added gradually as demand increases. This implies that the economies of scale

diminish above 10,000 tonne per year. Above these volumes the treatment costs will be in the range 1000 - 1400 DKK (130 - 190) per tonne *mixed* PVC waste.

3.4 Chemical treatment - the Stignæs process

The Stignæs process is a hydrolysis based technology which is not yet in operation but a plant is under construction and test runs have been conducted. The hydrolysis takes place in a 3800 meter long pipe reactor which is available because it was originally constructed for another purpose. An overview of the process is given in the process diagram in Figure 7 below.

The initial step is to granulate the PVC. Metals and dirt is removed by density sorting while the plastic fraction is maintained in the granulate compound, which now enters the reactor along with lye (NaOH). At a temperature of 250 °C the PVC is decomposed to hydrocarbon and hydrogen chloride (HCl). The hydrogen chloride reacts with the lye to generate salt (NaCl) and water (H₂O). The excess lye is neutralised by hydrochloric acid and the compound is led to a precipitation tank where solid fractions are separated from the salt solution. The salt solution is evaporated to dry salt. The solid fraction which is now free from chlorine enters a rotating heating unit which gradually decomposes the compound in oil, gas and a coke fraction. The gas is used for the heating and oil is sold. The coke residue is transferred to a nearby "Carbogrit" plant where it replaces metallurgic coke as raw material in production of sand blasting material.

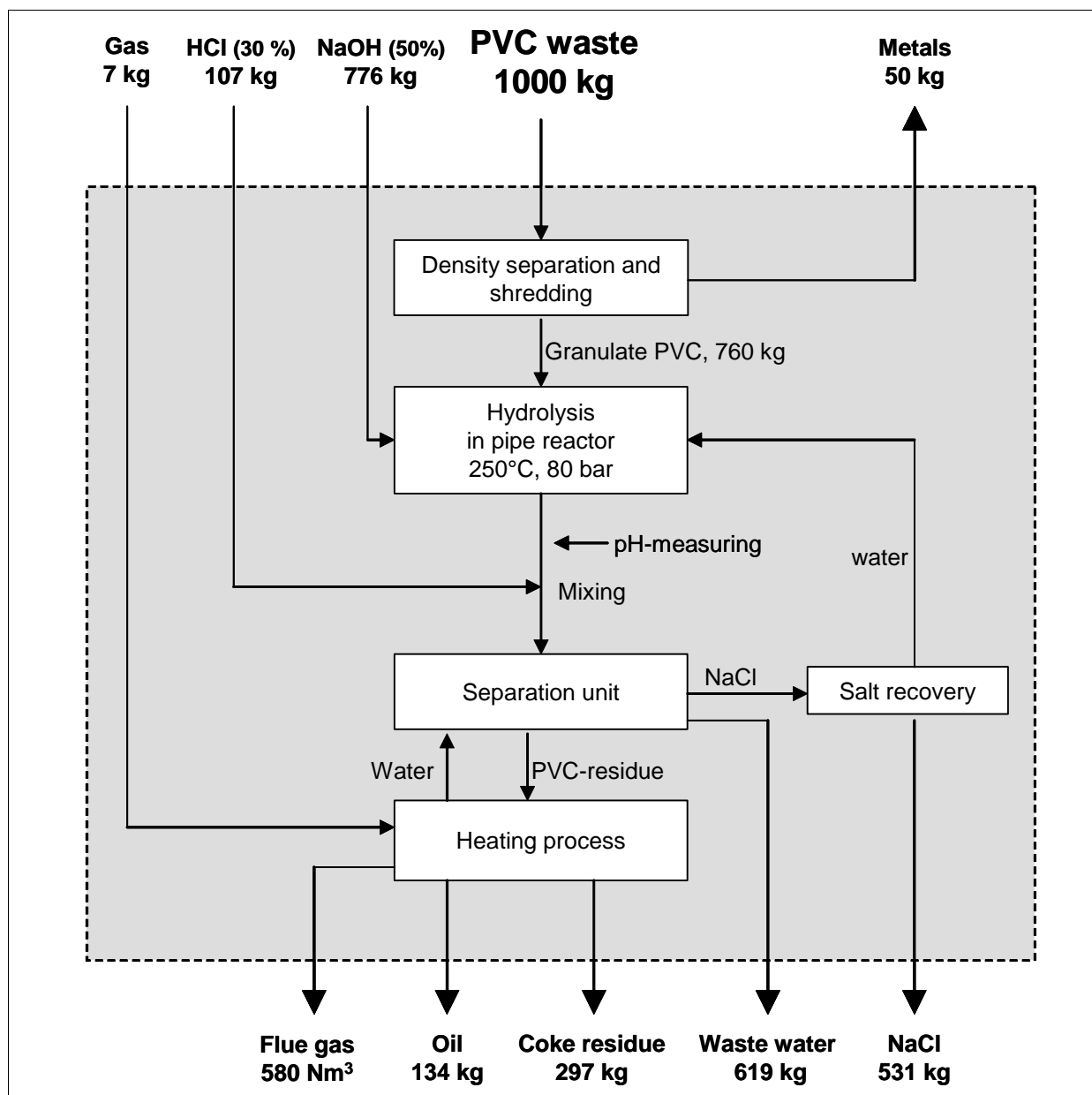
Environmental impacts

The gas combustion produces air emissions of CO₂, CO and NO_x. The coke contains the heavy metal from the PVC compound and also dioxin. The dioxin is subsequently destructed under the high temperature (1500°C) of the Carbogrit process. Air emissions of heavy metals from the Carbogrit plant are prevented by a flue gas filter. The flue gas filter cake is landfilled as hazardous waste with the same long-termed environmental risks as for the heavy metal residue from the Watech-process.

Costs

The costs per tonne for treatment of PVC waste by the Stignæs-process have been calculated along the lines of the Watech process using information from the process owner. The unit costs depends on the volumes processed, especially because a plant with substantial capacity is established from the start. The value of the existing pipe reactor for the hydrolysis is not included in the baseline assumption but is included in the sensitivity tests. Above 10,000 tonne per year the treatment costs will be in the range of 700 - 1300 DKK (85 - 160) per tonne *mixed* PVC waste.

Figure 7. Process diagram for the Stignæs process



4. Economic Analysis

4.1 Methodology

The economic analysis follows guidelines published by Danish Ministry of Energy and the Environment in 2000: *Economic Appraisal of environmental projects* [in Danish]. This means that the method of social cost benefit analysis is applied by calculating the net present value of each of the four scenarios as compared to the reference scenario. Evaluations of benefits and costs are made in market prices in year 2002 and the values of input factors are converted to market prices using a *net tax factor* which adds 25% to internationally traded goods and 17% to other goods. Future costs and benefits are discounted using a real discount rate of 3% per year. Capital costs are converted to annual costs over the assets service life assuming an alternative real rate of return of 6% per year. The chemical

treatment plants are depreciated over 20 years (buildings over 40 years) which is also the project horizon for the economic analysis.

Using the framework of social cost benefit analysis implies that all costs and benefits should to be monetarised and included in the net present value, irrespectively of whether a market price for the effects can be observed or not. Environmental costs and other non-marketed, such as waste producer's own efforts to sort out waste for chemical treatment, are monetarised to the extent possible by implicit valuation methods based on Miljøstyrelsen (2002c), ECON (2002) as well as own calculations.

In order to make the incidence of the alternative scenarios as transparent as possible the following main cost categories are used:

- Waste producer's costs
 - Sorting and collecting costs
 - Transport costs
 - Treatment costs
 - Tax payments
- Government budget (incl. changes in tax revenues)
- External costs

It is implicitly assumed that the chemical treatment plants are run with a profit including a normal risk premium reflecting the true financial risks of the investment in the plants. This implies that the costs for the waste producers include the increased costs related to the chemical treatment of the PVC waste. However, to the extent that waste tax exemption is granted for waste treated by Watech or Stignæs process some of the costs are financed the tax payers in general via the Government budget. Hence, changes in the waste producer's tax payments do not affect the overall result, but are merely transfers between waste producer's and the Government budget. In the results below the allocation of costs on waste producers and Government budget assumes that the waste tax will not apply to PVC treated at Watech or Stignæs.

Sorting and collecting costs are increased because of the waste users extra effort to sort out PVC as a separate fraction as well as organisation of an extra collection system (logistics, containers etc.). Transport costs are also increased by chemical treatment because all PVC waste has to be transported to one location in Denmark instead of local landfills or incineration plants.

In principle the analysis is limited to national approach so that only costs and benefits for the Danish population are included. However, costs of transboundary pollution are included to the extent that they are included in the Danish unit costs for air pollution.

In reality, a pragmatic approach has to be taken with regard to including monetary assessment of environmental impacts, because monetary values are not available for a very wide range of environmental effects. But as a minimum these effects should be assessed and quantified as far as possible in order to provide, as possible, a comprehensive picture of the full consequences of the scenarios to decision makers.

Also effects which *can* be monetarised are subject to substantial uncertainties, not only because of inaccurate monetarisation. Insufficient knowledge about technological factors and physical impacts are just as important causes of uncertainty. Because of the significant uncertainties it is crucial not to rely on only one calculation using baseline assumptions, i.e. best estimates of the uncertain

parameters. An essential part of a proper economic analysis is to perform a wide range of sensitivity tests of all uncertain parameters in order to identify the most critical factors for which the uncertainties have critical influence on the overall conclusions.

4.2 Results

Financial costs of the treatment of all PVC waste over the twenty year period are presented in Table 1 below. The costs for waste producers are split into cost components according to the description above. First column in the table shows the calculated net present value for the Reference Scenario, i.e. business-as-usual where all PVC waste is either incinerated or landfilled. For the four alternative scenarios the table presents the cost *differences* as compared to the Reference Scenario.

Table 1. Total financial costs of the four scenarios as compared to the Reference Scenario. 2000-2020

Net present value		Reference Scenario	1A: Watech Moderate	1B: Watech Maximum	2A: Stignsnæs Moderate	2B Stignsnæs Maximum
		mill. DKK	Change relative to Reference Scenario		Change relative to Reference Scenario	
Public sector costs	mill. DKK	-255	85	109	85	109
Treatment costs	mill. DKK	693	131	108	58	11
Collecting costs	mill. DKK	0	38	81	38	81
Transport costs	mill. DKK	48	30	40	30	40
Tax payments	mill. DKK	250	-88	-113	-88	-113
Waste producers costs	mill. DKK	991	110	115	38	18
Total financial costs	mill. DKK	736	196	224	123	127

In the Reference Scenario the waste producers' payments for treatment of their PVC waste have a net present value of 991 mill. DKK (130 mill.) for the 20 year period. These costs include transport costs as well as waste tax payments, which are reflected in corresponding revenues on the Government budget. It appears from the table that the Reference Scenario is the least expensive of the scenarios under the baseline assumptions. Both chemical treatment processes come out more expensive for the waste producers.

For the waste producers, the Stignsnæs process is least costly among the two chemical treatment technologies with extra costs of about 2-4% relative to the Reference Scenario whereas the cost increase for Watech is about 11%. The increase in total financial costs (which also includes the Government budget's loss of waste revenue) amount to 15-30% of the costs in the Reference Scenario. The differences between the "moderate" and "Maximum" scenarios are very small which reflects that the main economies of scale are already achieved in the Moderate Scenarios. A main precondition for this is that significant PVC waste volumes from abroad are assumed to be treated by the chemical treatment plants in all scenarios in addition to Danish PVC waste.

Between half and two third of the cost increase will be financed by the Government budget through reduced waste tax revenue.

Table 2 presents the social costs per tonne mixed PVC waste for each of the four treatment processes. These unit costs are calculated as the discounted cost flows for each cost component divided by the discounted volumes of PVC waste processes by each technology.

In addition to the financial costs the social costs include external costs and a welfare loss from distortionary effects of tax collection:

- *External costs* include air pollution costs and costs from risks of uncontrolled leachate emissions from landfills. All avoidance costs in terms of technologies to reduce environmental impacts are considered as treatment costs. Apart from these impacts a wide range of effects, primary long-termed environmental risks, have not been monetarised due to lack of methods and data. The important effects are emphasised in the conclusions
- *Welfare loss from taxation* is ignored in the baseline calculations. But as sensitivity test a welfare loss of 20% of the net reduction of the Government budget balance is included in accordance with guidelines in Finansministeriet(1999).

Table 2. Social costs per tonne mixed PVC waste treated by Incineration, Landfilling, Watech or Stignæs
DKK per tonne mixed PVC waste

	Incineration	Landfilling	Watech		Stignæs	
	All	All	1A: Moderate	1B: Maximum	2A: Moderate	2B: Maximum
Public sector costs	-421	-478	-27	-27	-27	-27
Treatment costs	1,526	568	1,262	1,232	877	840
Collecting costs	0	0	200	325	200	325
Transport costs	80	86	245	245	245	245
Tax payments	413	469	0	0	0	0
Waste producers costs	2,019	1,122	1,707	1,802	1,323	1,411
Total financial costs	1,598	644	1,680	1,775	1,296	1,384
External costs	345	23	160	136	81	72
Welfare loss from tax distortion	0	0	0	0	0	0
Total social costs	1,943	667	1,840	1,911	1,377	1,455

It appears from the table above that *direct treatment costs* amount to clearly the largest cost component for all four technologies. They are highest for **incineration** with 1526 DKK/tonne (205 /tonne) mixed PVC waste. If the external costs, primarily from emissions of CO₂, and heavy metals are added the total social costs per tonne are about 1943 DKK/tonne (260 /tonne). It should be noted that this figure includes benefits in terms of the produced heat and power and saved air pollution from alternative energy production.

Landfilling of PVC has the lowest direct treatment costs. Transport costs are almost the same as for incineration. The included external costs are relatively low and relate to CO₂-emissions from degradation of the plasticizers and to the roughly estimated environmental risks of leaching of phthalates. In total, social costs of landfilling PVC waste of 667 DKK (90) per tonne are the smallest among the four analysed treatment processes.

For **chemical treatment** processes costs are dependent of the amount of PVC waste processed. Higher capacity utilisation will reduce costs. Hence, treatment costs per tonne are lower in the Maximum than in the Moderate Scenario. However, in the baseline assumption the plants will also treat PVC waste from other countries of the same order of magnitude as Danish PVC waste volumes. As indicated in the table above this would imply that reduction of the treatment costs per tonne in the Maximum Scenarios (1B and 2B) are only 2% and 3% respectively.

The **Watech**-process have treatment costs of about 1250 DKK (170) per tonne mixed PVC waste which is less than for incineration but clearly more than for landfilling. Collection costs (separate collection system) and transport costs (longer distances are higher for the chemical treatments whereas external costs are in between incineration and landfilling. For Watech total social costs per tonne amount to 1900 DKK/tonne (250) which are more or less the same as for incineration.

However, a proper comparison with incineration should take into account that the marginal sorting and collection costs for waste transferred to chemical treatment from incineration are assumed to be substantially higher than from landfilling. The collection costs of 325 DKK/tonne in the Maximum Scenarios in the table is an average of 700 DKK/tonne (95 %) for PVC transferred from incineration and 200 DKK/tonne (25) for waste transferred from landfilling.

The **Stignæs**-process is very similar to Watech in economic terms. Treatment costs of 850 DKK/tonne (115 %) are some 30% lower than for Watech. External costs are also lower, primarily due to lower energy consumption per tonne treated, whereas sorting, collection and transport costs are assumed to be the same. Thus the social costs for Stignæs amount to about 1400 DKK (190 %) per tonne mixed PVC waste is lower than both incineration and Watech but clearly higher than landfilling. As for Watech comparison with incineration alone should take into account that the collection costs are assumed to be about 400 DKK per tonne higher than the average of 325 DKK per tonne in Table 2.

The average social costs per tonne for each of the four treatment processes are of course also determining the average social costs per tonne for the four alternative scenarios. Table 3 shows the results for the four alternative scenarios as additional costs per tonne pure PVC waste (i.e. excluding the plastic and metal fractions) treated by the chemical process.

Table 3. Social costs per tonne pure PVC waste for chemical treatment in the four scenarios
DKK per tonne pure PVC waste

	1A: Watech Moderate	1B: Watech Maximum	2A: Stignæs Moderate	2B Stignæs Maximum
	Change relative to the Reference Scenario		Change relative to the Reference Scenario	
Public sector costs	602	583	602	583
Treatment costs	925	579	413	57
Collecting costs	267	433	267	433
Transport costs	213	214	213	214
Tax payments	-625	-607	-625	-607
Waste producers costs	780	620	267	98
Total financial costs	1,382	1,204	869	682
External costs	187	52	83	-33
Welfare loss from tax distortion	0	0	0	0
Total social costs	1,569	1,256	952	648

In the Moderate Scenarios PVC waste is shifted to chemical treatment from landfilling only. Hence, costs of incineration do not influence the costs of the two Moderate Scenarios. In the Maximum Scenarios volumes for chemical treatment are increased with about 30% by transferring also a fraction from incineration. Hence, costs per tonne are also in the Maximum Scenarios dominated by the cost differences relative to landfilling.

Therefore, the extra costs per tonne are positive in all four alternative scenarios even though the Stignæs process appeared to be less costly than incineration (cf. Table 3). The additional costs as compared to the Reference Scenario varies between 650 and 1600 DKK (90 - 210 %) per tonne PVC waste transferred to chemical treatment and the costs are about 600 DKK (80 %) higher per tonne for Watech than for Stignæs. Similarly, net present value for the total additional costs over the 20 year period is 130 - 230 mill. DKK (18 - 30 mill %) higher than for continued incineration and landfilling.

Included environmental costs are also higher (although not very significant) for the chemical treatment scenarios, except for 2B: Maximum Stignæs. However, this result should be assessed

against the environmental effects which are not monetarised and therefore not included in the cost benefit analysis. These include i.a. the long-termed environmental risks of landfilling heavy metal containing PVC and flue gas residues from incineration of PVC.

The results presented above are not considering the uncertainties of input data for the calculations. In order to take into account these uncertainties in the overall conclusions a comprehensive sensitivity testing has been conducted for 40 important parameters based on assessment of the uncertainty range for each individual parameter. These calculations have been used to identify the most critical parameters and on the other hand parameters which are uncertain but where the uncertainties have very little influence on the results. The outcomes of the sensitivity tests have been taken into account in the formulation of the final conclusions from the analyses.

5. Conclusions

Treatment costs per tonne PVC waste

If we first look at the costs of the treatment process it can be concluded that landfilling is clearly the cheapest alternative. Incineration is about three times as expensive in spite of the fact that the energy content of the PVC is utilised for combined power and heat production. Apart from the high costs of incineration of waste in general this is also caused by the need for extra flue gas cleaning and disposal of the related residue in order to avoid emissions of the hydrogen chloride and heavy metals.

The Watech-process has treatment costs which are just below incineration, while treatment costs for the Stignsnes-process are about two third of incineration costs. Even if a cost reduction of about 50 mill. DKK from an existing pipe reactor at Stignsnes is included the treatment cost per tonne pure PVC waste is still two times the landfilling costs.

However, these conclusions is very dependent on the assumption that the chemical treatment plants will also receive imported PVC-waste which gives rise to utilisation of economics of scale following from higher PVC waste volumes.

In addition, several uncertainties about the costs of incineration and chemical treatment imply that the cost comparisons between these processes are not decisive. Important factor are

- the allocation of the capital costs of incineration on PVC waste and other incinerated waste
- the large variations in cost efficiency across incineration plants
- the value of the heat and power as well as the outputs from the chemical treatment

The overall assessment is, in spite of the uncertainties, that the chemical treatment technologies have higher treatment costs for the waste producers than the weighted costs of the existing treatment technologies, incineration and landfilling.

The extra costs for the waste producers over the 20 year period in the chemical treatment scenarios amount to 130 - 230 mill. DKK (18 - 30 mill.), which corresponds to 5-15% extra per tonne PVC waste produced.

Environmental costs

Landfilling also has the lowest environmental costs when considering the *monetarised* environmental impacts. These costs are clearly highest for incineration which is primarily due to CO₂-emissions and air emissions of heavy metals. Air emissions of heavy metals are considered to be

ignorable for both chemical treatments. Environmental costs for Watech are higher than for Stigsnaes which is primarily because of higher CO₂-emission per tonne PVC.

In total, monetarised environmental costs have in general relatively little importance as compared to direct treatment costs. On the other hand it should be noted that the treatment costs are to a wide extent related to prevention of environmental impacts.

To the above assessment of the environmental costs should be added that a number of environmental effects have not been converted into monetary units and are therefore not included in the calculated social costs of the alternative treatment technologies:

- For *landfilling* this is primarily environmental risks related to long-termed soil and water pollution from leaching of phthalates and heavy metals from potential uncontrolled leachate from the landfills due to possible failures of the membranes.
- For *incineration* it is similarly long-termed environmental risks from potential leachate of heavy metals from landfilled flue gas cleaning residues or slag residue which are landfilled or recycled as input in road construction.
- For *chemical treatments* it is primarily environmental impacts which are not monetarised: (i) Emissions of heavy metals from disposal of residues (including the filter cake from the Carbogrit plant). (ii) Reduced environmental impacts from displaced production of the products which are substituted by outputs from the chemical treatments (oil condensate, coke residue, NaCl and CaCl₂-solution).

However, the omission of these effects is assessed not to have decisive influence on the overall results, because the effects are very long-termed and with a very low probability of actually leading to extensive damages to human health or the environment in general.

Total social costs

The social costs of the disposal of PVC waste include also, apart from direct treatment costs and environmental costs, sorting, collecting and transport costs. All these costs are assessed to be higher for the chemical treatments than for both incineration and landfilling.

Consequently, total costs of chemical treatment are therefore also clearly higher than for landfilling. Comparison of chemical treatment and incineration is not decisive because it depends entirely on the assessment of the sorting and collection costs which are very uncertain.

The sorting and collection costs have been a critical issue in the analyses. On the one hand these costs are very difficult to quantify robustly, on the other hand they could be substantially higher for the chemical treatment plants than for incineration and also somewhat higher than for share the PVC waste which is landfilled.

The best estimate for the increase in collection costs is 300-400 DKK (40-50 %) per tonne depending on the degree of separate collection of PVC waste. The high estimate is about *four* times higher but this figure also includes a rough estimate of the extra costs for waste producers for sorting out their PVC waste in a special fraction. This great variation reflects that the basis for the assessment is weak: The costs of the effort of waste producers can not be observed and it is difficult to assess the collection costs for a hypothetical situation where separately collected volumes of PVC are significantly higher than today.

The transport costs for the chemical treatments are moderately higher, 200 DKK/tonne (25 /tonne), than for landfilling and incineration due to longer transport distances across the country to only one treatment plant.

To conclude ...

The four scenarios have compared transfer of realistic volumes of PVC waste from the current treatment technologies, landfilling and incineration, to one of the chemical treatments. The results indicate that the current treatment system in total is less costly than either of the scenarios with chemical treatment. Further, it should also be taken into account that if additional PVC waste is sorted out from incinerated volumes, this PVC waste could also be landfilled with lower costs than chemical treatment.

A decision about utilisation of one of the two chemical treatment processes should therefore in the last resort depend on a balancing of the documented extra costs as compared to the current treatment system against the political willingness to pay for avoiding the non-monetarised long-term environmental risks of soil and water pollution due to:

- possible leachate of phthalates and heavy metals from potential uncontrolled discharges due to failures of the environmental protection of landfills;
- possible leachate of heavy metals from landfilled flue gas cleaning residues and from the recycling of incineration slag in road construction etc.

Finally it should be emphasised that neither mechanical recycling nor substitution of PVC with other materials have been analysed as alternative technologies for reducing the environmental impacts from PVC.

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EFFICIENT TARGETING OF WASTE POLICIES IN THE PRODUCT CHAIN

By Richard C. Porter¹

1. Introduction

The efficient targeting of waste policies in the product chain means making sure that the actors at each phase of a product's life – from its birth to its death – face prices that reflect the marginal social costs of their actions. With waste, prices may not reflect marginal social cost. Price and marginal social cost can diverge for two principal reasons. One, waste handling often generates external cost, which means that part of the social cost is foisted onto unwilling or unknowing third parties. And two, waste handling is often subsidized, which means that part of the marginal private cost is paid for out of a government general fund.

The marginal private cost of waste disposal is readily observable. It consists of the extra costs of the equipment, the wages of the labor, and the opportunity cost of the land that are needed for the collection and disposal or recycling of one extra unit of trash. The marginal external costs are less visible. They consist of the noise, litter, dust, unsightliness, and potential air or groundwater pollution that are generated by one extra unit of trash collection and disposal or recycling. The marginal social cost is simply the sum of these two kinds of cost, the marginal private cost plus the marginal external cost.

When there is external cost, marginal social cost (MSC) exceeds marginal private cost (MPC).² When there is subsidy, MPC exceeds the price (P) that the waste generator pays. When MSC is greater than P, some actors on the waste stage will be undertaking activities whose MSC exceeds their willingness to pay (WTP) for them. For example, when someone gets a service for which he or she is willing to pay \$2, and the service costs society \$3 worth of resources to produce, then someone somewhere loses use of the \$3 worth of resources, and society is worse off by \$1 when the dollar gains and losses are summed (i.e., minus \$3 for the unlucky “someone somewhere” plus \$2 for the lucky “he or she”). In order to make MSC equal to MPC, efficient policies put taxes on activities that generate external costs.³ In order to make MPC equal to P, efficient policies avoid unintended subsidies.

In most of the United States, as in many other parts of the world, households pay nothing for their trash collection and disposal in the MPC sense. In the vast majority of cities and towns where some kind of collection is provided, its cost is covered either from general fund revenues or from a

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2. For the glossary, please see Annex 1.

3. Taxation is not the only way of handling external cost. For a fuller discussion of the policy instruments available, see Porter, 2002, Chapter 1, Appendix A. Many of the topics in this paper are treated in greater detail in that book..

time-based charge on residents by time-based charge, I mean that the charge for the service is so much per month or per year, an amount that is not related to the amount of trash being put out for pickup and disposal (Kemper and Quigley, 1976; Jenkins, 1993). It makes no difference with respect to household trash-generation incentives whether the general fund or a special time-based trash charge is used; the end result is the same – the marginal private cost of putting out an additional unit of waste is zero. And if a special time-based trash charge is used, it makes no difference whether the total revenue collected from households covers the total cost of household trash collection; the end result is still the same - the marginal private cost of putting out an additional unit of waste is zero.

Americans respond to this “free” service by generating large amounts of municipal solid waste (MSW). See Table 1. Per capita solid waste nearly doubled between 1960 and 1990, and it has not declined since then despite the great increases in recycling over the last two decades in the United States.⁴ This adds up to a lot of trash, over 200 million tons per year. America’s cities collect it and take it somewhere for further handling or disposal, almost all for free. Despite the largely free MSW disposal, households have begun to undertake recycling activities over the past 40 years. Starting from almost no recycling in 1960, recycling reached nearly half a kilogram per person per day by the mid-1990s. Recycling volumes, like MSW volumes, have leveled off in the past few years in the United States. Since many smaller cities are still in the process of initiating curbside recycling programs, this means that the recycling rate is falling in large cities that already recycle (Truini, 2002a).

Table 1. U.S. Municipal Solid Waste Generation and Recycling
(in kilograms per capita per day)

Year	MSW	Recycling (%)
1960	1.22	0.08 (6%)
1970	1.47	0.10 (6%)
1980	1.66	0.16 (9%)
1990	2.04	0.29 (12%)
1994	2.05	0.40 (16%)
1995	2.02	0.44 (18%)
1996	1.96	0.44 (18%)
1997	2.04	0.44 (18%)
1998	2.05	0.44 (18%)
1999	2.11	0.46 (18%)
2000	2.05	0.45 (18%)
2001	2.00	0.45 (18%)

Note: Recycling data excludes all composting and pre-consumer materials recovery.

Source: Franklin, 2002, U.S. EPA, 2003.

Most small businesses in the United States also use the MSW collection system. Nearly half of the total MSW collection is estimated to come from businesses, and they too usually pay no MPC for the service. In some cities, the time-based trash charge on businesses is set higher than the time-based charge on households, which means that the *average* private cost of trash is higher for businesses than for households, but the *marginal* private cost is still zero for both.

Many businesses and most manufacturers need more specialized or more frequent trash collection than the municipal MSW system provides, and they must hire private haulers or provide their own hauling operation. When they pay the price at the landfill or incinerator – called a “tipping fee” • they usually do cover the MPC of the disposal, though they may not be paying a fee that also reflects the external cost that landfills and incinerators generate.

4. MSW data are now collected or updated annually for the U.S. Environmental Protection Agency (U.S. EPA) by Franklin Associates (Franklin, 2002). No official figures were available before 1960, and only census-year estimates were made for the period, 1960-1990.

Most manufacturers, however, *completely escape* responsibility for some of the waste they generate – the waste created by the packaging of their products and the waste created by the products themselves when their useful life has ended. The producer passes on the problem of disposing of the packaging and the product to the consumer, and the consumer then passes these costs on to the municipality. Consumers end up paying the cost of producing the packaging and the product but not the cost of disposing of them. The social problem is to make producers and consumers aware of the costs they impose on society when they utilize unnecessarily bulky or unrecyclable packages and products. In the usually price-less world, neither producers nor consumers have any financial incentive to utilize less packaging or to recycle more of the packaging they use.

2. The Cost of Waste Market Price Failures

When we fail to price solid waste handling at the margin, we therefore create two kinds of market failure: 1) when $P < MPC$, we introduce a largely unintended subsidy to waste generation and thereby induce households to generate too much waste; and 2) when $MPC < MSC$ (*i.e.*, there is external cost), we deflect some of the waste handling costs onto third parties and thereby further induce households to generate too much waste.

A simple diagram lets us add precision to this general point. Figure 1 shows three lines, the demand for municipal solid waste collection and the MPC and the MSC of handling it. (Here, for simplicity, the downward-sloped demand curve is assumed to be a straight line, and each cost is assumed constant.) Currently, the price of such collection (P_0) is zero, and households react to that price by creating a large volume of waste (W_0). For much of this waste (*i.e.*, for $W_0 - W_2$), household WTP is less than the MSC of the disposal.⁵ It is socially inefficient to produce anything when the consumer's WTP does not cover the MSC of producing it. If the waste collection price were raised to equal MPC, less waste would be produced ($W_1 < W_0$). And if the price were raised still further to equal 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 what economists call a deadweight loss (DWL). The DWL is 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 shaded 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.

