# A Comparative Study on Economic Instruments Promoting Waste Prevention

## Final Report to Bruxelles Environnement

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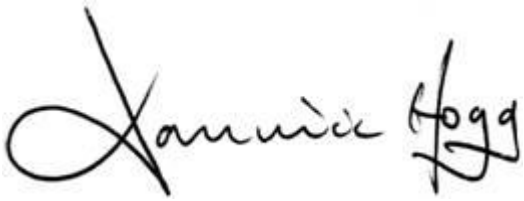

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# Contents

# 1.0 Introduction

This is the final deliverable (accompanied by the inventory spreadsheet) which is due to Bruxelles Environment as part of the comparative study on economic instruments promoting waste prevention.

This final report:

1. Outlines the list of economic instruments included in the inventory;
2. Highlights relative advantages and disadvantages for each instrument type included in the inventory;
3. Provides a justification for the instruments taken forward for further analysis;
4. Shows, through the meta-analysis, evidence of any waste prevention impacts that may be attributed to specific instruments;
5. Considers, through the meta-analysis, how novel instruments, that have not yet been widely used, could potentially be used to stimulate waste prevention; and
6. Identifies, for the economic instruments considered to have the potential to bring about waste prevention impacts, supporting actions or combinations with other instruments, which are likely to increase their effectiveness.

# 2.0 Inventory

The inventory (provided as a separate Excel spreadsheet) provides a list of examples of some key economic instruments which have either been used to stimulate municipal waste prevention, or that may have waste prevention impacts alongside their main intended effects. The inventory classifies these instruments and provides details, where they exist, on the waste prevention impacts. Each instrument is classified according to:

- Instrument type;
- Instrument sub-type;
- Country/region in which the instrument is operational;
- Responsible authority;
- The area of coverage (local, regional, or national);
- The material or waste stream which is covered;
- The sector covered; and
- Year of implementation.

The inventory of economic instruments, compiled by Eunomia, contains a total of 236 individual examples.

The most widely used type of instrument is 'Taxes, Fees and Charges' and under this category five sub-types were identified. These are shown in Table 1, along with the number of examples identified for each instrument in the inventory.

Two examples of tradable permit systems were identified.

Of the deposit-refund schemes, two sub-types were identified, with the majority of examples relate to beverage containers, with fewer examples relating to different types of products (e.g. for tyres, batteries and oil).

Fourteen examples of subsidies were included, with four sub types, identified.

Six examples of green procurement policies were identified, with no sub-categories being recognised under this instrument.

Please note, one example of a voluntary agreement was included in the initial inventory sent to Bruxelles Environnement, but after further consideration this has been removed.

As means of justification for this removal of voluntary agreements as a policy instrument type, it is important to refer to the project proposal in which it was stated that:

> *'The ITT states that all of the measures listed in Annex IV of the 2008 Waste Framework Directive are economic instruments. We disagree with this view, and for the purposes of this study, would, for example, seek specifically to exclude voluntary agreements from consideration, unless they are linked to an economic incentive, such as Government Green Procurement criteria, e.g. EPEAT in the United States.'*

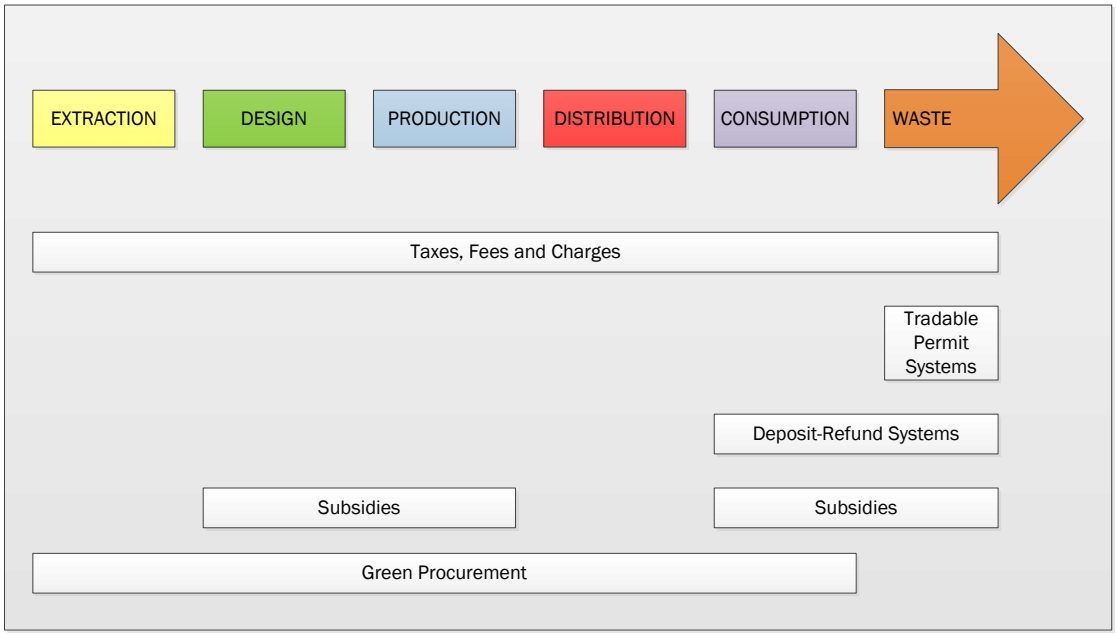
Under the classification system developed for the inventory green procurement has been identified as an instrument in its own right. As such, no voluntary approaches were identified which were linked to economic incentives. A number of voluntary agreements are in place which aim to improve resource efficiency; however, because of the nature of these agreements they fall outside of the realm of what can strictly be considered an economic instrument.

Table 1: Number of Examples Provided in the Inventory for each Instrument Type and Sub-Type

| Instrument Type | Instrument Sub-Type | No. of Examples in Inventory |
|---|---|---|
| Taxes, Fees and Charges | Disposal Tax | 26 |
| | Direct and Variable Rate (DVR) Charging | 18 |
| | Packaging Tax/Fee/Charge | 32 |
| | Product (excluding packaging) Tax/Fee/Charge | 103 |
| | Variable VAT Charge | 1 |
| | *SUB-TOTAL* | *180* |
| Tradable Permit Systems | Disposal | 1 |
| | Packaging | 1 |
| | *SUB-TOTAL* | *2* |
| Deposit-Refund Systems (DRS) | DRS for Beverage Containers | 24 |
| | DRS for Products | 10 |
| | *SUB-TOTAL* | *34* |
| Subsidies | Subsidised Home Composting Schemes | 5 |
| | Subsidies for Products | 3 |
| | Waste Prevention Subsidies (excluding home composting scheme subsidies) | 2 |
| | Loyalty Card Scheme | 4 |
| | *SUB-TOTAL* | *14* |
| Green Procurement | Green Procurement | 6 |
| | *TOTAL* | *236* |

Figure 1 represents the coverage of each instrument type across the entire waste stream. The following sections (Sections 2.1 to 2.5) describe each of the instrument-types in order to provide an introductory overview, and supporting text to the inventory itself.

Figure 1: Typical Areas of Coverage for Each Instrument Type along the Material Life-Cycle




## 2.1 Taxes, Fees and Charges

It is important to differentiate between a tax and a fee/charge. According to the OECD an environmental tax can be defined as:

> *'any compulsory, unrequited payment to general government levied on tax-bases deemed to be of particular environmental relevance'*.

The OECD states that

> *'Taxes are unrequited in the sense that benefits provided by government to taxpayers are not normally in proportion to their payments'*.[1]

Taxes frequently aim to internalise the externalities of certain activities or behaviours; for example, landfill taxes aim to balance the negative effects of landfilling on the environment and thereby incentivise alternative waste treatment options which are deemed to be more socially and environmentally desirable. They are paid irrespective of the level of service received from central government and are occasionally ear-marked for specific purposes.

Fees and charges, on the other hand, are levied in proportion to the costs associated with the provision of a service. For example, households are charged for a waste collection service or landfill sites apply a gate fee to cover the operational and management costs of the landfill site.

[1] OECD (no date given) *More Information on Environmentally Related Taxes, Fees and Charges*, Date Accessed: 2 June 2011, http://www2.oecd.org/ecoinst/queries/TaxInfo.htm

The rational choice model of behaviour change posits that human behaviour, at its simplest level, can be influenced by the provision of information together with an appropriate incentive to drive behaviour change. [2] The information provides the context while the incentive is meant to create the motivation to act on the information given. By acting as a clear financial incentive taxes and fees are widely used to modify people's (or industry's) behaviour. For example, in many – but not all – areas the use of direct and variable rate (DVR) charging systems for residual waste collections have been associated with marked decreases in waste arisings (DVR charging is also referred to as pay-as-you throw or pay-by-weight/volume). [3,4,5] Other examples include eco-taxes on products to incentivise consumers/producers to switch to using/producing more sustainable options. These include taxes on single use plastic carrier bags (e.g. Ireland, Italy, Denmark and South Africa), or taxes on disposable cutlery (e.g. Denmark).

## 2.2 Tradable Permit Systems

Only two examples were identified: the UK's Packaging Recovery Note system (PRNs) and the Landfill Allowance Trading Scheme in England (LATS), with the former scheme being focused predominantly on recycling and the latter influencing all tiers of the waste hierarchy.

The UK is, as far as we are aware, the only country which makes use of a system of landfill allowances. The Landfill Allowance Schemes (LASs) operate differently in the four countries of Northern Ireland, England, Scotland and Wales. [6] The LASs were designed specifically in response to the Landfill Directive and were intended to deliver compliance with the Article 5 targets under the Landfill Directive at least cost to the UK. The rationale, therefore, is specifically to reduce the amount of biodegradable waste being sent to landfill over time.

Allowances are issued to all local authorities with responsibility for waste disposal (waste disposal authorities, or WDAs). In England, the allowances have been made

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[2] Centre for Environmental Strategy (2005) *Motivating Sustainable Consumption: A Review of Evidence on Consumer Behaviour and Behavioural Change*, Report For the Sustainable Development Research Network, January 2005, www.c2p2online.com/documents/MotivatingSC.pdf

[3] Eunomia Research & Consulting (2006) *Impact of Unit-Based Waste Collection Charges*, Report for the Organisation for Economic Co-operation and Development, May 2006, www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV/EPOC/WGWPR(2005)10/FINAL&docLanguage=En

[4] Institute for Environmental Studies (2009) *Economic Instruments and Waste Policies in the Netherlands: Inventory and Options for Extended Use*, Report for the Dutch Ministry of Housing, Physical Planning and the Environment, March 2009

[5] Eunomia Research & Consulting (2006) *Financing and Incentive Schemes for Municipal Waste Management: Case Studies*, Report for the European Commission Directorate-General for Environment, http://ec.europa.eu/environment/waste/studies/pdf/financingmuncipalwaste_management.pdf

[6] At the time of writing (April 2009), the scheme has been suspended in Scotland.

tradable. Sanctions are to be applied in the event of non-compliance, these being set at punitive levels to deliver the required changes in management and infrastructure.

This scheme is due to be discontinued at the end of the 2012/13 financial year as the country's escalating landfill tax will be the main driver for landfill diversion at this point. [7]

In the UK the recycling of packaging material has been driven forward by the development of Packaging Recovery Notes (and Packaging Export Recovery Notes (PERNs)) and their subsequent trading as a means for providing verification that materials have been recycled. In essence, reprocessors issue PRNs as proof that a certain quantity of material has been recycled. These notes are then traded and can be purchased by organisations wishing to meet their targets. Historically, waste prevention has not been a priority of the PRN system which has focused largely on recycling. [8,9,10]

## 2.3 Deposit-Refund Systems

Deposit-refund systems (DRSs) are a particular form of product tax/recycling subsidy and have been defined as follows:

> *'A deposit-refund system is the surcharge on the price of potentially polluting products. When pollution is avoided by returning the products or their residuals, a refund of the surcharge is granted.' OECD, Glossary of Statistical Terms.* [11]

A DRS encourages the return of the materials into an organised reuse, recycling or treatment / disposal process. The producers typically finance the process through the payment of an administration fee on each container. Drinks containers are the most common target of DRSs, though economic theory suggests the schemes could be applicable to hazardous materials and other waste streams, subject to transaction costs being minimised. This instrument has also been used to promote the recovery of other products and materials, such as cars (Finland), tyres (Denmark, USA),

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[7] Department for Environment, Food and Rural Affairs (2011) *Government Review of Waste Policy in England 2011, June 2011*

[8] Department for Environment, Food and Rural Affairs (2011) *Advisory Committee on Packaging: Annual Report 2010/11*, June 2011, www.defra.gov.uk/publications/files/acp-report2010-11.pdf

[9] Perchards (2005) *Study on the Progress of the Implementation and Impact of Directive 94/62/EC on the Functioning of the Internal Market*, May 2005, www.perchards.com/files/documents/Final%20report%2020-6-05%20%28final%20v3%29.pdf

[10] European Environment Agency (2005) *Effectiveness of Packaging Waste Management Systems in Selected Countries: An EEA Pilot Study*, http://scp.eionet.europa.eu/publications/wp2005_2

[11] OECD (2001) *Glossary of Statistical Terms: Deposit-Refund System*, Date Accessed: 28 June 2011, http://stats.oecd.org/glossary/detail.asp?ID=594

batteries (Denmark, Mexico, Sweden, United States), WEEE (South Korea) and lubricating oil (Norway). [12]

The systems can encourage recycling and/or reuse where otherwise it is easy to dispose of containers with residual waste or for them to be discarded as litter. The same policy mechanism can also be used to target difficult to dispose of, or hazardous, items to ensure that these do not reach the residual waste stream.

DRSs are reported, in the literature, to have a range of possible environmental benefits. The key ones mentioned in the literature are:

1. Increasing the recycling of containers covered by deposits (for refill or recycling);
2. Reducing the extent of littering;
3. Increasing the use of / reducing the extent of decline in the use of refillables; and
4. Avoiding harmful chemicals being mobilised in the environment (usually not in beverage schemes, e.g. lead acid batteries, or pesticides).