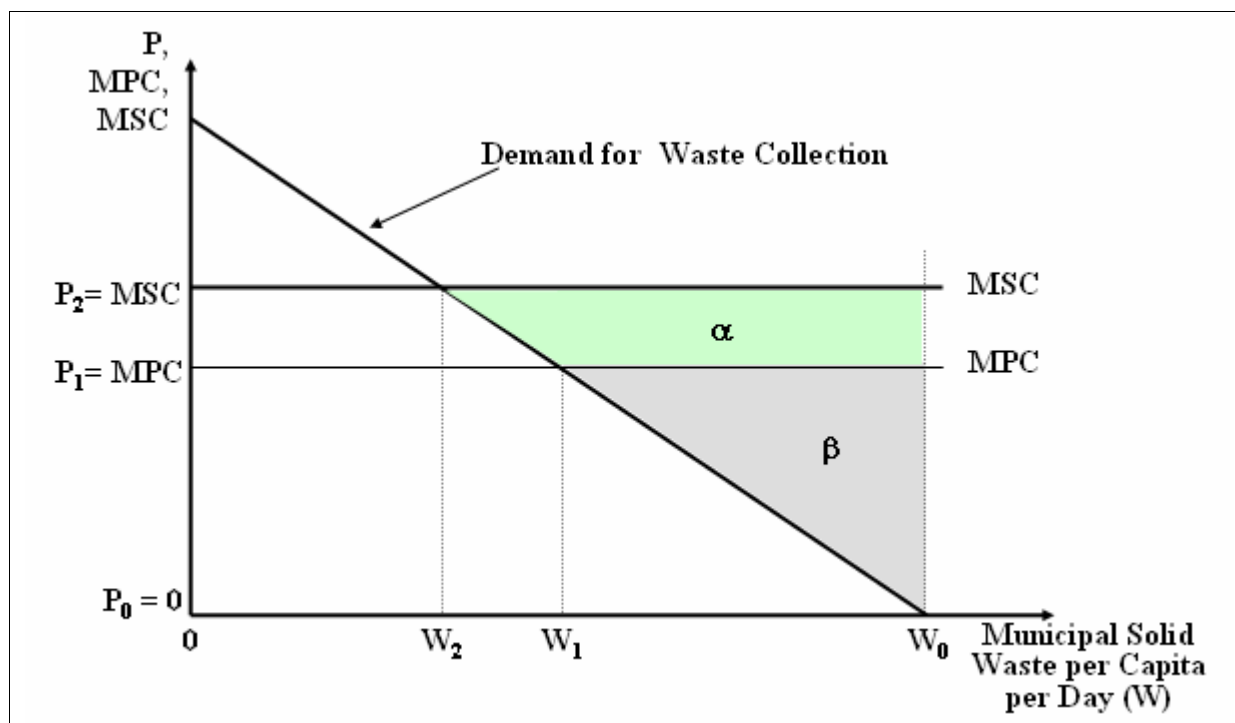
It is possible to put some empirical content into these concepts. In the United States today, each person generates about two kilograms of solid waste per day (*i.e.*, $W_0 = 2$ kg/cap/day). The MSC of collecting and disposing of waste – on average in the United States • is something like \$100 per ton (*i.e.*, $P_2 = \$0.10/\text{kg}$; Repetto *et al.*, 1992; Stevens, 1994). There are many estimates of the price elasticity of demand for MSW collection, and they cluster closely around -0.2 (Stevens, 1977; Skumatz, 1990; Jenkins, 1993; Reschovsky and Stone, 1994; Miranda *et al.*, 1994; Goddard, 1994; and Strathman *et al.*, 1995); although it is impossible to utilize this elasticity information in the neighborhood of a zero price, it is not unreasonable to guess that W_2 would be something like 1.5 kilograms (*i.e.*, $W_0 - W_2 = 0.5$ kg/cap/day).⁶ These numbers let us estimate the DWL of zero-pricing waste in the United States. The

5. Recall that the demand curve is also a schedule of WTP of households for waste collection (if we ignore the probably small income effects).

6. Where would these reduced 0.5 kilograms of waste go each day? We are going to be thinking a lot about that, but just to anticipate for now, it could go into various things – into somewhat reduced purchase of waste-producing products and packages, into somewhat longer lives of waste-producing goods, into somewhat reduced packaging volumes, into extensive recycling and composting of what was previously waste, and (alas) into illegal disposal – whether a little or a lot, we shall worry about soon. This illegal disposal comes in many forms • abandonment in public places, backyard burning,

DWL (= area α + area β in Figure 1) is \$0.025 per person per day, or \$2.4 billion per year for the United States. The fact that this DWL exists with zero-pricing tells us that it is possible to make everyone better off with optimal pricing.⁷

Figure 1. Deadweight Loss of Underpricing Municipal Solid Waste



In short, Americans put out too much trash for MSW pickup and disposal, simply because it doesn't cost anything at the margin. A welfare gain (*i.e.*, a reduction in the DWL) in the billions of dollars per year is waiting to be achieved. And only pricing waste can achieve it. The combination of education, exhortation, compulsion, and availability of curbside pickup of recyclable materials has worked in the past to increase household recycling, but it seems to be unable to get the recycling rate past 25% (on average for the entire United States).

Command-and-control approaches, long favored by bureaucrats, really cannot get at this problem. Try to think of some. For example, government could place a limit on the amount of waste that each family could put out for collection. Do we want to deny high-income families the right to exercise their income-elastic demand for waste even if they are willing to pay the full MSC for it? Or more realistically, do we want to drive well-off families to using a separate private-sector pickup of waste? That would be duplicative, and hence socially wasteful. Do we really want to enlarge the bureaucracy so that it could issue extra waste permits to large (or otherwise "waste-needy") families?

and surreptitious discard in commercial dumpsters (or skips). Together, these activities are called "midnight dumping" in North America or (more colourfully) "fly-tipping" in Britain. In this paper, I will often just use the term "litter" as a shorthand for all kinds of illegally disposed waste.

7. Later on, we shall find the proper pricing of waste disposal is much more complicated than it seems here, but this simple exercise makes the general point that *any* trash pricing system (where the price is no greater than the MSC of trash collection and recycling or disposal) can reduce the deadweight loss and save total trash costs. (This statement is subject to caveats about illegal disposal and administrative cost, which we will later treat more carefully.)

Here, a price-incentive solution is not only the more efficient way to reduce household waste, it may be the only sensible way to reduce household waste.

The next step is to search more carefully for the waste pricing scheme that comes closest to achieving an optimal allocation of resources in the waste-handling problem. This is a complicated search, and in an effort to make it more easily understood, I am going to go slowly, to “remove protective coating a little at a time.” We start (in the next section) by looking at a world where nothing is recycled and where nobody disposes of waste illegally. Consumers must decide only whether to reduce, or to reuse, or to dispose. In the two subsequent sections, we will first add the possibility of recycling to these choices, and then add the possibility of illegal disposal of waste.

3. Waste Pricing With No Recycling And No Littering

Once a package has served its purpose or a product has exhausted its useful life, something must be done with it. And that collection and disposal uses up resources, resources not usually paid for either by the manufacturer who made the package or product or by the household that puts it into the trash. The disposal costs are borne by others, either in the form of external costs, as with packaging that becomes litter, or in the form of implicit subsidies, as with municipal waste collection programs that are financed out of general fund taxation. To make manufacturers and consumers aware of these costs, it is necessary to estimate them and introduce taxes that reflect the marginal social costs that products and packaging impose at the end of their use.

In principle, such a tax could be levied in either of two ways. One, it could be levied on the manufacturer of the package or product as an advance disposal fee (ADF) at the time of sale. Or two, it could be levied on the consumer of the package or product as a marginal trash charge (MTC) at the time of disposal. Either levied on the manufacturer or levied on the consumer – but not on both. At this point in our theory, it doesn’t matter which party is taxed. To see this, consider the following example.⁸

A pencil. It could be sold without any package at all, or it could be marketed in a fancy plastic, cardboard, and paper package that will cost \$1 to collect and dispose of. We could tax the pencil manufacturer \$1 if he uses the fancy package. This ADF of \$1 would almost surely get passed on to the consumer in the form of a higher price for such packaged pencils, and a pencil in a fancy package would end up costing the consumer \$1 more than an unpackaged pencil. Or we could charge the consumer an MTC of \$1 when he puts the pencil’s package into the solid waste collection process. Either way, the rational, informed (or simply observant) consumer would realize that a packaged pencil ends up costing \$1 more than an identical unpackaged pencil, and the consumer would choose to buy a packaged pencil only if the extra convenience of the package is worth the extra \$1 that it costs. Consumers who are willing to pay the MSC of pencil-package disposal would buy packaged pencils; and consumers who are not willing to pay the MSC of pencil-package disposal would buy unpackaged pencils. Manufacturers would also respond to this consumer choice. If most consumers wanted packaged (though more costly) pencils, manufacturers would respond by producing mostly packaged pencils; or if most consumers wanted unpackaged pencils, manufacturers would respond by producing mostly unpackaged pencils. The MSC of collection and disposal would have been “internalized” into the decisions of manufacturers and consumers.⁹

8. For a full proof of the following proposition, see Fullerton and Wu, 1998.

9. This simple example offers only the binary choice of packaging or no packaging. In reality, manufacturers have many ways of reducing the ADF that they would have to pay. They can move, for a few examples, toward single-material packaging, fewer blister packs (i.e., packages consisting of a clear plastic overlay affixed to a cardboard backing for protecting and displaying a product), less

As long as we are dealing with such a simple production-consumption process – i.e., purchase the good, use it, put it in the trash, collect it, and landfill (or incinerate) it • it makes no difference whether the product carries an ADF, levied on its manufacturer, or an MTC, levied on its consumer. The outcome is very similar to the tax-incidence theory worked out in every introductory economics course, where it is shown that it doesn't matter whether you levy an excise tax on the supplier or on the demander • the end-result on the price of the product is the same.

Drawing from the identical incidence of the ADF and the MTC, some people suggest that a landfill tax would also have the same incidence as an MTC, and the landfill tax would be much cheaper to collect. Cheaper to collect, yes; same incidence, no. As long as the municipality picks up and delivers the trash to the landfill at no marginal cost to the household, the fact that the municipality's general fund is further depleted by a landfill tax will have no effect on household trash decisions. Landfill taxes where there are no household MTCs will affect businesses, which usually do pay a marginal cost for trash, but landfill taxes will not affect households. Indeed, this is exactly the result found in the United Kingdom with its introduction of a landfill tax (Davies, 2004).

At the risk of belaboring the obvious, this is a good time to introduce a diagram that we shall find useful in the more complex world when we consider recycling and litter. There are two equally appropriate means to handle the internalizing of the MSC of waste collection and disposal – the two means are shown in Figures 2 and 3.

This is the basic result so far – it doesn't matter at which level the waste tax is levied.¹⁰ The “Producer Pays Principle” contains no particular virtue even though it is much heralded by environmentalists. It takes two to make trash, a manufacturer to design, produce, and sell it and a consumer to buy, consume, and pitch it. It is as impossible to assign blame to one or the other as it is to decide which blade of the scissors cuts the paper. Anyway, guilt is irrelevant. The consumer usually ends up paying for the trash disposal no matter where the charge is levied.

Having made the point that the two approaches are identical in theory, we should now begin to look at the many ways in which they differ in fact, especially as these differences will become more important later, when we consider greater complexity in the waste disposal process:

1. Collection and disposal costs differ in different parts of the United States, as labor wages, population density, and land prices vary. In many industries, there are a few large factories serving regional markets, or even the entire national market. It would be extremely difficult to vary an ADF on such producers according to where the product (or its packaging) will end up being thrown away. An MTC on households, however, can readily consider these differences.
2. Collection and disposal costs also differ for different products. An ADF can vary according to the cost of collecting and disposing of the product, but an MTC – unless it is prohibitively costly to operate – must be uniform across products, so much per barrel or per bag or per kilogram. A uniform MTC will be too low on expensive-to-dispose-of materials and too high on cheap-to-dispose-of materials.

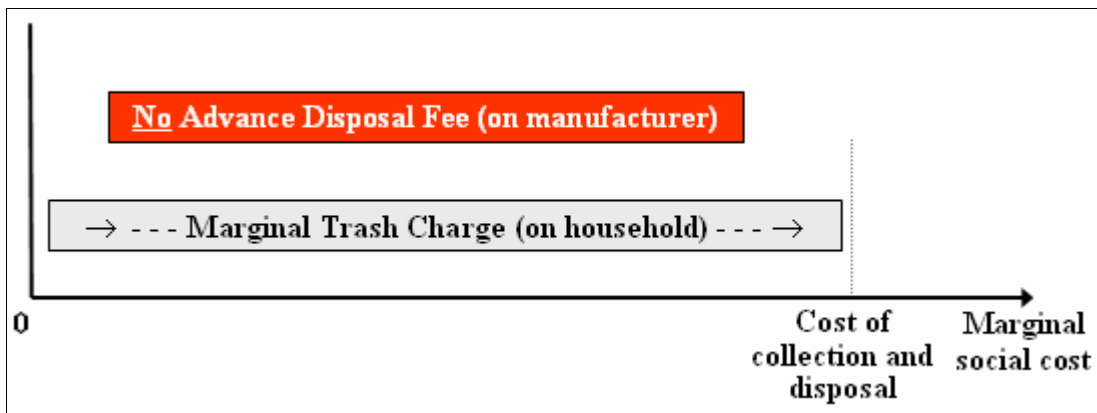
secondary or redundant packaging, lighter packaging, and the use of paper for dunnage (i.e., loose packing of bulky material put around a product for protection).

10. The charge could also be levied half as an ADF on the manufacturer and half as an MTC on the consumer – or indeed in any other proportions. To anticipate, we shall find reasons later on for levying some ADF on manufacturers and some MTC on households.

Figure 2. Means 1 – Charge Producer an ADF When Product or Package Is Sold



Figure 3. Means 2 – Charge Household an MTC When Product or Package Becomes Trash



3. ADFs would not have major impacts on the prices of most products. Estimates of marginal social collection-and-disposal costs typically run 1-2% of product prices (Pearce and Turner, 1992; Little, 1992; Ackerman, 1997). But ADFs would generate a lot of government revenue - a 2% tax would in effect be a significant increase in most U.S. state sales taxes - and this, curiously, presents a major problem.¹¹ Some politicians desperate for greater revenue might embrace high ADFs for the wrong reason, just to raise revenue. And other politicians strongly opposed to increased taxation might reject ADFs for the wrong reason, despite their desirable social purpose. Of course, ADFs could always be adopted along with a comparable cut in the traditional sales tax; then, the only effect would be to raise the prices of packaging-intensive products and lower the prices of little-packaged products. MTCs do not pose any of these problems. The government revenues from the introduction of an MTC system simply replace the property tax revenues previously needed to provide the trash service.
4. MTCs are difficult to levy on households that live in multi-family dwellings. Such residences usually have one large bin in the back to serve for all, and the residents treat it as a “commons” (Hardin, 1968). It is difficult, if not impossible, for landlords to charge individually for use of the bin, with the result that the imposition of an MTC does little or

11. The median sales tax of the 50 U.S. states is 5%. U.S. sales taxes are levied on consumer purchases of goods, but not services, and often exempt food and pharmaceuticals.

nothing to reduce the volume of trash, adds a new tax on multi-family dwellings, and forces the landlord either to raise rents or to accept a loss of income. In cities with much multi-family housing, MTCs may achieve little trash reduction and will face concerted (and justified) opposition from landlords.

5. An ADF raises the product price to the consumer by the same amount no matter how long the consumer keeps on using the product. So an ADF offers the consumer no inducement to keep and reuse the product. An MTC, on the other hand, does offer such an inducement because household reuse postpones the MTC. An MTC may not induce much reuse, but some reuse should be encouraged - a postponed call on collection and landfill (or incinerator) resources is a reduced use of resources in a present-value sense.
6. In almost all markets, there are fewer producers than there are consumers, and as a result, the administrative and monitoring costs of an ADF system may be significantly lower than those of an MTC system (Fullerton and Kinnaman, 1996).
7. An MTC invites tax evasion - i.e., illegal activities to reduce MTC payments on household trash • but an ADF cannot be evaded by the household.¹² We will think more about this shortly.

In short, the choice between an ADF and an MTC is not a simple one, even when the only consumer decisions are between reducing, reusing, and discarding. It gets even more complicated once the possibility of recycling is added.

4. Waste Pricing With Recycling But No Littering

Once recycling becomes a possibility, there are *three* policy tools available for use in the waste system: 1) an advance disposal fee, levied on the manufacturer at the time of sale; 2) a marginal trash collection charge (or refund), levied on the household at the time of disposal of the package or the remnants of the product itself; and 3) a recycling collection charge (or refund), levied on the household when materials are set apart for recycling.

Ideally, these three tools must be chosen in such a way that households buy the right amount of stuff, use (and reuse) that stuff for the right amount of time, and then make the right decision between disposing of the remnants as trash or for recycling. In order for the consumer to buy the right amount of stuff, the price ultimately must include not only the cost of producing the product and its package, but also the marginal social cost of disposing of the product and package. In order for the consumer to use the product for the right amount of time, the benefit to the consumer of reusing the product for an additional time period must equal the interest charge on the postponed disposal cost. And finally, in order for the consumer to choose correctly between trash and recycling, the difference in the price of the two disposal methods must equal the difference in the marginal social costs of the two methods.

All this sounds like a hard problem, but it can be solved. Indeed, no ADF is needed, simply an MTC that reflects the MSC of trash collection and disposal, and a recycling collection charge that reflects the cost of collecting and sorting recyclables (net of the revenue earned on those recyclables).¹³

12. Recall the difference between “evading” and “avoiding” taxes. Evasion is an illegal activity. Consumers can always avoid MTCs by buying fewer goods and more services or by buying less packaging-intensive products. Indeed, an MTC is meant to encourage avoidance!

13. Hereafter, when we want to refer to the cost of collecting and sorting recyclable material, net of the revenue earned on the sale of that recycled material, we shall simply call it the “net recycling cost.” The net recycling cost could, of course, be negative if the recycled material was sufficiently valuable. In that

Call this Scheme 1 (Fullerton and Kinnaman, 1995). Figure 4 shows these necessary fees. The figure is drawn on the assumption that the net recycling cost is lower than the trash cost (*i.e.*, the cost of collecting and disposing of trash).¹⁴

Moreover, there are two other pricing schemes that come very close to achieving the optimal household decisions. Schemes 2 and 3 also require the use of only two of the three available policy tools. These schemes are shown in Figures 5 and 6. These two schemes only “come very close” to optimality because neither offers the correct incentive for the household to reuse – in each of the two schemes, what the household saves by reusing a product is less than the interest on the MSC of its disposal, whether as trash or as recycling – indeed, with one of the two schemes, the household actually gains by rejecting reuse and putting the product out for recycling.

Up to the probably unimportant difference in the reuse incentive, the three pricing schemes are identical • in theory.¹⁵ Are there practical reasons for choosing one of the three over the other two?

Scheme 1. This is the correct theoretical scheme since it is the only one to offer the correct reuse incentive. Moreover, since it does not utilize an ADF, the MTC and the recycling collection charge can be tailored to local conditions. It does, however, require two different fees to be levied at the household level, which is an administrative nuisance (and perhaps nightmare). Since there are many more households than firms, the absence of an ADF means that a more easily collected tax goes unutilized. The MTC is the highest of the three schemes, being equal to the full MSC of trash collection and disposal, which means that this scheme creates the biggest incentive to illegal disposal. Finally, this scheme requires a recycling collection charge – provided the net recycling cost is positive, which it will be for almost all products for the foreseeable future – and this charge may anger many households, who feel that if recycling is a “good” thing, they ought not to be charged for doing it. Furthermore, in theory, this scheme calls for a different recycling collection charge for each product that is recycled, since the fee is supposed to equal the marginal net recycling cost. Since this net recycling cost depends greatly on the price of the collected recyclables, it will differ across products. But it is practically very difficult and costly to charge households different recycling charges for different recyclable materials. And levying a single sort-of-average recycling charge overencourages households to recycle low-value recyclable materials and underencourages households to recycle high-value recyclable materials.

case the appropriate recycling charge either becomes a recycling collection subsidy or is set at zero in the hope that private profit-seeking recyclers will make a market for the material.

14. Indeed, if the net recycling cost is not lower than the trash cost, recycling will probably fail a benefit-cost test. The “probably” in that sentence reflects two things we are ignoring here that a benefit-cost test would need to include: 1) some households have a WTP for the pleasure of recycling, and this should be counted as a benefit of recycling; and 2) starting to recycle when it has a high net cost may be socially profitable in a present-value sense if it hastens the day when net recycling costs become low (through learning-by-doing or induced invention and innovation).
15. There are of course also an infinite number of optimal policies when all three tools are used. We will consider only the three schemes pictured above, because two fees (or refunds) are easier than three to administer (and easier to examine pedagogically).

Figure 4. Scheme 1 – Waste Pricing With Recycling But No Littering

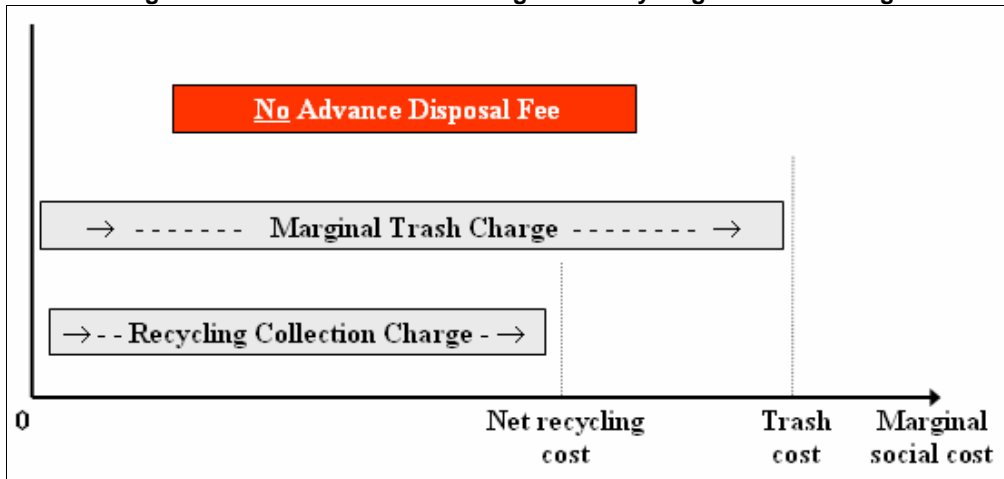


Figure 5. Scheme 2 – Waste Pricing With Recycling But No Littering

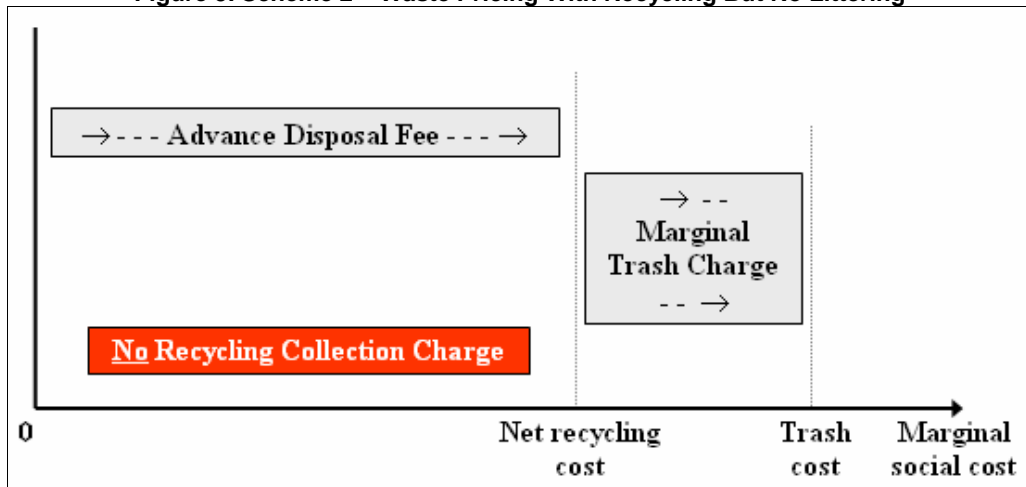
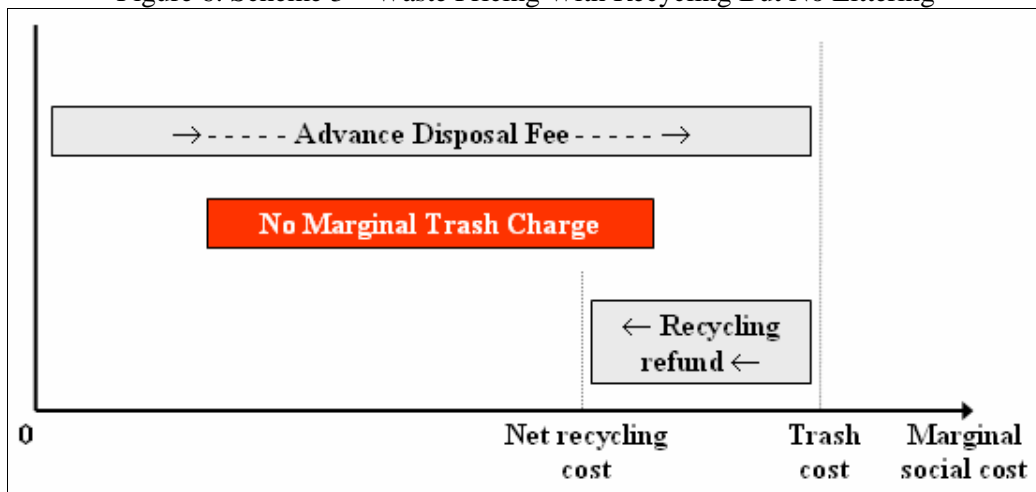


Figure 6. Scheme 3 – Waste Pricing With Recycling But No Littering



Scheme 2. This scheme utilizes the more easily administered ADF (equal to the net recycling cost of the material) as well as an MTC on the household. The MTC is smaller than in Scheme (1) and hence less of an inducement to illegal disposal, but it still provides some such bad incentive. The ADF must be uniform across regions, though not of course across products or packages, and so must be set at the average national net recycling cost of the material. This means that the ADF will be too high in urban areas where recycling collection is cheaper, and it will also be too high in places where prices of recyclable materials are high.¹⁶ There is neither fee nor refund for recyclable materials, which conforms with conventional values (and most current U.S. practice) and makes it easy to administer. But notice that the MTC in this scheme should, in theory, equal the difference between the marginal collection-and-landfill (or incineration) cost and the marginal net recycling cost. Since it would be administratively costly to vary the MTC across products, a single sort-of-average MTC would end up being chosen, underencouraging households to recycle high-value recyclables and overencouraging households to recycle low-value recyclables.

Scheme 3. The ADF in this scheme is large, equal to an average across the nation of the marginal collection-and-landfilling cost of the product. Thus, it is too high in some places and too low in others. This scheme (of these three schemes) most discourages reuse because it actually subsidizes recycling. Indeed, the recycling refund would be most difficult to administer since it should be different for different products, as well as at different places and different times. Again, use of a sort-of-average recycling refund would underencourage households to recycle high-value recyclables and overencourage households to recycle low-value recyclables. Moreover since the ADF would presumably be collected at a national level and the recycling subsidy paid out at the local level, this scheme would add budgetary burdens to the entire solid waste collection system at the municipal level unless an extensive system of federal grants to local communities was organized.

Without much empirical experience to tell us the magnitudes of these various good and bad features of the three schemes, we are left to make a judgment call. I like Scheme 2. It is administratively easy to operate, it conforms with common sense, and it should prove politically acceptable. Moreover, hidden in the preceding paragraphs is another reason for preferring Scheme 2. Only Scheme 2 gives manufacturers a *direct* incentive to make products and packaging more easily recyclable. Manufacturers can reduce the ADF that they must pay in four ways: 1) by using materials that are less costly to collect and sort for recycling; 2) by using materials that are more valuable when they end up on the recyclables market; 3) by actively entering and fostering the recyclables markets into which their products and packages ultimately enter; and 4) by leasing the product, and hence recapturing it for “remanufacturing,” such as is already being done in the United States - without any ADF-avoidance incentive - for toner cartridges, postage meters, office furniture, auto parts, and “disposable” cameras (Deutsch, 1998; Duff, 2001).¹⁷

If, however, one is pessimistic about the size of the net social benefit to recycling • which means that one thinks the net social cost of recycling most products is not soon going to be much less than the cost of landfilling or incinerating trash – then all three schemes become simpler, in theory and in fact. Look back at Figures 4, 5, and 6. When the trash cost and the net recycling cost are equal: Scheme 1 requires a single, identical charge on trash collection and recyclables collection; and Schemes 2 and 3 require only an ADF. Administratively, this is a serendipitous outcome, for it means that only one policy intervention is needed, and the ADF would be the clear choice from the

16. Why would the law of one price not apply? Recyclable materials are expensive to transport relative to their market value, so their prices will often differ a great deal across the United States. Aluminum cans, for example, sold (in the week ending 11 June 2003) for \$550/ton when delivered in Los Angeles (California) and \$792/ton when delivered in Atlanta (Georgia) (*Waste News*, 2003).

17. Producers will choose to lease, rather than sell, if the value of the recycled used-up product is greater than the cost of recapturing and recycling it *minus* the ADF (which is completely avoided by leased products). The introduction of an ADF thus encourages leasing and recycling.

viewpoint of administrative and monitoring cost. I, however, am optimistic about the net social benefit of recycling in the near future and would prefer a pricing scheme that gives tangible economic incentive to households to recycle.

How big might the ADFs be? According to Scheme 2, the ADF should reflect the net recycling cost of the material. Table 2 shows the ADFs actually charged for packaging in France and Germany. 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, 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.

Table 2. ADFs in France and Germany, 2002
(in 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

Note: These are the weight-based ADFs. There are also small per-unit fees and some rebates.
Source: Glachant, 2004; <http://195.126.62.227/de/lizenznehmer/neukunden/englisch/frame/news.htm>.

All the practical complications of the three schemes can be summarized easily. Any ADF would ideally be equal to some a_{ij} , where i is the product (or package) and j the place where it is sold (or more precisely, where it will be disposed of). But in fact, it would be difficult to vary the ADF by place, so it would end up being a_i , varying across products (and packages) but not across places. Similarly any trash collection charge (t) or recycling collection charge/refund (r) would ideally be equal to some t_{ij} or r_{ij} , also varying across product and place. But in fact, it is difficult to vary them across products since that would require extensive household sorting or lengthy collector examination, so these would end up being t_j and r_j , varying across places but not across products. In a sentence, it is in practice impossible to achieve a fully first-best policy – close is as close as we can come.

The possibility of illegal disposal introduces still more complexity to the waste pricing problem.

5. Waste Pricing With Recycling and Littering

Some illegal waste disposal occurs even without waste pricing. But we have to worry that illegal disposal will move from a minor eyesore and inconvenience to a major social problem if we add an MTC of \$0.50-4.00 per 30-gallon bag or can.¹⁸ Illegal disposal is a social problem because it increases the total social costs of collection and disposal above the cost of proper disposal in the trash at one's own curbside. Since illegal disposal is in its essence a way of making somebody else pay for one's own trash collection and disposal, all illegal disposal ultimately leads to costly government and private counter-measures.