## 2.4 Subsidies

Subsidies include any form of explicit financial assistance to polluters (grants, soft loans, tax breaks, accelerated depreciation, etc.). Subsidies are frequently used to fund innovation in design and production or to incentivise individuals to change the way in which they consume or manage their waste. There are many environmental subsides in place in countries across the globe, but few of them are directly targeted at promoting waste prevention. For many subsidies waste prevention may only be a small component of any particular programme and thus it is extremely difficult to determine the impact that these subsidies have on waste prevention *per se*.

A number of subsides exist to promote home composting in countries such as Canada, Italy, the United States and the UK. Recent research in the UK has suggested that over the course of a composting bins' operational lifetime, a typical local authority, rolling out such bins to residents, could realise a net saving of up to approximately €680,000. This is due to savings associated with disposal costs and gate fees through subsidising and promoting compost bins.[13]

Reusable nappies have also been widely promoted through the use of subsidies in countries such as Italy, New Zealand and the UK. Work by a UK charity, Go Real, has reported that approximately 3 billion disposable nappies, equalling 690,000 tonnes,

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[12] Kahhat, R., Kim, J., Xu, M., Allenby, B., Williams, E. and Zhang, P. (2008) Exploring E-Waste Management Systems in the United States, *Resources, Conservation and Recycling*, Vol.52, No.7, pp.955-964.

[13] Lets Recycle (2010) *Subsidising compost bins "could save £600,000",* Available: http://www.letsrecycle.com/news/latest-news/compost/subsidising-composting-bins-could-save-ps600-000, 20th July 2010.

are sent to landfill every year (this accounts for 4% of all waste sent to landfill). [14] It is reported further that if only 10% of UK households with new born children were converted to using reusable nappies the savings on disposal costs would amount to €6.2 million per annum.

## 2.5 Green Procurement

Green Public Procurement (GPP) policies focus on a range of goals including resource efficiency, energy and water conservation and waste reduction. They often include a recommended list of products or suppliers (often identified through eco-labelling), detailed language to be included in bid specification documents, a minimum recycled content requirement for particular products, a monitoring process and staff training requirements. As well as the environmental benefits (e.g. reduction of material extraction due to increased use of recycled material products). There are also economic benefits through creating a market pull for new environmental technology to be successful on the marketplace.

Using public procurement to secure certain social ends has a long history, starting in the UK and USA as early as the 1840's. Since this time government procurement has been used all over the world as a means of addressing social inequalities. In the 1990's this spread to the use of procurement as a means of securing environmental ends. 'Green' procurement policies sprung up as a means of addressing the growing concerns about sustainable development, and in the last decade have seen significant growth both in Europe and abroad. [15]

In 2004 the European Commission published two directives which detailed the requirements for public procurement within the EU (Directives 2004/17/EC and 2004/18/EC). These Directives laid the groundwork for the implementation of green public procurement (GPP) by enabling the inclusion of various environmental objectives in procurement processes. In 2008 the Commission published a Communication on *'Public Procurement for a Better Environment'* which outlined how the public sector could use its purchasing power to drive issues of sustainable consumption and production. In this Communication the European Commission set out a voluntary target for member states, which aimed to achieve a target of 50% green procurement by 2010.[16]

This communication defined GPP as *'a process whereby public authorities seek to procure goods, services and works with a reduced environmental impact throughout*

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[14] Go Real (2010) *Reducing Disposable Nappy Waste: Review of Waste Policies - Call to Evidence*, October 2010, www.goreal.org.uk/media/documents/Call_to_Evidence_v2_041010.pdf

[15] McCrudden, C. (2004) Using Public Procurement to Achieve Social Outcomes, *Natural Resources Forum*, Vol.28, 257-267.

[16] Commission of the European Communities (2008) *Public Procurement for a Better Environment - Communication from the Commission of the European Parleament, the Council, the European Economic and Social Committee and the Committee of the Regions*, July 2008, http://ec.europa.eu/environment/gpp/gpp_policy_en.htm

*their life cycle when compared to goods, services and works with the same primary function that would otherwise be procured'*.

Green procurement requires that specific criteria are developed for defining a product as green. There are frequently issues with regards to this and outside Europe no standardised approach seems to exist at present. Ideally, products should be compared on the basis of detailed life cycle assessments (LCAs) and a suitable product labelling scheme developed in parallel to GPP policies to help purchasing officers identify a product's relative performance. Green procurement is thus heavily depended on the presence of eco-labels to help identify favourable products. (In Europe a report was published in 2007, as part of the Union's GPP programme, which compared the life cycles of 'green' and 'non green' products to assist procurement decisions). [17]

[17] Oko-Institute e.V. and ICLEI (2007) *Costs and Benefits of Green Public Procurement in Europe - Part 1: Comparison of the Life Cycle Costs of Green and Non Green Products*, Report for the European Comission Directorate-General for Environment, July 2007, http://ec.europa.eu/environment/gpp/studies_en.htm

# 3.0 Advantages & Disadvantages of Each Instrument Type

A summary of the high level advantages and disadvantages associated with each instrument type are provided in Table 2. These are discussed in more detail in the sections which follow and should act to provide some deeper insights into the pros and cons of each instrument type. What is presented here is a broad level discussion that focuses on the overall aspects of each instrument and not on the intricate details of the numerous subsidiary instruments that may fall under each type.

It is widely acknowledged that economic instruments tend to be more economically efficient than conventional regulations (sometimes referred to as command-and-control regulations). However, it is also widely recognised that these instruments are not a panacea and frequently require strong supporting regulations or targets to ensure that they function efficiently (for example, the setting of competitive targets in cap-and-trade schemes).

Five different types of economic instruments were identified as part of this study and the advantages and disadvantages of each are now discussed in turn.

## 3.1 Taxes, Fees & Charges

Environmental taxes are widely used to raise revenue and change behaviours. Taxes have a clear advantage in that they can provide a strong incentive to the market and consumers to change their behaviour. Whether consumers or the market respond to this stimulus will be dependent on the level and degree of exposure to the tax, the presence of reliable information, and the availability of alternatives. For example, many countries in Europe have implemented landfill taxes which have caused industry and businesses to seek alternatives far more quickly than households. A review of the effectiveness of the landfill tax system in Holland – at the time the highest in the EU at €85 per tonne - suggested that:

> *'...the landfill tax did not have a significant direct impact on the generation of household waste, nor did it affect the choice for household waste disposal options. However, there may be an indirect effect if municipalities pass on the higher costs of landfilling to households by means of a unit-based charge (instead of a 'flat fee') on household waste (in 2004 29% of Dutch municipalities applied such a differentiation)'*.[18]

[18] Institute for Environmental Studies (2005) *Effectiveness of Landfill Taxation*, Report for VROM, November 2005, http://www.ivm.vu.nl/en/Images/Effective%20landfill%20R05-05_tcm53-102678_tcm53-103947.pdf

Table 2: Summary of the Key Advantages and Disadvantages Associated with Each Instrument Type

| Advantages | Disadvantages |
|---|---|
| **Taxes, Fees and Charges** | |
| ➢ Taxes, fees and charges provide clear incentives to which economically rational individuals can respond<br>➢ There are many examples of increased taxes and charges being associated with waste prevention at the household level (e.g. charge on carrier bags, direct and variable rate charging for waste collections, or taxes on disposable products)<br>➢ Compliance costs are likely to be lower than for tradable permit systems | ➢ There are arguments that poorly formulated taxes may distort the market<br>➢ Concerns over unfair competition and additional financial burdens on companies having to compete in a global market |
| **Tradable Permit Systems** | |
| ➢ Provides flexibility in the way targets are met and can lead some firms to innovate to a greater extent than they would under a tax | ➢ In many instances other policy or fiscal instruments will be driving change and thus all parties will ultimately have to make investments to ensure that they achieve their targets (this negates the benefits of having tradable allowances)<br>➢ Compliance costs may be higher than for taxes |
| **Deposit-Refund Systems** | |
| ➢ Encouraging the use of reusable beverage containers can aid in delaying the point at which an item becomes waste<br>➢ Deposits on products can potentially act as an incentive to increase an item's durability<br>➢ Such schemes may also function to promote improved designs for easier recycling at the end of a product's life | ➢ There is not always a strong waste prevention component to deposit-refund schemes<br>➢ Deposit-refund schemes for beverage bottles have begun focusing increasingly on the recycling of one-way containers<br>➢ Deposit systems for other products also frequently focus on the return of items for recycling |

| Advantages | Disadvantages |
|---|---|
| ➢ Deposits can also help to reduce litter and prevent abandonment of products (e.g. cars and tyres) | |
| **Subsidies** | |
| ➢ Can encourage behaviour change at the household level by providing information and removing barriers to more sustainable environmental actions (e.g. the provision of reusable nappies together with the offer of a subsidised laundry service).<br>➢ Subsidies for 'good' behaviour can theoretically act as incentives to reduce waste | ➢ Requires extensive and holistic behaviour change campaigns to ensure that households take up the desired behaviour (e.g. home composting or the use of reusable nappies) and persevere for an extended period of time<br>➢ In communities with very mobile populations investments by municipalities in behaviour change may be 'lost' as people move away (e.g. subsidised home composting) |
| **Green Procurement** | |
| ➢ Green public procurement can comprise a significant proportion of total expenditure within a country; thus, moves to enforce green procurement measures can have extensive impacts on the design, composition, production and distribution of products/materials.<br>➢ There is clear evidence suggesting that GPP is environmentally beneficial; however, links to waste prevention are more limited | ➢ Issues with competition and some concerns have been raised about the effect of green procurement on market efficiency<br>➢ The criteria by which products are deemed to be 'green' may sometimes be somewhat dubious and unclear<br>➢ Environmental outcomes can be strongly dependent on the provision of clear guidance and purchasing criteria<br>➢ Waste is often only a small component of a product's 'green' criteria and thus in many instances it is difficult to quantitatively assess what impacts green procurement has on waste prevention<br>➢ Where government expenditure within an economy is only a small fraction of total purchasing power they will have to seek to obtain cooperation from individuals and the private sector (a government's ability to drive innovation is related to their overall purchasing power) |

In this instance the implementation of a unit-based charge for waste collection ensures that the economic signal of the tax is more effectively passed on to households. In many instances the use of unit-based charging has been shown to be an effective means of reducing household waste.[19]

The fact that some actors may choose not to respond to financial incentives is a potential weakness of market-based mechanisms. This has been picked up in the literature and highlighted as a potential limitation of such approached relative to 'command-and-control' policies with strict regulations and imposed limits. The latter, however, have their own issues and have been blamed for causing distortions in the market. Taxes allow individual operators to make their own decisions as to the most economically viable options for mitigating their environmental impacts.[20] A lengthy debate between the two approaches will not be entered into here, where the main task remains to highlight the broad advantages and disadvantages of environmental taxes and charges.

If designed effectively taxes clearly have an important role to play in any integrated waste management package; however, a number of concerns are frequently raised with regards to their implementation and operation:[21, 22]

- Concerns over the effect of taxes on competition and placing additional financial burdens on companies having to compete in a global market;
- Issues surrounding market interference;
- There may be distributional impacts (i.e. some social groups may be disproportionately affected);
- Taxes applied to products with low price-elasticity (e.g. fuel) may be ineffectual ;
- There may also be uncertainty in what the final outcome may be – there is no guarantee that a desired rate of pollution abatement or waste diversion will be achieved;

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[19] Eunomia Research & Consulting (2006) *Impact of Unit-Based Waste Collection Charges*, Report for the Organisation for Economic Co-operation and Development, May 2006, http://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV/EPOC/WGWPR(2005)10/FINAL&docLanguage=En

[20] Bailey, I. (2002) European Environmental Taxes and Charges: Economic Theory and Policy Practice, *Applied Geography*, Vol.22, No.3, pp.235-251.

[21] Fullerton, D., Leicester, A. and Smith, D. (2010) *Environmental Taxes*, In *Dimensions of Tax Design*. Ed. Institute for Fiscal Studies (IFS). Oxford: Oxford University Press, 2010, http://works.bepress.com/don_fullerton/37

[22] Institute for Fiscal Studies (2006) *The UK Tax System and the Environment*, October 2006, www.esmeefairbairn.org.uk/docs/tax_system_environ.pdf

## 3.2 Tradable Permit Systems

Tradable permit systems have been used widely to control the emission/discharge of potential pollutants from industry; however, they have seen little application within the waste sector. The only two examples of which we are aware come from the UK, and include the PRN/PERN system and England's Landfill Allowance Trading Scheme.

Baily has compared the packaging regulations in Germany and England in 2002. [23] The UK chose to implement its packaging obligations by using a tradable permit scheme to achieve the minimum targets set out by the EU, while Germany chose to rely far more heavily on direct regulation. The two approaches provide an interesting comparison and Bailey concludes that:

> *'Whilst recycling in Germany has undoubtedly benefited from high Green Dot charges and the co-ordinating powers of the DSD [Dual System Deutschland], the absence of competitive pressures in the system has exacerbated economic inefficiencies. By contrast, Britain has increased recycling at significantly lower relative cost by adopting market-based pricing'.*

The European Environment Agency has summarised the PRN/PERN system as follows:

> *'It is a particularly complex system…due to DEFRA's attempts to design a system that was supported by as many branches of industry as possible. The system was designed to enable businesses to comply with their obligations at the lowest possible cost, and this aim appears to have been fulfilled: the financing need per tonne of packaging waste recovered is low compared with other countries. To achieve this, PRN prices are governed by supply and demand for the recycling and recovery of packaging. If recovery capacity falls below that needed to meet recovery targets, PRN demand exceeds supply and prices rise. However, prices are very unstable and do not include other costs such as those associated with data collection, registering with the Agencies and dealing with fraud'*.[24]

Thus, it is evident that while the use of tradable permit systems may be economically efficient they do not necessarily result in high recovery rates without ambitious targets being set. Indeed, the lack of ambitious targets in the past has resulted in much volatility in the value of PRNs. Prices have tended to rise as target deadlines approached, only to crash soon after it was clear that they had been achieved.