The possibility of illegal disposal adds a new branch to the consumer's waste decision tree. Until now, in order to create less solid waste and thereby avoid MTCs, consumers could do (some or all of) three things: 1) reduce their purchases of waste-making products and packages; 2) postpone their

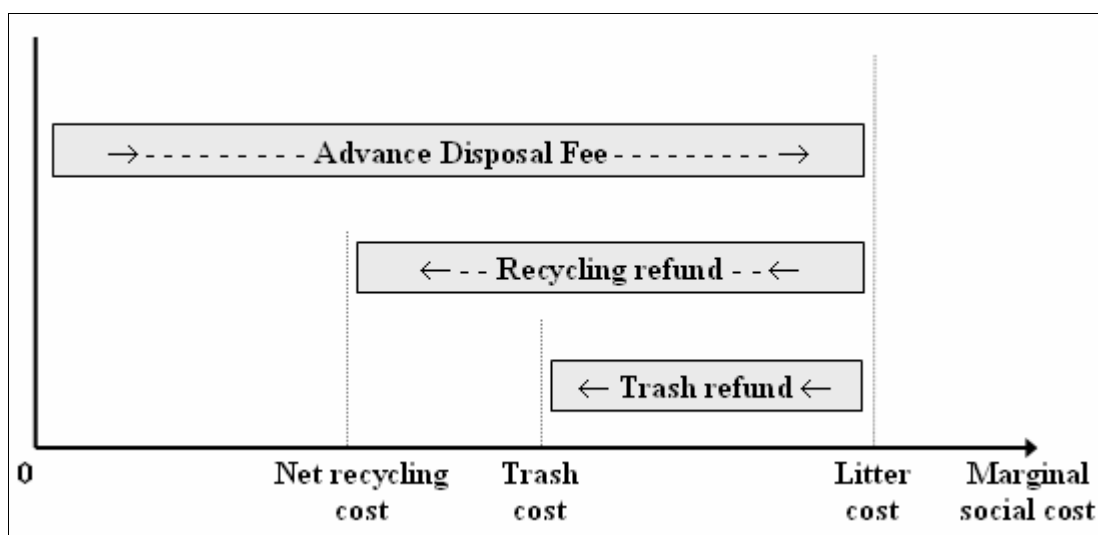
18. 30 U.S. gallons is the typical bag or can size in the United States; it is the equivalent of about 110 liters. \$0.50-4.00 is the range of MTCs currently utilized in the United States; most are toward the lower end of that range (U.S. EPA, 1999).

trash creation by reusing products; and 3) recycle things that would otherwise become trash. We now consider a fourth possibility, disposing of the waste illegally in order to evade the MTC.

If households can undertake costless illegal disposal - and are willing to undertake it - then it becomes impossible to charge a fee for any kind of legal disposal. There can be no charge for trash, and there can be no charge for recycling (Dinan, 1993; Fullerton and Kinnaman, 1995; Palmer *et al.*, 1995; Palmer and Walls, 1997).

Indeed, if households can litter without cost to themselves, then they must be encouraged by subsidies to dispose of trash and recyclables by socially more desirable means. An ADF must be added to the product, *equal to the marginal social cost of disposal by littering*, and refunds must be offered both for proper trash disposal and for recycling. Each of these refunds must be large enough to reduce the net private cost of proper disposal to the MSC of that means of disposal. The only feasible pricing scheme is shown in Figure 7.

Figure 7. Waste Pricing with Recycling and Illegal Disposal



Just to glance at this pricing structure is to see its weaknesses:

1. While huge revenues may be gained at the federal or state level from ADFs, there are huge new expenditures on refunds at the municipal level. Massive funds transfers would have to be organized.
2. The refunds for both trash and recyclables give a perverse incentive *not* to reuse. Indeed, worse than that, they encourage overloading the trash barrel with non-trash in order to get the subsidy. And they encourage misplacing trash into the recyclables container in order to get the larger subsidy there.
3. The ADF can vary across products but not across places, and the social cost of illegal disposal will vary across places. Similarly, the trash and recycling refunds can vary across places but not across products, and the net recycling cost (if not the trash cost) will vary across products.
4. The greatest objection, however, is a non-economic one. In most democratic societies, we do not like to subsidize "good" behavior, but rather we expect it and prefer instead to penalize "bad" behavior.

Is there an alternative to subsidies for discouraging illegal disposal? Yes. For activities that generate external costs, Pigovian taxes are appropriate.¹⁹ In this case the proper tax would be equal to the probability of apprehension and conviction times the damage done by the illegal disposal – *i.e.*, the expected damage – which consists of both the “eyesore” damage of the litter on the ground and the later pickup costs of that litter.²⁰ Unfortunately, while most U.S. states do advertise high fines for littering, very few “litterbugs” are actually apprehended and fined. Law enforcement officers seem to have more pressing things to do than to guard against litter. Once we multiply the probability of being caught and convicted times the amount of the fine, we find the *expected private cost* of illegal disposal is close to zero, no matter what the level of fines is.

Of course, one could always make the *expected* fine high by offsetting a low probability of being caught and convicted with an extremely high fine – say \$1,000 or even \$10,000. But such a penalty structure would not prove tolerable in the United States.²¹ If litterers were rarely apprehended but then threatened with huge fines, the courts would overflow with contested cases, juries would refuse to convict, and judges would balk at sentencing. In most democracies, people feel that the punishment should fit the crime.

Overall, in theory, there is something to be said for either approach to illegal disposal, subsidizing legal disposal or penalizing illegal disposal – or both (Sullivan, 1987). But the two approaches are hardly identical. Subsidizing legal disposal lowers the cost of producing (and later legally disposing of) waste, so it encourages greater waste; but penalizing illegal disposal raises the average cost of waste and discourages its production.²² Subsidizing legal disposal costs government dollars but not real resources; but penalizing illegal disposal eats up resources – investigators, police, lawyers, courts (and possibly jails). Both approaches are subject to diminishing returns of a sort. Doubling the enforcement resources will not usually double the number of litterers detected. And doubling the subsidy rate means that ever more legal disposers are receiving unnecessary inframarginal rents. These diminishing returns suggest that, in theory, some of each approach might be optimal.

Whenever trash collection charges are imposed or increased, anti-litter enforcement resources may have to be expanded. Useful ways to expend additional resources include: 1) cleaning up rapidly any areas that attract illegal dumping since the presence of previous litter reduces people's guilt feelings about adding new litter; 2) dedicating cameras or police for surveillance of potential illegal dumping sites or for searching illegally dumped trash for clues about the owner's identity – as the U.S. Coast Guard already does for boat-litter in the Great Lakes; 3) encouraging snitches by providing

19. Named after A. C. Pigou, who first suggested taxation as a remedy for external cost, a Pigovian tax on an activity should be equal to the marginal external cost generated by that activity (Pigou, 1920). When people considering the activity take into account this Pigovian tax, they are in effect taking their external costs into account. (For more on Pigovian taxation and for a discussion of other means of handling external costs, see Porter, 2002, Chapter 1, Appendix A.)

20. I use the word “eyesore” as a catchall for all the damages that littered trash causes before it is picked up • not only esthetic displeasure but also hand and foot cuts, farm equipment damage, animal injury, etc.

21. The only country where steep fines for littering *do* seem tolerable is Singapore – often called a “fine” city. First-time littering offenders face a fine of roughly \$500, and for repeat offenders, a fine of \$1,000 *and* a Corrective Work Order (CWO), which requires a few hours picking up litter in a park. Moreover those on CWO must display their status by wearing special brightly colored jackets and sometimes endure local media coverage. Litter is *not* a problem in Singapore (<http://www.singapore-window.org/sw01/010306nz.htm>).

22. Subsidizing legal disposal can be made equivalent to taxing illegal disposal if an appropriate output tax is also utilized (Fullerton and Mohr, 2003).

a litter tip hotline or by sharing any resulting fines with the tipster – New York City gives half the fine to the tipster if he or she testifies (U.S. EPA, 1998, p. 28; and 4) fining the *landowner* when illegally disposed litter has to be cleaned up on a property – unfair but effective (Gantert, 2002a and 2002b).

All of this so far assumes that illegal disposal of trash would become a serious problem if MTCs were imposed. In the United States, it is still not clear whether illegal disposal does in fact become a serious problem with MTCs. One early, careful study of MTCs examined a sample of households in Charlottesville (Virginia; Fullerton and Kinnaman, 1996). Before the city's MTC was initiated, the sampled households averaged 0.71 kg per capita per day of trash and 0.24 kg per capita per day of recyclables.²³ After the city introduced a \$0.80 price per bag for trash (with no fee for recycling), the households averaged 0.10 kg less trash per capita per day and 0.04 kg more recyclables. The difference, 0.06 kg, represents either source reduction and reuse or illegal disposal. This study guessed that illegal disposal rose by roughly 0.03 kg per person per day, which is 30% of the reduction in trash (though only three% of the total trash-plus-recyclables generated). The authors conclude that "the social cost of noncompliance can be large" (*ibid.*, p. 980).²⁴

The vast majority of studies of trash collection charges, however, conclude that illegal disposal is not a serious problem (Deisch, 1989; Goldberg, 1990; World Wastes, 1993; Bender *et al.*, 1994; Miranda *et al.*, 1994; Miranda *et al.*, 1996; Skumatz *et al.*, 2001; <http://www.epa.gov/epaoswer/non-hw/payt/tools/tools4.htm>). The problem with this "vast majority" is that their results are largely based on hearsay, often from those who are biased toward MTCs. There is also a question of causation. It is very possible that the more environmentally concerned communities, who are usually the first to embrace MTCs, are also the more law-abiding communities

Another possibly serious drawback to MTCs is the extra administrative cost of implementing the system. Tags (or bags) must be printed and distributed and the system requires increased monitoring and paperwork. Earlier (in Section 2), we suggested that an MTC of \$0.10/kg would reduce trash from roughly 2.0 kg/capita/day to 1.5 kg/capita/day, reducing the deadweight loss (DWL) of zero-pricing by \$0.025 per person per day. The remaining 1.5 kg/person/day would have to cost less than \$0.017 per kilogram more to collect with the MTC or the added administrative burden would more than offset the reduced DWL. The Fullerton and Kinnaman, 1996 study estimated this administrative cost at about \$0.024 per kilogram.²⁵ They concluded that MTCs do not pass their benefit-cost test.

Surprisingly, few other studies have even asked whether the added administrative cost of an MTC system is significant or not. Most studies content themselves with just estimating the trash decreases and recycling increases. Anecdotal evidence and comments of solid waste administrators,

23. Trash per capita per day in this sample is much lower than the national average of around 2 kg per capita per day (Table 1) because the sample excluded small businesses and multi-family dwellings and oversampled educated and high-income families.

24. See also Fullerton and Kinnaman, 1995, and Kinnaman and Fullerton, 2000.

25. They estimated the added cost to be \$0.193 per bag, and I have converted that at a rate of eight kilograms per bag. The introduction of MTCs increases the weight per bag since the MTCs are almost always based on the volume, not the weight, of the trash. This weight increase is achieved by means of the "Seattle stomp" (named after the Washington city where it was first observed) • households buy and use home compactors to reduce the volume and hence the cost of their trash. This is socially inefficient since the trucks that pick up the trash do that same compacting job over again, better and cheaper, so every home compactor purchased solely to save trash fees is a complete *social* waste of resources. The way out of this problem is of course weight-based charges, and this has been tried in various cities (McLellan, 1994; Skumatz *et al.*, 1994; Andersen, 1998). But weight-based charges mean weighing, and this requires scales on the collection trucks, it slows collection (by 10% in Seattle), and it necessitates a complicated billing system.

however, suggest that MTCs do not add much to the cost of a MSW program. After all, the addition of a tag or the substitution of a different bag changes a city's overall solid waste system very little.

A final concern of MTCs is their effect on the income distribution. A switch from property-tax financing of trash collection to MTC financing will take a larger fraction of a poor family's income than of a rich family's income. Of course, this unfortunate effect on income distribution also applies to most other publicly provided services, such as electricity, telephones, gas, and water, and we almost always charge for these according to the quantity purchased. But if equity concerns arise with the introduction of MTCs, they can be alleviated by "lifeline pricing" – for example, not charging for the first bag each week. One costless bag would be all most poor families would need. Since one bag would also be what many not-so-poor families would need each week, the first-bag exemption would much reduce the administrative costs of operating the MTC system.

Possibly the best evidence that the advantages of MTCs outweigh the drawbacks in many U.S. cities is the rapid growth in the number of American municipalities that have adopted and retained MTCs. Before 1986, only 126 municipalities put any kind of marginal price on trash collection. Today, something like 6,000 municipalities do (Siskos 1999; www.epa.gov/epaoswer/non-hw/payt/). In only four of the 50 American states does no community at all utilize some kind of MTC (Alabama, Kentucky, Mississippi, and Wyoming). But the fact that MTCs make sense in some U.S. cities does not mean that they are sensible in all cities. Cities with any or all of the following characteristics would be poor candidates for MTCs: 1) administering an MTC system would be high-cost; 2) illegal dumping would be a serious problem; 3) income distribution concerns would be irremediable; 4) multi-family housing makes up a significant percentage of residences; and/or 5) net recycling costs are high.

We turn next to a form of MTC that changes the waste disposal/recycling system quite radically – mandatory deposits.

6. Mandatory Deposits

For some products, the temptation to litter is particularly high or the resulting wrongly discarded trash is particularly obnoxious – so high or so obnoxious that we do not wish to rely solely on traditional anti-litter ordinances to ensure proper disposal. One example of illegal disposal for reasons of aesthetics is the littering of beverage containers in public places, and several U.S. states and most Canadian provinces have reacted to this by imposing mandatory deposits on such containers – a special fee that consumers pay at the time of purchase that is rebated when the container is properly returned.

For other products, their hazardous contents make it particularly important not only that they are not illegally dumped but also that they are kept out of MSW landfills (or incinerators). Examples of mandatory deposits on products where the external cost of illegal, landfill, or incinerator disposal is great are also beginning to appear in the United States – notably, automobile and household batteries, pesticide containers, tires, and motor oil, all of which can cause serious health or environmental damage if casually discarded into MSW landfills (U.S. EPA, 2001).

How do mandatory deposits differ from an ADF coupled with a household trash or recycling refund? The critical distinction is in the re-collection of the product at the end of its life. With the trash or recycling refund, the disposed product is collected through the regular municipal trash or recycling collection system; with a mandatory deposit, on the other hand, a special, separate collection path is established. When the consumer returns the product to a retailer, the product is then returned to the wholesaler, who arranges its proper disposal or recycling. If the intention of the deposit is to keep the product out of a landfill or incinerator, this special collection path may be essential. But if the intention

of the mandatory deposit is only to prevent litter and/or to increase recycling, then the special collection path is an unfortunate additional cost of deposits.²⁶

The theory of mandatory deposits is straightforward. The deposit should be set equal to the extra social cost of improper disposal over the net recycling cost (assuming that there is already an ADF on the manufacturer equal to the net recycling cost; look back at Figure 5). Then, if a person disposes of the product improperly, that person pays the external cost of improper disposal by forgoing the deposit. The threat of a foregone deposit becomes a Pigovian tax equal to the marginal external cost of littering.²⁷

Another way of viewing the mandatory deposit is as a system of Pigovian taxes and subsidies. When the consumer buys the product, the deposit is a tax levied on the assumption that the residual material will be improperly disposed of. And then, should the buyer return the product for proper disposal, the redemption is a subsidy for rejecting the easier, but socially undesirable, option of improper disposal.

While the theory is straightforward, the actual workings of mandatory deposits are often complex. It is worth our time to look closely at an example of the use of mandatory deposits, one that has been applied in parts of the United States and Canada for several decades • mandatory deposits on beer and soft-drink containers.

For the first half of the twentieth century, there were always deposits on beverage containers. But these deposits were not mandated by law. They were undertaken voluntarily by the beverage producers themselves in an effort to recapture the relatively expensive bottles for re-use. The bottles were much too valuable to throw away – they were heavy, durable, and reused some 15-20 times before breaking, getting lost, or chipping to the point where they were finally discarded. Steel cans for beverages appeared during World War II to provide beer and soft drinks for troops overseas, and they moved into the consumer market soon after the war (Bingham *et al.*, 1989). The trend to “no-deposit-no-return” cans and bottles was rapid, driven by many forces. The “one-way” container not only was convenient to consumers, it provided marketing advantages for the aggressive oligopolists of the beer and soft-drink industries. One-ways were becoming steadily cheaper and lighter, they reduced transport costs, and they obviated the need for the labor-intensive sorting, washing, and inspecting of used containers. By the 1980s, the returnable, refillable container with a deposit had all but disappeared (Franklin, 1991; Saphire, 1994).

A byproduct of this disappearance of non-mandated deposits was greatly increased beverage container litter in public places. Such litter provides two kinds of external costs • the obvious flow cost in the sense that someone else must dispose of the illegal trash, but also a stock cost in the sense that the public must endure the esthetic disutility, equipment damage, and health costs before the litter is picked up. A return to deposits, this time government-mandated deposits, was seen as a means of ending this widespread illegal disposal of beverage containers. Moreover, with the advent of the energy crisis in 1973, mandatory deposits were also envisioned as a means of encouraging the

26. Similar to a deposit-refund system, but with lower re-collection costs, is a combined tax-subsidy system. This combines a tax paid by the producers of a product with a subsidy paid to those who later collect and recycle the used-up products (Palmer and Walls, 2002). This system raises the initial price to the consumer and, unlike with mandatory deposits, this price increase would almost surely not be fully refunded at the end of the product's life and perhaps not refunded at all. As Palmer and Walls (2002) put it, “a program that lacks incentives for consumers to return products is destined to be either a recycling failure or a very expensive ‘success’” (*ibid.*, p. 39).

27. What is the illegal-disposal cost implied by a five-cent deposit on an aluminum beverage container? \$0.05 times (roughly) 60 containers per kilogram times 1,000 kilograms per ton yields a figure of \$3,000 per ton of illegally disposed beverage containers.

return to the reusable container, which uses less energy per beverage delivery than do single-use containers. Re-use also addressed the worry of the “doomsday theorists” of the early 1970s who were forewarning us that the planet was rapidly running out of resources. Laws mandating deposits on beer and soft-drink containers began to pass, first in British Columbia in 1970 and Oregon in 1972 and eventually in eight of the ten Canadian provinces and eleven of the 50 U.S. states.²⁸

The experiences of these ten states provide clear evidence of the impact of mandatory deposits on beverages and their containers (Porter, 1978 and 1983; Bingham *et al.*, 1989; Franklin, 1991; Saphire, 1994; Ackerman, 1997):

1. *Return Rates and Litter Rates.* With 5-cent or 10-cent deposits to be redeemed, consumers *do* return their beverage containers. Studies consistently show that 85-90% of the beverage containers are redeemed, nearly double the rate at which containers are recycled in non-deposit-law states (Michigan, 1998, p. 28).²⁹ And of course the byproduct is a decline in container litter, by nearly 80%, and a decline in overall litter by nearly half. Municipal solid waste also declines, by a few percent. Mandatory deposits *do* achieve what they are primarily intended to achieve.
2. *Beverage Prices, Costs, and Consumption.* Studies consistently find that beverage consumption, both of beer and of soft drinks, goes down by 5-10% as a result of the introduction of mandatory deposits. This decline is the rational response to the fact that the *total* price to the consumer goes up, where by total price is meant the sum of three components: 1) the actual retail money price; 2) the expected foregone deposit (because even conscientious consumers lose or break some); and 3) the inconvenience cost of returning empties. With mandatory deposits, all three components increase, although the actual retail price increase may be small because recycling revenues and unreclaimed deposits offset, to some extent, the increased cost of beverage delivery.
3. *Return to Reusable Containers?* One of the anticipated side-benefits of mandatory deposit laws was a reversal of the trend away from refillable glass containers. Indeed, mandatory deposit laws are still often called “bottle bills” because of this expected return to refillable bottles. Once bottlers and brewers were forced to re-collect the containers, it was thought, they would choose to re-collect reusable rather than non-reusable containers. Not so. Mandatory deposits may have slowed the tide toward one-ways, but not much and not for long – by the 1990s, refillable beer bottles accounted for 13% of the market in deposit-law states and 3% of the market in non-deposit-law states (Saphire, 1994). Efforts to tailor deposit legislation to stimulate a revival of refillables have so far failed to achieve that goal.³⁰ The only way to stop

28. Oregon, Vermont, Maine, Michigan, Iowa, Connecticut, Massachusetts, Delaware, New York, California, and Hawaii • in chronological order of implementation. These states include about 30% of the U.S. population. Exactly which beverage containers are covered by mandatory deposits varies from state to state. All include beer and carbonated soft drinks; some include mineral waters; others include wine coolers; Maine includes juices and tea; Delaware exempts aluminum containers from its mandatory deposits (U.S. EPA, 2001). For details of U.S. state and Canadian province deposit systems, see http://www.bottlebill.org/what_are_b-bills.htm.

29. In Michigan, where the basic deposit is twice as high as in any other deposit-law state (\$0.10 versus \$0.05 per less-than-48 centiliter beverage container), the return rate averaged 98% during the 1990s. This astonishingly high return rate is probably explained by illegal returns of non-deposit bottles from neighboring states (Seinfeld, 1996; Truini, 2002c).

30. Under the mandatory deposit law of New Brunswick and Nova Scotia (Canada), consumers get back the full deposit if the container is refillable but only get back one half the deposit if the container is not refillable (*i.e.*, if it is a plastic container, a metal can, or a one-way glass bottle). But this difference has not stopped the trend to one-way containers • while the law has been in effect, refillable containers

the trend toward “one-ways” seems to be the Prince Edward Island (Canada) approach, to completely ban the sale of all containers except refillable glass bottles (www.gov.pe.ca/fae/env/refillablebottles.php3).

4. *Recycling Impact.* Although increased recycling was not the original intention of mandatory deposits – the intention was to get back to refillable, reusable containers – once wholesalers were forced to re-collect their one-way containers, they sought ways to avoid the cost of landfilling them. Recycling was not widespread in the 1970s, but markets for the recyclable aluminum and glass quickly sprung up. Today, the eleven deposit-law states supply almost half the glass bottles, aluminum cans, and plastic bottles that are recycled in the United States (<http://www.container-recycling.org>). Before enthusing about this serendipitous boost to recycling, however, we should note that re-collection of containers by their producers is a *very expensive* way to recycle. One study concluded that recycling beverage containers under a mandatory deposit system costs \$320 per ton, while the average cost of all other recycling programs is only \$120 per ton (Flynn, 1999). The bottom line is that mandatory deposits are an effective anti-litter program, but they are a costly recycling program.

One question about mandatory deposit laws was never asked until California contemplated its deposit law: What is the optimal number of redemption centers (*i.e.*, the number of places at which empty containers can be returned for redemption of the deposit)? In almost every deposit-law state until California, any retailer who sold a particular brand of beer or soft drink was required to accept back empty containers of that brand. Implicitly, legislatures were maintaining that the optimal number of redemption centers was equal to the number of stores selling the product. But just because the market decides to have *N* stores selling Whoopie Cola does not mean there should be *N* redemption centers for Whoopie Cola containers.

When California initiated its deposit law, it recognized that not all retailers need also be a redemption center. If a state-certified redemption center is located within one half mile of a store, the store is not required to redeem empty containers. This greatly reduces the number and cost of redemption centers without greatly adding to consumer inconvenience. While the California return rate is much lower than in other mandatory deposit states, this is due not so much to the fewer redemption centers as to the lower redemption value³¹. Several deposit-law states have now adopted this idea, permitting a retailer to refuse to redeem containers for deposit if an authorized redemption center exists within a certain distance.

California’s law also solves another of the inefficiencies of mandatory deposits, that redemption centers have to sort out returned containers by brand in order to recapture their deposits from the brand’s wholesaler. From the social viewpoint, this incurs totally unnecessary cost, since all the containers are going to the same recycling places in the end. In California, the State, not the manufacturer, effectively terminates the deposit chain. The redemption center has no need to sort its containers by brand since the State, and the recyclers with which it deals, couldn’t care less which

have gone from a few percent of the total to virtually zero percent (Ezeala-Harrison and Ridler, 1994). Efforts to force the return to refillable bottles have not ceased. Since the start of 2003, Germany has required deposits of \$0.25 per non-refillable (small) container on beverage sellers whose sales over the previous two years have not reached a target of 72% refillable containers (and this new deposit is roughly three times the current deposit on refillable containers). By the way, a consumer must return the empty container to the same store from which it was purchased • and produce a sales receipt as proof • in order to redeem the deposit (Landler, 2003).

31. The basic California redemption rate was only \$0.025 until recently, when it was raised to \$0.04. In another interesting California twist, the state intends to further raise the redemption rate if the beverage container recycling rate does not reach 75% by 2006 (Saphire, 1994; Truini, 2003).

brands are being returned. One study concluded that these cost-cutting changes resulted in a collection cost under California's redemption system of \$0.002 per container (i.e., \$120 per ton) while the cost in other deposit-law states is greater than \$0.020 per container (i.e., \$1,200 per ton; Ackerman *et al.*, 1995).³²

When mandatory deposits on beverage containers were enacted in many U.S. states in the 1970s and early 1980s, there were almost no recycling programs with curbside pickup in the United States. The question today is whether mandatory deposits are sensible in an era when lower-cost municipal recycling programs have become widespread. Even recycling enthusiasts who were in the vanguard of those proposing mandatory deposit laws 20 years ago are now asking this question, especially because such deposits seriously hurt the financial viability of municipal recycling programs (Hawes, 1991; Ackerman *et al.*, 1995). Where there is recycling but no mandatory deposit law, beverage container scrap provides nearly half of the recycling revenue (U.S. GAO, 1990). The *one city* in the United States that had adopted mandatory deposits on beverage containers (Columbia, Missouri) recently repealed its law because a recycling alternative is now available, because retail beverage prices were significantly higher, and because the city recycling operation needed the aluminum revenue (Toloken, 2002).

Adding mandatory deposits to an existing recycling process increases both the total collection costs and the total recyclable tonnage collected. The relevant question is whether the marginal benefit (of landfill/incinerator cost avoided) exceeds the marginal cost (of collection and processing, net of revenues earned) when mandatory deposits are added. The net marginal cost of collecting more recyclables by adding mandatory deposits to a system that already recycles has been estimated to be much higher than the net marginal cost of collecting recyclables through the curbside recycling service (Franklin, 1988; Ackerman and Schatzki, 1991). A recent estimate of how much higher is implicit in a (rare) collaborative study undertaken jointly by a group of bottlers and environmentalists. In ten mandatory deposit states, 71.6% of the beverage containers are collected, at an average cost of \$0.0153 per container (net of recycling revenues from the recovered material); in the 40 non-deposit-law states, 27.9% of the containers are collected, at an average cost of \$0.0125 per container (also net of revenues; Beck, 2001).³³ Assuming the states are otherwise identical, these figures imply that the marginal cost of going from 27.9% collection to 71.6% collection is \$0.0171 per container.³⁴ This means \$889 per ton (since on average in the United States, there are 52,000 aluminum, glass, and plastic containers per ton of recycled beverage containers). The marginal benefit of the added tonnage is the collection and disposal cost of those containers at a landfill or incinerator – certainly far less than \$889.³⁵

Greater recycling through adding mandatory deposits to a curbside recycling system yields benefits other than avoided landfill or incinerator costs. By recycling, we reduce air and groundwater pollution, decrease global warming emissions, save energy, economize on land use (for landfills), and conserve natural resources by reducing demand for virgin raw materials. If the marginal social cost of these were equal to their prices, no market failure would occur; recycling would already take into account these benefits. To illustrate, consider just one, groundwater pollution. Suppose the landfill tipping fee does not fully reflect the expected marginal external cost of groundwater pollution due to landfilling. Then the tipping fee is too low – *i.e.*, the landfill price is below the marginal social cost of landfilling. The first-best response to this is to tax landfills, which would raise the tipping fee, raise the

32. See footnote 27 for the method of converting dollars per container to dollars per ton.

33. Hawaii was included as a non-deposit-law state since its law did not become effective until after the Beck, 2001, study was done.

34. $0.0171 = [(0.716) \cdot (0.0153) - (0.279) \cdot (0.0125)] / [(0.716) - (0.279)]$.

35. The National Soft Drink Association has declared this study “flawed ... biased ... sloppy research ... manipulated” even though the findings implicitly provide a strong argument against the mandatory deposits U.S. bottlers have long and vehemently fought (Truini, 2002b).

marginal trash charge, and encourage lesser beverage consumption or greater recycling of beverage containers. Introducing mandatory deposits on beverage containers is a very indirect and second-best way of encouraging recycling. Why second-best? Because it only encourages beverage-container recycling - not recycling in general - and it initiates a new, separate, and high-cost collection system - instead of utilizing the already-installed, cheaper curbside-collection system. The same sort of story applies to the other supposed benefits of mandatory deposits listed at the start of this paragraph. They are all second-best, at best.

Another benefit of mandatory deposits on beverage containers, or more generally of any means of greater recycling, is often said to be job creation. Even the U.S. EPA regularly turns out statements like “recycling is estimated to create five times as many jobs as landfilling” (U.S. EPA, 1994, p. 1; and U.S. EPA, 1997, p.1), and “for every 100 recycling jobs created, ... just 10 jobs were lost in the solid waste industry, and three jobs were lost in the timber harvesting industry” (U.S. EPA, 1995a, p. 11).

Several things are wrong with seeing “jobs” as a benefit of greater recycling. 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 and 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 what government spending is decreased when spending on recycling is increased.³⁶ 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 not be sure that recycling was the best way of achieving this outcome. I still recall vividly (and sadly) the debate between the Nixon and McGovern forces in the 1972 U.S. Presidential election campaign over whether our involvement in Vietnam increased employment by hiring labor-intensive infantry or decreased employment by employing capital-intensive helicopters – as if the answer to that question would decide the merits of our involvement there. The fact that there are unemployed people is a market failure, but it is not one that will be cured by recycling.

Only one first-best argument remains for adding or retaining mandatory deposits after recycling has become widespread. Mandatory deposits are an anti-litter policy, which recycling is not. Neither the availability of free curbside recycling nor an MTC does anything to prevent litter, and taxing litter directly is impossible. If beverage container litter is considered to be a serious problem, then mandatory deposits are a much more effective policy than either the availability of recycling or an MTC, and the co-existence of all three may make sense.³⁷

Two bottom lines to mandatory deposits. One, if a product is toxic and should be recovered and recycled carefully, then mandatory deposits are an excellent way of enlisting consumer assistance in keeping the product unlittered and out of landfills (or incinerators). And two, mandatory deposits are a very expensive anti-litter program even when there is no recycling, and mandatory deposits become more expensive when and where there is curbside recycling. Moreover, if the loss of recycling revenue delays or undermines the operation of socially profitable recycling programs, the final cost of any litter reduction will be even higher (Ackerman and Schatzki, 1991). This is not just speculation. One

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

37. Where both recycling and mandatory deposits exist side by side, it is important to remember that these are two separate policies aimed at two separate goals. The amount of the deposit should be a function of the marginal social cost of illegal disposal – there is no good reason for the deposit to be lower for more widely recycled materials or lower for potentially refillable containers, as is often suggested (Cohen *et al.*, 1988). California’s plan to raise redemption rates on beverage containers if recycling goals are not met is misguided unless the recycling rate is interpreted as a close proxy for the “not-littered rate” (see footnote 31).

empirical study found that where mandatory deposits were in effect, communities were 18% less likely to start curbside recycling collection (Kinnaman and Fullerton, 2000).