England's Landfill Allowance Trading Scheme has also had a number of issues since its launch in 2005. As a trading scheme it has not functioned very well as most players in the market have faced the same costs in setting up residual waste treatment facilities. Undoubtedly, some local authorities are ahead of

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[23] Bailey, I. (2002) European Environmental Taxes and Charges: Economic Theory and Policy Practice, *Applied Geography*, Vol.22, No.3, pp.235-251.

[24] European Environment Agency (2005) *Effectiveness of Packaging Waste Management Systems in Selected Countries: An EEA Pilot Study*, 2005, http://scp.eionet.europa.eu/publications/wp2005_2

others in terms of their waste management infrastructure, but due to the nature of the market this has not seemed to provide them with a competitive edge.

A further drawback of both trading systems described above is that they are not specifically focused on waste prevention. The PRN/PERN system looks to promote recycling, while the LATS only aims to divert biodegradable waste form landfill for alternative treatment.

## 3.3 Deposit-Refund Systems

DRSs are reported, in the literature, to have a range of possible environmental benefits.[25] The key ones mentioned are:

- Increasing the recycling of containers covered by deposits (for refill or recycling);
- Reducing the extent of littering;
- Increasing the use of / reducing the extent of decline in the use of refillables; and
- Avoiding harmful chemicals being mobilised in the environment (usually not in beverage schemes, e.g. lead acid batteries, or pesticides).

Many theoretical studies have recommend DRSs as economically efficient mechanisms to increase rates of recycling.[26] However, there have been some concerns about the administrative costs of running such schemes, although these are also said to be offset by higher recovery rates.

## 3.4 Subsidies

An advantage of subsidies is that they can be used constructively to change behaviour or improve the design and production of goods (e.g. by designing out the use of hazardous substances, or increasing the durability of a product). The choice of

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[25] Eunomia Research & Consulting (2010) *Have We Got the Bottle? Implementing a Deposit Refund System in the UK*, Report for Campaign to Protect Rural England (CPRE), September 2010, www.cpre.org.uk

[26] See, for example, Dinan, T.M. (1993) Economic Efficiency Effects of Alternative Policies for Reducing Waste Disposal, *Journal of Environmental Economics and Management*, 25: 242-256.; Fullerton, D. and Kinnaman, T. (1995) Garbage, Recycling and Illicit Burning or Dumping, *Journal of Environment Economics and Management*, 29: 78-91; Pearce, D.W. and R.K. Turner (1993) Market-based approaches to solid waste management, *Resources, Conservation and Recycling* 8: 63-90. Porter, R.C. (1978) A Social Benefit Cost Analysis of Mandatory Deposits on Beverage Containers, *Journal of Environmental Economics and Management*, 5: 351-375; Sigman, H. (1995) A Comparison of Public Policies for Lead Recycling, *Rand Journal of Economics* 26: 452-478; Thomas Skinner and Don Fullerton (1999), The Economics of Residential Solid Waste Management, *NBER Working Paper 7326* http://www.nber.org/papers/w7326; Palmer, K. and Walls, M. (1999) Extended Product Responsibility: An Economic Assessment of Alternative Policies, *Discussion Paper 99-12*, January 1999, Washington DC: Resources for the Future; Don Fullerton and Amy Raub (2003) Economic Analysis of Solid Waste Management Policies, in OECD (2004) *Addressing the Economics of Waste*, Paris: OECD.

which materials or sectors of the economy a government should subsidise requires extensive information: detailed knowledge of the broader environmental and economic implications is essential. Accordingly, subsidies can be used to counteract market distortions, such as when governments choose to subsidise healthier low fat foods in order to adjust for the fact that the costs of obesity are being disproportionately carried by the population at large. [27]

It has been shown by Calcott and Walls that downstream taxes or charges on waste disposal do not in themselves improved design measures being taken further upstream. [28] This is due to markets functioning imperfectly, and here too subsidies can play a role by distributing some of the revenue from these taxes to fund waste prevention measures further upstream.

Subsidies can be given to single or multiple products/sectors, they can be geographically contained (as is the case with municipal waste prevention schemes), or focused on a particular production factor, such as labour or capital. The use of subsides to promote environmental goals has on occasion led to accusations of market distortions. It is unclear, however, to what extent this applies to waste prevention initiatives at the local level. It is possible, for example, that there could be resistance from the producers of disposable nappies if reusable nappies were suddenly heavily subsidised by a large number of municipalities (a rather unlikely scenario). Subsidies for particular industry sectors can also be controversial and may be subject to strong opposition from those who do not qualify for the additional support.

A further general concern with subsidies is that it is sometime difficult to know whether the environmental benefits of the subsidy is justified in relation to the cost of the investment. In countries which have a landfill tax that accurately reflects the externalities of landfilling there are likely to be strong economic incentives for reducing waste arisings. In such instances it may be easier to justify relatively large subsidies aimed at waste prevention measures, as the costs of disposal are high. Where the costs of disposal are minimal it may be necessary to rely on life cycle assessments to justify the subsidy. This is frequently more ambiguous and far more difficult to substantiate. For example, a life cycle assessment conducted by the UK's Environment Agency compared reusable and non-reusable nappies and found that reusable nappies only outperformed their competitors when managed under certain quite specific conditions (e.g. washing at 60 $^{o}$C in A-rated washing machines, minimising the use of tumble driers etc.).[29] Therefore, based on the premises of this

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[27] Finkelstein, E., French, S., Variyam, J. N. and Haines, P. S. (2004) Pros and Cons of Proposed Interventions to Promote Healthy Eating, *American Journal of Preventive Medicine*, Vol.27, No.3, Supplement 1, pp.163-171.

[28] Calcott, P. and Walls, M. (2000) Can Downstream Waste Disposal Policies Encourage Upstream "Design for Environment"?, *The American Economic Review*, Vol.90, No.2, pp.233-237.

[29] Environment Agency (2008) *An Updated Lifecycle Assessment Study for Disposable and Reusable Nappies*, October 2008, http://randd.defra.gov.uk/Document.aspx?Document=WR0705_7589_FRP.pdf

study, it would only be worth switching to reusable nappies if it could be reasonable assumed that they were managed within the given parameters set out by the study. Such uncertainty is undesirable and can make it difficult to justify subsidies in certain instances (this does not apply in the case of the UK, where landfill taxes are set to rise to £80 per tonne by 2014/15 and nappies are said to comprise up to 4% of the waste stream).[30]

## 3.5 Green Procurement

Some of the main advantages and disadvantages of green public procurement (GPP) will be outlined below, with particular emphasis being given to the instrument's ability to contribute to waste prevention. A report on green public procurement published by the OECD in 2003 stated that although total government spending – as a proportion of GDP – may be high, it was frequently distributed between central government (30% – 35%) and numerous other government bodies (65% - 70%). Given this distribution in spending it is suggested that:

> *'At most, procurement policies co-ordinated across all levels of government will directly affect, on average, only 20% of purchases in a targeted market. Similarly, policies co-ordinated at the central government will directly affect, on average, only 5 to 7% of purchases in a targeted market'.*[31]

As a result of this limited influence over the purchasing market it is argued that the *'potential environmental benefits of GPP may be small'*. However, in examining a number of case studies on GPP in various countries it was found that many succeeded in meeting their desired goals or achieving positive environmental impacts. Nevertheless, it was emphatically stated that the *'general lack of environmental data to assess GPP is alarming. Particularly since these programmes and policies [i.e. those included as part of the OECD review] are "advanced" compared to those not selected for this review'.* The report goes on to say that: *'this lack of concrete data could very well jeopardise the credibility of GPP in the future if efforts are not made to measure and demonstrate their environmental effectiveness'*.

There has been a growing focus on GPP over the last decade and developing indicators to assess the impact of such policies. [32] In light of this it has been reported that in Europe GPP policies contributed to an *'average reduction of $CO_2$ emissions of*

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[30] Go Real (2010) *Reducing Disposable Nappy Waste: Review of Waste Policies - Call to Evidence*, October 2010, www.goreal.org.uk/media/documents/Call_to_Evidence_v2_041010.pdf

[31] OECD (2003) *The Environmental Performance of Public Procurement: Issues of Policy Coherence*, September 2003, www.oecdbookshop.org/oecd/display.asp?lang=EN&sf1=identifiers&st1=9789264101555

[32] PricewaterhouseCoopers, Significant and ECOFYS (2009) *Collection of Statistical Information on Green Public Procurement in the EU: Report on Methodologies*, Report for the European Comission Directorate-General for Environment, January 2009, http://ec.europa.eu/environment/gpp/studies_en.htm

*25% in 2006/2007 when purchasing green'* (this was for the ten product groups covered in the cited study). According to the authors, *'This means that public purchasers have the possibility to substantially reduce $CO_2$ emissions through GPP'*.[33] Evidence also seems to suggest that such policies can drive technological innovation and force producers to spend more on research and development (R&D). [34, 35] In addition, and in contrast to common perceptions, it has also been reported that GPP in Europe has actually led to an overall decrease in procurement costs. [36, 37]

A number of studies have looked to quantify the environmental benefits of green procurement programmes, but these have tended to focus on defining these benefits in terms of greenhouse gases. [38, 39] There is very little published material on how these schemes have contributed to waste prevention.

The flexibility of green procurement policies to respond to local environmental conditions is a distinct advantage. However, GPP has a drawback in that it is frequently bound by international and European trade agreements which do not allow preferential treatment of local companies (e.g. the WTO's Government Procurement Agreement). Countries that are not bound by these agreements are free to favour their own local or national industries; for example, China has reportedly instituted a procurement policy which actively discriminates against innovation being sourced from foreign companies.[40] With environmentalist's increasing interest in 'localism' this may be seen as a disadvantage by some.

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[33] PricewaterhouseCoopers, Significant and ECOFYS (2009) *Collection of Statistical Information on Green Public Procurement in the EU: Report on Methodologies*, Report for the European Comission Directorate-General for Environment, January 2009, http://ec.europa.eu/environment/gpp/studies_en.htm

[34] Edler, J. and Georghiou, L. (2007) Public Procurement and Innovation - Resurrecting the Demand Side, *Research Policy*, Vol.36, No.7, pp.949-963.

[35] Oko-Institute e.V. and ICLEI (2007) *Costs and Benefits of Green Public Procurement in Europe - Part 3: The Potential of GPP for the Spreading of New/Recently Developed Environmental Technologies - Case Studies* , Report for the European Comission Directorate-General for Environment, July 2007, http://ec.europa.eu/environment/gpp/studies_en.htm

[36] Oko-Institute e.V. and ICLEI (2007) *Costs and Benefits of Green Public Procurement in Europe - Part 3: The Potential of GPP for the Spreading of New/Recently Developed Environmental Technologies - Case Studies* , Report for the European Comission Directorate-General for Environment, July 2007, http://ec.europa.eu/environment/gpp/studies_en.htm

[37] See Footnote 12 above

[38] PricewaterhouseCoopers, Significant and ECOFYS (2009) *Collection of Statistical Information on Green Public Procurement in the EU: Report on Methodologies*, Report for the European Comission Directorate-General for Environment, January 2009, http://ec.europa.eu/environment/gpp/studies_en.htm

[39] Efektia Ltd (2005) *Measuring the Environmental Soundness of Public Procurement in Nordic Countries*, Report for Nordic Council of Ministers, Copenhagen, 2005, www.norden.org/sv/publikationer/publikationer/2005-505

[40] Edler, J. and Georghiou, L. (2007) Public Procurement and Innovation - Resurrecting the Demand Side, *Research Policy*, Vol.36, No.7, pp.949-963.

Green procurement programmes also require that purchasing officers are suitably trained and that good guidance on the procurement/tendering process are widely disseminated.[41] This is widely acknowledged and many of the better functioning schemes have incorporated within them clear guidance documents and sufficient training regimes to ensure that tendering processes are carried efficiently and in accordance with the green objectives of the policy.

In 2003 the OECD outlined a number of barriers to effective and efficient GPP:[42]

- Management and organisational cultural barriers
    - As discussed above the lack of reliable information on green products limits the effectiveness of any programme;
    - Procurement decisions are becoming increasingly decentralised making coordinated responses more difficult and diluting spending power;
    - Procurement officers seldom use the equipment they purchase and thus training is needed to ensure an efficient and effective green procurement process; and
    - GPP policies are frequently voluntary and are therefore seldom subjected to detailed public scrutiny.
- Budget and financial barriers
    - It is frequently felt that there are insufficient funds to pay the premiums on green or eco-labelled products (green products frequently have higher initial costs, but lower long-term operating cost; for example, due to increased energy efficiencies or increased lifespans)
    - Many contracts are awarded on “lowest price”; and
    - Issues have also been raised about the differentiation of capital and operational costs within different departments.
- Informational barriers
    - In many instances the lack of targets and indicators are a significant drawback to green procurement; and
    - Comprehensive LCAs of different products are required in order to set accurate targets and enable detailed reporting.

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[41] Li, L. and Geiser, K. (2005) Environmentally Responsible Public Procurement (ERPP) and its Implications for Integrated Product Policy (IPP), *Journal of Cleaner Production*, Vol.13, No.7, pp.705-715.

[42] OECD (2003) *The Environmental Performance of Public Procurement: Issues of Policy Coherence*, September 2003, www.oecdbookshop.org/oecd/display.asp?lang=EN&sf1=identifiers&st1=9789264101555

The European Commission has done much to address these issues and a number of reports have been published as part of the Commission's programme of promoting the uptake of GPP within member states.[43]

# 4.0 Instruments for Further Assessment

The inventory of economic instruments provides an extensive list of specific examples, each classified under one of five instrument types and further still under 12 possible sub-types (see Table 1)

The list of instruments was reviewed in detail to determine which instrument sub-types showed the greatest promise in terms of the selection criteria outlined in Section 4.1 below. The preferred instruments will be subject to a more detailed meta-analysis as part of the next phase of research.

## 4.1 Selection Criteria

The decision regarding which instruments to recommend for further analysis was based on a number of factors, as described in the four key points below.

1. **Strength of evidence / accessibility of data regarding the impact of the instrument on municipal waste prevention.**

The extent to which the impact can be quantified is reliant upon the ability to access data from which the waste prevention impact can be derived. Whilst access to data may be linked with the frequency with which the instrument has been implemented, the search should not be narrowed to only the most popular of instruments, as there may be some potentially innovative, less-known instruments of interest.

In many instances data on waste prevention is very limited as measurement of this has only become a significant priority over recent years. This shifting priority has been reflected in the increased focus on developing appropriate indicators to accurately measure waste prevention impacts. Over the last decade, significant work has been undertaken on developing indicators for measuring the impact of waste prevention policies and projects at various scales and for different waste streams (i.e. municipal, industrial, construction and demolition waste etc.). The OECD has been actively involved in this area since 2000 and more recently the European Commission engaged consultants for a study which looked at developing a standardised set of European indicators. [44,45]

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[43] For more details see: European Commission, Environment (2011) *Green Public Procurement*, Date Accessed: 26 July 2011, http://ec.europa.eu/environment/gpp/index_en.htm

[44] Organisation for Economic Co-operation and Development (2004) *Towards Waste Prevention Performance Indicators*, September 2004, www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV/EPOC/WGWPR/SE%282004%291/FINAL&docLanguage=En

[45] BIO Intelligence Service (2009) *Waste Prevention: Overview on Indicators*, Report for the European Comission Directorate-General for Environment, November 2009, http://ec.europa.eu/environment/waste/prevention/pdf/WPG_indicators.pdf

Given the changing context of waste prevention indicators in recent years, and the lack of a standardised approach, comparison of different instruments can be challenging as indicators vary and can either be focused on outputs or outcomes.

Output indicators relate directly to the size or level of participation in an activity (e.g. the number of households choosing to undertake home composting or the number of local authorities with green procurement policies in place), whereas outcome indicators assess actual changes in the mass flow (e.g. changes in municipal waste tonnages). With respect to these two types of indicators a report by Arcadis *et al* states that:

> *'If an output indicator depends upon a direct measurement of the application of an instrument, you have detailed information on the instrument but you do not know the real impact of this instrument on the environment. If an outcome indicator measures the impact directly, you have detailed information on the impact but you are uncertain on the relationship between the instrument and the impact. Both categories of indicators cannot be integrated but they are both necessary to make meaningful judgements on the applied prevention policies'*. [46]

2. **The potential scope of the instrument (geographical coverage and range of materials)**

In short-listing the instruments for more detailed analysis, it is not essential, but would be more interesting, to focus on instruments which look at a variety of materials (e.g. organic waste, dry recyclables, niche materials) and which also cover a range of geographical scopes (e.g. local-level and national-level implementation).

3. **Point(s) of intervention, i.e. the point(s) at which the instrument focuses its activities, for example, at the waste, consumption or production phase**

As shown in Figure 1, the instrument types impact across different areas of the waste stream. Arguably, there is greater benefit from those instruments impacting during the earlier stages, than those impacting most significantly in the later stages. However, it may be interesting to consider a range of instruments that focus on different stages of the material life cycle.

4. **Transferability and applicability of the instrument to other regions or countries / Potential for novel application**

There is thought to be greater value in investigating instruments which can theoretically be implemented in a range of regions/countries. Investigating instruments of this nature will result in findings which are applicable to a far wider audience. In addition, there may be examples of instruments that have not been

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[46] Arcadis, VITO, Umweltbundesamt and BIO Intelligence Service (2010) *Analysis of the Evolution of Waste Reduction and the Scope of Waste Prevention*, Report for the European Commission DG Environment, October 2010, www.eu-smr.eu/wasterp/; pp. 35.

widely used, and for which evidence of impacts is lacking, but where it appears that the potential exists for their application in new areas to bring about waste prevention.

## 4.2 Short-listed Instruments

The above selection criteria provides a series of limiting factors to the selection process which, in an ideal world, would all be met and complied with. Using the selection criteria as a framework we have looked at the instruments in as broad a fashion as possible to choose a representative cross section of possible approaches to promoting waste prevention at the municipal level.

The following grading system has been applied:

- Data accessibility
  - data accessibility poor ✓
  - data accessibility acceptable ✓✓
  - data accessibility good ✓✓✓
- Scope of the instrument

  *The aim is to ensure that the final package of recommended instruments collectively cover a wide scope (in terms of both targeted materials and geographical scope).*
  - Scope of instrument poor ✓
  - Scope of instrument acceptable ✓✓
  - Scope of instrument good ✓✓✓
- Point of intervention
  - Whilst there is arguably greater environmental benefit through preventing waste early in the material life-cycle, it is clear from Figure 1 that it is more common for economic instruments impacting household-level waste prevention to occur later in the material life-cycle. This selection criteria is therefore aiming to ensure that the final package of recommended instruments collectively cover the relevant stages of the life-cycle. To this end, there is no grading system as such, but a sense-checking process to ensure this is achieved.
- Transferability / Potential for Novel Application
  - Economic instrument transferability / potential for novel application poor ✓
  - Economic instrument transferability / potential for novel application acceptable ✓✓
  - Economic instrument transferability / potential for novel application good ✓✓✓

Table 3 shows the results of this selection process. It can be seen that six instrument sub-types are to be taken forward for further analysis and investigation, and seven were felt not to be suitable for detailed analysis. Where there is sufficient data

availability, a quantitative analysis will be undertaken, to the extent possible. This is the case for Direct and Variable Rate (DVR) Charging, Product (excluding packaging) Taxes/ Fees/ Charges, and Subsidies for Products, all denoted by (Q) in Table 3. Where there is insufficient data on waste prevention effects, the analysis will be more descriptive. This will be the case for Variable VAT Charge, where despite a lack of evidence, the instrument is thought to have considerable potential for novel application, and the analysis will focus on how this might be undertaken. In the case of DRS for Beverage Containers, and for Packaging Tax/Fee/Charge, while the focus is very much on recycling and recovery, (and possibly preparation for reuse in the case of washable glass bottles), Bruxelles Environnement requested that these be investigated, and consideration will be given to the wider effects of these instruments.

Short sections outlining the justifications for these recommendations are in Sections 4.3 and 4.4.

Table 3: Selection of Instrument Sub-Types which Eunomia Suggests Taking forward for Further Analysis

| Instrument Sub-Type | Selection Criteria | | | | To be taken forward for further analysis? | Brief Justification |
|---|---|---|---|---|---|---|
| | Data Accessibility | Scope of Instrument | Point of Intervention | Instrument Transferability / Potential for Novel Application | | |
| Direct and Variable Rate (DVR) Charging | ✓✓✓ | ✓✓✓ | Disposal | ✓✓✓ | Yes (Q) | Extensive and reliable data; widely implemented; flexible over range of scales |
| Product (excluding packaging) Taxes/ Fees/ Charges | ✓✓✓ | ✓✓✓ | Consumption | ✓✓✓ | Yes (Q for bags) | As part of this sub-group we propose investigating taxes applied to products with the intention of shifting consumer choices to 'greener' reusable options. Taxes/charges provide clear incentives and have been associated with reducing consumption |
| Subsidised Home Composting Schemes | ✓✓ | ✓✓✓ | Consumption /Disposal | ✓✓✓ | No | Data on waste prevention impacts available; potentially very beneficial for the environment; very flexible instrument that can easily be implemented. However, it is already considered by Bruxelles Environnement to be an established and accepted method for waste prevention requiring no further analysis. |
| Subsidies for | ✓✓ | ✓✓✓ | Consumption | ✓✓✓ | Yes (Q) | Data on waste prevention |

| Instrument Sub-Type | Selection Criteria | | | | To be taken forward for further analysis? | Brief Justification |
|---|---|---|---|---|---|---|
| | Data Accessibility | Scope of Instrument | Point of Intervention | Instrument Transferability / Potential for Novel Application | | |
| Products | | | /Disposal | | | impacts available; potentially very beneficial for the environment; very flexible instrument that can easily be implemented |
| DRS for Beverage Containers | ✓✓ | ✓✓ | Consumption /Disposal | ✓✓ | Yes (D) | Where reusable containers are still used (e.g. Germany, Finland, Netherlands) can promote preparation for reuse; large scale. Not focused on waste prevention *per se*. |
| Green Procurement | ✓✓ | ✓✓✓ | Consumption (& longer term subsequent upstream impacts) | ✓✓✓ | No | Data on waste prevention is limited; potential for large and far reaching impact in terms of broader environmental considerations |
| Disposal Tax | ✓ | ✓✓✓ | Disposal (& longer term subsequent upstream impacts) | ✓✓✓ | No | Little evidence directly linking instrument to waste prevention at the household level. However, it should be recognised that this instrument is likely to be extremely important in understanding the context for application of other instruments. |

| Instrument Sub-Type | Selection Criteria | | | | To be taken forward for further analysis? | Brief Justification |
|---|---|---|---|---|---|---|
| | Data Accessibility | Scope of Instrument | Point of Intervention | Instrument Transferability / Potential for Novel Application | | |
| | | | | | | Implementation of economic instruments which directly impact on the householder may potentially be driven by a national level disposal tax. It is therefore anticipated that dependencies between disposal taxes and other economic instruments will be significant. |
| Packaging Tax/Fee/Charge | ✓✓ | ✓✓ | Design, production, consumption | ✓✓✓ | Yes (D) | Instruments focused predominantly on recovery and reuse, rather than waste prevention |
| Variable VAT Charge | ✓ | ✓ | Consumption | ✓✓✓ | Yes (D) | Not widely used; no data on waste prevention effects; however, could conceivably have a waste prevention effect if applied in certain situations |
| Tradable Permit Systems | ✓ | ✓ | Disposal | ✓ | No | Only two examples identified and no strong links to municipal waste prevention |
| DRS for Products | ✓ | ✓ | Disposal | ✓✓✓ | No | These instruments typically focus on incentivising the return of products for recycling |
| Waste Prevention Subsidies (excluding home composting scheme subsidies) | ✓ | ✓ | Consumption /Disposal | ✓✓✓ | No | Subsidies are used to fund various activities, depending on local needs; no studies were identified which sought to assess the impacts of |

| Instrument Sub-Type | Selection Criteria | | | | To be taken forward for further analysis? | Brief Justification |
|---|---|---|---|---|---|---|
| | Data Accessibility | Scope of Instrument | Point of Intervention | Instrument Transferability / Potential for Novel Application | | |
| | | | | | | these subsidies (such subsidies may be used to promote composting or the use of reusable nappies) |
| Loyalty Card Schemes | ✓ | ✓✓ | Consumption /Disposal | ✓✓ | No | There appears to be no data demonstrating the effects on waste prevention; these schemes can be costly to operate. There is possible evidence to suggest a 'rebound' effect. |

## 4.3 Instruments Chosen for Analysis

The following sections describe those instruments which have been chosen for analysis, along with a summary table outlining the results of the qualitative selection framework criteria.

### 4.3.1 Direct & Variable Rate (DVR) Charging

This instrument was chosen predominantly because a significant body of research exists on the subject and much of the research suggests that DVR charging schemes lead to reductions in household waste arisings. The schemes are typically implemented locally (e.g. Belgium, Germany, Sweden, USA), but in the case of South Korea a nation-wide DVR charging scheme has been in place since 1995. Their wide application, flexibility and extensive use means that such schemes have been well tested and many lessons learnt.

Eunomia has undertaken extensive work in the area of DVR charging and will draw on this experience to provide a sound assessment and analysis of such schemes. [47,48]

### 4.3.2 Product Taxes / Fees / Charges

Taxes, fees and charges have been applied to numerous products and are widely used as a means of promoting recovery, recycling and driving consumer choices towards more sustainable product options. Two tax/charging schemes have been included under this sub-type. These are:

- Taxes on disposable items to encourage the use of alternatives (e.g. taxes on disposable cutlery and single use plastic carrier bags); and
- Fees paid for extended producer responsibility (EPR) schemes (e.g. WEEE, oils, tyres, etc).