7. Setting Recycling Targets

The missing markets in waste policy distort the incentives of businesses and households, and some combination of an ADF on business products and packaging and an MTC on household trash collection does much to correct this distortion. In the United States, however, there are still few MTCs and almost no ADFs. Governments have preferred to counter the absence of adequate incentives for business and households to recycle by imposing recycling targets on *municipalities*, quite the wrong target for correcting the market failures. With a recycling target, if recycling fails to exceed a certain percentage of solid waste generation by a certain date, the municipality may be penalized. The U.S. Environmental Protection Agency has set a national recycling goal • 25% by the year 1992, announced in a speech in 1988 by then Assistant Administrator of EPA, Winston Porter, and this percentage was later and quietly boosted to 35% by the year 2005 (Porter, 1988; Truini, 2002e). Almost every U.S. state has also set targets, and they range widely, up to 70% for Rhode Island.³⁸

The fundamental error in setting recycling targets is that it suggests that the more recycling the better. That is just plain wrong. There are many materials that ought not to be recycled – now, at least. (Landfills can always be “mined” later.) Two of the papers in this volume take the trouble to ask carefully what should be recycled (Kristensen, 2004; Fitzsimons, 2004). The social cost of landfilling PVC waste in Denmark is significantly lower than the three alternative treatment methods studied (Kristensen, 2004). Examination of eight comparative studies of the choice between incineration and re-refining of used oil finds no clear indication which is socially preferred (Fitzsimons, 2004). Other studies have pointed out that the proper collection of used oil is what is socially important, and if re-refining used oil adds to the costs, it may lead to a reduction in the amount of used oil collected (CONCAWE, 1996; Sigman, 1998).

Setting a recycling target probably does little harm if the state then does nothing to implement it. Many of the states back their recycling targets with little more than a requirement that cities develop recycling plans or introduce some minimal form of recycling. Others intend to levy fines on municipalities that fail to meet a specified target by a specified date. As of mid-1999, only 69 of California’s more than 450 municipalities were meeting the state’s year 2000 target – a 50% reduction in landfilled solid waste. Did the State really fine the other 381 towns and cities \$10,000 per day starting on 1 January 2001 (San Francisco Chronicle, 1999)? Of course not. Extensions were granted until the year 2006, leaving plenty of time to change the law. Or grant new extensions. Or redefine waste – California does not actually measure waste generation but estimates it through a formula involving the state’s income and population (Johnson, 2000).

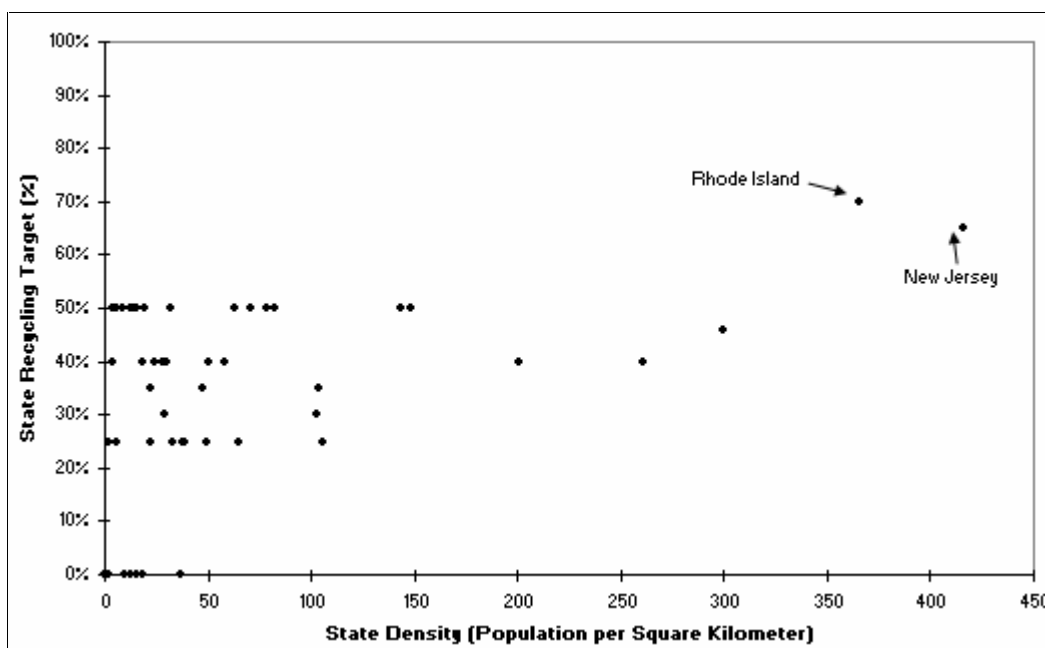
But even targets without teeth have a way of inducing serious policies to achieve them. Setting a wrong target may well lead to wrong recycling policies. Careful economic analysis is needed to even guess at the answer to the question of how much recycling is desirable. But very little such analysis has gone into the setting of state targets. Neighbor states with similar living standards and population densities have often set very different targets, strongly suggesting that one – or both – of the targets is wrong. We should expect that states with dense populations, and hence more costly landfills and less costly recycling collection, would set higher recycling goals, but the correlation is very low between

38. The current state recycling goals can be seen at www.afandpa.org (then click on Environment & Recycling, Recycling, and State Recycling Goals). The states’ definitions of the recycling rate differ widely. Indeed, some states target the recycling rate, while others target the waste reduction rate (which counts reduction and reuse as equivalent to recycling). To further complicate state comparisons, some states count incineration as recycling, and some states do not count composting as recycling (Rabasca, 1995).

the 50 states' recycling targets and the population densities ($R^2 = 0.10$). A scatter diagram of these two variables is shown in Figure 8 – if you cover up the two states to the far right (New Jersey and Rhode Island), almost no correlation remains ($R^2 = 0.01$).³⁹ When the state then urges its chosen target on each of its cities and towns, it is also failing to recognize that the optimal degree of recycling will vary across municipalities of different densities and locations.

If the state targets were serious and binding on municipalities, we should see a close relationship between state recycling targets and state recycling achievements. There is a relationship, as Figure 9 shows, but it is weak ($R^2 = 0.29$ for 49 states, but only $R^2 = 0.11$ if the seven states with no target are omitted).⁴⁰ And there always lingers the possibility that achievements determine targets rather than the reverse – that is, states that do (or can) recycle cheaply set high recycling targets and that states that can only recycle expensively set low (or no) targets. The fact that only seven of the 43 states (that have set recycling targets) have actually met those targets suggests less than serious goal-setting.

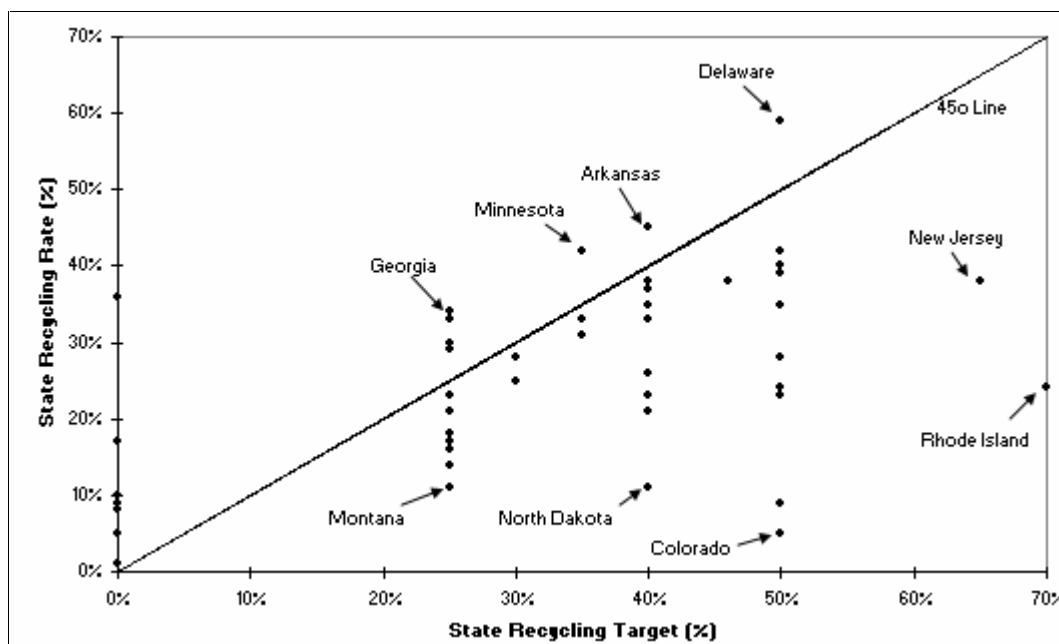
Figure 8. Relationship of State Recycling Target to Population Density



39. The source of the state recycling goals in Figure 8 is that given in the previous footnote. The source for state population density is the *U.S. Statistical Abstract*, various years.

40. The source of 46 states' actual recycling rates in Figure 9 is Goldstein and Madtes, 2001. The rates for three of the states omitted in that source are estimated from various other sources. I could find no recycling estimate for one state (Idaho). The state recycling targets are the same as in Figure 8.

Figure 9. Relationship of Actual State Recycling Rates to Target Rates



The basic failing of a state recycling goal is that it targets the state's municipalities, but the municipalities are not the reason for inadequate recycling. The reason is that the generators of waste - who also provide the potential for greater recycling - are businesses and households, and their incentive to recycle is not increased by a state recycling target. To see how misdirected targets are, consider the following hypothetical: Having found out that Americans are eating too much and are on average 20% overweight, the government then mandates that the average weight of the residents of every city should be reduced by 20% or else the *city government* will be fined!

Where states have taken serious steps toward increasing the amount of recycling, these steps have been of two basic kinds, demand-side and supply-side. Demand-side policies are those that stimulate demand for recyclable materials or for products made from recycled materials. Supply-side policies are those that stimulate greater collection and processing of recyclable materials. We now look at each.

8. Demand-Side Policies

If we think only about first-best policies for trash collection and handling, there is no role for demand-side policy. The market failure is supply-side, that businesses and households are not paying ADFs and MTCs that reflect MSC, and hence they are not recycling enough. In a second-best world, encouraging the demand for recyclable waste and recycled products may be an appropriate way to counter irremediable supply-side failures. Furthermore, the use of virgin raw materials is favored by public policies in many ways – both by direct subsidies and by government failure to correct external costs. And anything that favors virgin materials takes demand away from substitutable recyclable materials. Again, in a first-best world, government subsidies to virgin material growth or extraction would be removed, and the external costs of using virgin materials would no longer go uncorrected. In a second-best world, however, where virgin material policies cannot be corrected, stimulating demand for recyclable materials and recycled products may be appropriate.

The higher the price of recyclable materials, the greater will be the optimal amount of recycling. The higher the price of products made of recycled materials, the higher will be the price of those recyclable materials, and the greater will be the optimal amount of recycling. Demand-side policies

seek to artificially raise the price of recyclable materials and recycled products in order to stimulate greater recycling. In the perfect world of competitive equilibrium, artificially stimulating these demands would overstimulate recycling, but in a world where the supply of recyclables is artificially held back by the absence of ADFs and MTCs, increasing demand may be a sensible second-best approach.

Most products made from recycled materials are at least somewhat inferior - in consumers' eyes - to products made from virgin materials either because recycled material can never be completely cleaned of extraneous matter or because the previous use has somehow depreciated the material. Often, consumers can see this inferiority in the color or feel of the recycled product. Often sheer unfamiliarity scares consumers away. As Frances Cairncross nicely noted:

Many people love recycling. It seems to meet some deep need to atone for modern materialism by saving some of the materials from the rubbish bin. Unfortunately, people do not feel quite the same craving to buy products made of recycled materials. (Cairncross, 1993)

Economists who want to make welfare judgments based on consumer sovereignty are reluctant to suggest "educating" consumers, but if consumers are overreacting to the inferiority or unfamiliarity of products made from recycled materials, then demand-side policies might be warranted.

In any case, demand-side policies abound in the United States. States, counties, and cities regularly mandate minimum recycled-input percentages for products such as the use of old newspapers in making new newsprint, and governments at all levels establish procurement preferences for recycled products such as office paper, re-refined oil, and re-treaded tires. Some simply mandate a minimum percentage for purchases of recycled products - often a minimum percentage of 100% - while others give a 5-10% price preference to recycled products. The percentage price preference is the better of these two policies, because it limits the potential for inefficiency.

It is worthwhile taking a long paragraph to see what's wrong with mandated recycled-content percentages. Consider a product made from virgin material, recycled material, and labor (Palmer and Walls, 1997). Because households are not charged for their solid waste, they do not recycle the socially optimal amount. As a result, the firm gets too little recycled material and uses too much virgin material. Mandating a higher recycled-content percentage can offset this, forcing the firm to offer a higher price for recycled material. This higher price might induce cities to do more recycling and could, in theory, establish the socially optimal ratio between recycled and virgin inputs.⁴¹ But it has also pushed up the costs of the firm and made labor relatively cheaper than material inputs. The firm's total output may fall, and its labor use may rise relative to its material inputs. All kinds of indirect, unwanted, and unpredictable secondary effects may take place, calling in turn for corrective output or labor taxes (or subsidies).

Aside from the second-bestness of demand-side policies, there are other problems. For one, those whose purchasing decisions are forced to change will resist the mandates and may find ingenious ways to circumnavigate them - governments themselves regularly fail in their efforts to get their own purchasing agents to buy more expensive recycled products. But a second problem is that it is all so *ad hoc*. For those recycled products for which there is a means of stimulating demand, demand is stimulated, but for others no stimulation is available. There is no guarantee that the right products - from the viewpoint of net social benefit - will be the ones to get the stimuli.

41. However, the higher price of recyclable material will not change household behavior if it continues to be charged zero for both its trash and its recyclables. Scheme 2, on the other hand, would encourage greater recycling by increasing the marginal trash charge when the market value of recyclable materials rises.

If demand-side policies must be used, they should be applied through marketable permits. Consider newsprint. The government mandates, say, an increase to 40% recycled content. Newspapers comply. We have no explicit picture whether the target was easy to reach and perhaps could readily be raised still further, or whether the target was hard to reach and greatly raised the cost of newspapers. If, however, the 40% target was marketable, newspapers in big cities near recycled newsprint mills could raise their recycled content beyond 40% and sell their surplus allowances to newspapers near virgin pulp newsprint mills (Bingham and Chandran, 1990). The 40% target would be reached *on average*, but at lower total social cost. Moreover, from the price of the marketed percentage, we would get a clear idea about the marginal cost of achieving the target.

9. Supply-Side Policies

The market failure that affects recycling is basically a supply-side failure – households supply too little of their waste for recycling because they are able to use the municipal solid waste disposal service for free, and businesses create hard-to-recycle products and packages because they are not charged for the disposal or expensive recycling of those products and packages. The first-best public response to this is also supply-side, charging for solid waste service so that households and businesses have incentives to seek waste handling techniques that involve less landfilling and incineration.

Without pricing waste disposal and gaining the stimulus it provides to recycling, other supply-side policies are almost inevitably going to be at best partially effective. Let's look at a few of the supply-side approaches that have been tried in the United States:

1. About half the states offer either tax credits or direct subsidies to encourage municipal recycling operations (Sparks, 1998). There seems to be no limit on the imagination – tax-exempt bond-financing for recycling facilities, property tax exemptions on land and buildings involved in recycling, subsidized land sales for recycling facilities, grants or low-interest loans to finance the purchase of recycling equipment, etc. (O'Leary and Walsh, 1995). Notice, however, what all these policies have in common – they affect things connected with recycling, but they do not directly stimulate recycling. Each relies on indirect linkages to recycling. In each case, some input price is distorted, and so each stimulates a distortion in the input choice. Recycling operations become too capital-intensive or too land-intensive, or facilities get located where the tax advantages are greatest, not where resource costs are lowest.
2. Products may be taxed if delivered to in-state landfills, or in the extreme case, completely banned from in-state landfills (Rabasca, 1995; Allison *et al.*, 2002). The hope, of course, is that these products will, as a result, be recycled. But the incentive offered by the tax or ban is not an incentive to recycle, but rather an incentive to *not landfill*. Illegal dumping, exporting, and incineration are also stimulated. Since landfill arrivals are often sized or weighed, taxing overall landfill volume or weight is not a big administrative problem, but taxing or banning individual items in those deliveries requires an expensive inspection system. States that have tried it have generally repealed or simply not enforced the ban or tax.
3. The construction of new landfills and incinerators may be banned, as in Massachusetts in 1990 (Johnson and McMullen, 2000). This has the advantage over an immediate ban in that it gives waste handlers time to adjust to the new situation, namely, whatever time is left in the existing landfills or whatever life is left to the existing incinerators. But eventually, this too becomes a ban on landfilling and incineration, and it then becomes a stimulus not only to recycle but also to export or to dump illegally. In Massachusetts, for example, within the last five years, interstate exports of solid waste have risen from practically none to one fifth of the total MSW generated (NEWMOA, 2000; Repa, 2002). Not surprisingly, as its landfills and

incinerators reached full capacity, Massachusetts was finally forced to lift these bans in 2000 (Goldstein, 2000; Daley, 2000).

4. Households may be required to recycle. Mandatory recycling, without any financial incentive to recycle, may “work” because people are basically law-abiding, but it works against the self-interest of most households.⁴² Mandating recycling means that most people recycle because they are told to, not because they are better off if they recycle and hence want to recycle.⁴³ An MTC would be the equivalent of a fine for not recycling, much more effective than largely unenforced recycling mandates.
5. The sale of products that are not recyclable may be forbidden. Several communities, including Berkeley (California) and Portland (Oregon), have banned the use of polystyrene foam in fast-food outlets, despite the fact that substitutes are not only more expensive but also little recycled (Eckhardt, 1998). In 1989, Minneapolis (Minnesota) banned the sale of products in packaging that could not be locally recycled – not surprisingly, it proved impossible to enforce.⁴⁴ Massachusetts and Oregon nearly passed referendums requiring all packaging be at least 50% recycled. Laws like these ignore, or discount to zero, the fact that non-recyclable packaging may serve an important other purpose. For example, shrink-wrap and other plastics containing medical supplies or foodstuffs usually cannot (now) be recycled, but they may be necessary for the sterility or security of the product.⁴⁵

In each of these supply-side approaches being practiced in the United States, recycling is encouraged by discouraging some substitute activity. Rarely has there been any effort to *directly* force manufacturers to stimulate or undertake recycling of their products and packages. But discussion is growing of this possibility, called producer take-back responsibility.

10. Producer Take-Back Responsibility

Producer take-back responsibility means that manufacturers are required to assume responsibility for recycling (or properly disposing of) any product or package that they put into the marketplace. In strictest terms, this means that they must *physically* take it back from the consumer and dispose of it themselves – or reuse it or recycle it – at their own expense or hire someone else to do that. As Reid Lifset has noted, such take-back responsibility changes the transaction between the producer and the consumer from a purchase to, in essence, a lease (Lifset, 1998). Producer take-back is

42. The word “most” in this and the next sentence is necessary because, for many households, the psychic benefits of recycling are sufficiently great that no price is needed.

43. Why, one might wonder, would any city force recycling on those who do not want to do it? The answer lies in volume. Mandatory recycling achieves larger volume and at very little extra *money* cost – the average money cost per ton of collecting recyclables comes down a lot, according to one study (converted to year 2002 dollars), from around \$160 with 25% of the households participating to \$130 with 75% participating (Miller, 1995). Sorting costs at the materials recovery facility, however, rise since reluctant recyclers do not prepare their curbside trash and recyclables with as much care. The psychic costs to unwilling participants or nervous lawbreakers also should be counted. In one survey, households preferred MTCs to mandatory recycling by a three-to-one margin (Fullerton and Kinnaman, 1996).

44. Cf. <http://www.mtas.utk.edu/public/ORDINANC.NSF/0/6dd8d1c19792c1f685256413007de99e?OpenDocument>.

45. The correlation between the amount of packaging and amount of food waste is highly *negative* (Alter, 1989). The World Health Organization (WHO) maintains that in the developing countries of the world, 30-50% of all food decays before reaching consumers, while in the United States and Western Europe, only 1-3% of food decays. Better food packaging (and refrigeration) has probably contributed importantly to the dramatic decline in the incidence of stomach cancer in the United States over the past century (Nomura, 1996).

intended to make producers consider the cost of disposal or recycling when they decide how much packaging and what kind of packaging to use.

Producer take-back responsibility certainly does encourage recycling - it is like a mandatory deposit system without the deposit. Without such take-back responsibility, the producer only tries to recapture and recycle the used-up product if its value in recycling exceeds the cost of recapturing and recycling it. With such responsibility, the producer must recapture it in any case, and hence will recycle it if the value in recycling exceeds the cost of recycling it *minus* the cost of disposing of it. That gives two additional reasons for recycling the recaptured product. One, with take-back responsibility, the cost of recapturing it has become a sunk cost; and two, recycling makes it unnecessary to pay for disposing of the product. More used-up products would get recycled.

The problem is that it is a very expensive way to do it. It is expensive because it foregoes the advantages of economies of scale in waste collection. The cheapest way to collect waste is to have a single truck pass by each house and pick up all the household's waste as it passes. Or with curbside recycling, the cheap way is to have two trucks pass each house, one to pick up all waste and another to pick up all recyclables. Even having two trucks pass by, one to pick up for disposal and another to pick up for recycling raises the cost a great deal. But take-back responsibility increases the costs even more because, in theory, one truck passes to pick up the empty Campbell soup cans, another to pick up the empty Crest toothpaste tubes, yet another to pick up the empty Reebok shoe boxes, and on and on. Of course in reality, each manufacturer would not actually pick up or take back each discarded package. But that is beside the point - the law requires that *someone* outside the municipal solid waste system make the special effort to collect all packages and take them back to the original producers or to proper recycling or disposal sites.

In fact, where take-back responsibility has been implemented, producers quickly realize the enormous expense of collecting their own packaging, and they band together to pick up collectively. This reduces the cost, but not very much compared to a well-run municipal solid waste collection system because the manufacturers' collection system at best no more than supplements an already operative municipal system and effectively duplicates it for no good reason.⁴⁶

Furthermore, unlike a mandatory deposit system, take-back responsibility provides no incentive for households to participate in the return of packaging. Indeed, if it requires extra effort by the household, there is an incentive not to participate. For some products, special disposal care is important and for these, take-back responsibility may be appropriate, but household incentives must also be attended to.

In short, the idea of producer take-back responsibility is correct - to make manufacturers consider the disposal costs of their packaging when they design their packages - but the requirement that they actually physically take back the package leads to enormous and unnecessary costs. To see how producer take-back responsibility works in practice, let us examine briefly the actual operation of one such program, the German "Green Dot" system.

In 1991, Germany made the manufacturer of each product responsible for the collection and recycling of the packaging of its product. In principle, the consumer was to return each package to the retailer, who would return it to the manufacturer, who would recycle it (Shea and Struve, 1992; Fishbein, 1994; Scarlett, 1994; Reynolds, 1995; OECD, 1998). The potential cost of many small and independent recycling systems was so staggering that German manufacturers quickly came up with an alternative. They formed a company, *Duales System Deutschland* (DSD), to collect all the packaging waste of its members and arrange for its recycling. The member firms paid a fee on each

46. The manufacturers' collection cannot replace the municipal collection since there is much waste for which no single producer can be made responsible.

package they put into the economy and then placed a “green dot” on the package to indicate to consumers that the package could be returned to any of the widely distributed yellow DSD collection bins.

Effectively, the Green Dot introduced a new collection system to the already existing municipal waste collection and recycling systems. This duplication has been, by general agreement, very costly. The fee levied on participating manufacturers was initially based on their (self-reported, and often under-reported) volume (Louis, 1993). The original fees ignored any differences in the weight per volume and in the recyclability of the material, and hence gave no incentive to the manufacturers to produce lighter or more easily recycled packaging, but in 1993, fees began to vary with recyclability, ranging from \$0.09/kg of glass to \$1.76/kg of plastic (OECD, 1998).

The program succeeded in greatly reducing the amount of packaging and packaging waste; packaging volumes had been increasing at 9% per annum in the years immediately preceding the introduction of the Green Dot, but actually declined (by 7%) over the decade after the Green Dot appeared (Quoden, 2002). Moreover, packaging shifted significantly from plastic to glass (Schmid, 2001). However, the program also shifted a lot of waste and recyclables from one set of bins to another. And the new bins were much more expensive ways of collecting recyclable materials than the traditional municipal programs - some estimates say more than twice as expensive per ton of recyclables (Brisson, 1993; Fishbein, 1994). One study found that the Green Dot collection system cost nearly \$500 per ton of recyclables (Boerner and Chilton, 1994). Another found that the total cost per ton of collecting and recycling material through the DSD system was over \$400, “effectively approach[ing] the costs of handling a tonne of hazardous waste” (OECD, 1998, p. 33).

In those municipalities where households paid a marginal trash collection charge (MTC) for their solid waste curbside pickup, they tended to use the DSD bins for all their waste – the German government found that nearly half of the contents of the DSD bins was not packaging waste and much of it was not even recyclable (Gehring, 1993; Jaeckel, 1998). And those firms that contracted to collect the material in the DSD yellow bins had no incentive to monitor the use (or abuse) of the bins because they were being paid by the ton. Finally, the firms to which DSD sells the collected material (or pays to have it taken) must guarantee to recycle it, but there is evidence of corner-cutting here, ranging from warehousing it, to sending it surreptitiously to landfills, or to exporting it.

The German “Green Dot” program is the ultimate supply-side policy. In principle, every producer is required to physically take back and recycle every product and package it produces, without regard for the cost of collection or the value of the recycled material. In short, supply-side policies *do* stimulate recycling, but they fail to consider cost, and they rarely stop to ask, what *should* be recycled. I once heard the German solid waste policy described as “trial and horror.”

Producer take-back responsibility is not the only grandiose supply-side policy. Government regulation of packaging so as to increase its recyclability has also been suggested.

11. Regulating Packaging

One of the purposes of an advance disposal fee (ADF) is to encourage manufacturers to utilize packaging materials that are smaller, lighter and more readily recyclable. We could achieve this more directly by using a command-and-control mechanism, regulating the size, shape, and composition of packaging. No state is doing this yet in the United States, but some are getting close. California, for example requires plastic packaging manufacturers (of other than food, beverages, and cosmetics) to use at least 25% recycled content, show at least a 10% reduction in container weight, or make reusable containers (*Packaging World*, 1998). Should we seriously consider such quantitative regulation of packaging?

One answer is administrative complexity. It is not too difficult to determine the ADFs to be paid by various kinds of packaging, but it would require a huge bureaucracy to specify to each manufacturer what kind and amount of packaging should be used. But there are theoretical reasons as well for preferring an ADF.

To simplify, let us assume that all packaging consists of a single unrecyclable material, each pound of which costs c to collect and dispose of. There are two shippers who use this packaging, one of marbles and one of eggs. Each can reduce the amount of packaging used, but the egg shipper can reduce it only at high cost - e.g., customer dissatisfaction, messy cleanups, and the need to replace damaged product. The marble shipper can easily reduce the packaging. To regulate by quantity, we would need two special studies, one of egg shipping and one of marble shipping, to determine the optimal quantity of packaging for each. In each study, we would need to discover the marginal cost of packaging reduction, equate that to the marginal benefit of packaging reduction - i.e., the avoided marginal cost of collecting and disposing of packaging (c) - and promulgate a regulation requiring that optimal volume of packaging.

An ADF of c per pound of packaging achieves the same result without the special studies of eggs and marbles. Each shipper now considers c to be a part of the marginal private cost of packaging. The marble shipper will increase profit by cutting back on packaging a lot, while the egg shipper will find it profitable to pay a lot of ADF rather than ship broken eggs to customers.⁴⁷ The shippers themselves figure out what is their optimal amount of packaging!

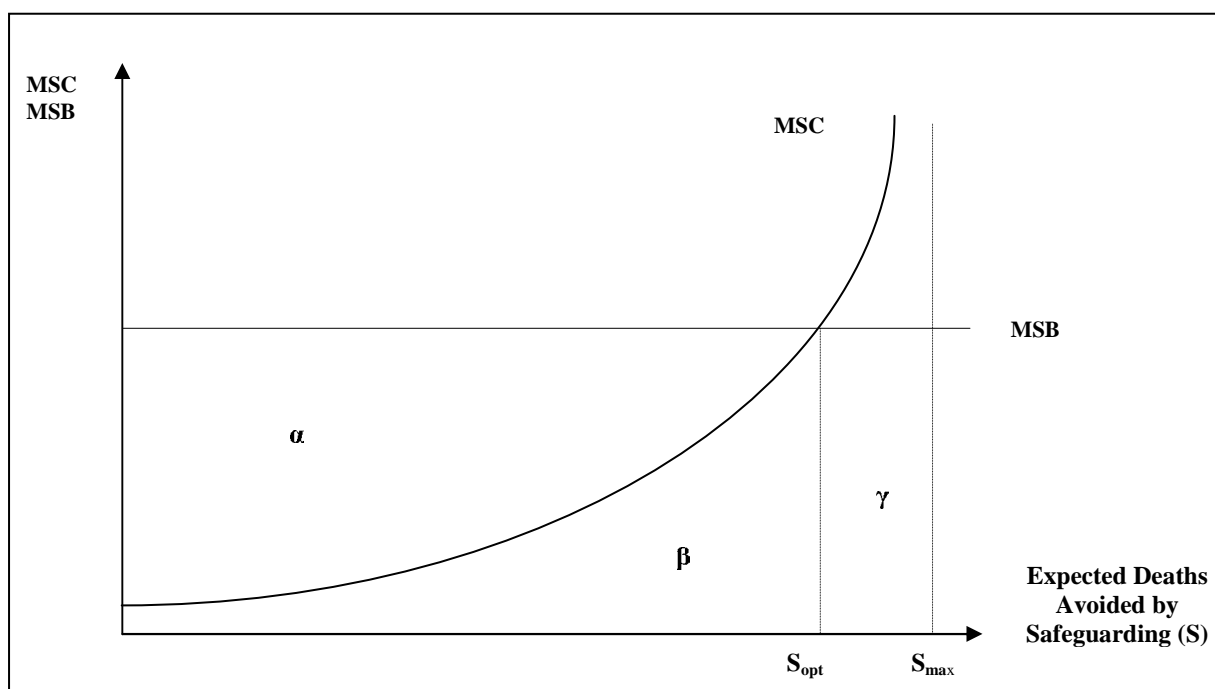
The real world of course is much more complex than that. But the conceptual difference between price and quantity regulation of packaging applies just as well - indeed, better since the real world consists of thousands of products and packages, each of which would require special study before optimal quantitative regulations could be issued. General George Patton once said, "Never tell people *how* to do things - tell them *what* to do and they will surprise you with their ingenuity" (Patton, 1947, p. 357). Advance disposal fees essentially follow Patton's advice. Tell manufacturers how much they can save by not using a pound of packaging - they can save the ADF on each pound not generated - and let them surprise us with their ingenious ways of optimally reducing packaging - while they think they are just avoiding taxes.

12. Restricting Solid Waste Trade

Another favorite way of targeting solid waste policies is to limit interregional or international trade. This is surprising - to an economist, at least - because the welfare gains from free trade are so well known, in theory since David Ricardo worked it out nearly two centuries ago and in fact since the developing countries began providing innumerable real-world experiments a half century ago. If the real resource costs of disposing of waste are higher in Region A than in Region B (and higher by more than the transport costs between them), then the total disposal costs will be lower if the waste of Region A is sent to Region B for disposal. If Region A pays a price lower than its own MSC of disposal but higher than Region B's MSC of disposal, both regions will be made better off by the trade. If there are no market failures such as monopolies or uncorrected external costs in the disposal of waste, there is no reason for opposition to such trade.