As part of this sub-type we propose investigating taxes applied to products with the intention of shifting consumer choices to 'greener' reusable options. A particular case study that may be of interest is Ireland's plastic bag levy as a number of studies have looked to assess the impact of this policy which has been in place since 2002. [49,50]

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[47] Eunomia Research & Consulting (2006) *Impact of Unit-Based Waste Collection Charges*, Report for the Organisation for Economic Co-operation and Development, May 2006, http://www.oecd.org/officialdocuments/publicdisplaydocumentpdf/?cote=ENV/EPOC/WGWPR(2005)10/FINAL&docLanguage=En

[48] Eunomia Research & Consulting (2006) *Modelling the Impact of Household Charging for Waste in England*, Report for the Department for Environment, Food and Rural Affairs, December 2006, http://archive.defra.gov.uk/environment/waste/strategy/incentives/documents/wasteincentives-research-0507.pdf

[49] AP EnvEcon Limited (2008) Regulatory Impact Analysis on Proposed Legislation to Increase Levies on Plastic Shopping Bags and Certain Waste Facilities, November 2008, www.environ.ie/en/Legislation/Environment/Waste/WasteManagement/FileDownLoad,21599,en.pdf

Attention will also be given to difficulties arising when the alternative choices on the market potentially have a greater environmental impact than the item being targeted by the tax. For example, a recent life cycle assessment has suggested that some substitutes for single use carrier bags may have to be used a significant number of times before the impact of the plastic carrier bag it intended to displace is offset (e.g. a cotton carrier bag has to be used 131 times if a plastic bag is used only once).[51,52]

The second of the two instruments listed above, fees payable for EPR schemes, is typically used to fund recovery and recycling operations and it is therefore suggested that this is not subject to further analysis. For example, the European WEEE Directive (2008/34/EC) has resulted in increased recovery of electrical and electronic items, but there is no data on how these schemes are contributing to reducing waste arisings; indeed, in many instances WEEE arisings are actually increasing year on year. [53] In 2008 the European Commission stated that:

> *'New waste management legislation, notably the revised Waste Framework Directive with its waste prevention provisions, may influence WEEE arisings and collection rates in the longer term. However, as these measures are still in a very early stage of development there is insufficient evidence for any quantitative estimates'.*[54]

It is this lack of focus on waste prevention which suggests that the instrument is not yet ideally suited for further analysis, coupled with the difficulty associated with assessing this at the household level. It should also be noted that reductions in the use of hazardous substances in electrical appliances, which is also recognised as a waste prevention measure, has been driven by legislation rather than economic pressures and is thus not considered here.[55]

Other EPR schemes, such as the system set up by Eco TLC for waste textiles in France's, have also focused more heavily on recovery and reuse with very little data

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[50] Simon McDonnell and Susana Ferreira (2007) The most popular tax in Europe? Lessons from the Irish plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, no. 1, pp. 1-11

[51] ENDS Report (2011) Reusing Plastic Carrier Bags is Greenest, Finds Life-Cycle Study, March 2011, Issue 434, pp. 19-20

[52] Environment Agency (2011) Life-Cycle Assessment of Supermarket Carrier Bags, http://www.environment-agency.gov.uk/research/library/publications/129364.aspx

[53] Ongondo, F. O., Williams, I. D. and Cherrett, T. J. (2011) How are WEEE doing? A global Review of the Management of Electrical and Electronic Wastes, *Waste Management*, Vol.31, No.4, pp.714-730.

[54] Commission of the European Communities (2008) *Directive of the European Parliament and of the Council on Waste Electrical and Electronic Equipment (WEEE) Impact Study*, 2008, http://ec.europa.eu/environment/waste/weee/index_en.htm

[55] Arcadis and RPA (2008) *Study on RoHS and WEEE Directives*, Report for the European Commission DG Enterprise and Industry, March 2008, http://ec.europa.eu/environment/waste/weee/pdf/rpa_study.pdf

being recorded on the waste prevention impacts. [56,57] As such, the focus here will be on eco-taxes rather than extended producer responsibility schemes.

### 4.3.3 Subsidies for Products

Such subsidies have been used by municipalities in various countries across the world and have been found to have substantial benefits in terms of diverting waste from landfill. These schemes are also relatively inexpensive to operate and where landfill taxes are in place such measures may help authorities to reduce their disposal costs and meet diversion targets for sending biowaste to landfill. We propose examining examples in the UK using data from studies funded by the country's Waste & Resources Action Programme (WRAP).

In the UK it has been estimated that reusable nappies account for up to 4% of all waste sent to landfill and with the help of the Go Real campaign many local authorities across the UK have started offering subsidised reusable nappies. [58] WRAP has developed an online Waste Prevention Toolkit which can be used to determine the likely waste diversion rates for families who choose to use non-disposable nappies. [59] Eunomia believes the use of subsidies to promote the uptake of reusable nappies will provide an interesting avenue for further investigation.

### 4.3.4 Deposit-Refund Systems for Beverage Containers

Over recent years many deposit-refund systems (DRS) for beverage containers have shifted towards the use of one-way bottles. In these situations the deposit is merely aimed at incentivising the return of the bottles for recycling, and not reuse. Despite this growing trend a number of the more traditional DRS schemes, which focus on the collection of beverage containers for reuse, are in place. Deposit-refund schemes have been used as a means of encouraging reuse for many decades, both in Europe and beyond. However, due to the growing use of disposable bottles many of these schemes have either ceased to operate or have been forced to modify their approach to keep up with changing consumer trends.

Eunomia has extensive experience in the field of deposit refunds, having recently completed a project on implementing a UK deposit-refund system for the Campaign to

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[56] Emmaüs France (2008) *The Textile Contribution: What's New?* Accessed 24th February 2011.

[57] Oakdene Hollins (2009) *Maximising Reuse and Recycling of UK Clothing and Textiles*, Report for the Department for Environment, Food and Rural Affairs, October 2009, www.pbmsolutions.co.uk/11%20Knowledge%20Sharing%20Centre/DEFRA%20Minimising%20Reuse%20and%20Recycling%20of%20UK%20Clothing%20and%20Textiles.pdf

[58] Go Real (2010) *Reducing Disposable Nappy Waste: Review of Waste Policies - Call to Evidence*, October 2010, www.goreal.org.uk/media/documents/Call_to_Evidence_v2_041010.pdf

[59] WRAP (2011*) Waste Prevention Toolkit*, Date Accessed: 21 July 2011, www.wrap.org.uk/applications/waste_prevention_toolkit/restricted.rm

Protect Rural England (CPRE). [60] At present we are also conducting an extensive European-wide study to understand the options and feasibility of a European deposit refund system for metal beverage cans. [61] We will draw on this experience to provide a detailed assessment of this policy instrument and we suggest analysing more closely the deposit scheme which is currently operating in Germany. The German deposit system has been chosen because the DRS for reusable beverage containers operates alongside a deposit system for one way containers and has been responsible for high rates of recovery.

However, it is necessary to highlight the area of the waste hierarchy upon which this instrument impacts. The focus of this research is directed towards the first tier of the waste hierarchy, '*waste prevention and reuse'*, whereas this economic instrument is directed towards '*preparation for reuse'* since upon being returned through the deposit system, the bottles will require cleaning, and therefore preparation, before being suitable for reuse. It is known that Bruxelles Environment is keen to investigate DRSs more closely, despite this diversion from the project remit, in terms of instrument impact.

### 4.3.5 Packaging Tax/Fee/Charge

This sub-type includes taxes and charges paid on various packaging materials (including beverage containers), generally as part of extended producer responsibility programmes aimed at driving behaviour change, efficiencies in resource usage, or improved recovery and recycling. Some of these instruments have been effective in limiting the growth of packaging materials, but much of the focus still appears to be on promoting recovery and recycling. There does not appear to be a significant amount of data on the effects that these instruments have had on reducing municipal waste arisings. However, Bruxelles Environnement has expressed enthusiasm for a descriptive consideration of this instrument sub-type, despite the diversion from the original project brief.

### 4.3.6 Variable VAT Charge

The use of variable VAT rates is mentioned fairly widely, but under current EU regulations there is little scope for varying VAT rates for different products - i.e. there is limited flexibility for influencing consumer choices through the preferential use of the lower or standard VAT rate. [62, 63] However, under the European VAT Directive

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[60] Eunomia Research & Consulting (2010) *Have We Got the Bottle? Implementing a Deposit Refund System in the UK*, Report for Campaign to Protect Rural England (CPRE), September 2010, www.cpre.org.uk

[61] European Commission (2011) *Options and Feasibility for a European Refund System for Metal Beverage Cans*, Date Accessed: 21 July 2011, http://eunomia.co.uk/ecdrs/intro.html

[62] Institute for Environmental Studies (2008) *The Use of Differential VAT Rates to Promote Changes in Consumption and Innovation*, Report for the European Commission Directorate-General for Environment, June 2008, http://ec.europa.eu/environment/enveco/taxation/pdf/vat_final.pdf

[63] Copenhagen Economics (2008) *Reduced VAT Rates for Environmentally Friendly Products*, December 2008,

(2006/112/EC) labour intensive services can be subject to a lower rate of VAT. Belgium has taken advantage of this for labour intensive activities associated with reuse and repair (e.g. repair of shoes, clothing and bicycles). European VAT legislation is now under review and it is expected that the results of this consultation process will be published towards the end of 2011.[64] There has been much debate about modifying the VAT system to enable it to deal more flexibly with promoting 'green' environmental choices, but it is not clear if this will find its way into new legislation. [65]

The relationship between a reduced rate of VAT and increased reuse is unknown. However, it is thought that there could be some novel applications where a variable rate of VAT may lead to waste prevention effects. Such potential applications will therefore be considered further.

## 4.4 Instruments not Chosen for Analysis

### 4.4.1 Subsidised Home Composting Schemes

Some local authorities provide fully or partially funded home composting bins to households wishing to undertake their own composting. In the UK, WRAP has undertaken some studies on the diversion rates which can be achieved by the uptake of schemes in the UK in addition to the research funded by compost bin producers themselves. [66, 67, 68]

However, Bruxelles Environnement has stated that they feel that the waste prevention impacts of home composting are already well proven, and are therefore not to be considered further in this study.

### 4.4.2 Green Procurement

Green Public Procurement policies focus on a range of goals including resource efficiency, energy and water conservation and waste reduction. They often include a recommended list of products or suppliers (often identified through eco-labelling),

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http://ec.europa.eu/taxation_customs/resources/documents/taxation/gen_info/economic_analysis/economic_studies/study_on_reduced_vat_for_environmental_friendly_products_en.pdf

[64] EurActiv (2010) *Brussels Launches Overhaul of VAT*, Date Accessed: 27 June 2011, www.euractiv.com/en/euro-finance/brussels-launches-overhaul-vat-news-500187

[65] EurActiv (2009) *Green VAT Proposal Likely to be Scrapped*, Date Accessed: 27 June 2011, www.euractiv.com/en/energy-efficiency/green-vat-proposal-scrapped/article-180000

[66] Resource Futures (2010) A Cost-Benefit Analysis of Local Authority Home Composting Support Programmes, Report for Straight Plc., July 2010, www.eastmidlandsiep.gov.uk/uploads/Waste-%20Becky%20/Straight%20-%20Cost%20benefit%20analysis%20home%20composting.pdf

[67] Resource Futures (2009) Home Composting Diversion: District Level Analysis, Report for Waste & Resources Action Programme, September 2009, www.wrap.org.uk/local_authorities/research_guidance/waste_prevention/

[68] WRc (2009) Home Composting Diversion: Household Level Analysis, Report for Waste & Resources Action Programme, September 2009, www.wrap.org.uk/local_authorities/research_guidance/home_composting/index.html

detailed language to be included in bid specification documents, a minimum recycled content requirement for particular products, a monitoring process and staff training requirements. Green procurement is of great interest because of the significant scale over which it operates and the extent to which it could have an effect on preventing waste due to the purchasing budgets that the public sector commands.

For example, an early analysis of green procurement in Europe (under the RELIEF programme) estimated that green procurement could potentially be responsible for reducing the amount of waste PCs in the public sector by 163,767 tonnes per annum (this value assumes one year of operation and can be multiplied by the number of years for which computers are actually in use). [69]

However, experience suggests that there are likely to be data availability issues for this area. Previous research into the impact of green procurement at the EU level undertaken by Eunomia for the OECD has resulted in predominantly qualitative outcomes, or at best, output based, rather than the preferred outcome based data. However, ongoing monitoring, such as the RELIEF studies, may provide more recent data which will enable a more quantitative analytical approach to be taken. [70]

### 4.4.3 Disposal Tax

Landfill taxes have been reported to have a limited direct effect on household waste arisings - even at very high rates of tax. [71] With the focus of the study being at the household level this is not thought to be a suitable instrument for further analysis.

A reduction in the amount of waste being sent to landfill as a result of the tax is inevitable. However, at the household level any change in behaviour is unlikely to be as a result of the tax because this is not directly fed back to householders, but rather an indirect result. In other words, a national-level tax will impact local authorities who in turn will educate or incentivise householders. In this case there can be a number of confounding factors which prevent correlations from being drawn between the tax and the household level waste arisings. Without being made explicitly aware of the impact of the tax rise on the amount charged for the management of their waste (as is possible through pay-as-you throw schemes) households are unlikely to respond by reducing their waste.

Other aspects, such as socio-economic status, access to recycling/reuse facilities, and the broader economic environment will also be impacting on waste arisings in any given area and will make it difficult to attribute causality. These inherent difficulties, together with the lack of a substantial body of evidence, and the clear

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[69] Pierrard, R. (2003) *Chapter 9: Results of the European Calculation*, In *Buying into the Environment: Experiences, Opportunities and Potential for Eco-procurement*, Sheffield, UK: Greenleaf Publishing Limited. pp. 164 - 192.

[70] Procura+ (2011) RELIEF – European Research Project on Green Purchasing , Available: http://www.procuraplus.org/index.php?id=8104 , Accessed July 2011.

[71] Institute for Environmental Studies (2005) Effectiveness of Landfill Taxation, Report for VROM, November 2005, http://www.ivm.vu.nl/en/Images/Effective%20landfill%20R05-05_tcm53-102678_tcm53-103947.pdf

focus on instruments impacting at the household level, have meant that this instrument has not been chosen for further analysis.

Despite not *directly* impacting upon the householder, it is important to highlight the potential dependencies associated with disposal taxes. Disposal taxes can act as a very strong driving force for change, which may result in the implementation of economic instruments which *do* directly impact upon the householder. As such, although disposal taxes are not recommended as a possible case study, it is inevitable that in cases where they have been implemented, their impact will be fundamental to understanding the context setting and potential inter-dependencies between economic instruments.

### 4.4.4 Tradable Permit Systems

Tradable permit systems are not widely used in the waste sector. The only two examples identified for inclusion in the inventory came from the UK, where the instrument is being used to promote the recycling of packaging materials and to ensure the diversion of biodegradable waste from landfill. The latter policy had been implemented through the Landfill Allowance Trading Scheme which has been effective in diverting biodegradable waste from landfill, but there is no evidence of its impact on waste prevention (this scheme is due to be discontinued at the end of the 2012/13 financial year as the country's escalating landfill tax will be the main driver for landfill diversion at this point).[72]

In the UK, the recycling of packaging material has been driven forward by the development of Packaging Recovery Notes (and Export Recovery Notes) and their subsequent trading as a means for providing verification that materials have been recycled. In essence, reprocessors issue Packaging Recovery Notes (PRNs) as proof that a certain quantity of material has been recycled. These notes are then traded and can be purchased by organisations wishing to meet their targets. Historically, waste prevention has not been a priority of the PRN system which has focused largely on recycling.[73, 74, 75]

Due to the small number of tradable permit systems which were identified, and their lack of focus on waste prevention, it seems that there will be little benefit in examining this instrument in any more detail.

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[72]Department for Environment, Food and Rural Affairs (2011) *Government Review of Waste Policy in England 2011, June 2011*

[73] Department for Environment, Food and Rural Affairs (2011) *Advisory Committee on Packaging: Annual Report 2010/11*, June 2011, www.defra.gov.uk/publications/files/acp-report2010-11.pdf

[74] Perchards (2005) *Study on the Progress of the Implementation and Impact of Directive 94/62/EC on the Functioning of the Internal Market*, May 2005, www.perchards.com/files/documents/Final%20report%2020-6-05%20%28final%20v3%29.pdf

[75] European Environment Agency (2005) *Effectiveness of Packaging Waste Management Systems in Selected Countries: An EEA Pilot Study*, http://scp.eionet.europa.eu/publications/wp2005_2

### 4.4.5 Deposit-Refund System for Products

In compiling examples of economic instruments for waste prevention for the inventory a number of local, regional and national deposit schemes were found for items other than beverage containers. These items included: tyres (e.g. in Denmark, United States); batteries (e.g. Sweden, Mexico and United States); end-of-live vehicles (Finland); and lubricating oils (Norway). Again, these programmes appear to be predominantly focused on capturing these potentially hazardous products for recycling or safe disposal, rather than reuse. Given this, it is suggested that these instruments will not be considered further here.

### 4.4.6 Waste Prevention Subsidies

Two regional waste prevention subsidies were identified and recorded in the inventory. These subsidies are offered to municipalities in the regions of Flanders and the Mantua (Italy), for promoting and implementing waste prevention initiatives. The effectiveness of these programmes is not known as the subsidies would have been used to fund other local initiatives such as the provision of reusable nappies. No studies could be identified which have sought to assess the impacts of these subsidies on waste prevention. Given the different ways in which these subsidies can be directed and the limited number of examples, together with the lack of data it is suggested that there will be little benefit in examining these general waste prevention subsidies further.

### 4.4.7 Loyalty Card Schemes

Loyalty cards schemes have been trialled in local areas in a number of European countries. Four such schemes were identified and these included:

- NU-Spaarpas – City of Rotterdam, Netherlands (only operated for 11 months after a two year preparation phase – stopped in 2003); [76, 77]
- Umwelt.plus.karte – City of Heidelberg, Germany; [78]
- Credit scheme for 'green' products - Flemish municipalities in the province of Limburg: Overpelt, Diest, Hechtel-Eksel, Houthalen-Helchteren, Leopoldsburg, Lommel, Zonhoven;[79, 80] and

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[76] NU-Spaarpas (2004) *NU-Spaarpas: The Sustainable Incentive Card Scheme*, January 2004, www.nuspaarpas.nl/www_en/pdf_en/NUspaarpasENGCH1.pdf

[77] OVAM (2008) Analysis of Innovative Environmental Policy Instruments Towards the Realisation of Environmentally Responsible Production and Consumption, February 2008, www.ovam.be/jahia/Jahia/cache/offonce/pid/176?actionReq=actionPubSearch&searchLanguage=english&sort1=type&sort2=title&sort3=date&sort4=cost&showTable=1

[78] Umweltdirect (2011) *Die Umwelt.plus.karte*, Date Accessed: 22 June 2011, www.umweltpluskarte.de/umweltpluskarte/karte/

[79] Groupe One (2009) Etude Sur Les Expériences de Systèmes Visant à Encourager des Comportements Spécifiques au Moyen de Monnaies Complémentaires, Report for Bruxelles Environnement, December 2009

- Credit scheme for reusable beverage containers – Zonhoven, Belgium; [81, 82]

All of the schemes above are based on the concept of 'earning' points/credits for all purchases of selected 'green' goods made at participating outlets. In reviewing reports on the different schemes no clear indications were found which suggested that waste prevention had been a significant outcome or, in the case of the first three schemes, that it was even a primary objective at all. The last of the four schemes appears to be most suited to encouraging waste prevention as it aims to promote the use of reusable bottles in exchange for an exemption on the customers annual waste tax (up to a maximum of €8.75). However, if taken forward, the DRS case study will cover much of the same ground as this initiative.

All of the schemes aim to increase the consumption of 'sustainable'/'green' products and there is some uncertainty as to the overall environmental benefits of this. [83] It may also be possible that the money saved by using these points (i.e. money saved if points are reimbursed for commonly used products or services) is then spent on activities or products which have a high environmental costs. Research by Druckman *et al* suggests that there may be 'rebound' or 'backfire' effects from households who spend money saved from environmentally sound activities on high impact products or services (e.g. flights abroad). These authors have demonstrated how money saved from activities such as walking instead of driving or installing additional home insulation can be spent on activities which act to counterbalance some, if not all, of the expected reductions in carbon dioxide. [84]

Given these uncertainties and the lack of focus of the above mentioned schemes on waste prevention *per se* it is felt that these instruments should not be taken forward for further analysis at this point.

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[80] OVAM (2008) Analysis of Innovative Environmental Policy Instruments Towards the Realisation of Environmentally Responsible Production and Consumption, February 2008, http://www.ovam.be/jahia/Jahia/cache/offonce/pid/176?actionReq=actionPubSearch&searchLanguage=english&sort1=type&sort2=title&sort3=date&sort4=cost&showTable=1

[81] Bruxelles Environment (2010) Mapping Report on Waste Prevention Practices in Territories within EU27 - Pre-Waste: Improve the Effectiveness of Waste Prevention Policies in EU Territories, October 2010, www.bruxellesenvironnement.be/uploadedFiles/Contenu_du_site/Professionnels/Formations_et_s%C3%A9minaires/Conf%C3%A9rence_Pre-waste_2011_(actes)/p3-%20prewaste-mapping-report.pdf

[82] OVAM (2004) *Economic Instruments to Steer Eco-consumption Involving Local Authorities in the Flemish Region*, Presentation by OVAM published on 28th October 2010, http://www.ovam.be/jahia/Jahia/cache/offonce/pid/176?actionReq=actionPubDetail&fileItem=771

[83] Alfredsson, E. C. (2004) "Green" Consumption - No Solution for Climate Change, Energy, Vol.29, No.4, pp.513-524.

[84] Druckman, A., Chitnis, M., Sorrell, S. and Jackson, T. (2011) Exploring Rebound and Backfire Effects in UK Households, *Energy Policy*, Vol.39, No.6, pp.3572-3581

# 5.0 Analysis of Impacts

The analysis of impacts seeks to establish not just *which* instruments have been successful in preventing waste, but to explain, through reviewing secondary data, *why* they have worked. This will enable the development of recommendations, drawing out lessons that can be used in the design and implementation of future economic instruments for waste prevention.

The analysis is therefore retrospective to a certain degree, by evaluating the waste prevention impacts of a number of instruments that have been implemented in a number of situations over several years. However, it is also forward looking, not only in making recommendations for future implementation, but also in considering novel applications of economic instruments without a track record of implementation.

The mode of analysis therefore varies based on the extent of available information. For three instruments, where there are a number of examples of their use, we will undertake quantitative analysis, based on secondary data, to the extent that this is available. This will be the case for:

- Direct and Variable Rate (DVR) Charging;
- Product Taxes / Fees / Charges; and
- Subsidies for Products.

For a further three instruments, data availability on waste prevention effects is known to be poor. For two of these, the instruments themselves are designed to increase levels of recycling and recovery, and if working well, will not have a waste prevention effect. These instruments are:

- Deposit-refund Systems for Beverage Containers; and
- Packaging Taxes / Fees / Charges.

For these instruments we will undertake a descriptive analysis, looking at the way in which they operate and any evidence of wider effects beyond the intended goals of stimulating recycling and recovery, for example in the prevention of litter.

For the final instrument, there is very little actual implementation, and no studies evaluating impacts. This instrument is:

- Variable VAT Charge.

As this instrument is, however, of interest to Bruxelles Environnement, we will consider ways in which it might be implemented in future to bring about waste prevention impacts. Consideration of the application of this instrument is, of necessity, a speculative exercise.

# 6.0 Direct & Variable Rate (DVR) Charging

DVR charging schemes, sometimes called pay-as-you-throw (PAYT) schemes take a variety of forms:

- ***Bin Volume-Based Schemes*:** under these schemes, typically, households are asked at the beginning of a particular year to say which sized bin they would like to use. The charge is then related to the size of bin used. This type of scheme provides little by way of continuous, marginal incentive. It is generally seen as important to offer a good *range* of bin sizes (from smallest to largest), with a range of choice between the maximum and minimum size. An important decision in this type of system is whether, and if so, how often, to allow the choice of bin size to be changed. The costs of allowing frequent changes are obvious (in terms of the stock of bins and the need to make the replacement). These schemes are popular in the US, and have been popular in the past in Europe. It would seem that there has been some move away from this type of system in favour of other variants (see below) in some countries;
- ***Frequency-Based Schemes*:** these schemes are based upon the frequency of service provided to the household. Two possibilities exist:
  - The household subscribes for a particular service frequency (in which case, the marginal costs of waste generation are low); or
  - Either tags, or electronic chips, are used to record when bins are emptied following their being presented in a specific way.

  The electronic approach is increasingly common, and is widely used in the Netherlands and Belgium, as well as in part of Germany. The frequency-based aspect is important from the perspective of collection logistics, since the costs of collection are generally linked to the frequency of set-out (more than they are to weight). However, households may seek to reduce charges by 'stomping' (compacting) waste;
- ***Volume and Frequency Based Schemes*:** these schemes are usually based around the use of bins and as with the frequency-based schemes, they can be 'subscription based', or based upon the number of emptyings of the bin. In this case, as well as an incentive to reduce set-out frequencies, there is some incentive for reduction implied by the choice of bin size. However, the strength of the latter incentive is likely to be limited once the choice has been made;
- ***Sack-based schemes*:** sack-based schemes are also, essentially, volume based schemes. However, since the space available for refuse is not 'fixed', as with the volume-based bin schemes, there is a stronger incentive to reduce waste and recycle more. In these schemes, either a) specific (readily identifiable) sacks are sold to households, or b) tags / stickers are sold to households, which must be attached to the sacks. It is a good idea, generally, to offer different sack sizes for residents.

- ***Weight-based schemes*:** in these schemes, bins are usually equipped with a transponder which is read by software on the collection vehicle as the bin is loaded. The bin is weighed when it is loaded on the vehicle. Weight-based schemes have a good incentive effect for many materials, but the implications for waste collection logistics might not always be significantly affected by these schemes. Collection inefficiencies will be experienced where small quantities are collected on a frequent basis. However, where the marginal benefits of avoided disposal are significant, weight-based elements are clearly useful;
- ***Bin volume, frequency and weight-based schemes*:** schemes are becoming more sophisticated as technology develops. It is possible to have '3-D' schemes, with charges varying by bin size, set-out frequency and weight. In this way, there is an up-front choice to be made to reduce bin volume, an incentive to reduce set-out rates (so as to improve collection logistics) and a weight-based element to reflect the marginal benefits of avoided disposal. However, allowing the customer to choose the bin size and then regularly change this size is likely to incur additional costs. In frequency and weight-based schemes, the frequency based incentive works to optimise logistics, whilst the weight based incentive works to minimise set out of refuse (and biowaste in some schemes).

The emphasis tends to be on the charges levied for door-to-door collections of refuse. However, the overall waste management system needs to be considered. It is now common to see charges levied on:

- ***Biowaste collections*.** The rationale for this is to reduce the extent to which new material is drawn into the collection scheme, and to encourage home composting. Where charges are levied on biowaste, they are always lower than for residual waste (reflecting, usually, the difference between the lower costs of treating biowaste and the higher costs of disposal);
- ***Recycling collections*.** The rationale for this may be to recover costs, but also to convey an incentive for waste prevention. Some argue that 'free recycling' does not do enough to encourage prevention;
- ***Waste delivered to CA sites*.** It is becoming common for CA sites (containerparks) on the continent to charge for some waste fractions delivered to CA sites (not only residual wastes). Evidently, if CA sites offer free disposal services for residual waste, the risk is that residents simply avoid the charge by taking refuse to CA sites;
- ***Bulky wastes*.** It is already common to see bulky waste charges in England. Some communities have related these to increased incidence of fly-tipping. What appears to be important where charging schemes are introduced is to have a quality bulky waste collection service in place.

Evidently, the aim should be to develop a coherent system of charges to encourage the desired behavioural change, and to prevent those 'leakages' from one part of the system to another (on the basis of price differentials) which are not considered as desirable. The 'waste system' is like a balloon - squeeze it in one place and it tends to appear somewhere else.

## 6.1 Case Studies

There are a number of case studies which have been considered in the literature. Good examples can be found in:

- Bauer and Marie Lynn Miranda (1996) *The Urban Performance Of Unit Pricing: An Analysis Of Variable Rates For Residential Garbage Collection In Urban Areas*, Report prepared for Office of Policy, Planning and Evaluation, U.S. Environmental Protection Agency, Washington, D.C. 20460, April 1996;
- Danish Environmental Protection Agency (2000) Fordele og ulemper ved gebyrdifferentierede indsamlingssystemer for husholdningsaffald. *Miljøprojekt nr. 576*, 2000. (Study on the advantages and disadvantages of fee-differentiated waste collection schemes for domestic waste from households);
- Dijkgraaf, E., and Gradus, R. (2003) Cost Savings of Unit-Based Pricing of Household waste, the case of the Netherlands. Rotterdam: OCFEB;
- Ferrara, Ida, and Paul Missios (2005) Recycling and Waste Diversion Effectiveness: Evidence from Canada, Environmental and Resource Economics (2005) 30, pp.221-38;
- Hogg, D. (ed.) (2002) *Financing and Incentive Scheme for Municipal Waste Management: Case Studies*, Final Report to DG Environment the European Commission;
- Hogg, D. (2006) Impact of Unit-based Waste Collection Charges ENV/EPOC/WGWPR(2005)10/FINAL, Paris: OECD;
- Hong, S., Adams, M., and Love, S. (1993) An economic analysis of household recycling of solid waste: the case of Portland, Oregon. *Journal of Environmental Economics and Management*, **25**, pp 136-146; and
- Linderhof, V., P. Kooreman, M. Allers and D. Wiersma (2001) Weight-based pricing in the collection of household waste: the Oostzaan case, *Resource and Energy Economics* 23, 359–371.