47. The ADF will end up affecting the relative prices of marbles and eggs. The relative price of marbles will fall, as it should since marbles will be shipped with little socially costly packaging. The relative price of eggs will rise, as it should since eggs are packaging-intensive and the subsidy to packaging disposal has now been removed. For a fuller version of the theory of price and non-price policies in a world of uncertain marginal costs and marginal benefits of pollution abatement, see Baumol and Oates, 1988, Chapter 5.

Figure 10. Optimal Safeguarding in Waste Disposal



In fact, however, market failures do accompany waste disposal, and these offer justification to resistance to imports of waste. The market failure that chiefly affects attitudes toward waste imports involves the safeguarding of the waste disposal (whether in landfill or incinerator). Consider Figure 10, which shows the marginal social cost (MSC) and marginal social benefit (MSB) of different degrees of safeguarding. To keep it simple, think of a unit of safeguarding as one expected cancer death prevented through groundwater protection in landfills.⁴⁸ The MSB of safeguarding is a constant, and that constant is the amount society is prepared to spend in different life-saving policies to prevent one expected death.⁴⁹ The MSC of ever greater safeguarding is ever more costly, as the low-cost safeguards are undertaken first. S_{\max} is the number of expected deaths that will occur if no safeguarding is undertaken. Perfect and complete safeguarding would prevent all of these S_{\max} deaths but would be very expensive - perhaps impossible. The optimal safeguarding process is to undertake all those safeguarding measures for which MSB exceeds MSC. The optimal safeguarding is therefore S_{opt} .

Suppose, again for simplicity, that this landfill imports all its waste from other regions, and that the landfill owner will earn revenue of R (net of all landfill costs other than safeguarding). Does any $R > 0$ mean that this region should happily accept the waste? Certainly not.

First, consider the worst case, where no safeguarding is undertaken (*i.e.*, $S = 0$). In this case, the dollar value of the expected deaths will be the sum of the three areas, $(\alpha + \beta + \gamma)$. If we consider the region as a single entity, then the waste should be accepted only if $R > (\alpha + \beta + \gamma)$. But, if R goes to the

48. In a full empirical analysis of optimal safeguarding, we would want to also count many other things, such as illnesses prevented and environmental damage prevented.

49. This figure is often called “the value of life” though it has nothing to do with the value of anybody’s life. It simply recognizes that we do not have sufficient resources (or know-how) to prevent all deaths and hence that we should try to maximize the number of expected deaths prevented (with a limited budget). This requires setting a limit on how much society should spend to avoid one *expected* death. For a fuller discussion of this, see Porter, 2002, Chapter 1, Appendix B.

landfill owner, possibly not even a resident of the region, and the cancer deaths are suffered by residents of the region, then the net benefit to the residents of the region may be as low as *minus* $(\alpha + \beta + \gamma)$.

Suppose now that it is possible to force the landfill to safeguard optimally (at S_{opt}). Now the profit of the landfill operator is $(R - \beta)$ since the operator must incur the costs of this optimal safeguarding. The operator will be anxious to accept the waste if $R > \beta$, but for the region as a whole, the waste should only be accepted if $R > \beta + \gamma$. And again, if the landfill operator is not a local resident, the net benefit to the residents is still negative, *minus* γ .

In short, it is possible that the residents of a region are made worse off by market-driven import of waste into the region.

Where a landfill serves local waste needs *and* imports waste, there are even more reasons why the local residents might want to treat the imported waste less favorably than their own local waste. Consider three:

1. The municipal landfill may have been deliberately underpriced (*i.e.*, $P < MSC$) in order to discourage illegal dumping by locals. In this case, importing waste at this same subsidized price is foolish from the local viewpoint • though it also may discourage illegal disposal, it does so elsewhere and therefore provides no local benefit. Banning waste imports, or at least charging such imports the full MSC, may be good policy *from a strictly local viewpoint* (Copeland, 1991).
2. Even where the municipal landfill sets the price of local waste correctly (*i.e.*, $P = MSC$), if the landfill has any monopoly power at all in its waste import market, *local* welfare is enhanced by charging a higher price for imported waste (*i.e.*, the price where the *marginal revenue* from imported waste equals MSC). Charging non-residents a higher price for services makes sense from the residents' viewpoint. Indeed, charging a higher price for out-of-staters has a long and unquestioned history in the United States for services provided by the states • such as parks, hunting and fishing licenses, and university education • and extending this to imported waste is sensible for the city or state, although inefficient from the national viewpoint.⁵⁰
3. If the local landfill is privately owned and waste is permitted to be imported from other regions, either the landfill price will be higher or the landfill will fill up and close sooner, requiring that local trash be sent to a more distant and more expensive landfill. Even if the local citizenry do not fear health damage from imported waste, they may fear monetary damage (Ley *et al.*, 2002).

There are probably even more reasons why people might want to ban - or at least charge a higher price for - waste that is imported from other regions. However, in the United States, the Supreme Court has almost always found such bans or price discriminations to be violations of the Interstate Commerce Clause of the U.S. Constitution (U.S. EPA, 1995b). The Court's decision is consistent with a national view of welfare - if waste is unwanted locally, then local and imported waste should be equally unwelcome (and hence equally priced). Local discomfort with imported waste, because of real or wrongly perceived risk, is sufficiently great that there are constant efforts to get around this

50. This argument raises the logical question: Doesn't the landfill probably have even greater monopoly power over local waste? Yes, it probably does, but a correctly priced municipal landfill would not exploit its own citizens by charging a price above MSC.

Commerce Clause deterrent. Some are pernicious, and three currently popular proposals in the United States are particularly pernicious:

1. The U.S. Congress could explicitly permit states to control their own interstate waste trade. The Congress rarely gives this right to states, but it could. The states would then create a jumble of regulations on what kinds of waste and what volumes of waste could pass through or be landfilled in each state. This would dramatically increase the cost of waste disposal, either through big increases in the costs of transporting waste or through the increased landfilling of waste in inappropriate places.
2. On grounds of health or security, which the U.S. Supreme Court recognizes as legitimate causes of interference with interstate trade, the states could limit the ways in which waste may be transported to landfills, carefully eliminating those ways that are preferred for waste traveling long distances. For example, Virginia periodically considers banning barge transport of waste in order to deter the import of New York City waste into Virginia landfills (Timberg, 1999). This could cause New York to choose either a socially more costly landfill elsewhere or a socially more costly transport method (e.g., heavily subsidized and air-polluting semi-trailers on the heavily congested and accident-prone highways between New York and Virginia).⁵¹
3. Specific items could be forbidden in a state's landfills, which means that states that routinely landfill those items must begin to sort them out if they want to use the banning state's landfills. Massachusetts, for example, bans yard waste, batteries, tires, white goods (i.e., large appliances), metal, glass, plastic, and paper from its landfills and incinerators (U.S. EPA, 1999, p. 20). This of course puts an equal burden on the banning state using its own landfills • except in cases where the banning state is already fully recycling the banned item. For example, states with mandatory deposits on beverage containers could ban the landfill disposal of such containers, since they already are nearly 100% recycled. This is not just theory: Iowa has done this, and Michigan is considering it (Truini, 2000). It is a clever way of preventing imports, though from a national viewpoint, it is also not efficient (unless the banned items generate higher external costs).

There is, fortunately, one very sensible way to handle all this. Recall that the basic reason for the NIMBY (i.e., “Not In My Back Yard”) attitude toward waste imports is that residents of the area around the landfill or incinerator will bear uncompensated external costs (even if the operation is optimally safeguarded). The landfill or incinerator should pay a “host fee” to the neighboring community high enough for these residents to feel adequately compensated for the discomfort of living near potential environmental problems. (In the language of Figure 10, the host fee would have to be at least as high as γ , the cost of the remaining expected local deaths despite optimal safeguarding.) This host fee becomes a part of the costs of the disposal operation and becomes incorporated into a single tipping fee for all, both local waste and imported waste.

With host fees, while all waste pays the same tipping fee, not all regions make the same *net* expenditure (per ton of waste) for waste disposal. For the host community of the landfill or incinerator, the net cost is the waste payment minus the host fee. Distant waste producers pay in cash only, while neighbors pay not only in cash but also in potential future health and environmental damages. The host fee recognizes that and thereby reduces the cash part of the payment.

The search for the correct host fee is elusive. Nevertheless, such reimbursements are increasingly being offered. Generous host fees can greatly reduce local opposition to the siting and operation of

51. Per ton of waste transported, barges use one-ninth the fuel and emit one-seventh the air pollution as do large trucks (Timberg, 1999).

incinerators and landfills. Since local opposition means for the facility the loss of time and the expenditure of legal resources, host fees can even *add* to profit (Simon, 1990). While a benefit-cost analysis seeks to find out if the benefits of a facility are great enough that everyone's costs *could* be compensated, we can only be *sure* that the social benefits of a solid waste disposal facility outweigh the social costs if everyone *actually is* compensated. The successful negotiation of a host fee with the neighboring community at least guarantees that a majority of the neighbors are in fact better off.

U.S. states are beginning to formalize these host fee negotiations. In Wisconsin, for example, once a waste facility has been approved, the permitted facility must negotiate with "affected municipalities" about social and economic issues raised by the new or expanded facility. Both operating methods and host fees may be negotiated. Compensation can be based not only on possible declines in property values but also on declines in the quality of life through noise, traffic, and ecological destruction. Payments have taken many forms – one community demanded a new school bus. If the local negotiating committee and the waste facility applicant are unable to reach agreement, the state serves as arbitrator. The two parties each submit a final offer on compensation and operating methods, and the state selects the one it considers the fairer of the two offers. Of the first 195 such negotiations, only five failed to reach agreement and were submitted for arbitration (Bacot *et al.*, 1998; personal communication with Patti Cronin, Executive Director, Wisconsin Waste Facility Siting Board).

13. Conclusion

The handling of solid waste is filled with market failures. These are principally of two types, external costs (where marginal private cost is less than marginal social cost, $MPC < MSC$) and subsidies (where price is less than marginal private cost, $P < MPC$). Efficiency requires that $P = MSC$, but these two market failures ensure that $P < MSC$. Exploring ways to correct (some of) these market failures - and ways *not* to correct them - has been the main business of this paper.

Two missing prices were explored at length. One, manufacturers are usually given no incentive to consider the ultimate disposal cost or recycling cost of their products and packages, and the result is products and packages that are too heavy, too complex, and too difficult to recycle. And two, households are usually given no incentive to consider the disposal cost or recycling cost of the solid waste they generate, and the result is too much waste, too little reuse, and too little recycling.

Various pricing mechanisms were examined in Sections 3-5 for correcting these failings. The one I prefer - Scheme 2 - requires two new charges: 1) an advance disposal fee (ADF) on products and packages equal to their net recycling cost;⁵² and 2) a marginal trash charge (MTC) on households equal to the excess of the collection-and-landfill disposal cost over the average net recycling cost of household solid waste. The ADF encourages manufacturers to make products and packages more cheaply recyclable. The MTC encourages households to reduce, reuse, and recycle.

While logic urges some combination of ADFs and MTCs, practical considerations may make either or both of them less desirable. The ADF should be fairly easy to develop and administer since it is simply a matter of estimating the net recycling costs of various packaging materials and applying them as an excise tax. ADFs on products themselves probably would have little effect on the composition of products and hence may not be worth the administrative cost of developing and

52. Recall that the net recycling cost is the cost of collecting and sorting a recyclable material minus the revenue earned on its sale.

assessing ADFs. But ADFs on packaging could at least stem the recent proliferation of the amounts and types of plastics used in U.S. packaging.⁵³

There are, however, a number of circumstances that would make the application of MTCs less beneficial:

1. If illegal disposal is, or is expected to become, a serious problem, any MTC would probably increase the volume of such costly disposal. Where illegal disposal is expected, therefore, the optimal MTC will be lower, and perhaps even zero (or negative!).⁵⁴
2. If administrative and monitoring costs are high, such costs could more than offset the reuse, reduce, and recycle benefits of an MTC.
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, then it may be appropriate not to initiate an MTC system.
4. If multi-family dwellings dominate the municipal landscape, the dumpsters/skips behind these apartment houses essentially become a “commons” where each individual resident has almost no personal incentive to reduce his or her trash (Hardin, 1968). Then, an MTC on multi-family dwellings would be essentially just a new tax on such dwellings, with little, if any, solid waste effect.
5. If net recycling costs are nearly as high as (or higher than) trash collection and disposal costs, then the appropriate MTC is small (or even negative!), and hence little or no efficiency gain would be achieved by applying an MTC.⁵⁵

One of the things I have tried to stress throughout this paper is that there is no easy formula for deciding what mix of taxes and subsidies is appropriate for waste handling. It is therefore tempting to forget them and rely entirely on non-price policies. Indeed, non-price policies may have quicker, surer impact when recycling is just getting started. But all non-price policies suffer two huge defects.

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. Banning new landfills or banning certain items from landfills are not so much non-price policies as infinite taxes on the banned activity, and neither hidden subsidies nor external costs usually call for an infinite Pigovian tax response. When states mandate recycling targets, it is only the fear of the fine that induces cities to react. Low-interest loans for recycling facilities may be snapped up, but only because cities love low-interest loans, not because it makes them love recycling. And requiring purchasing officers to buy recycled materials will be ferociously resisted unless the recycled materials are priced competitively. There is also no easy formula for choosing non-price policies.

53. Some observers think that this proliferation of packaging plastics is confusing consumers about which plastics are recyclable and is responsible for the recent decline of the U.S. plastics recycling to less than 10% – even the quite homogeneous plastic beverage container has experienced a serious decline in recycling, from 40% in 1995 to 22% in 2001 (U.S. EPA, 2003; Truini, 2002d).

54. I am indebted to Thomas Kinnaman for this observation.

55. This is not just a conceptual possibility. In the United States today, prices earned on the sale of recyclable materials are very low and, except in the northeast part of the country, landfill tipping fees are also very low (partly because so much landfill capacity was created in the 1980s and 1990s in anticipation of landfill scarcity!).

The second defect is that there is no limit to the inefficiency that a badly chosen non-price regulation can cause, while price policies self-limit their damages, no matter how badly chosen. An example may illustrate this. Suppose paper and cardboard are the packaging materials that are cheapest to recycle. With a price policy, they would receive a relatively low ADF, and manufacturers would be greatly encouraged to use paper and cardboard for their packaging. Manufacturers who could easily switch to paper and cardboard would quickly make the switch in order to reduce their ADF costs. But those manufacturers who needed packaging of plastic (or some other material) for the security, safety, or sanitation of their product would not switch and would pay the higher ADF on plastic. A non-price ban on packaging other than paper and plastic could not make this distinction - 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 ADFs and producer take-back responsibility.

If price policies are superior to non-price policies for waste handling externalities, it is curious that they are so little used, while non-price policies abound. I have four suggestions as to why non-price policies are so often preferred. One, they are easier for policymakers to apply - one can ban or require something without the need for a difficult empirical search for marginal benefit and marginal cost and optimal prices or taxes. Two, they are easier for non-economists to comprehend - mandating that recycling increase by x% makes more immediate sense than, say, raising the price of trash disposal by y%. Non-economists understand when you say a policy will reduce waste x% or increase recycling y%. Saying a tax or price will reduce trash *optimally* or increase recycling *optimally* means nothing to non-economists. Three, many waste professionals and policymakers do not believe there is price elasticity in the waste decisions of manufacturers and households, which would mean that changing prices would not change behavior. And four, every non-price policy hides the cost (as well as the benefit) of the policy - no explicit tax or higher price is imposed on anyone. As a result, unfortunately, these non-price policies become acts of faith, or lack of it, and they lead the waste-policy focus to the poles of nothing-discarded-everything-recycled and nothing-recycled-everything-discarded. The optimum, I am certain, is somewhere in-between, and only a greater emphasis on price policies can lead us toward it.

ANNEX

Glossary

ADF	Advance disposal fee (on manufacturers)
DWL	Deadweight loss
MPC	Marginal private cost
MSC	Marginal social cost
MSW	Municipal solid waste
MTC	Marginal trash charge (on households)
P	Price
U.S. EPA	U.S. Environmental Protection Agency
WTP	Willingness to pay

All monetary figures are reported in U.S. dollars. Euros are converted to dollars at the rate of 1 Euro = US\$1.17.

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TARGETING LEAD IN SOLID WASTE

By Hilary Sigman¹

1. Introduction

Exposure to lead may present one of the most prevalent and serious environmental threats to human health. Recent research has shown that lead is toxic even at very low doses; indeed, there may be no blood lead level completely without adverse effects. With levels of lead exposure common in some U.S. cities, children and fetuses are at risk of IQ deficits, impaired reaction time, and other neurological problems (Banks *et al.*, 1997). Studies have found that children exposed to lead have higher school drop-out rates, lower career achievement, and higher proclivity to crime, so the range of social harms and victims from lead pollution is great.

Many countries have made significant efforts to reduce exposure to lead, most importantly phasing out lead additives in gasoline. Other policies target exposures from lead in paint, in food containers, and in drinking water from old lead-bearing pipes. There is also interest in reducing lead in solid waste because of the concern that lead waste may give rise to human and environmental exposure after wastes are incinerated or disposed in landfills. Different policy approaches have been suggested and adopted: this paper provides a comparison of some of the most prevalent and promising of these approaches.

The paper begins with background on the sources of lead in municipal solid waste and its implications. The second section discusses policy options to reduce lead discards. In the third section, I quantify the effects and costs of several incentive policies for battery recycling in the U.S. and suggest how the results might differ if the policies were extended to other uses of lead. The empirical evidence suggests that incentive-based policies can be successful in encouraging recycling and reducing disposal, but there are substantial differences in the cost effectiveness of various incentive policies. The fourth section discusses earlier research that raises questions about the desirability of any policy to reduce lead in solid waste in the U.S. A final section briefly concludes.

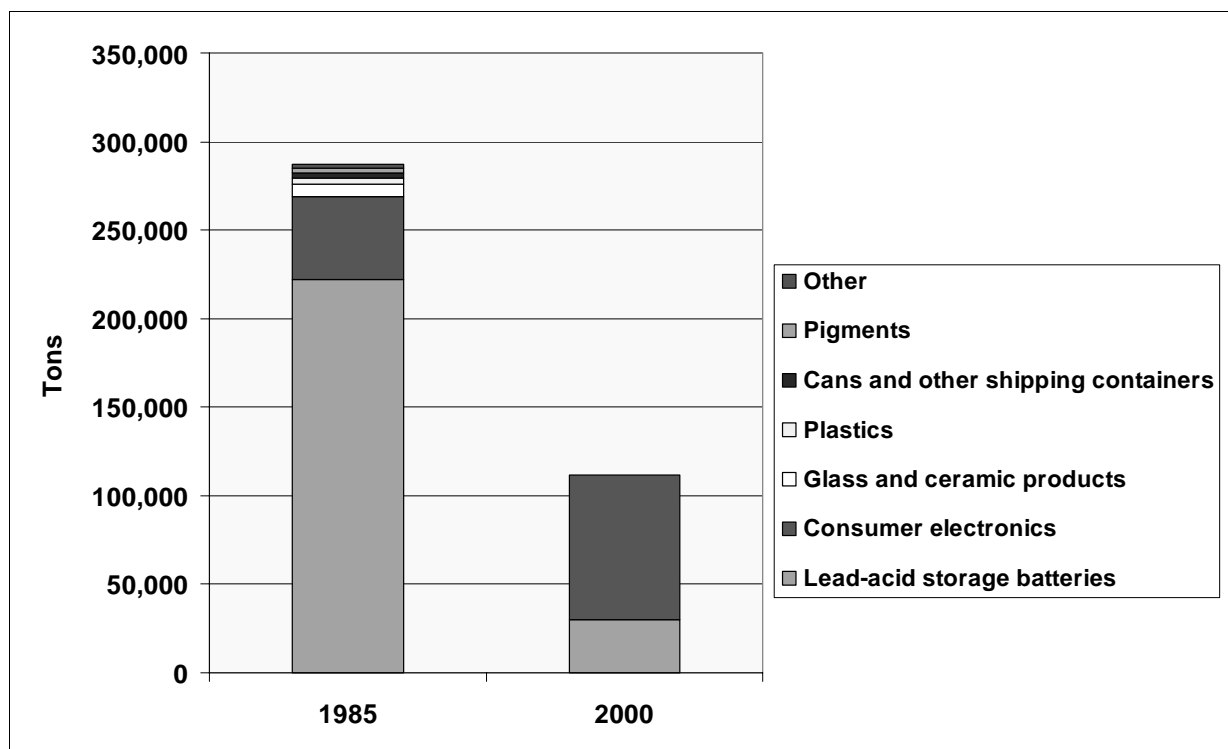
2. Lead in municipal solid waste

Figure 1 presents the available information about lead in discards in the United States in 1985 and 2000 (Franklin Associates, 1989; U.S. Environmental Protection Agency (EPA), 2002).² The figure

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1. Associate Professor at Rutgers University, New Brunswick, New Jersey, USA. I am grateful to Huiying Zhang for research assistance and to Nils Axel Braathen and Richard Porter for helpful comments.
 2. There is considerable uncertainty about these figures. They are estimated based on sales figures from an earlier period, with the lag being determined by an average of the life-span of the relevant products. They assume that all of these defunct products enter municipal solid waste, when many households end up storing rather than disposing these products. A 1990 survey found that 20% of U.S. households had at least one old car battery stored in their house (EPA, 1991). Even more dramatically, a 1995 study in the U.S. suggests that 75% of defunct cathode ray tubes are in storage (cited by

shows that two sources, lead-acid batteries and consumer electronics, dominated in 1985, the most recent complete accounting. In that year, batteries composed 77% of lead in municipal solid waste and consumer electronics, 16%. A few other sources, including glass, plastics, and cans represented cumulatively about 6% of lead discarded. For a more recent perspective, Figure 1 presents data for batteries and consumer electronics in 2000, but the 2000 values exclude smaller sources so understate the total.

Figure 1. Lead in municipal solid waste in the United States, 1985 and 2000



Sources : Franklin Associates (1989), all sources of lead, 1985; U.S. EPA (2002), batteries and consumer electronics only, 2000.

Figure 1 suggests a dramatic decline in lead in municipal solid waste during this 15 year period. The source of this decline is an increase in the recovery of lead from batteries. As a result, batteries have shrunk from 77% of lead discarded to less than 30%. At the same time, there has been an increase in lead disposed in consumer electronics, in particular the lead in cathode ray tubes (CRTs) from televisions and computer monitors. Although these represented only 16% of lead in 1985, they now far surpass batteries as a source of lead. These figures are for the U.S. which recycles more batteries and uses both more storage batteries and electronics than most other countries; however, the relative change over time is probably similar in other developed countries. Thus, any policy that aims to reduce lead in municipal solid waste should focus on these two sources, which the next two subsections discuss in more detail.

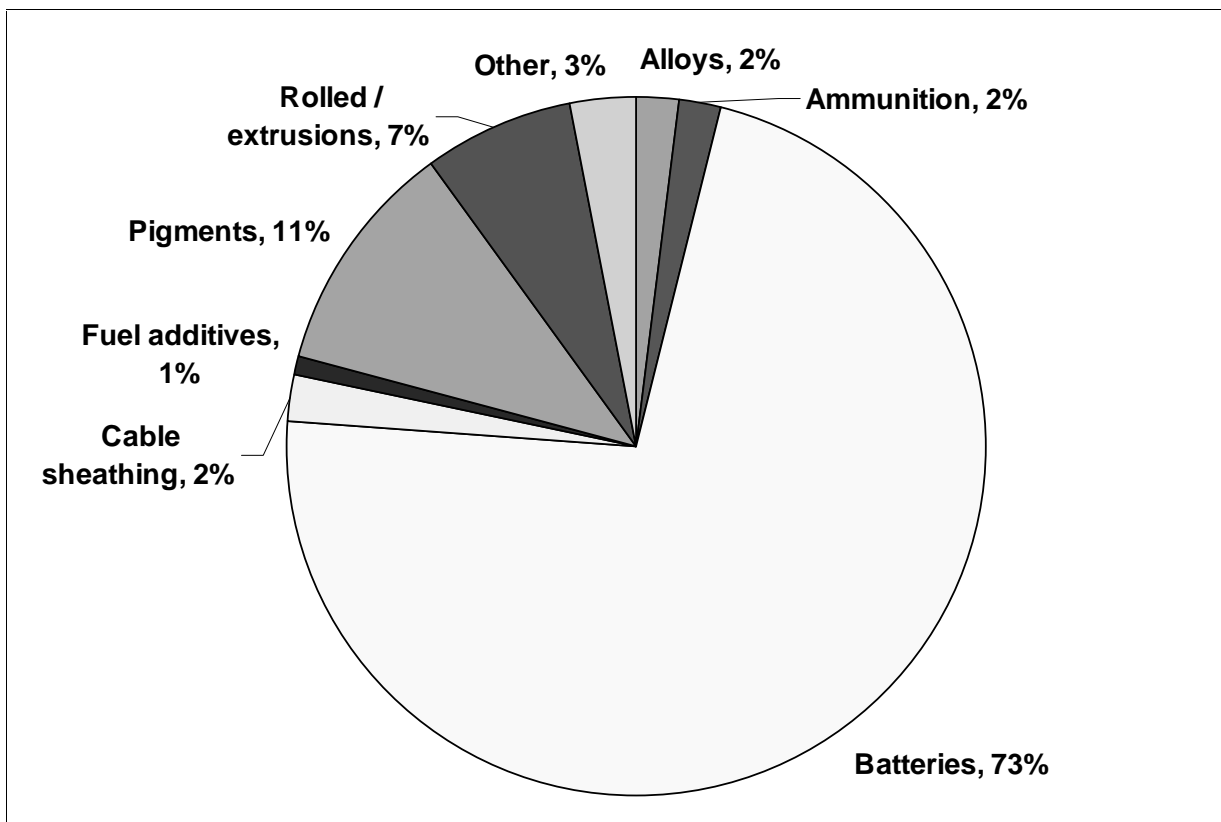
Macauley *et al.* 2001). Similarly, in Australia, 69% of obsolete computers were stored in 2001 (Meinhardt Infrastructure and Environment Group, 2001). Although these stored products will probably ultimately be disposed, they suggest the difficulty in evaluating lead discards in any one year.

2.1 Lead-acid batteries

Lead's primary use is for lead-acid batteries. Figure 2 shows use in 1997 of lead in countries that are members of the International Lead and Zinc Study Group (ILZSG), which account for about 80% of global lead consumption. As the figure shows, 73% of lead is used for batteries. In the U.S., batteries are even more predominant, consuming 1.4 million metric tons in 2001, 87% of the total consumption of lead. Most lead-acid batteries are used as starting-lighting-ignition batteries for motor vehicles (78% in 1992). The remaining batteries are used for motive power in electric vehicles, such as in-plant forklifts, and other industrial uses, such as uninterruptable power supply for large computer systems and standby power supply for emergency lighting.

Demand for batteries will probably continue to grow for a few reasons. First, increased vehicle use will raise demand. Second, battery lives are declining because of higher vehicle underhood temperatures, greater use of vehicles in warmer climates where their life-span is shorter (averaging 40 months in the U.S. South relative to 58 months in the North), and more electrical gadgets in cars (*Purchasing*, 1997). Finally, storage batteries are seeing expanded use in electric and hybrid vehicles and as backup power for computer and electrical systems.

Figure 2. International uses of lead, 1997



Source: Thorton et al. (2001), from ILZSG data.

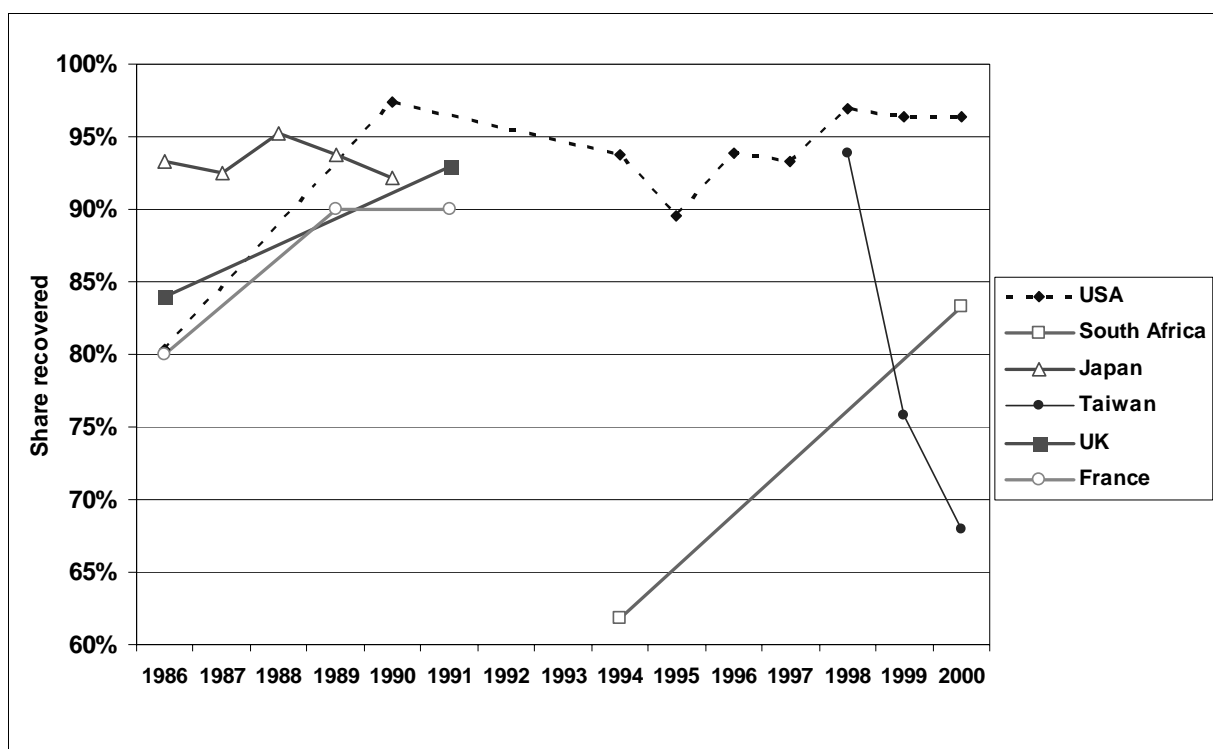
Although batteries represent the dominant use of lead, their contribution to lead in municipal solid waste is not as great because most batteries are recycled. Secondary lead accounted for 79% of refined lead production in the U.S. and possibly as much as 43% of production internationally in 2001 (Smith, 2003). Used batteries constitute most recovered scrap, 92% in the U.S. in 1998 (Smith, 2002). The remaining recovered lead includes 5% “new scrap” — scrap recovered from facilities that use lead

as an input — and about 3% old scrap lead from metal sources such as casting, sheets, solder and fabricated metal products.

In the U.S., retail battery dealers collect used batteries from consumers and typically discount purchases of new batteries in exchange. Discounts have varied considerably: in general, they have been in the range of \$4 to \$7 (BCI, 2003). In the early 1980s when refined lead prices were especially low, however, some dealers charged a fee of \$.50 for the disposal of batteries (Putnam, Hayes and Bartlett, 1987). Scrap battery dealers purchase used batteries and sell them either to battery “breakers,” who remove the plastic cases and drain battery acid, or directly to the secondary smelters who break the batteries and then re-refine the lead. Secondary lead from battery scrap is often used to make batteries (some battery manufacturers operate secondary smelters) and is a very close substitute for primary lead in other uses.