Below, we give some further detail on two examples which have been explored in some detail, those of Bjuv in Sweden, and Landkreis Schweinfurt in Germany.

### 6.1.1 Bjuv, Sweden

This case study is based upon work undertaken for the European Commission, and led by Eunomia, in 2001.[85] It is, therefore, an 'old' example, but it does draw out specific issues in respect of charging systems.

[85] Hogg, D. (ed.) (2002) *Financing and Incentive Scheme for Municipal Waste Management: Case Studies*, Final Report to DG Environment the European Commission

#### 6.1.1.1 Rationale for Scheme

In Sweden, the municipalities are responsible for collecting household waste and taking charge of it in an environmentally sound way. A wide range of collection and handling methods are used to cater for the varied demands for recycling and waste minimisation. Several municipalities in Sweden are using a weight-based fee system for household waste collection, where a part of the collection fee is based on the amount of waste collected. This weight-based fee is intended to give direct economic incentives for households to prevent waste, and to recycle.

Starting in April 2000, the municipality of Bjuv implemented a weight-based system and at the same time, introduced kerbside collection of 11 waste fractions. The results of this initiative have far exceeded expectations when it comes to waste minimisation and recycling, even though there have been some uncertainties and drawbacks that have to be dealt with, as the case study will make clear.

#### 6.1.1.2 Background Information

Bjuv is a small municipality with around 14 300 inhabitants in the north-western part of Skåne, in the south of Sweden. At the time the study was undertaken, the municipality had 4,100 household subscriptions for waste collection from detached housings, and 369 household subscriptions from apartment blocks. The average number of persons per household was 3.1.

Waste collection and transport in Bjuv is the responsibility of the municipality and was carried out by a contract firm, Sita (this is still the case). An important actor in the area was the jointly owned regional waste company NSR, which apart from Bjuv also serves five other municipalities in the region with planning, design, building and operation of the regional plants for reception and treatment of household and industrial waste.

#### 6.1.1.3 Waste Collection Schemes in Bjuv Municipality

Up until 2000, Bjuv had a fixed collection fee and kerbside collection of residual waste, with collection every week. Recycling fractions were collected at 10 recycling stations around the municipality.

In April and May 2000, the system was more or less revolutionised: kerbside collection of residual waste, organic (not garden) waste, newspapers, 6 fractions of packaging waste (cardboard, hard and soft plastics, coloured and uncoloured glass, metals), and an option for kerbside collection of garden waste were introduced. The collection fee was changed to from a flat fee basis to a flat fee alongside variable elements based upon the weight of residual waste, and organic waste, with an option to have collection every two weeks (see Table 4). At the same time, 7 of the existing bring recycling stations were closed down.

An on-vehicle weighing system was used. Containers were tagged with an ‘intelligent chip’ (to avoid switching, voluntary or involuntary, between neighbours) and weighed both before and after emptying. The difference in weight was the basis for the charge to the household.

This change led to the cost for the waste collection system almost doubling. The fee paid by households was also expected to increase substantially (although this was, of course, dependent on how much sorting the household conducts).

Table 4: Subscription Fees for Household Waste Collection (2001)

| Compulsory fees | Collection every 7 days | Collection every 14 days |
|---|---|---|
| Fixed fee for residual waste, 140-l container | €59.6 /year | €29.79 /year |
| Per kilo residual waste | €0.39 /kg | |
| Per kilo organic waste | €0.23 /kg | |
| **Additional services** | | |
| Collection of recycling fractions | €16.8 /year | (newspapers, cardboard, hard plastics 12 times/year, metals, coloured and uncoloured glass, soft plastics 6 times/year) |
| Collection of garden waste | €53.19 /year | (including container rent, collection 20 times/year) |

All households use the organic waste (as this was compulsory), and some 3,000 households subscribed to the collection of recycling fractions, and some 2,100 for the collection of garden waste. Thus, 75% of the households had kerbside collection of the recyclable fractions.

#### 6.1.1.4 Equipment

The households in Bjuv were given a 140-l container for residual waste (a 240-l container can be obtained at no extra fixed cost) and a 140-l container for compostable waste. For recyclables, 100-l fibre sacks were provided for each fraction. A container of 370 l was provided if the garden waste service is subscribed for. As discussed above, bins were tagged with an intelligent chip to record data concerning collected weights from specific households.

#### 6.1.1.5 Information

Prior to the change to a weight-based collection fee, there were significant information campaigns. Due to political disquiet concerning the existence of the system, no further information was provided for households thereafter. The households seemed to manage the new system perfectly well following the initial information campaign.

#### 6.1.1.6 Effect of Weight-based Collection Fee in Bjuv

The introduction of the weight-based fee led to a dramatic change in waste streams (see Table 5). We have also included figures for 2007 from Bjuv's Waste Plan for 2009-2013.

The amount of recycled waste (compostables and recyclable fractions) almost doubled in the first year of weight based fees. At the same time, the total amount of waste dropped by almost 20%, leading to a reduction in residual waste of some 45%.

In the period to 2007, waste quantities have increased again. By 2007, though, total waste quantities were still lower than in 1999, whilst residual waste had fallen by 50%. Recycling rates were higher in 2007 than in 2000.

Table 5: Waste Amounts per Household, kg/year, in Bjuv 1999-2001

| | 1999 | 2000 | 2007 |
|---|---|---|---|
| Total amount of waste* | 302 | 245 | 273.5 |
| Residual waste | 246 | 136 | 123 |
| Compostable waste and recycling fractions | 56 | 109 | 150.5 |

** including waste from kerbside collection and recyclables from recycling stations, not including garden waste*

The increased amounts of collection of recyclables and compostables can be explained by the introduction of the weight-based fee in combination with the increased sorting possibility for households with kerbside collection of recyclables and compostables.

There are a number of possible reasons for the reduction in total waste amounts:

- Waste may be dumped elsewhere. There has been an increase in residual waste collected from recycling stations (households are not supposed to leave residual waste there). The waste collected from cleaning the municipality amounted to 62 tonnes in 2001 (compared to the total amount of waste in the municipality of some 3000 tonnes). No earlier statistics were available for comparison, but it was reported that there has been no notable difference in the littering in Bjuv compared to neighbouring municipalities following the charging scheme's introduction;
- Waste may be burned in private fireplaces/stoves etc.;
- Waste may be composted to a larger extent than before in private gardens; and
- The total amount of waste may actually have decreased.

Concerning the recyclables and compostables, there has been no discernible change in the quality of these fractions after the change to weight-based collection fee.

#### 6.1.1.7 Lessons Learned in Implementation

In Bjuv, the weight-based fee scheme has been implemented for all households, including apartment blocks. The implementation in apartment blocks proved to be no more difficult than for detached housing areas. However, as Bjuv is a relatively small municipality, the implementation in a major urban settlement may lead to a different result.

During the start-up of the system, there were problems concerning the weighing equipment. These problems are considered to be transitory.

The most significant problem is that it has been difficult to balance the budget, since the reduction in residual waste (which provides most of the variable revenue) was far

greater than expected. The weight-based collection fee, in combination with kerbside collection of recycling fractions, has led to dramatically increased recycling rates and a dramatic reduction in overall waste amounts. This has presented some budgetary problems (see below).

The collection system proved to be relatively expensive – almost twice as expensive as the previous collection system. As the recycling rates increased far more than was expected, the collected fees do not now cover the costs for the municipality. To reflect this, Bjuv municipality was intending to increase the fixed fee substantially (no amount fixed at date) while decreasing the variable fee from 3.65 SEK/kg to 2.40 SEK/kg. This change makes the system less vulnerable from the budgetary point of view. It will be interesting to see whether the rates of source separation fall, and residual waste collection increases.

Weight-based fee systems require a higher degree of administration, as waste amounts and fees have to be registered for every household.

The system may lead to increased incentives for illegal behaviour from households, (for example, where waste is burned in private, or dumped illegally). There is no obvious indication that this has been a problem in the Bjuv municipality.

### 6.1.2 Landkreis Schweinfurt

In Germany, municipalities have responsibility for waste collection, but in Bavaria, collection tends to be organised on a county wide basis. The county of Landkreis Schweinfurt has a population of 116,000 inhabitants. The area excludes the main town, and covers only the peri-urban area surrounding it.

#### 6.1.2.1 Pre-scheme Situation

In the late 1980s, the system started changing from a one bin to a three bin system. By 1994, the doorstep collection of residual waste, biowaste and DSD fractions, supported by separate collections for glass, cans, and paper and card at bring sites, was delivering a recycling rate of 58.6%.

At this time, households only paid for residual waste collections. The system was based around a purely volume-based scheme where households paid an annual subscription based upon the volume of the bin they chose. This was therefore effectively fixed for the year, and variable only through that ‘one-off’ decision.

#### 6.1.2.2 Rationale for Scheme Introduction

In 1997, the County decided it wanted to take an additional step. It wanted to reduce the quantity of waste for disposal. It was not keen to change the existing system, but was interested in improving incentives for improved management / reduction of waste by households.

In the year 2000, incineration of refuse would have cost the county €250 per tonne. The landfilling of waste at the time cost €80 per tonne. Before the DVR scheme, a household using a 120l bin was charged €170 per year. The fee covered all the costs of waste management, including fortnightly refuse collection and fortnightly collection of the biotonne (the biowaste container). Costs of collecting the DSD fraction were not covered by the municipality (this was covered by producer fees). It was expected that

without any change in the performance of the system, the fee would rise to €210 per household.

The key aims of the change sought were:

1. Improved sorting of waste, and reduction in overall waste, leading to reduced residual waste collection;
2. A fairer system of charging; and
3. A reduction in costs (from anticipated levels).

The County undertook two pieces of work to understand what might be done:

1. In the first instance, it undertook an analysis of the composition of residual waste. Even though a doorstep collection of biowaste was in place, and this was collecting 110 kg / inhabitant, the proportion of residual waste which was organic waste was estimated at 33%. Paper was also a significant component at 12%; and
2. Secondly, a review of three systems was undertaken:
   a. Smaller bins;
   b. Tag scheme; and
   c. Weight based.

Of these, the weight-based scheme was deemed most likely to give the greatest reduction in waste.

It should be noted that prior to the system's introduction, the quantity of residual waste per inhabitant was already extremely low by international standards at 120kg/inhabitant. This makes the system's objectives all the more challenging given the already low level of residual waste in the system, and illustrates the commitment of the staff to the project.

#### 6.1.2.3 Preparation for the Charging System

In 1997, a trial across a community of 3000 persons was carried out. This enabled knowledge to be gained about the system and any shortcomings. Interestingly, it was not so straightforward to start the trial. The intention was to implement the whole system, including the new charging system (see below) in this trial. The first community approached complained and did not want to be the group being experimented upon. The second community happened to be the one in which the political leader of the county lived, and this community was willing to be the trial area.

Successful trials led to a decision being taken, in 1998, to introduce the scheme across the county. The scheme was not suddenly 'rolled out', however. Across the 29 component communities, information and publicity campaigns were undertaken. In each community, a 13 week programme was implemented. Fourteen months elapsed between the decision being made (1998) and the scheme being rolled out across the whole county (2000).

#### 6.1.2.4 Pre-implementation Issues

There were three major concerns that managers had prior to the scheme's introduction. They were:

a) Concerns that some fractions which should have been set out as residual waste would contaminate separately collected fractions;
b) Concerns (expressed by households) that users would fill up their neighbours' bins; and
c) Concerns regarding fly-tipping.

Social issues were not considered a major concern since an aim was to make charging fair, and since also, one of the aims was to reduce costs to householders.

The issues were addressed in the following way:

1. With regard to the issue of contamination, the charging structure was deliberately designed so as not to generate enormous differentials between the residual waste charge, and that for biowaste. Since the other collected fraction was paid for through purchases of products, there was no need to charge for these fractions. The intention was to apply the weight-based element of the charges *only* on the basis of the costs of incineration of residual waste, and the costs of composting of biowaste. On the basis of anticipated costs, this would have led to charges for residual waste of €250/tonne, or €0.25 /kg, and charges for biowaste collection of €30/tonne, or €0.03 /kg. It was felt that this level of differential was too high, and could lead to contamination of the biowaste collection. Hence, a decision was made to set the biowaste weight-based charge at 60% of that for residual waste. Note also that the charging structure was such that the weight-based element of the charging system did not constitute a major proportion of the overall charge paid by householders (see below).

   The other approach taken to addressing issues of contamination was that of free bulky waste collection. Prior to the system's introduction, this service was provided to all householders 2 times per week. After the system was introduced, the collections were provided only on request, and again, 2 times per annum. The aim was to enable some control to be exercised over the collection of this fraction – since one would know to whom the waste belonged, the possibility that residual waste (or other materials such as biowaste) might be delivered into the bulky waste collection was reduced.

2. With regard to the issue of neighbours using the bins of others, a gravity lock was offered to households as an optional extra, this at a cost of €0.50 per month (or €1 per month for both the residual waste and biowaste bins). The charge remains at this level today. This is sufficient to cover the cost of such locks at about €40 each. Before the system was implemented, households were given the opportunity to select this option so that when bins were delivered, the nature of the bin reflected that choice. There has been strong demand for this system – approximately 40% of households elected for such bins.

3. Regarding illegal waste disposal, it was felt that the only thing to do was to keep observing the level of tipping and to patrol and enforce the system.