This recycling chain usually captures a large fraction of the used storage batteries. Figure 3 shows estimated recycling rates for batteries in a few countries for which recent estimates are available. These rates cannot be measured directly because the number of defunct batteries in a given year is unknown; it is estimated based on sales from a period of a few years prior, using a typical failure interval for different types of batteries. The rates in Figure 3 are the share of batteries recycled, which includes not only the lead but also a substantial volume of plastic casings; however, the share of battery lead recovered is similar in magnitude and trend.

Figure 3. Lead-acid battery recycling rates in selected countries, 1986–2000



Sources: Japan, UK, France, OECD (1993); USA, U.S. EPA (2002); South Africa, Joseph and Verwey (2001); Taiwan, Lee (2003).

Although the data are somewhat patchy, Figure 3 suggests a few conclusions.³ First, battery recycling rates are high in high income countries. By the early 1990s, rates in the UK, Japan, and the

3. Some of the volatility in the data, especially for Taiwan, may be noise rather than rapid shifts in actual conditions.

U.S. exceeded 90%. Second, rates have generally risen over time. Public policies, discussed below, may have partly driven the increase in recycling rates.

Another source of information about battery recycling for the U.S. is a survey of generators of small quantities of hazardous waste in 1983 (Abt Associates, 1985). In that survey, the generators of used batteries (primarily vehicle repair shops) sent 89% of batteries they collected to off-site recyclers, with the remainder mostly sent to solid waste landfills. This percentage is probably higher than total recovery because some households change and dispose their own batteries. Thus, the survey points to somewhat lower recycling rates than the mass-balance calculations above.

There is substantial trade in scrap batteries. The US exported 152,000 tons of scrap battery lead (either in drained or undrained batteries) and imported 27,000 tons per year on average from 1997 through 2001 (BCI, 2003b). The export figure represents about 14% of the battery lead recovered domestically.

2.2 Consumer electronics

U.S. EPA (2002) estimates that consumer video products contain 7% lead by weight and information products contain about 3% lead by weight. The vast majority of this lead is in CRTs used for televisions and computer monitors, although electronics also contain some lead in solder on circuit boards. Figure 1 suggests that the total lead in municipal waste from consumer electronics has grown dramatically since 1985. This change results from increased use of these products, especially computers, and also changes in their lead composition. In particular, monochrome computer monitors contain substantially less lead than color monitors (Macauley *et al.*, 2001).

In contrast to lead-acid batteries whose role and design are fairly stable, this technology is in flux, so historical data provide less of a guide for public policy. A few technological changes may be especially important in the immediate future. First, flat panel display monitors and televisions do not have the lead content of CRTs. Flat panel displays are still considerably more expensive than traditional technologies, but if their sales rise, current lead disposal in monitors may turn out to be a temporary phenomenon. On the other hand, the advent of high definition television may encourage widespread replacement of older televisions and, at least temporarily, shorten the life-span of these products.

The recycling chain for electronics is more complicated than for batteries because of a range of possible end-of-life fates. Used electronics may be immediately reused (as with some charitable donations), refurbished and resold, demanufactured for spare parts, or sent to facilities that recover raw materials, or experience some combination of these fates. The National Safety Council (NSC, 1999) provides the best data on recycling of electronics in the United States. The report finds a substantial discrepancy between the number of CRTs reported processed by recyclers and the number reported received by glass recovery facilities and secondary smelters. The NSC report hypothesizes the discrepancy arises because firms fail to report all the used CRTs that they export (or sell to brokers for export). Despite this discrepancy, it is clear that recycling rates are low. The NSC report concludes that 1.5 million computer monitor CRTs were recovered in 1998, compared with 15.8 million that became obsolete, for a recovery rate of under 10%. In Australia, recovery rates for computers (not just monitors) are estimated to be even lower, less than 1.5% (Meinhardt Infrastructure and Environment Group, 2001). The difference between the U.S. and Australia may be largely definitional because the Australian data uses a broader product category and inclusion of reused computers in the denominator.

The reason for low recovery of CRTs is the high cost of recycling. In the US, the NSC report indicates that smelters charge \$.10 per pound to accept intact CRTs and \$.07 per pound for crushed CRTs (about \$2 to \$3 per unit). Thus, policies for CRTs must confront a very different environment

than policies for lead-acid batteries. For CRTs, collected scrap goods are not valuable without government intervention.

2.3 Consequences of lead in municipal solid waste

The environmental harms from lead in municipal solid waste depend on whether waste is sent to a landfill or a combustion facility, such as an incinerator or waste-to-energy facility. In a landfill, the concern is that battery cases break and CRT glass becomes pulverized, so that liquids moving through the landfill can become contaminated. Contaminated leachate may reach groundwater when leachate containment systems fail or where wastes are landfilled without such safeguards.

The extent of the danger from landfilled lead is uncertain. Past history does suggest some risk. Of the 158 municipal landfills on the U.S. priority list for contaminated sites, 22% have released lead (ATSDR, 1988). However, these landfills predate modern leachate containment and the decline in lead discards, so they overstate the likelihood of harm from current practices. Macauley *et al.* (2001) cite releases of lead to leachate from .0035 pound to under a billionth of a pound per CRT; the wide range in these values results from uncertainty about whether CRT glass is typically pulverized or just broken in landfills. Given leachate containment and inexpensive alternatives to use of any contaminated groundwater, Macauley *et al.* conclude that land disposal of CRTs imposes few health costs in the U.S. However, the long-term fate of lead in landfills and the success of post-closure care assurances are unknown.

Combustion poses a bigger immediate risk. Both batteries and consumer electronics can potentially be separated from other wastes before the wastes enter a municipal waste combustion facility. However, many facilities do not have suitable sorting areas before materials are placed into furnaces; even those that do may require visual inspection of the waste, which is a leaky process. Where combustion is used, therefore, it is likely that a substantial share of lead discarded goes into the incinerator along with other less hazardous waste.

When lead enters combustion facilities, it elevates the toxicity of residual ash and, more harmfully, may be emitted to the air. According to estimates by Pacnya and Pacnya (2001), global air emissions of lead from waste disposal were 821 metric tons per year in the mid-1990s. This is a tiny share of total air emissions of lead because the vast emissions of lead fuel additives. However, excluding fuel additives, municipal waste disposal accounts for about 4% of global lead emissions to air in their assessment.

Nonetheless, Macauley *et al.* (2001) find fairly low costs from combustion of computer monitor CRTs in the United States. Using air emission rates of .00026 pounds per CRT incinerated and assuming that the share of consumer's monitor CRTs incinerated equals the national average share of all municipal solid wastes incinerated, they conclude that the health costs have a dollar value of \$2.67 million annually. This value may be an overestimate because it gives no role to sorting at the facility. On the other hand, it captures only costs from computer monitor CRTs and reflects U.S. air pollution controls, so harms in less stringently regulated countries may be greater.

Setting aside the health costs of disposal, households that dispose lead may impose costs on other households by raising the average cost of safe waste management. Lead discards make municipal waste more hazardous. In response to this more hazardous waste, authorities may have to choose more costly waste management designs and locations. Thus, it might reduce costs to discourage discards of such hazardous products, even if environmental standards are always high enough to ensure that lead discards do not harm anyone's health.

3. Analysis of public policies

In response to these costs and risks, several public policies have been proposed and implemented to reduce lead disposal.⁴ Two general approaches can be taken, one more direct than the other.

3.1 *Direct approaches*

Direct policies place restrictions on lead disposal. The most common direct response is a ban. In the U.S., 42 states have banned lead-acid battery disposal by households (BCI, 2003). One state, Massachusetts, banned disposal of CRTs in 2000. Similarly, large business users of batteries and CRTs may fall under hazardous waste regulations, which require that they dispose the products in special hazardous waste landfills.

Another direct approach has the government charge fees for disposal of lead-containing products that reflect its environmental costs. Unlike bans, this price-based policy would allow some battery disposal to continue if the cost of alternatives is higher than the fee. Such fees have not seen widespread use.

These direct approaches may impose high costs if policy enforcement is imperfect. If users dump wastes surreptitiously because the policy precludes legal disposal, a ban may increase environmental damages. There are no estimates of how much dumping occurs in response to current restrictions on auto batteries. However, I studied reported incidents of dumping of used oil — another waste largely handled by auto repair shops — in the United States from 1987 through 1994 (Sigman, 1998). States that banned disposal of used oil experienced a 28% increase in the number of dumping incidents, suggesting a substantial adverse effect of disposal bans.

3.2 *Indirect approaches*

More indirect policy approaches may reduce disposal without these adverse consequences. These approaches tend to discourage use of the lead in products and promote recycling. Although there are a large number of possible policies, I will consider four incentive approaches in particular: (i) deposit-refunds; (ii) taxes on lead; (iii) subsidies for recycled lead and (iv) recycled content standards. For clarity, this section ignores non-recyclable uses of lead; the implications of these uses are discussed later.

Deposit-refund. Deposit-refunds are second to disposal bans as the most common type of public policy in place to curb lead disposal. Nine states place deposit-refunds on vehicle batteries, at a rate of \$5 or \$10 per battery (BCI, 2003).⁵ Several countries, including Mexico, Denmark, and South Africa, also have deposit-refunds on car or lead-acid batteries (OECD/EEA, 2003; Johnson and Verwey, 2001).

The deposit-refund requires a fee for the initial use of the lead that is rebated when the lead is recycled (Bohm, 1981). It raises the cost of lead to consumers who do not recycle and thus end up

4. For economic comparisons of alternative waste management policies, see Dinan (1993), Fullerton and Kinnaman (1995), Palmer and Walls (1997) and Porter (2002).

5. The states have mandatory deposit-refunds are: Arizona, \$5 per battery; Arkansas, \$10; Connecticut \$5; Idaho, \$5; Maine, \$10; Minnesota, \$5; New York, \$5; South Carolina, \$5; and Washington, \$5. Wisconsin and Rhode Island have voluntary charges. All these policies were implemented by 1989 and 1991.

paying the deposit without receiving the refund. It also raises the value of recycling used batteries. Thus, it provides incentives for both the reduction of lead use and for recycling.⁶

A deposit-refund may be imposed at different levels. The most common approach — placing the deposit and refund on the consumer — may help make the public aware of battery recycling, but has significant drawbacks. A better deposit-refund would be imposed at the producer level, with a charge for lead use in production and subsidy for recovered lead.⁷ Assuming well-functioning markets, the deposit would be passed forward to consumers, in the form of higher prices for goods containing lead. In addition, because secondary lead producers receive a subsidy, scrap lead has more value. Some of this value will be passed to consumer; consumers who return batteries may get higher prices for their batteries (sometimes seen as a discount on a new battery) or more convenient collection.

This producer-based deposit-refund has a number of benefits. First, it is likely to lower the administrative costs of the deposit-refund because the government does not need to assure that all retail transactions have the deposit and refund properly administered. Second, it provides greater incentives to assure that lead scrap collected from consumers is recovered because the refund is not payable otherwise. A consumer-level deposit-refund may result only in the collection of batteries and their subsequent disposal by retail stores or collection centers — with some benefit in perhaps getting them disposed as hazardous waste rather than municipal waste — but without the full benefit of reducing lead disposal.

Taxes on lead. An alternative policy is to tax lead. One approach applies the tax to all lead. By raising the cost of lead, such a tax may discourage its use, but would not have a direct effect on recycling. Although in principle such a tax could apply to all refined lead or to all products containing lead, in practice most of these taxes are restricted to batteries. Four U.S. states and several European countries have charges on batteries, ranging in the U.S. between \$1 and \$3 per battery (BCI, 2003; OECD/EEA, 2003). Most of these charges are fixed per battery and thus do not provide incentives for use of lower lead batteries, unlike a pure lead tax, which would vary with the lead content of the product.⁸

Another tax approach — a virgin material tax — applies only to primary lead. The U.S. has considered such a tax (*Environment Reporter*, 1991). A virgin material tax raises the price of primary lead to its users. Because primary and secondary lead are very close substitutes, users will purchase recycled lead at nearly the same price, implying that the tax will also raise the price of recycled lead. As a result, the virgin material tax is similar to a deposit-refund in its effect. Under both programs, recycling consumers receive a rebate. The rebate is the direct refund with a deposit-refund; with a virgin materials tax, the tax increment is returned in the form of higher prices for lead scrap.

6. Not only does a deposit-refund not create incentives for illegal disposal, it may even encourage the collection of illegal dumped and stored lead products because consumer can collect refunds by returning products. Collection of the illegally dumped products creates substantial environmental benefits. Collection of stored products, especially old computers, probably does not have large environmental benefits and may add to the cost of program by requiring handling of products that would not face environmental release for some time.

7. See Fullerton and Wolverton (1999) for their discussion of “two-part” instruments in environmental policy, with a producer-level deposit-refund as an example.

8. Texas does distinguish high and low voltage batteries, which will reflect their lead content. Below, I discuss whether the difference between fixed charges per battery and charges based on the lead content of the battery will make much difference empirically.

Subsidy to recycled lead. An alternative program would provide a direct subsidy to recycled lead production. This direct approach does not seem to have been used in practice. However, there are indirect subsidies to recycled lead. For example, many U.S. states provide investment tax credits (which offset income taxes) to firms that produce recycled goods (U.S. Office of Technology Assessment, 1989). Such indirect subsidies may have similar effects to a direct subsidy in encouraging lead recovery; however, they will also have additional effects (not analyzed here) on the recyclers' production decisions that raise the costs of these policies relative to direct subsidies. Thus, the paper examines direct subsidies as a lower bound on the costs of such indirect subsidies.

The effect of a subsidy is to lower the costs of recycled lead relative to virgin lead. This should have the effect of reducing virgin lead production. However, the price of lead to its users declines because recycled lead has become cheaper. Thus, although the recycling subsidy may encourage recycling, it also creates an incentive for *increased* consumption of the lead.

Recycled content standard. A recycled content standard stipulates a ratio of recycled lead to total lead used. In the United States, Congress has considered a recycled content standard for batteries before recent increases in the recovery rate. Early versions of the EU's Directive on Waste Electrical and Electronic Equipment (WEEE) contained a recycled content standard that was dropped by the time of the final version.

A government may implement a recycled content standard on a variety of scales. One approach would require that each individual product has a given recycled lead content, for example 80%. This policy would not give firms flexibility to choose product lines for which recycled lead was most appropriate and would require them to engage in costly monitoring of the use of specific shipments of lead. A more flexible approach is to set the requirements at the firm level, so a firm would have to assure that its products overall contained 80% secondary lead, but some products or batches might have higher or lower recycled content.

An even more flexible approach applies the recycled content standard to all users of lead. The government would set a requirement that 80% of all lead consumed be secondary lead and leave it to firms to assure that this level was obtained. Firms could meet this requirement by trading recycled content: a firm using a larger fraction of recycled material than the standard can trade its surplus with firms using too little, so the standard applies to the industry as a whole. If the permit market operates well, this tradable recycled content standard accomplishes the 80% level (or whatever level is chosen) at least cost.⁹

It is useful for analysis to see that a tradeable recycled content standard is equivalent to a tax-subsidy combination. Suppose that a permit entitles its holder to use one unit of virgin lead. This permit must be traded for enough units of recycled lead that the recycled content for the industry conforms to the standard, call it r^* . This trade-off holds if a permit can be created by the use of an additional $r^*/(1-r^*)$ units of recycled lead. Thus, for example, with the 80% standard, a firm could use 1 ton of virgin lead if it found other firms willing to use 4 tons of recovered lead. This arrangement means there is a cost equal to the price of a permit (whatever that turns out to be) for using virgin lead, like a tax. There is an effective subsidy for using secondary lead, equal to the value of the permits that are created: with a 80% content standard, the subsidy for each ton of secondary lead would be 25% of the price for a permit to use a ton of virgin lead. Thus, it is as though the government collected a virgin material tax equal to the permit price and used all the money it collected to give a recycling subsidy. This tax-subsidy combination is revenue-neutral, unlike the deposit-refund, which raises revenue for the government on any lead that is not recycled.

9. For more discussion of tradeable recycled content standards, see Dinan (1993) for paper and Macauley *et al.* (2001) for CRTs.

A recycled content standard should increase the recycling rate. It also likely makes lead more costly. The reason it raises the lead costs is that firms must expend resources either to buy permits for the use of primary lead or to use secondary lead when primary lead might be cheaper. However, the net effect of the price of lead to its users is smaller (at least for a tradeable recycling content standard) than for a deposit-refund or virgin material tax.¹⁰ As a result, the reduction in lead use will be less than under these other policies.

Producer responsibility requirements. Under producer responsibility, the government sets rules requiring that producers, individually or through proxies, take back products. The government may set varying types of requirements for the nature of the take-back, the share of the products the producer successfully take-back, and the required fate of products collected. The principal example of this sort of policy for lead is the EU's WEEE Directive, which sets take-back and recovery requirements for various white goods and consumer electronics, including CRTs.

Given the complexity of producer responsibility, it deserves more extensive discussion than accorded here. However, it is worth mentioning that the requirements have some features of the tax and subsidy combinations discussed here. Producer responsibility creates large effective subsidies to recycling because of the funds the producers must spend on collection of scrap products and their recovery. To cover the costs of these subsidies, producers will increase the prices of their products to reflect the additional costs that selling a product now entails. In most markets, prices will increase by most or all of the costs of the expected collection and recovery costs. Thus, there is also an effective tax on the purchase of the product. The WEEE Directive allows producers to collect this cost as what the Directive calls a "visible fee" for the first 8 to 10 years. Thereafter, the price premium will continue to exist, despite the elimination of the label. Consequently, despite their very different legal form, producer responsibility requirements will give rise to the same types of social costs as comparably stringent tax and subsidy rules. In particular, without government price controls, there is no way to keep the consumers from bearing a substantial share of their costs.

3.3 Comparison of private costs

Reducing lead in municipal solid waste will require firms and consumers to bear costs and exert effort. Collectively, these costs are the "private costs" of the policy. The costs include the time spent returning products for recycling, resources used for transporting, storing and disassembling scrap goods, and the costs of rerefining lead. They also include costs from lost opportunities as a result of source reduction. These private costs may be worthwhile to avoid the environmental consequences of battery disposal, but one goal of good policy is to make them as low as possible for any given reduction in lead disposal.

Table 1. Summary of effects of indirect incentive policies

Policy	Effect on:	
	Use	Recycling rate
Deposit-refund	↓	↑
Tax on all lead	↓	–
Virgin material tax	↓	↑
Recycling subsidy	↑	↑
Recycled content standard	↓	↑

Note: Bold arrows indicate effects that will typically be larger than other arrows.

10. The reason that the lead price increase is smaller is most easily seen by considering the tax-subsidy equivalent to a tradeable recycled content standard. Under this policy, all of the "revenue" from primary lead use is returned as an effective subsidy to secondary lead use, with the latter pulling down the lead price. For the deposit-refund, the effective subsidy to secondary lead is smaller because the government keeps some revenue, so its downward effect on lead price is smaller as well. See Sigman (1995) for a mathematical exposition.

Private costs will be lower under a deposit-refund than under a tax on all lead, a recycling subsidy, or a recycled content standard. Table 1 shows the reason for these cost differences. By encouraging users to take advantage of both options for source reduction and recycling, the deposit-refund and virgin material tax will reduce lead with the least cost to the society. However, approaches such as a subsidy or tax on all lead leave some low cost options for lead disposal reduction options unexploited and, thus, require higher costs to accomplish the same disposal reduction. This ranking will hold regardless of whether the policies are restricted to batteries or also encompass consumer electronics and other recyclable lead uses.

4. Empirical effects of battery recycling policies

To illustrate the magnitude of differences in the costs, this section will focus on application of the policies to lead in batteries. As discussed above, batteries are the most common application of these policies so far. The end of the section discusses how the results would differ if other uses of lead were also subject to the policies.

4.1 Responsiveness to price

To estimate the effects of the policies, we need information about how participants in the market respond to changes in price — that is, price elasticities. The first question is how sensitive recycling rates are to prices. Sigman (1995) estimates the responsiveness of battery recovery to lead scrap prices in the United States from 1954 through 1992. In this research, a 10% increase in the scrap lead price increases recovery rates by about 2% (which would be about a 1 percentage point increase in the recovery rate in 1988).¹¹ Thus, there is evidence that recovery of lead from batteries is responsive to price, providing support for incentive-based policies as an effective approach to encourage recycling.

A second price elasticity that is important is the responsiveness of primary lead supply to prices. In Sigman (1995), a 10% increase in refined lead prices increases primary lead supply by 8% in the long run.

Finally, the calculation requires an estimate of the sensitivity of demand for lead in batteries to price. Sigman (1995) did not find a statistically significant elasticity of lead in batteries to price. This insensitivity may reflect the absence of any good substitute for lead and is consistent with the lack of attention devoted to the lead content of batteries in engineering papers like Salkind *et al.* (1984). However, it also suggests that consumers do not adjust their consumption of batteries to lead costs. One might expect more rapid replacement of old batteries when they are cheaper. Earlier studies do estimate some sensitivity of battery lead demand to its price. Moroney and Trapani (1981) estimate a 2% reduction in demand for lead in batteries for a 10% increase in price and Anderson and Spiegelman (1976) estimate 2.1% reduction in demand for all lead (not just lead in batteries) for 1949–72. The calculations below use a 1% reduction in demand for the 10% price increase because it is a midpoint of the studies and reflects the likelihood of at least some response in battery demand.

The sensitivity of lead demand to price is critical to comparison of the policies. If demand is insensitive to price, then there are no opportunities for source reduction. In this case, the tax on lead is ineffective. In addition, recycling subsidies do not have any perverse effect on lead use and thus are not any more costly than the other policies.

11. That is, the recovery rate elasticity is 0.2. The long-run elasticity is estimated at 0.1, lower perhaps because of depletion of stored batteries. For comparison, Anderson and Spiegelman (1976) find a supply elasticity for all secondary lead of 0.48 for the period 1954 through 1972 and Fisher, Cootner, and Bailey (1972) find a long-run supply elasticity for secondary copper of 0.31 • 0.33.

4.2 Empirical effects of the policies

Table 2 calculates the impacts of three policies for reducing disposal of lead in batteries in the United States. The table uses 1988 price and quantity data: the price of refined lead is \$.3714 per pound and the recovery rate is .665.¹² I use data from 1988 because it predates the state taxes and deposit-refunds and thus shows the change from the market without lead disposal policies. Table 2 compares the policies when implemented to achieve a 20% reduction in lead disposal. It is worth highlighting that the appropriate target is not the recycling rate, but rather the amount of lead incinerated or disposed because it is these activities that potentially create health and environmental harms.

Table 2. Effects of policies for a 20% reduction in battery lead disposal, United States in 1988

	Private costs (million \$)	Revenue (million \$)	Intervention level (\$ per pound of lead)
Deposit-refund	34	136	Deposit-refund = \$.24
Recycling subsidy	59	- 483	Subsidy = \$.30
Tradable recycled content standard	38	0	Permit for virgin lead = \$.17

Calculations use a recovery rate elasticity of 0.2, primary lead elasticity of 0.8, and lead demand elasticity of -0.1, and 1988 prices and quantities. See Sigman (1995) for the equations used to calculate the values.

Deposit-refund. Table 2 indicates that the deposit-refund rates necessary to achieve moderate reductions in lead disposal are consistent with those currently in use. A deposit-refund of 24 cents per pound of lead amounts to about \$5 per battery, which is the level of most US deposit-refunds. The private costs of this deposit-refund would be about \$34 million. The uncollected deposits amount to \$136 million, which the government could collect unless the law allows manufacturers or retailers to keep this windfall.

However, this figure may underestimate the true costs of accomplishing disposal reductions through these policies because of implementation issues that are omitted from the calculation. First, the deposit-refunds used in the U.S. do not vary with the lead content of the battery, unlike the deposit-refund shown here. A fixed deposit-refund per battery provides weaker incentives for source reduction than a variable one because the policy does not encourage consumers to choose lower lead batteries. Thus, the costs of achieving the reduction could be greater. In practice, however, the lack of these incentives may not be that important; the discussion of price responsiveness above suggests that reducing the lead content of batteries does not have much potential as an avenue of source reduction.

Second, Table 2 fails to consider the administrative and compliance costs of the policies. These costs may be substantial for a deposit-refund if it is administered at the retail level because large numbers of firms must comply. Administering the program at the level of battery producers could make administrative costs much lower.

Recycling subsidy. Table 2 suggests that a subsidy has significant disadvantages relative to a deposit-refund. For a reduction in disposal of 20%, a subsidy has private costs almost twice as high as the least cost approach, \$59 million compared to \$34 million. Because the indirect subsidies actually in use generate additional distortions over this direct subsidy, subsidy costs in practice are even higher.

The revenue requirements for a subsidy might present an even more formidable obstacle. Whereas the deposit-refund would collect \$136 million in revenue, the subsidy would require expenditures of \$483 million. Subsidy expenditures are high even for small reductions because of the large amount of lead recovered initially.

12. This recovery rate is my estimate (Sigman, 1995). It is somewhat lower than the recovery rate estimated by EPA described above.

On the other hand, the administrative costs of this program are not likely to be substantial. A limited number of firms reprocess battery lead or purchase this lead. In addition, reported quantities of recycled lead are easily verified, making this subsidy less vulnerable to sham operations than other pollution abatement subsidies.

Recycled content standard. In Table 2, the recycled content standard costs only slightly more than a deposit-refund. For a 20% reduction, it costs \$38 million compared to \$34 million for the best approach.¹³ The recycled content standard produces only a small subsidy to recycled lead (because initial recovery rates are high) and therefore does not differ dramatically from a deposit-refund.

However, Table 2 shows the best possible case for the recycled content standard by allowing trading of permits. Costs without trading will probably be higher because firms have different opportunities for substitution of recycled lead for virgin lead.

Even if the government does allow trading to meet a recycled content standard, the costs could exceed the estimates in Table 2 because of the transactions costs that have hampered other environmental permit markets. However, recycled content permits are likely to be subject to fewer transactions obstacles than other permit markets. Firms are already familiar with their patterns of reliance on recycled and virgin lead, unlike pollutants to which permits have been applied previously. In addition, there should be little need for intrusive regulatory oversight of the market given the ease with which regulators can monitor lead usage. Thus, transactions cost may not greatly elevate the costs of a tradeable recycled content standard.

4.3 *Results with other uses of lead*

Although the figures above address policies restricted to batteries, this section makes some qualitative observations about how the effects and comparison of the policies would change if the policies were applied to additional uses of lead. First, I consider extending the policies to other recyclable uses of lead, such as consumer electronics and wheel weights in motor vehicles. The overall ranking of the policies will not change with this extension, but the relative costs of the policies may change. Second, I discuss how the ranking may change if policies apply to nonrecyclable uses of lead, such as gasoline additives or ammunition.

Additional recyclable uses of lead. In comparing the policies, the principal effect of adding other recyclable uses of the lead will be to change the price elasticities of lead demand and recycling. Demand for other uses of lead, such as consumer electronics, is likely to be more sensitive to price because of more available substitutes. For example, higher lead costs may raise the price of CRTs and encourage the substitution of flat panel displays. Consumers may also choose to extend the life of televisions and computer monitors, unlike batteries for which replacement is often forced by the demise of the old battery. A somewhat dated estimate supports the notion of greater price sensitivity in other uses: Wise (1979) estimates a reduction in demand of 3% for a 10% increase in price for lead uses other than batteries, ammunition, and gasoline additives in the United States. If demand is more sensitive to price, the relative costs of policies that do not take advantage of demand reductions will rise: the recycling subsidy and the recycled content standard will be more costly relative to the other policies.

13. The inefficiency of the tradeable permit system may come as a surprise. The result arises because of the specific permit system analyzed, namely one based on recycled content standards. It would be possible to design a tradeable permit system more comparable to the deposit-refund. Such a system would require permits for lead discards rather than virgin lead use. It would, however, face the enforcement problems of other direct approaches.

In addition, there are fewer low cost recycling opportunities for sources of lead other than batteries. In particular, it seems that lead prices will have to rise dramatically for recycling of CRTs to become economical. The lack of recycling options also favors policies, such as the deposit-refund and lead taxes, that can take advantage of source reduction relative to recycling subsidies and recycled content standards.

Another issue that arises in considering other uses of lead is the loss from using a fixed deposit-refund relative to a deposit-refund that varies with the lead content of the product. For batteries alone, this loss might not be very great because changes in product design will account for little source reduction. However, with other products, product design changes may be important ways to reduce lead. A deposit-refund that is invariant with the lead content of batteries provides no incentive for this substitution and thus will be much substantially more costly for any reduction in disposal than a deposit-refund that varies with lead content.

Non-recyclable uses of lead. Expanding the policies to encompass non-recyclable uses of lead complicates the policy comparison. With these other uses, we can no longer justify a simple cost-effectiveness analysis, which compares the policies for a given reduction in lead in municipal solid waste. To compare the broader policies, we also need an assessment of other environmental exposures to lead.

For example, a deposit-refund for which the deposit was paid on all uses, but the refund for recycling only, would not only decrease disposal of lead but also other releases to the environment. This breadth could be an advantage of the policy if it reduced the air pollution from leaded fuel or wildlife exposures to lead ammunition. However, it could also be a disadvantage if it unnecessarily discouraged use of lead in applications, such as construction or X-ray shielding, for which there was little risk of environmental exposure. A broad policy could also be excessive if non-municipal solid waste releases of lead are already adequately addressed by other environmental policies, such as policies for air pollution. For example, a recycling subsidy could dominate an excessively broad deposit-refund because the subsidy will affect only lead in municipal solid waste. If so, restricting the deposit-refund only to recyclable uses of lead would reestablish its superiority.

Non-recyclable uses of lead also affect the best design of a recycled content standard. Recycled content standards may apply only to the lead content of recyclables or to all uses of lead. A standard solely for recyclable products may be ineffective, however. It may shift recycled lead used by other industries into manufacture of batteries and other recyclable products. Such a standard may not reduce disposal at all. To accomplish reductions, therefore, it may be necessary to apply the recycled content standard to all lead.

5. Desirability of reducing lead in municipal solid waste

Finally, we should consider the overall desirability of policies to reduce lead in municipal solid waste. Two previous studies raise questions about whether even the most cost-effective policies would have a beneficial effect. First, Walker and Wiener (1995) summarize a 1991 U.S. EPA assessment of increasing the recycling of lead-acid batteries.¹⁴ As reported in Table 3, EPA found that increasing the battery recycling rate from 85 to 95% dramatically reduced air emissions of lead because of a great reduction in lead emitted from municipal waste combustion facilities and a smaller

14. The documents are U.S. EPA "Lead Acid Battery Recycling Risk Assessment" and "Cost Analysis of Options to Enhance Lead Acid Battery Recycling," both drafts from 1991, but my discussion is based on Walker and Wiener's summary.

reduction in emissions at primary smelters.¹⁵ Offsetting these gains is an increase in emissions from secondary smelters.¹⁶

Table 3. U.S. EPA estimates of lead air emissions and children at risk from increased battery recycling

Pollution source	85% recycling	95% recycling	Change
	Air emissions of lead (tons)		
Primary smelters	498	398 ^a to 473 ^b	-100 ^a to -25 ^b
Secondary smelters	509	560	+51
Municipal waste combustion	736	282	-454
	Children with blood lead $\geq 10\mu\text{g/dl}$.		
Primary smelters	226	198 ^a to 219 ^b	-28 ^a to -7 ^b
Secondary smelters	525	563	+38
Municipal waste combustion	–	–	-2

Notes: The range in emissions and number of children at risk from primary smelters reflect different assumptions about the change in primary production:

^a Assumes primary lead production falls by the full amount of the increase in secondary production.

^b Assumes primary lead production falls by only 25% of the increase in secondary production (with the rest perhaps exported).

Source: Walker and Wiener (1995).

When the EPA examined the number of children predicted to have high blood lead, however, the gains from increased recycling look less clear. The reduction in lead emissions from combustion has little effect on the number of children with high blood lead for two reasons. First, the populations near the facilities had low levels of blood lead, meaning that few children cross the 10 $\mu\text{g/dl}$ threshold. Second, although the change eliminates many tons of air emissions, the reduction is spread over 186 facilities, so the change in exposure at any one location is small.

In addition, the trade-off between primary and secondary smelting does not appear favorable. Secondary smelting puts more children over the threshold because secondary smelters are located near people (who supply their raw materials), whereas primary smelters tend to be located near mines and are thus more remote. As a result, increased recycling may harm more children than it helps.¹⁷

An important caveat in interpreting these results is that the EPA does not include any source reduction in the change it analyzes. Indeed, total U.S. lead output increases in the scenario that leads to the worst case. Such an increase might occur with a recycling subsidy, but would not characterize the other policies discussed here. A policy such as a deposit-refund that also has the potential to reduce lead use might result in greater health gains.

15. It also considered reduced exposure to contaminated groundwater from recycling more batteries. However, the EPA concluded that this pathway did not pose a significant health risk even when batteries are sent to older landfills.

16. These emissions occur despite existing U.S. air pollution regulations and might be worse in countries with less stringent regulations. Engineering estimates suggest that all environmental regulations in effect in 1988 added \$0.068 per pound (18% of its price) to the cost of secondary lead (U.S. Office of Technology Assessment, 1989).

17. The figures shown here presume that the increased secondary smelting occurs in the U.S. The EPA also presented a variant in which this secondary smelting occurs abroad, in which case the figures look more favorable for increased recycling, but exposure of children outside the U.S. is not taken into account.

As a first step toward extending this analysis to other countries, Table 4 presents my calculations of the population near secondary and primary lead smelters around the world.¹⁸ The results do suggest substantially higher population densities near secondary smelters than primary smelters. On average 393 people per square kilometer live near primary lead smelters, but more than twice as many, 915, near secondary smelters. Thus, there is reason for concern that the trade-off identified by the EPA applies internationally as well. However, this calculation is only suggestive; it does not follow that exposure would actually increase because the lead emissions and control technologies for primary and secondary smelters differ.

Table 4. Population densities near lead smelters around the world

	Primary smelters	Secondary smelters
Number of smelters	74	84
Number located for density calculation	46	55
Population density (people/km ²) within 20 km	393	915

Source: Author's calculations.

Macauley *et al.* (2001) also evaluate the overall desirability of policies to reduce lead in waste. They study the overall costs and benefits of several policies that address end-of-life computer monitor CRTs. They find that a complete ban on land disposal and incineration of monitors (assuming it is enforceable) would provide benefits of about \$4 million per year, as a result of \$2.7 million in health improvements from avoided exposure to air emissions and \$1.4 million savings in waste handling costs. However, the ban costs nearly \$300 million per year. Although some of these costs are from increased recycling, Macauley *et al.* calculate that most users would store their old monitors, unless recycling receives substantial subsidies.

Macauley *et al.* find that none of the policies they consider passes a cost-benefit test. The most favorable policy in their analysis is the most targeted to the health harms: excluding CRTs from incineration. Even this limited policy costs \$38 million for benefits of only the \$2.7 million in health improvements.

6. Conclusion

Governments wishing to reduce lead in waste have a variety of policy options. This paper considers policies that can reduce waste disposal when direct restrictions are too difficult to enforce. Successful policies address disposal at two levels, encouraging recovery of lead and discouraging its consumption. The deposit-refund and virgin material tax are low cost policies because they create both types of incentives. In contrast, subsidizing recycling is more costly because it decreases the price of lead to users and thus encourages lead consumption.

The paper presented empirical analysis of programs for recovery of lead from batteries, using supply and demand conditions from the U.S. This analysis suggests two conclusions. First, price-based recycling policies can effectively increase lead recycling. Second, the policies differ substantially in the costs of accomplishing a given reduction of lead in waste. A recycling subsidy entails nearly twice the private costs of a deposit-refund, with a recycled content standard intermediate in costs. The general ranking presented also applies to policies aimed at other sources of lead in municipal solid

18. Alliance to End Childhood Lead Poisoning (1994) provides the names of the town in which lead smelters are located. We were able to attach latitude-longitude coordinates to these names for 46 primary and 55 secondary smelters, representing 62% and 65% of the total respectively. Population densities within 20 kilometers were then calculated using ArcGIS and data from the Gridded Population of the World, Version 2 (Center for International Earth Science Information Network, 2000).

waste, including consumer electronics. Indeed, expanding the sources covered probably increases the importance of source reduction and thus the cost differences among the policies.

Despite the effectiveness of price-based policies, earlier studies surveyed here suggest the need for caution in pursuing policies that reduce lead in solid waste. For countries with already high recovery rates of lead from batteries, such as the U.S., it may be that the environmental gains of reduced lead disposal are not high enough to merit the cost and environmental consequences.

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CHANGING PRODUCT CHARACTERISTICS TO REDUCE WASTE GENERATION

by Matthieu Glachant¹

1. Introduction

The paper deals with policies that may efficiently encourage innovation reducing waste at source through changes in product characteristics. There is a growing consensus that modern waste policies should not only tackle with end-of-pipe issues such as the collection, recycling and disposal of household waste. Municipal waste is a by-product of the consumption of goods designed upstream by producers. Waste policies should thus seek to influence the behaviour of consumers, retailers, and producers. In this respect, as waste production is tightly correlated with product characteristics, a key goal is to foster product changes.

Anecdotic examples suggest that there exists a large technical potential there. A 1.5 litre plastic bottle used by Danone in 1993 for mineral water weighed 42 g while its 2003 equivalent weighs 32 g implying a waste reduction rate of about 25%. However, product change is not only a technical matter. It is primarily an economic process in which firms introduce on the markets new products with design characteristics reducing waste at the post-consumption stage. A first challenge for waste management policies is thus to provide producers with the appropriate incentives to innovate. Furthermore, less waste-intensive products also need to be bought by consumers. In the end, the (difficult) question for waste management policies is how to create market conditions favourable to the production and consumption of these goods. In the end, such policies necessarily interact more tightly with economic processes than traditional end-of-pipe waste policies.

The paper builds on the results of the economic literature on innovation and waste policies and discusses the policies suitable to promote product change and re-design. Note that the specific economic literature dealing with the relationship between waste policy, source reduction and product innovation is still scarce.² However, economists have extensively worked at a more general level on the issue of innovation and its relationships with policy instruments. The paper basically uses both types of result. It remains exploratory and the conclusion includes a set of questions and issues on which further work is needed.

The structure of the paper is as follows. A first section describes the types of innovation that waste policies intend to induce and discusses some economic properties of the products concerned. A second section describes two key general economic features of product innovation and derives policy lessons. The third section is dedicated to the Extended Producer Responsibility policy concept and we discuss its potential impact on product design. Then we analyse the ability of different waste policy instruments (advance disposal fees, recycling standards, tradable recycling certificates, etc.) to

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2. Exceptions are recent papers by Fullerton & Wu (1998) and Walls & Palmer (2000).

promote product innovation. A concluding section summarizes the policy implications and identifies the topics on which further work is necessary.

2. Product characteristics and waste production - Facts and statistics

2.1 What product changes?

This paper focuses on the way waste policies can influence product characteristics leading to less waste generation. A preliminary point to consider is the type of changes in product characteristics waste policies should seek to promote. These changes are basically defined by the policy objectives they potentially contribute to. They are changes in product design or characteristics leading to at least one of the three consequences:

- To reduce the quantity of waste generated by consumption
- To reduce the toxicity of the waste generated
- To facilitate recycling or re-use

These objectives may contradict each other. For instance, substituting plastics by glass in beverages packaging definitely reduces recycling costs but increases packaging weight. To tackle with these possible contradictions, any waste policy should necessarily establish a clear and non-ambiguous hierarchy between the different objectives. In following sections, we will come back on this issue. The practical ways of contributing to these goals are very diverse. Here are some examples:

- Lightening packaging or opting for more easily recyclable materials.
- Lengthening the lives of products such as tires.³
- Reducing the variety of plastics used (in a car, a computer).
- Avoiding painting and putting labels on recyclable parts.
- Using a modular architecture for durable goods to make them easier both to upgrade and to disassemble at end-of-life.
- Minimising or eliminating embedded metal threads in plastics.

The diversity of the practical solutions is in fact determined by the variety of the products generating household waste. In this respect, it is possible to roughly distinguish two broad product categories differing in waste-related design patterns.⁴ A first group gathers packaging and non-durable goods. For these products, the challenge is mainly lightening, reducing the development of small containers, and substituting material by lighter and/or more easily recyclable material. The second group is constituted by durable products such as electronic equipments, household appliances and cars. A significant proportion of their metal contents is already recycled (about 75% for white goods, almost 100% for cars). In this context, complete product redesign might be a critical part of

3. Note that, when adopting a broader perspective, increasing product durability does not necessarily prevent waste. The reason is that lengthening the life of a product delays the purchasing of a new product. It thus reduces the consumption of that particular product and consumers save money. But it is very likely that they will use this money to consume other goods. Hence, increasing durability simply implies consumption substitution between two products. In the end, the waste impact of such a substitution will depends on the relative waste intensity of the two products. This point was initially made by Nils Axel Braathen from OECD.

4. Annex 1 presents US statistics on the relative contribution of different categories of product on overall municipal waste production.

reaching waste reduction goals (Palmer and Walls, 2002). This requires radical innovation as opposed to incremental changes that are at stake for non-durables and packaging. A further difference is that durable products are much more complex and embody a higher number of materials. In the end, these differences cannot but imply differentiated policy consequences depending on product category.

2.2 “Business-as-usual” product change

Another critical point of context is that products continuously change in market economies, regardless of the eventual existence of waste policies promoting innovation. A recent paper by Bernard et al. illustrates the extent of such business-as-usual BAU changes in the USA (2003). In average, about 70% of US manufacturing firms change their product mix within a 5-years period. In terms of output, product switching is even more important since the firms changing their products represent more than 90% of total output while the new products account for 50% of firm's current output in value.

The impact of BAU product change on waste streams is assumedly far from being negligible even though it certainly varies across product category. A study has recently analysed this aspect over the period 1979-1999 in France. Table 1 gives a very aggregated view of the results of the study distinguishing foodstuffs from other products. While the consumption of foodstuffs is quite stable in weight over the period (+4%), the quantity of waste generated has significantly increased in the meantime (+15%) essentially because of the development of packaging.

By contrast, the consumption per household of non-food products in weight has dramatically decreased. In fact, this is essentially due to a dramatic reduction in the consumption of coal which has been substituted by electricity of which weight is zero [each household was consuming 305 kg of coal per year in 1979 and the 1999 consumption is now 40 kg]. If we exclude coal, the consumption of non-food products has in fact increased by 28%. In Annex 2, we report in Table A.2 product by product the evolution of the consumption. The most impressive increase is observed for brown goods (+140%), tyres (+235%), drugs (+239%), oils and lubricants (+266%), car batteries (+1100%) and telephone equipments (+2260%). Despite such increases in the consumption, the quantities of waste generated by non-food products are almost stable over the period (-4% or +1% when we exclude coal). This is essentially due to the lightening of durable goods. Overall, waste intensity, excluding the specific “coal effect”, has been reduced by 21%. More precisely, the unit weight of “white goods” – household appliances such as refrigerators, ovens and washing machines – was reduced by 18%. For “brown goods” – HIFI equipments, TV - the reduction rate is even 40%.

Extrapolating from these statistics, the overall trend seems thus characterized by (i) a dramatic increase in packaging use for non-durable goods, (ii) a very important weight reduction of durable goods, but associated with an increasing product complexity, an increasing diversity of the embodied materials, and a decreasing recyclability due to the substitution of metal by plastics. Once again, the difference between durables and non-durables appears quite sharp.

Table 1. Change in consumption and waste generation over the period 1979 – 1999 in France

	1979	1999	Variation 79-99
<i>Food stuffs</i>			
Annual consumption per household	916 kg	955 kg	+ 4%
Annual waste production per household	211 kg	243 kg	+ 15%
Waste generated per kg of consumption	230 g	255 g	+11%
<i>Non-food products</i>			
Annual consumption per household	674 kg	420 kg	- 38%
Annual consumption per household excluding coal	370 kg	474 kg	+28%
Annual waste production per household	304 kg	292 kg	- 4%
id. excluding coal	286 kg	289 kg	+1%
Waste generated per kg of consumption	450 g	695 g	+54%
id. excluding coal	773 g	610 g	-21%

Source: adapted from ADEME, 2001.

In the policy literature on innovation, a further distinction is often made between so-called “demand pull” and “technology push” innovation. In the first case, innovation is driven by the necessity to meet the changing preferences of the consumers. As a result, commercial and marketing considerations are the key determinants of the innovative process. In the case of technology push innovation, the driver is continuous technological progress. The typical example is the market for PCs in which the very fast renewal of products is basically driven by technological progress in the semiconductor industry that continuously increases the processing speed of chips. Clearly, the fact that innovation focuses on processing speed is not demand driven. In fact, the demand-pull versus technology-push classification does not coincide with the classification between durable versus non-durable goods we have emphasized before. For instance, while innovation in the computer industry is definitely “technology push”, the car manufacturing industry probably belongs to the demand-pull category. By contrast, industries producing non-durable goods are all characterized by intense incremental product innovation and product differentiation pulled by demand.

To sum up, the context is marked by BAU product changes having dramatic impacts on waste generation. Therefore, the goal of waste policies is clearly not to initiate product change. Instead, the challenge is to modify the pattern of business-as-usual product change in order to position goods on less waste-intensive innovation trajectories. When designing these policies, it is essential to take into account these BAU trajectories and the fact that they are industry specific.

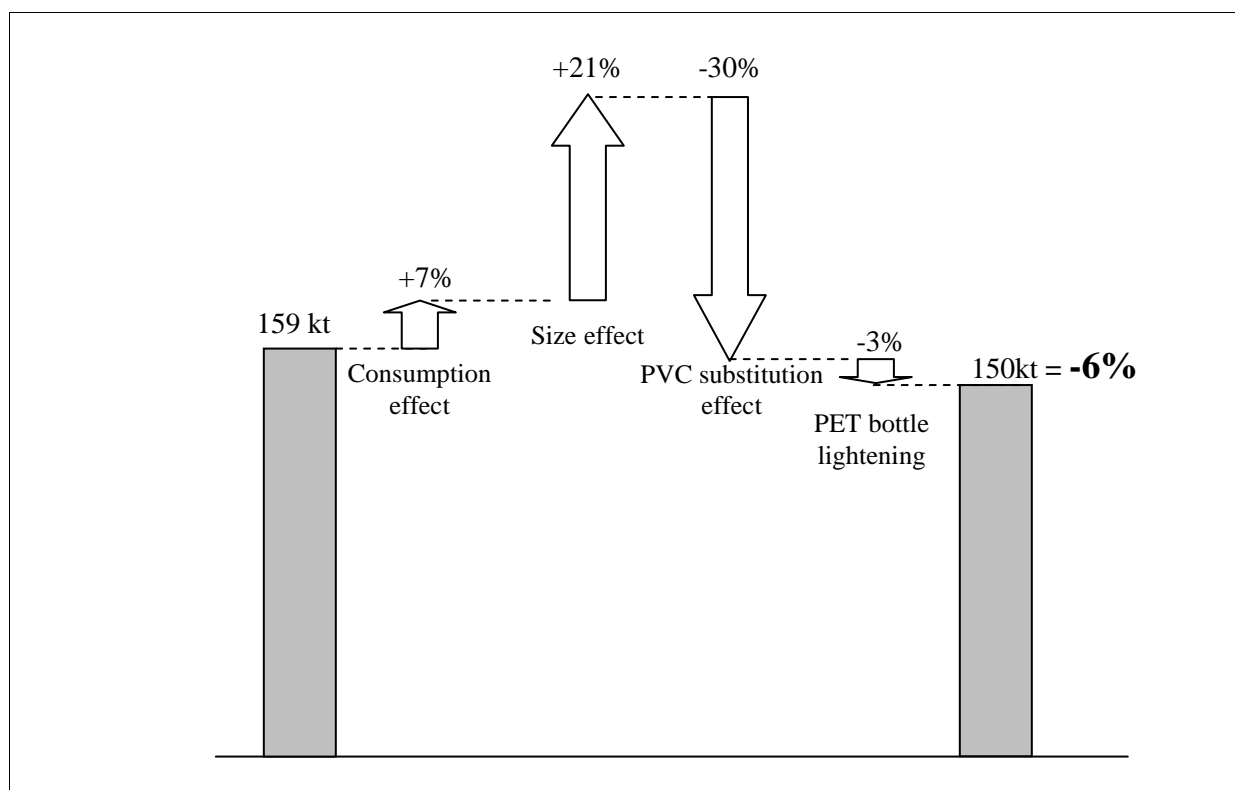
2.3 A case study: plastic bottles for mineral water in France

The factors driving product change and product waste intensity are probably extremely variable and local. As an illustration, this section describes and analyses the evolution of mineral water plastic bottles on the French market over the period 1994-1997 and its consequences on waste generation. As shown in Figure 1, the quantity of waste generated by mineral water packaging in France has slightly decreased by 6% over the period. However, behind this apparent macroscopic stability, Figure 1 shows that the determinants of waste generation dramatically changed. First of all, the consumption of mineral water slightly increased leading to a 7% increase in waste quantity. Second, the market of mineral water saw a very fast development of small bottles (1 l, 50 cl and 33 cl). This “size effect” induced a 21% increase in waste production in only 3 years time! This BAU change was basically demand driven.

The third determinant had even larger consequences on waste: -30%! PET almost completely substituted PVC. The quantity of PVC used for bottle packaging was reduced by 81,5% in the 1991-2001 period. This had positive environmental consequences downstream: PVC is for instance the first source of chlorine in incineration outputs (between 38 and 66% of the total). Moreover PVC is heavier than PET. As a result, the substitution led to an impressive 30% reduction of waste. The last factor is PET bottle lightening. For instance, the food company Danone reports the following figures for its standard 1,5 litre PET bottle. In 1994, they were using a PET bottle of 42g. In 1995, a new design brought down the weight of the bottle to 35 g. Finally, new optimisation techniques were implemented in 2000 leading to further progress. The bottle now weighs 32 g. However, over the whole sector and over the period 1994-1997, bottle lightening only had a limited impact on waste streams: -3%.

What general remarks can be made on the basis of this example? The first one is obvious but it is still useful to insist on: product changes may have dramatic impacts on overall waste flows (e.g. PVC substitution leading to a 30% decrease!). Among the drivers of product change, market forces can be extremely powerful (e.g. +21% for the size effect). A second remark is that anecdotic evidence of product change (e.g. the impressive lightening of Danone's bottles) does not necessarily imply significant macro-impacts on the waste flow (e.g. overall lightening only accounts for a modest 3% reduction). A corollary is that the diffusion of innovation in the market - enabling macro impacts - is as important as the innovation itself. In the next section, we will see that it is challenging for public policies to conciliate innovation and diffusion.

Figure 1. Change in plastic bottles for mineral water in France over the period 1994-1997



Source: ADEME, 1999, reported in Barbier Frinault & Associés, 2003.

Finally, the PVC substitution story illustrates a very interesting property of the relationship between waste policies and product design. Substitution was certainly motivated by commercial and

marketing considerations. For instance, consumers prefer PET to PVC because of its transparent and bright appearance. But a further reason was also that PVC had already been substituted in the vast majority of the neighbouring countries. Some countries had even banned the use of PVC for bottles (e.g. in 1993 in Switzerland). As a result, in 1995, French facilities of Nestlé were producing PVC bottles for the French market and PET bottles for export (about 30% of the sales). In this context, PVC substitution in the French market was saving production cost. This highlights a further aspect of product design. Design decisions are made at the level of the market. If the market is global (e.g. computers), decisions are made at the global level. If the market is regional (e.g. mineral water), decisions are made at the regional level. By contrast, the size of the waste policies' jurisdictions is often smaller since policy decisions are essentially made at the municipal level (e.g. unit based waste disposal and collection charges) and at the national level (e.g. take back mandates, advance disposal fees). A major policy implication follows: for waste policies to be effective in promoting product innovation, an implicit or explicit coordination is necessary to coordinate and to integrate municipal and national policies. Policy coordination is particularly crucial when the scope of the product market is international.

3. Economic properties of innovation and implications for waste policies

Economic works on the specific issue covered in this paper – waste policies and product innovation – are very scarce with notable exceptions by Calcott and Walls M. (2000), Eichner and Pethig (2001), or Fullerton and Wu (1998) that will be covered in the next sections. By contrast, innovation in general is a major topic in economics. Using this literature, the section describes two economic properties of innovation – the fact that it is a risky activity and the fact that it has public good features – and derives implications for waste policies.

3.1 Innovation is risky

The economic argument. In market economies, producers undertake innovation activities because it increases their wealth, meaning that their expected benefits are higher than innovation costs. The economic analysis of innovation stresses that these costs and benefits shares many features that all imply a high level of risk and uncertainty on the profitability of innovation. The first of these features is obviously that the success rate of innovation projects is typically low. A study by Mansfield et al. (1971) on research projects in the pharmaceutical, electric and chemical industries show for instance that the rate of technical success ranges between 52-68%. But only 8-29% were commercially successful. Figures are rather old but give an order of magnitude.

The uncertainty of the innovation outcome is not the only source of risk. Perhaps more importantly, innovation is an *investment*. This precisely means that the innovator should bear innovation costs now in order to obtain future benefits. The typical time span between expenditures and returns is several years. In the meantime, the economic environment evolves; competitors launch new products changing more or less drastically the conditions under which the initial innovation decision was made. A further source of risk is that innovation costs are *sunk* in the sense that they cannot be recovered should the innovation project be withdrawn. The consequences of innovation failures are thus worse than those of classical investments. For instance, creating a new airline Paris-Toulouse requires investing in a new aircraft. In case of commercial failure, however, the company can hope to recover a share of the value of the initial investment by selling the aircraft or by transferring it on another profitable airline. When innovating, the resources used are lost in case a commercial product fails to come out at the end of the process.

Policy implications. In the end, many factors lead innovation to be a particularly risky economic activity: outcomes are uncertain; while costs are immediate, gains are mid or long-term; costs are sunk. This has implications for waste policies aiming at promoting innovation. In order to be effective, policies should reduce as much as possible the level of risk surrounding product innovation. It

primarily requires a long-term stability of the policy signals, or at least a predictability of their changes. In this regard, policy goals are key. We have already pointed out that waste prevention policies may pursue different goals – to reduce the quantity of waste generated by consumption, to reduce the toxicity of the waste generated, or to facilitate recycling or re-use – and that these objectives may contradict with each other. In this context, it is essential that waste policies include a clear a stable hierarchy among them. This is a precondition for producers to innovate.

3.2 *Innovation is a public good*

The economic argument. Another economic property is that innovation outcomes may benefit to others, in particular competitors, through imitation. Imitation is particularly of concern for the type of innovation we are concerned with in this paper, that is product innovation as opposed to process innovation which improves production processes. The concern is that the resulting innovation is embodied in the product which is sold in the market. It is then fairly easy for any imitators to exploit the innovation by analysing the product – a process called reverse engineering. In the case of process innovation, imitation is more difficult because the innovator has the possibility to keep secret the innovation.

The economic analysis uses the notion of public good for denoting such “goods” which may benefit to anybody by contrast to traditional private goods of which benefits are only appropriable by the owner. The public good feature has multiple consequences. First of all, imitation threatens the market benefits that the innovating firm can expect. Put differently, innovators cannot appropriate all the benefits of their innovation. Therefore, they have reduced incentives to innovate. This is detrimental to the society as a whole since innovation is the basic ingredient of economic growth. However, it is not the end of the story. If imitation hurts the general interest by reducing innovation incentives, it has positive impacts on another ground. It participates to the diffusion of the innovation enabling to spread over benefits in the economic system. For example, if say DELL invents a new modular architecture for PC chassis that reduces disassembling costs by 30%, the adoption of this innovation by SONY, COMPAQ and others PC manufacturers will dramatically rise the benefits for society through an overall reduction of dismantling costs. There is a trade-off here. On the one hand, the regulator should protect innovation incentives. On the other, he should promote innovation diffusion.

There are three generic strategies to solve the dilemma. First, the regulator can grant intellectual property rights such as patents and copyrights. These rights give an exclusivity on the innovation to the innovator for a certain period of time. Basically intellectual property rights transform innovation into an appropriable private good through legal means. Note that the duration of the period of exclusivity – usually 18 years for a patent – is the variable on which the trade-off is solved: the innovator is given the exclusivity and the corresponding monopoly position on the market but only for a limited duration. Afterwards, others can freely adopt the innovation. Another property of patents is that they include the obligation to publicize the technical characteristics of the innovation. In this way, patenting enables others to exploit new ideas on which the innovation is based before the termination of the patent period.

The second solution is public research or publicly funded research. In fact, public R&D suppresses the dilemma. There is no longer any needs to stimulate private innovators since research is directly made by non-profit public organisations which publicize their results contributing to their diffusion.

The third one is a private solution. Innovators can form R&D joint ventures (JV) or research consortia. This solution is based on the idea that imitators and innovators generally belong to the same sectors. They are frequently competitors on a given market producing the same kinds of products facing similar problems and thus searching for similar innovative solutions. In this context, one

possible solution for them is to cooperate on R&D. It has the advantage of mitigating the imitation problem. In fact, imitation is limited to JV outsiders since insiders directly benefit from innovation outcomes. A further advantage is that it saves innovation costs via economies of scale. Innovation costs are indeed partly fixed in the sense that costs do not vary with the quantity of the innovation-associated goods that will be sold on the market. In this context, pooling innovation efforts diminishes innovation cost per unit of good. Given these advantages, cooperation in R&D is very widespread in certain industrial sectors. For instance, it is pervasive in the pharmaceutical industry.

Implications for waste policies. In the case of waste policies, cooperation among firms is observed and usually takes the form of so-called Producer Responsibility Organisations (PROs). These entities, as represented by, among others, Duales System Deutschland AG (DSD) in Germany, fulfil collectively on behalf of the producers the requirements associated with an Extended Producer Responsibility (EPR) programme. They are very widespread for a variety of products: packaging (e.g. France, Germany, the Netherlands), batteries (e.g. the Netherlands, the United States), cars (e.g. the Netherlands), Electrical and Electric Equipments, EEE (e.g. Switzerland, Norway, the Netherlands, Sweden) and used oils (e.g. Canada). Most PROs are based on a liability principle according to which the individual producers (or retailers in some cases) are responsible for fulfilling certain obligations related to waste prevention, recycling or other waste management aspects. These obligations may take different forms: a take back requirement, a minimal rate of recycling to be met or other quantitative targets. The key point is that individual producers can partly or completely escape from their individual liability by participating to a PRO which collectively fulfils the EPR requirements.

One can perfectly imagine these entities being active in cooperative R&D on waste prevention. In practice, it happens not to be the case. Tojo and Hansson (2002) have recently reviewed the functions performed by PROs. They include the management of physical waste management infrastructure (for collecting, sorting or recycling), fund-raising beside members through fees to finance PRO activities, information provision to the public about the performances of the EPR programme, reporting and monitoring to the regulator, etc. Cooperative R&D seems very limited. Tojo and Hansson only mention the example of the Swedish car manufacturer association BIL which conducted a joint research programme for the design of end-of-life management and of the Japanese EEE manufacturers which established a common pilot plant to examine the feasibility of various recycling options. However, in these two particular cases, cooperative programmes were in fact not implemented by a PRO but an industry association. In practice, it thus seems that PROs do not have clear mandates to set up cooperative product innovation. The economic literature suggests that it could be one objective for PROs.

It should be noted that these organisations could also play a role in coordinating the individual product design decisions made by their members. The fact that waste of different products is generally jointly processed creates externalities between producers. In this context, one way of cutting recycling and dismantling costs downstream is to standardize certain design parameters of the products. PRO may provide the adequate forum for the discussion and the adoption of such standards.⁵

What is the potential of public R&D in our case? Probably, very limited. Public research focuses on science - the production of generic knowledge - as opposed to innovation. By contrast, waste-related product innovation is tightly related to commercial, marketing and industrial issues and should be mainly undertaken by private actors. However, it does not exclude publicly funded private research, the idea being not to fully finance private R&D but to provide financial incentives to compensate the weakness of market incentives. Another advantage of the involvement of public

5. The author thanks Professor Stephen Smith from University College London for suggesting this point.

bodies in the financing of private innovation is the possibility to orient research directions in technological paths that are more favourable from a social point of view.

4. Extended Producer Responsibility and product innovation

We have already started to evoke the issue of Extended Producer Responsibility when discussing PRO. In this section, we investigate in a more systematic way how this policy concept can contribute to waste-related product innovation. By definition, an EPR programme is an environmental policy approach in which a producer's responsibility physical and/or financial for its product is extended to the post-consumption stage of the product's life cycle. Such programmes are seen by their promoters as the ultimate tool to achieve product innovation since – contrary to traditional downstream waste policies – they directly target the upstream producers who make product design decisions. In fact, one key goal of EPR is, by making producers paying for the costs of recycling and disposal, to create incentives for cheaper recycling and waste prevention. In the following, we discuss the impact on product innovation of two key features of such programmes – whether they are collective or individual schemes and whether the responsibility is completely or partially shifted on producers.

4.1 Collective versus individual EPR programmes

One of the major disputes during the designing process of the EU Waste Electrical and Electronic Equipment (WEEE) Directive (2002/96/EC) was whether the producers should fulfil their financial responsibility on their own, or collectively. The industry itself was divided on that issue. Some of the prominent EEE manufacturers strongly supported individual responsibility to reward the efforts of the manufacturers that produce products with higher recyclability and lower recycling costs (Tojo and Hansson, 2002). The underlying argument was that collective schemes would suppress – or would reduce – the individual incentives to change products.

In fact, as argued by van Beukering and Hess (2002), confusion exists on the exact meaning of collective versus individual EPR programmes. For the sake of clarity, they suggest making a distinction between two functions performed by EPR programmes, that is the manner in which wastes are collected and processed on the one hand, and the source of financing and the management of collected funds on the other hand. As shown in Table 2, collective collection and processing schemes – PROs – are most widespread in Europe. PROs' primary task is to set up and manage the infrastructure that organise the collection and processing of waste on behalf of their individual members. They often try to use the existing infrastructure and thus negotiate and contract with a variety of entities previously engaged in collection, transportation, and recycling. By contrast, individual collection and processing schemes whereby each firm organizes on their own waste collection and treatment are very rare. Possible illustrations include XEROX that reuse their copiers and IBM who recycle servers. According to Beukering and Hess (2002), these individual schemes are only feasible because they operate on a business-to-business basis, limiting the number of actors involved.

Table 2. Distinction between collective and individual EPR schemes

	<i>Collection and processing</i>	<i>Financing</i>
<i>Individual</i>	Each company organizes the collection and treatment of the waste generated by its own products (e.g. Xerox with copiers, IBM with servers)	Brand-based costing on the basis of specific product's waste collection and processing costs (e.g. ICT in the Netherlands, sales packaging in Germany)
<i>Collective</i>	End-of-Life managers are collectively established by industry (i.e., most consumer goods in Europe)	Costing on the basis of market share or on the basis of other variables without any direct links with individual firm's waste management costs

Source: van Beukering and Hess (2002).

The fact that collection and processing schemes are collective may raise efficiency concerns. In particular, the existence of a dominant PRO in a country generally prevents the establishment of an alternative scheme. As a result, it can create competition distortions on waste management markets. DSD has long been under the scrutiny of the Bundeskartellamt – the German antitrust regulator - for these reasons. But, it is difficult to see how the way the PRO behaves downstream on the market for end-of-life management can affect directly or indirectly the upstream competition between the producers who finance the scheme. A potential adverse impact on product innovation due to the collective nature of the collection and processing scheme is difficult to conceive.

On the contrary, the intensity of the upstream competition between producers and product innovation is at stake as soon as the financing role of PROs is concerned. First, note that the funding characteristics of the PROs are more difficult to classify into the collective and individual categories. For instance, all known PROs manage funds collectively. The crucial point on which the distinction is based is how individual producer's contributions are calculated. Under the individual regime, each individual producer contributes on the basis on its *own* products' collection and processing costs. Under the second regime, contributions are based on variables such as market shares which have no direct relationships with individual producers' product characteristics.⁶ As a result, they fail to provide producers with incentives to alter their products.

Individual financing schemes is common in PROs dealing with packaging waste. Such PROs are generally financed by an advance disposal fee, that is a fee paid on each unit of product put on the market which reflects waste management costs. In Box 1, we describe the packaging fee of the French PRO Eco-Emballages in charge of sales packaging. The fee includes a variable part which is product specific; more specifically, it depends on packaging unit weight by material. It thus potentially provides each producer with an incentive to modify its product through packaging lightening or material substitution. The key element in the system is that the PRO is able to link the size of the fee with waste-related characteristics of a specific product sold on the market by a given producer.

In the case of packaging, the unit weight by material of a specific branded product is relatively easy to monitor; furthermore, it provides incentives for lightening and material substitution which are a key part of packaging waste prevention. In the case of more complex products, designing incentive fees is far less feasible. In particular, many more materials are embodied in durable goods like cars or computers and the dismantling ability is a key factor of waste prevention and recycling. As a result, waste prevention generally requires a complete redesign of product and it is difficult to imagine a product fee providing the adequate incentive.

It is thus not surprising that “collective” financing schemes are more widespread for these products. For instance, in the case of EEE, there exist product fees in Belgium, the Netherlands and Switzerland of which rate depends on product category. There is thus no relationship between the size of the fee and the characteristics of individual products. If incentive product fees are not feasible for EEE, alternative individual financing instruments are possible. In this regard, the case of the Netherlands is particularly interesting to consider. In this country, the EPR programme dealing with EEE involves two PROs because of persisting disagreements among producers on financing aspects (van Beukering and Hess, 2002). ITC producers have organised themselves into *ICT Milieu* of which financing is individual. More specifically, waste is sorted according to brand once they have been collected, enabling to charge each producer for the transport and the treatment of its own waste. In addition, producers finance an added amount for free riders and orphaned products. The second PRO *NVMP* gather producers of “white” and “brown” goods. This PRO implements a collective financing scheme through a fee depending on product category.

6. Note that no schemes are known that make no differentiation between companies.

Box 1. Fee structure of the packaging PRO Eco-Emballages in France

A fee is paid by the producers on each unit of sales packaging they put on the market (i.e., an advance disposal fee). This is a two-part tariff. A first part is variable and depends on the material and the weight of the packaging. Rates are given in Table 3. Differences between materials reflect the cost born by Eco-Emballages to manage the different types of material. As an illustration the contribution for plastics and for glass differ from a factor 50 due to the difference in recycling cost. The second part of the tariff is fixed and equal to 0.1 cents of euros per unit of packaging. The fixed part is much simpler than its German DSD equivalent which depends on the packaging size.

Table 3. Packaging fee's variable part of Eco-Emballages (in cents of euro per kg)

Material	Rate
Glass	0,33
Paper-cardboards	11,10
Steel	2,06
Aluminium	4,12
Plastics	16,17

* adopted on 1/4/2002

The Dutch example suggests that it is always theoretically possible for PROs to design financing instruments providing producers with individual incentives. But it entails a trade off that should be explicitly recognised: these instruments usually lead to high administrative costs. It is a first general conclusion of this analysis. The second lesson is that cooperation, on which PROs are based, and competition-based financing structure are not necessarily contradictory as illustrated by sales packaging PROs or the Dutch *ICT Milieu* dealing with EEE. PROs are perfectly able to design financing instruments fostering competition among their members on product design. It is possible but not systematic and one role for the public regulator is to encourage PROs to proceed this way.

4.2 Full financial producer responsibility or shared responsibility with local governments?

We have seen that a precondition for a PRO financing scheme to provide producers with individual incentives is to link the size of the fee with individual producer's waste collection and processing costs. But it is not sufficient. As a matter of fact, most observers of the French system Eco-Emballages agree that the size of the advance disposal fee described in Box 1 is not sufficient to induce significant product changes. The problem does not lie in the fee structure, but the rate which is very low. In comparison, the German DSD fee rates are 10-20 times higher. The reason of this difference is ultimately related to the underlying responsibility distribution under the two systems. In Germany, the producers are financially (and technically) responsible for the *total* waste collection, treatment and recycling costs of consumer packaging. In France, Eco-Emballages only finances a share of the post-consumption costs, leading to a partial cost internalisation. Moreover, the burden-sharing rule between producers and local governments is ambiguous leading to continuous disputes. By contrast, in Japan, the *Container and Packaging Recycling Law* includes a very clear rule: the local

governments are financially responsible for waste collection and disposal costs whereas producers must finance recycling.⁷

According to economic theory, what is the most efficient distribution of responsibility? The guiding principle is the concept of social cost internalisation. Producers will make efficient waste-related product design decisions if they bear the total social costs associated with waste collection, disposal and recycling of their products. The general intuition is quite simple: when all costs are internalised, the producers bear all the consequences of their decision; they are thus able to make the efficient decision. Full financial responsibility in EPR programmes is certainly a first step towards the internalisation of waste social costs. However, this might not be sufficient if external (environmental) costs associated with waste collection and processing are not properly internalised in waste management costs through disposal taxes or regulations.

By contrast, a partial financial responsibility generally contradicts economic efficiency since the producer only bears a share of the costs. However it should be recognised that, in theory, it is not true in all cases. In particular, if all households face a unit-based collection and treatment charge reflecting the remaining share of the social costs, the market may transmit upstream the missing incentives to the producers, restoring efficiency. In practice, unit-based pricing is not very developed in many countries wherein the essential channel influencing producers is EPR programme. In these countries, partial financial responsibility is definitely not economically efficient.

5. Waste policy instruments and product innovation

In this section, we assess the instruments which can be used to meet waste policy goals. In the OECD literature, there exists a well-established list of assessment criteria including environmental effectiveness, economic efficiency, political acceptability, fiscal effects, competitiveness concerns, etc. In this paper, we focus on two criteria: (i) *dynamic effects on product innovation*; (ii) *administrative feasibility and other institutional aspects*; the latter is justified on the ground that administrative costs are often a critical factor of the selection of the instruments promoting product changes as evoked before.

5.1 Taxes or standards?

Many waste policy instruments fall in one of two categories: standards or taxes. Examples of waste taxes are advance disposal fees paid on each unit of product sold in the market which reflect the disposal cost of the product, unit based user charges paid by the households to the municipalities to have their waste collected and disposed, or the various taxes constraining waste disposal facilities. Examples of standards are requirements prescribing a minimum content of recycled material for certain category of products (e.g. newsprints), or the various emission standards constraining waste disposal facilities.

The economic literature has developed theoretical arguments supporting the claim that taxes are more effective in inducing innovation than standards.⁸ The intuition is very simple. When using a standard, no additional effort is needed once the standard requirements have been met. By contrast, such a threshold does not exist when using a tax. Whatever the environmental progress already made by the polluter, he continues to pay the tax on residual waste. As a result, there always exists an incentive to make further progress. In this way, taxes create a “moving target”, leading to dynamic effects and continuous environmental efforts and innovation. Note that the superiority of taxes over standards only holds true all other things – in particular, the strictness of the policy scenarios – being

7. In OECD (2001), annexes 8-9 pp. 135-37 describes different examples of responsibility distribution rules in various EPR programmes.

8. For a recent review, see for instance Jaffe et al. (2003).

equal. Of course, a very low rate tax is less likely to induce innovation than a very strict standard – e.g. a ban of the use of PVC in packaging – with which compliance requires a radical change in comparison with the current state of the art.⁹

5.2 Upstream taxes or charges

The primary justification of upstream approaches is the expectation by policy makers that they might provide producers with incentives to alter their products (OECD, 2001). It has just been argued that taxes are more effective in promoting innovation. Moreover, they are much more widespread than standards at the upstream level of the product chain. One essentially finds upstream standards in certain US States prescribing minimum recycled content. Those are two reasons for focusing the analysis of upstream instruments on taxes and charges.

Impacts on innovation (and administrative costs)

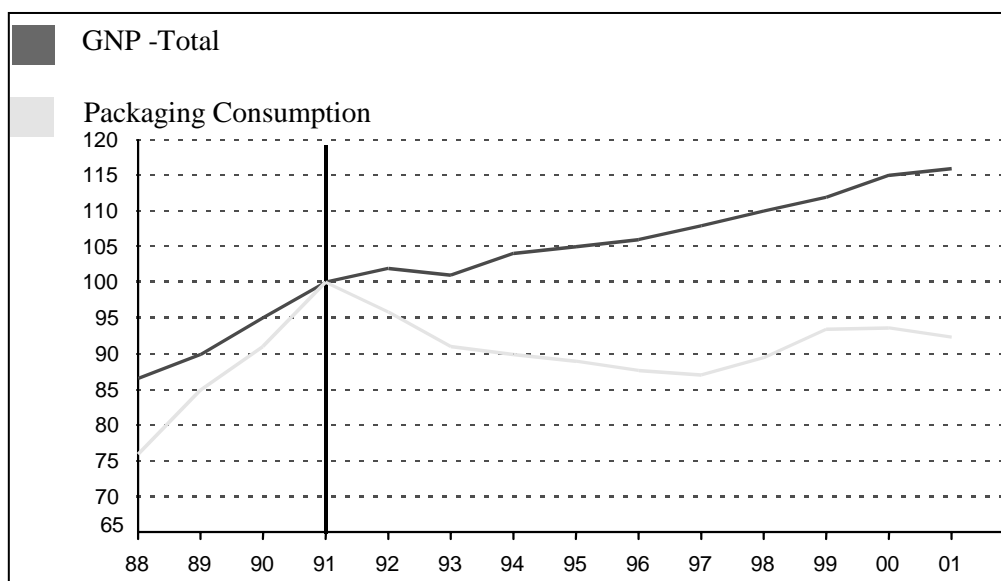
Does upstream taxes induce product changes in practice? In fact, ex post evidence is very scarce. Results are however available on the impacts of the emblematic EPR programme DSD set up in Germany for sales packaging. Figure 2 respectively depicts GNP and packaging consumption in weight by German consumers. One central objective of DSD was to reverse the positive correlation that existed between economic growth and packaging consumption. Given that the Dual System was created in 1991, the graph neatly suggests that DSD was successful. Despite a very significant increase in the number of households from approximately 35 million to 38 million, the use of packaging is stagnating since 1991. The comparison between the amounts of packaging used in 2000 with a hypothetical trend in the absence of the Dual System shows a reduction of 18% (Quoden, 2002).

A first probable explanation of this success is related to the DSD fee structure. Like the French packaging fee presented in Box 1, the size of the fee paid by each producer depends on the packaging weight and on the material used. This is not sufficient to explain the performance of DSD since Eco-Emballages, using a similar fee, has not experienced the same development over the period. The other explanation is that German fee rates are 10 to 20 times higher than the French counterparts. In turn, this difference is ultimately rooted in the fact that the French EPR programme is based on a principle of shared responsibility between the producers and the local governments limiting the producers' financial efforts while producers are fully responsible for sales packaging waste in the Dual System.

As argued in section 4, designing product fee providing incentives is much more problematic for more complex products – like durable goods – in which the number of product parameters influencing waste generation and recycling is much higher (durability, dismantling ability) and which embodies many more materials (e.g. many different plastics). For these goods, a possible alternative set up by Dutch ITC producers for instance is to sort waste according to brand and to charge each producer collection and treatment cost of its own waste. In fact, as underlined before, there always exists the possibility to design upstream pricing schemes that provides individual incentives to innovate but it entails a trade off between improved incentives and administrative costs.

9. This does not imply, however, that the enactment of unrealistic policy goals is a good way to foster innovation. As developed previously, innovations are long-term decisions which are only made when policy signals are credible.

Figure 2. Real GNP and Packaging Consumption of private consumers in Germany 1988 – 2001, Indices 1991 = 100.



Adapted from Quoden (2002); source: StaBuAmt GVM.

Institutional aspects

Upstream instruments may be implemented in two very contrasted institutional environments. First, advance disposal fees and other product taxes are often implemented by PROs to finance their activity in the market for end-of-life management. As previously discussed, it is a widespread option for packaging, for EEE, batteries or used oils. In this configuration, the upstream instrument is part of complex policy mix involving a take back mandate, collective recycling targets, and a collective PRO. It should be emphasized that no known PROs set standards to achieve their targets. In practice, PROs are thus exclusively associated with *economic* instruments.

In a more traditional way, public bodies can instead implement the same upstream taxes or standards. Possible examples are the standards mandating a minimum content of recycled paper in newsprint that exist in some States in the USA. An example of advance disposal fee is the tax on used oil in France which is collected by the Government. It is worthwhile noting that, like taxes managed by PROs, the tax revenue is pre committed – or earmarked – to collection and recycling financing.

In section 4, we have already discussed the concern of PROs being unable to design financing instruments that foster competition among their members. We have already provided counter-examples to this claim. But one pending question is whether cooperation-based PROs have a higher propensity to design low powered incentives than public agencies. Our opinion is that there is no general answer to this question. On the one hand, there exists the risk that PROs provide a forum for producers to collude on waste-related issues and to mitigate product competition through non-variable fees. But it has been stressed for a long time by the economic literature that collusion is not necessarily a stable equilibrium. The public regulators can thus try to prevent the emergence of such collusive outcomes.

Under the alternative regime, public agencies are also able to design schemes with poor incentives. This is so because the public regulator frequently sees waste taxes, or green taxes more generally, as funding mechanisms. Having this perspective in mind, he might not be keen on designing taxes with optimal incentives.

5.3 Downstream instruments

Promoting waste-related product innovation and re-design is a primary role for upstream instruments since they target producers who make product design decisions. It does not imply that downstream instruments should not be considered as useful tools influencing product innovation. One important reason is that, in market economies, the ultimate impact of product design on waste streams depends on the commercial success of re-designed products. In this way, consumers play a crucial role and downstream instruments better target them.

A second, probably more important, reason to consider these instruments is that existing waste policies are still essentially made of downstream instruments even though the development of EPR programmes is changing the landscape. One finds incineration or landfill taxes in many countries; over the last decade, regulatory constraints on waste disposal have been considerably reinforced using increasingly stricter emission standards. Recycling has been promoted by recycling subsidies in other countries. In this context, pragmatism requires wondering what impacts these instruments might have upstream on producers and whether they should be modified in this aim.

The importance of unit-based user charges

A necessary condition for downstream policies to influence upstream design decisions is to provide consumers with incentives to modify their purchasing behaviour on the product market. There exist “soft” means to do so such as awareness-raising campaigns communicating on the influence of consumption choices on waste generation or product labelling signalling environmentally friendly products to the consumers. But, if one seeks significant impacts, the only possibility is to implement unit-based waste disposal user fee to households. Note that in this model, waste disposal taxes and emission standards constraining waste disposal facilities are integrated in waste disposal and recycling cost; they are thus reflected in the unit charge paid by households so that all downstream policy signals are transmitted up to the producers.

Under very restrictive conditions we will discuss later on, it can be shown in theory¹⁰ that a combination of a recycling subsidy with a unit-based waste disposal user fee to households for the non-recycled waste fraction could lead to the same effects on producers as an upstream approach. This is so because households paying according to the quantity of waste they dispose off or recycle internalise in their purchasing decisions waste collection, recycling and disposal costs. In turn, it creates a demand on the product market for products with less waste which lead producers to alter their products in order to meet the demand. However, this result holds true in an ideal world which differs from reality. A first difference is that illegal disposal is of concern in reality; certain households might illegally burn or divert their own waste to avoid paying unit charge. It generates damages that may well outweigh the benefits of unit pricing in some cases. Second, obtaining an effect which has the same magnitude on product design decisions as that induced by an upstream approach is only possible if *all* the households intervening on a given product market are subject to unit pricing. In practice, choosing unit pricing is of the responsibility of local governments. For the moment, only a small minority of them have opted for this solution. This holds true even in countries promoting the approach; as an illustration, only 6,000 municipalities in the USA or 4% of the Danish population are covered by such pricing solutions. We will probably wait for decades before the generalisation of this option. In the meantime, the influence of households subject to unit pricing on producers is *de facto* limited by their dilution among the vast majority of consumers facing flat rate waste collection charge.¹¹ Third, as emphasized by Calcott and Walls (2000), recycling markets only function

10. For instance, this result is stated in Fullerton and Wu (1998) or Choe and Fraser (1999).

11. A further condition of the applicability of the above theoretical result on the equivalence of downstream and upstream approaches is the product market being competitive. Unit pricing creates a demand for less waste intensive products but product design is a matter for producers, not consumers.

imperfectly. In particular, it is too difficult and costly for recyclers to pay prices that perfectly reflect products' recyclability. Given the existence of such market transaction costs, the market fails to transmit upstream to the producers the right signal on recyclability.

Combining downstream and upstream approaches

Unit pricing and recycling subsidy thus cannot lead alone to efficient upstream design decisions. They are nonetheless irreplaceable in that there are the only instruments involving the households in the achievement of waste policy goals. And households are necessarily key actors of any waste policies since they are those purchasing and consuming the products, sorting waste in various separate collection programmes. It follows that one should combine both approaches. This is probably the most important message in the economic literature dealing with waste policies. How can it be done in practice?

First, note that the canonical EPR programme involving a PRO in charge of a take back mandate generally implements such a combination: an upstream tax paid by producers which finance downstream activities. One strong limit of these experiences is that they fail to significantly involve the households. In this respect, their sole actions take the form of information campaigns on the necessity of separate collection. Can producers do more under this regime? No, since the relevant instrument is unit based pricing of which adoption is the responsibility of local governments.

Traditional deposit-refunds that exist in many countries for beverages packaging (e.g. in the USA, Germany, Denmark, the Netherlands)) are other instances of upstream-downstream combinations. Under these systems, a fixed fee – the deposit – is charged to a consumer for purchase of a container of a given size and the fee is given back when the container is returned. What impacts can they have on product design? In fact, they are likely to be extremely limited. Because the fees are fixed, the system does not provide any incentives to weight reduction.

A further possibility, called the upstream combination tax/subsidy model (*UCTS*) has been proposed in the economic literature (see Walls, 2002). Under the *UCTS* system, producers of goods pay a weight-based tax and collectors of used products receive a weight based recycling subsidy. The product tax encourages source reduction while the downstream subsidy promotes recycling. The advantage of allocating the two policy goals – source and reduction and recycling – to each of the two instruments is administrative simplicity. In particular, the upstream tax does not need to take into account the products' recyclability and both taxes are simply weight based.

5.4 Other instruments

Tradable permits

Tradable permits are not widespread in waste policies. The only known experience is in UK where producers that face recovery and recycling requirements can either collect the proof that they have met their obligations, or they can buy from waste processors an official "Packaging waste Recovery Note" (PRN). PRN certifies that a certain waste quantity has been processed. If the supplier of waste packaging does not require the PRN, waste processors can sell it. In this way, the possibility to trade PRNs introduces flexibility while ensuring that the overall recycling objective is met.

If they are in a monopoly position, or more generally if the competitive pressure is low, producers might decide not to meet this demand. Is this condition met in practice? More or less, the answer is probably affirmative even though generalities on the intensity of market competition should be drawn cautiously. The main industries concerned – the car industry, EEE sectors, the food and beverages industry, etc. – are sectors in which the competition is tough.

What upstream incentives does such a system provide? The key point is that each producer must take care of its own waste, which can be done either directly – by having its own waste processed – or indirectly – by purchasing PRN from waste processors or from other producers. It thus provides the adequate incentives because the system tracks each producer's waste. In this respect, it seems similar to the system implemented by the PRO *ICT Milieu* gathering ITC producers in the Netherlands. Of course, this incentive capability is not without cost since it demands significant monitoring efforts. In the end, the UK experience suggests that tradable permits can perfectly promote product innovation if they are adequately designed.

Voluntary approaches

They are a category covering a wide variety of arrangements. They might be negotiated agreements achieved through bargaining between the industry and public authorities like the Dutch Packaging Covenant. Or, they might be unilateral commitments made by the industry as the well-known example of Responsible Care programme set by the chemical industry which embodies the very concept of product stewardship.

Given this variety, it is difficult to draw general conclusions on their potential for promoting innovation. For instance, when Veerman assesses the performances of Dutch EPR programmes (2002), he does not point out a significant difference between mandatory programmes (e.g. cars, car tyres, white and brown goods, batteries) and voluntary programmes (e.g. packaging, agricultural plastic films, waste paper/cardboard). By the way, differences between both types of programmes in terms of types of target, instruments or enforcement schemes appear to be minimal in the Netherlands.

In the USA, EPR or product stewardship is generally implemented on a voluntary basis (Palmer and Walls, 2002). They might be firm-level initiatives such as the Nike's Reuse-a-Shoe Program or the various recycling programmes implemented by manufacturers of computers (Dell, IBM, Gateway, HP-Compaq, etc.) or industry-led coordinated initiatives (e.g. the Rechargeable Battery Recycling Corporation, the Vehicle Recycling Partnership). A last possibility is to involve multiple stakeholders and public authorities (e.g. the Minnesota Electronics Recycling Initiative, the Carpet Stewardship Memorandum of Understanding). Whatever the form it takes, Palmer and Walls express some doubts on the possibility of these approaches to go beyond the business-as-usual trend (2002). This does not exclude positive “soft effects” associated with information sharing between participants on recycling costs and re-design possibilities.

6. Conclusion

Aiming at inducing upstream product change and innovation is a new challenge for waste policies. This paper is an exploratory analysis of the way to proceed to meet this goal. It builds both on the general economic literature on innovation and waste policies and on the OECD literature on EPR programmes. The major policy lessons are the following.

Products change continuously, regardless of the potential impacts of waste policies. In this context, the goal of waste policies is clearly not to initiate change. Instead, the challenge is to modify the pattern of business-as-usual product change in order to position goods on less waste-intensive innovation trajectories. When designing these policies, it is essential to take into account these BAU trajectories and the fact that they are industry specific.

It is worthwhile stressing the differences between packaging and non-durable goods on the one hand, and durable goods such as electronic equipments, household appliances and cars on the other hand. Concerning non-durable products, the challenge is mainly lightening, reducing the development of small containers and, substituting material by lighter and/or more easily recyclable material. The second group gathers more complex products involving more parameters influencing

waste generation (e.g. dismantling ability). For this second group, complete product redesign might be a critical part of reaching waste reduction goals. This requires radical innovation as opposed to incremental changes that are at stake for non-durables and packaging.

Clear and stable policy goals. Innovation is intrinsically highly risky. In this context, waste policies should not add further uncertainty. In particular, waste prevention involves several, possibly contradictory, policy sub-goals such as reduction in weight, in toxicity or increase in products' recyclability. Any policies aiming at promoting product innovation should establish a very clear and stable hierarchy among these different sub-goals.

Taxes and charges are more likely to induce innovation than standards. This is so because taxes or charges always provides firms with incentives to innovate. By contrast, such incentives vanish once regulatory thresholds are met.

There exists a trade-off between administrative costs and impacts on innovation. Waste policy instruments that promote upstream product innovation should provide producers with individual incentives to alter their products. In general, it implies a sophisticated design which is likely to rise administrative costs. For instance, in the case of waste packaging, advance disposal fees should take into account the packaging unit weight and the type of material used. The existence of such a trade-off between innovation incentives and administrative simplicity should be recognised from the outset.

Collective EPR programmes involving PROs may be useful tools to promote innovation. A first point is that *PROs are perfectly able to design financing schemes that provide producers with individual incentives to alter their products* as illustrated, among others, by the packaging fee structure implemented by many PROs in charge of packaging or by the Dutch PRO ICT Milieu dealing with EEE. However, counter-examples also exist and one role for the public regulator is to prevent PROs to implement financing instruments annihilating the competition between members on product characteristics.

A second point is that *PROs may themselves directly undertake cooperative R&D on waste prevention*. The economic advantage of cooperation is that it may reduce innovation costs by avoiding the duplication of R&D efforts. Furthermore, it helps mitigating the problem associated with product imitation arising when R&D is carried out on an individual basis. This general problem is related to the fact that other producers can very easily imitate product innovation. This reduces individual innovation incentive since the innovator only partly appropriate innovation benefits. R&D cooperation mitigates the problem by restricting the number of potential imitators to cooperation outsiders. Note that, in practice, the involvement of existing PROs in R&D is very limited.

A third point is related to *the liability principle underlying the EPR programmes*. In countries wherein unit-based pricing is not widespread, the advance disposal fee paid to the PRO is the only significant economic incentive influencing product design decisions upstream. In this context, the economic principle of social cost internalisation requires the fee rate to be made equal to the marginal social costs associated with waste management. Practically, a necessary condition for that is that the EPR programme should hold producers fully financially responsible of waste management costs. In countries wherein unit-based pricing is widespread (e.g. Switzerland, South Korea), producers also perceive the economic signals emitted by consumers on the product markets. In these countries, the principle of social cost internalisation can be compatible with a rule of shared financial responsibility between producers and local governments. The reason is that the advance disposal fee and the downstream unit based pricing are substitutes. Hence, when unit based pricing is implemented, the role of the ADF is to complement the incentive provided by unit based pricing if necessary, that is if unit based pricing delivers a price signal lower than the marginal social cost.

Combining upstream and downstream approaches. Promoting waste-related product innovation and re-design is a primary role for upstream instruments since they target producers who make product design decisions. It does not imply, however, that downstream instruments are useless in influencing product innovation. One important reason is that, in market economies, the ultimate impact of product design on waste streams depends on the commercial success of re-designed products. In this way, consumers play a crucial role and downstream instruments may better target them. 'Soft instruments' based on awareness raising campaigns are probably not sufficient and unit-based user charges whereby households pay waste collection and disposal services depending on the quantity of waste they generate individually appear irreplaceable.

Product design decisions are made at the market level. In practice, the scope of the market is frequently regional or global. This poses a problem to waste policies which are historically national, or even municipal to a large extent. If policy makers aim to influence product design, explicit or implicit coordination between national policies is necessary. Otherwise, national policy signals have little chance to be effective except if the country is very large.

Issues on which further work is needed. As the paper is exploratory, it is finally useful to identify future areas for research and analysis. A first point regards the ex post evaluation of existing EPR programmes. These programmes of which a primary goal is product change have been implemented in many OECD countries during the last decade. Nevertheless, ex post evidence on their impacts on waste prevention is hardly available. In our opinion, the second need is to develop product specific analysis of the relationships between product innovation and waste policies. We have underlined that the products generating waste differ dramatically in terms of technical, market or economic characteristics and waste-related issues. Also, we have argued that these specificities necessarily have important waste policy implications but we have not gone very far in this direction in this paper. Much remains to be done product by product, including on very practical questions (e.g. fee structure).

How can competition between producers co-exist with cooperation (e.g. in PROs)? It is another possible research question. We have seen that both mechanisms promote innovation. However, there exists the risk that cooperation undermines competition even though instances suggest that it is not systematic. In this regard, what are the possible policy safeguards?

ANNEX 1. PRODUCTS IN MUNICIPAL SOLID WASTE IN THE USA (2000)

Table A.1 provides recent US data on the contribution to waste production of different types of goods. Such figures differ from one country to another but US data is indicative of the general patterns. Four categories of products are distinguished: (i) durable goods (e.g. appliances, furniture, electronic equipments, cars); non-durable goods (e.g. newspapers, clothing); containers and packaging; and other wastes gathering essentially food scraps and yard trimming. The range of products concerned is in fact potentially very broad.

Table A.1 Generation of products in US municipal solid waste, 2000

	<i>Weight generated (Millions of tons)</i>	<i>Percent of total MSW</i>
Durable goods	36.3	15.7 %
Non-durable goods	63.7	27.5 %
Containers and packaging	74.7	32.2 %
Other wastes (food scraps and yard trimmings)	57.1	24.7 %
TOTAL	231.9	100.0 %

Source: Franklin Associates, Ltd.

ANNEX 2. CHANGE IN THE CONSUMPTION OF NON FOOD PRODUCTS OVER THE PERIOD 1979-1999 IN FRANCE

Table A.2 Consumption in weight of non-food products in France in 1979 and 1999

	1979	1999	Variation 1979-1999 (%)
Tobacco	5,3	4,1	-3,8
Clothing, shoes	27,3	20,4	-7,1
Coal	304,6	39,9	-83,7
Furniture, bedding	73,7	59,5	0,5
Textile household goods	2,9	4,0	72,3
Refrigerators	5,9	8,0	68,1
Washing machines	6,0	6,0	23,7
Dish washers	2,0	2,0	25,7
Stoves	1,8	1,7	19,6
Ovens, microwave	2,2	2,2	24,8
Batteries	0,9	0,8	19,4
Cleaning and hygiene products	25,9	29,7	42,4
Other hardware	12,3	9,8	-0,2
Drugs	2,6	7,2	239,2
Cars	87,3	103,9	48,2
Motorcycles	4,7	3,1	-17,3
Tyres	7,6	20,6	235,7
Oils, lubricants	0,5	1,4	266,7
Car batteries	0,1	0,8	1120,0
Telephone equipments	0,0	0,1	2262,2
Other white goods	1,4	1,7	52,5
Brown goods	3,5	6,7	140,0
Grey goods		1,3	
Leisure	21,7	21,2	21,7
Newsprint	68,9	58,9	6,4
Books	5,4	5,3	23,5
Total	674,3	420,4	-37,7
Total excluding coal	369,8	473,5	28,0

Source: ADEME, 2001.

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OECD PUBLICATIONS, 2, rue André-Pascal, 75775 PARIS CEDEX 16
PRINTED IN FRANCE
(97 2004 03 1 P) ISBN 92-64-10618-9 – No. 53387 2004