The degree to which these issues became problems is discussed below.

#### 6.1.2.5 Charging System

The charging system was calculated in such a way that on average, householders would pay the same cost after the system's introduction as before. At the end of the trial, some were paying more and some were paying less.

The system is based upon a three part tariff. These are:

1. A fixed fee. This was intended to cover the fixed costs of the collection infrastructure, including the bulky waste collection, the collection of tyres, fridges, special wastes etc. The annual cost for this fixed element does vary with the size of residual waste bin chosen (the fixed fee is only linked to the refuse bin). For a 120l bin, the fee in 2002 was €8 per month, and for a 240l bin, the fee was €16 per month. The minimum bin size is 120l. It is felt that smaller bins are unlikely to lead to optimised collection of the different fractions. It should be noted that these fees are lower today than in 2002 (€5.30 and €10.60 for 120l bins and 240l bins, respectively);

2. A fee per emptying of any bin. The basis for the 'emptying charge' is the amount saved by not emptying a bin. This was calculated as €0.20 per emptying. This fee remains the same today. However, a minimum number of emptying (7 for residual waste, 13 times for organic waste) are charged; and

3. A weight-based fee. This was set at €0.25/kg for residual waste and €0.15/kg for biowaste. These figures have actually declined over time and are €0.14 and €0.07 respectively.

The billing scheme works through an annual invoice, which calculates a bill based upon the previous year's performance by the household. Each year, an adjustment is made to the preceding year's bill based upon the performance of the household relative to the beginning of year estimate. The bill is paid in 4 installments.

#### 6.1.2.6 Effects of Scheme

Under the scheme, though collections are only fortnightly, for several bin types, the set out rate fell close to 50%. In other words, on average, for many householders, bins are being set out approximately once a month. Interestingly, the set out rate tends to be lower for those using smaller bins. For those using larger bins, the materials tend to be collected approximately once every three weeks. This change in set out rates has led to reduced staffing levels. The materials are collected on side-loading vehicles, and the pre-scheme situation, in which these were operated with a driver plus one crew, was changed such that the vehicles operate with a driver only.

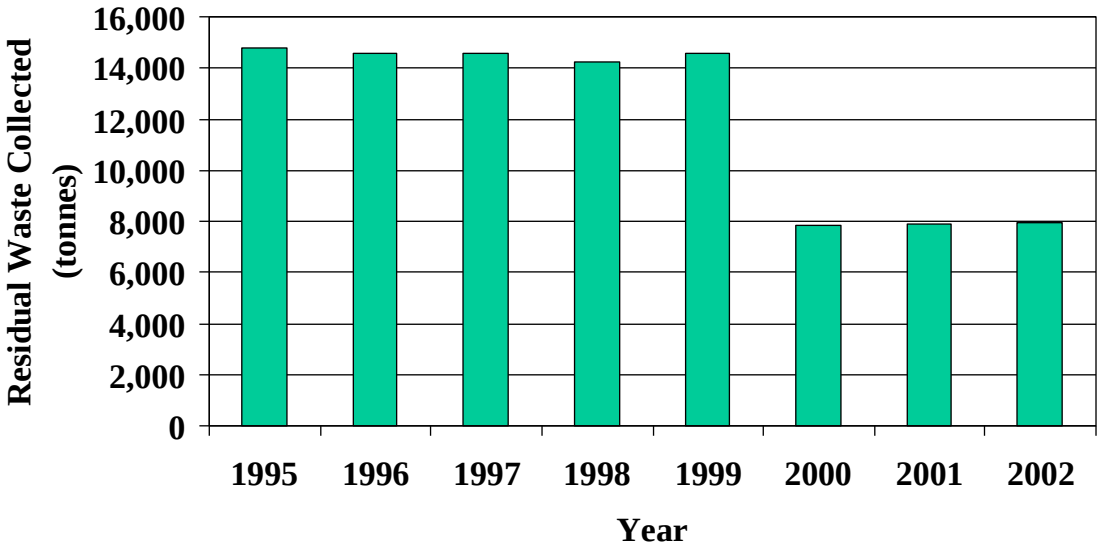
**Residual Waste**

In understanding the effects of the scheme, one must understand the linkages between what has happened to collections at the doorstep, and what has happened at other collection routes.

Figure 2 shows the effect of the schemes on the quantities of material collected at the doorstep. The effect is clearly dramatic, with collections falling by more than 40%

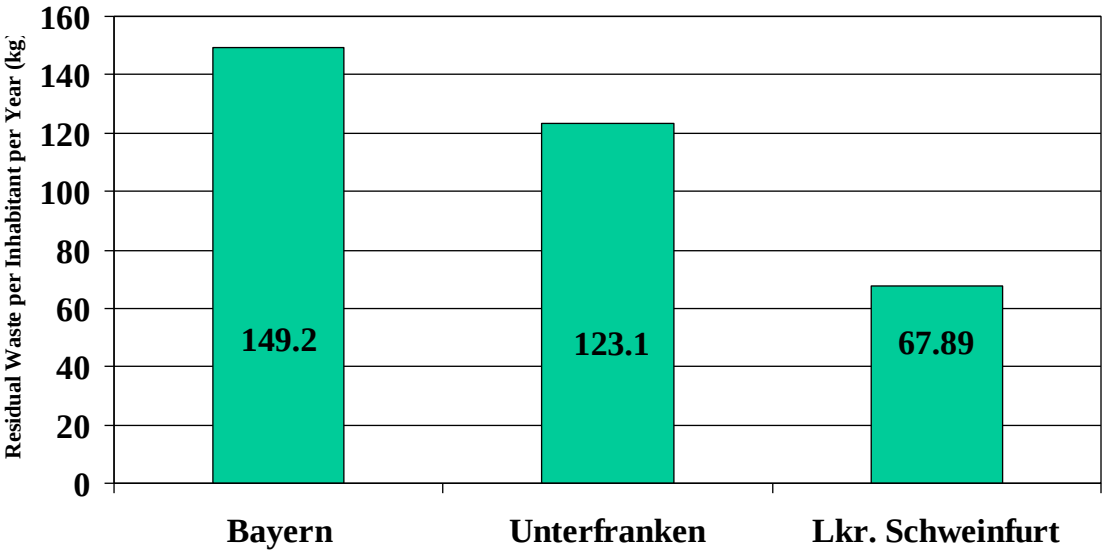
of the original quantity. Figure 3 clearly shows that this is an exceptional result, even when compared with well-functioning schemes elsewhere in Germany.

Figure 2: Quantities of Residual Waste Collected Through Doorstep Collections




*Source: Landratsamt Schweinfurt*

Figure 3: Comparison of Residual Waste Collected at Doorstep, Landkreis Schweinfurt and Others




*Source: Landratsamt Schweinfurt*

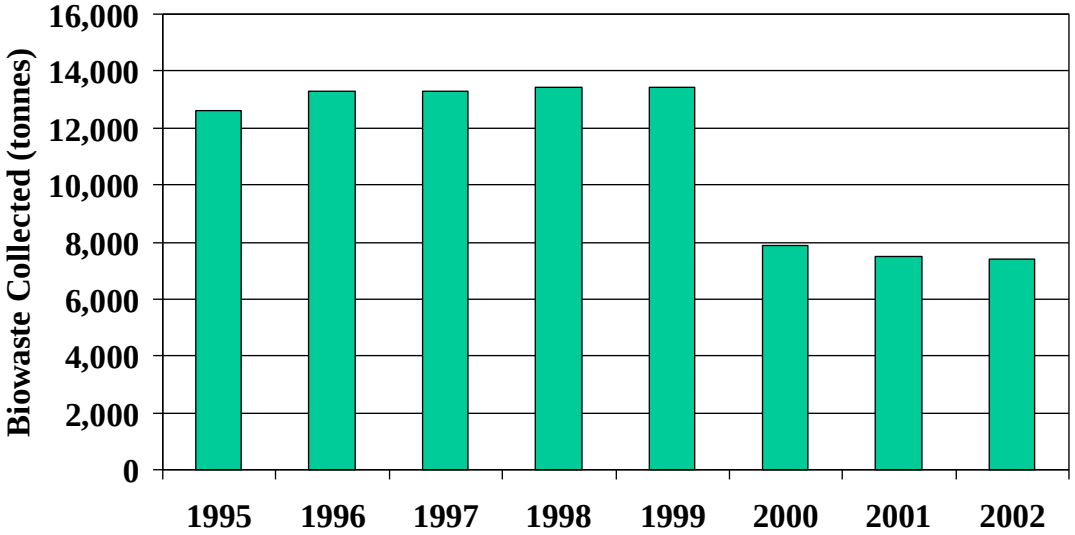
### Separately Collected Biowaste

It might be expected that the decline in residual waste quantities would be explained through an increase in source separation. However, the quantity of material collected on the door-to-door biowaste collection also fell by more than 40%. In absolute terms, the fall was about 5,500 tonnes.

Part of the biowaste fraction simply moved into a different collection outlet. The County operates a network of sites where citizens can bring material from the garden

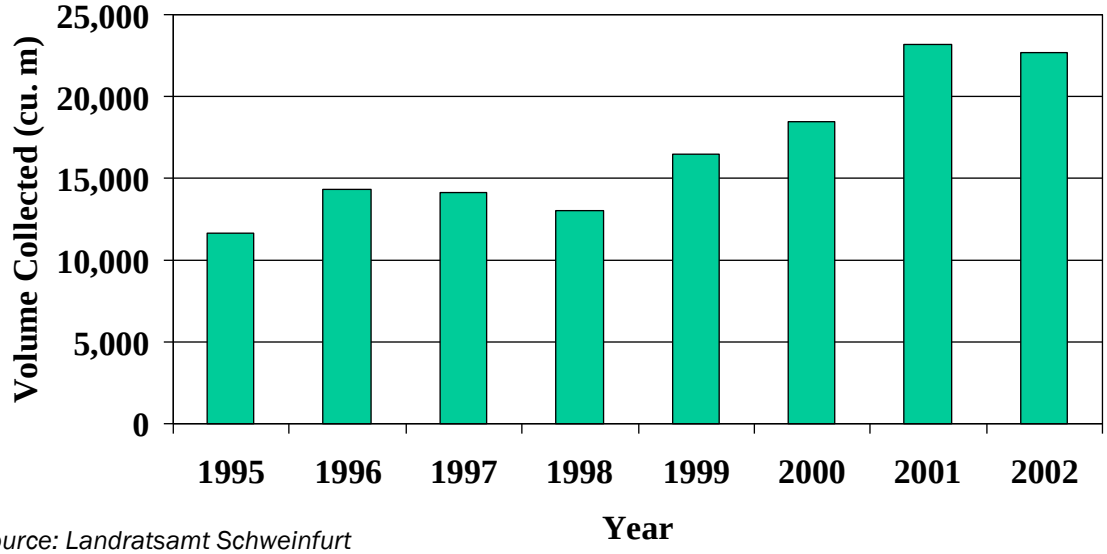
for chipping. There was an increase of around 2,800 tonnes of this material (estimated on the basis of a volume of approximately 7,000 $m^3$ when shredded). Hence, this does not completely explain the reduction in biowaste collections, which suggests an increase in home composting. A net reduction of around 2,700 tonnes (23kg per inhabitant) of biowaste still remains.

Figure 4: Biowaste Collections, Door to Door System




*Source: Landratsamt Schweinfurt*

Figure 5: Change in Material Received at Chipping Stations (volume)

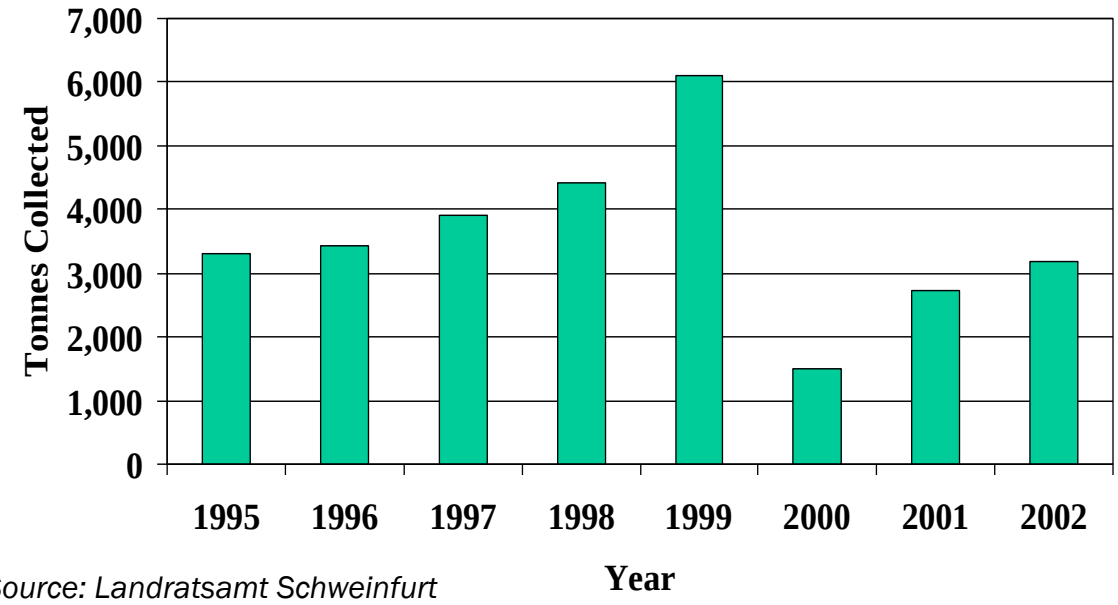



*Source: Landratsamt Schweinfurt*

### Bulky Waste Collections

The bulky waste collections show the effects of the expectations of a change in the system. The amount collected showed a sharp increase just before the change, and a drop immediately after. This suggests that many households had a clear out prior to the scheme's introduction.

Figure 6: Bulky Waste Collections in Schweinfurt

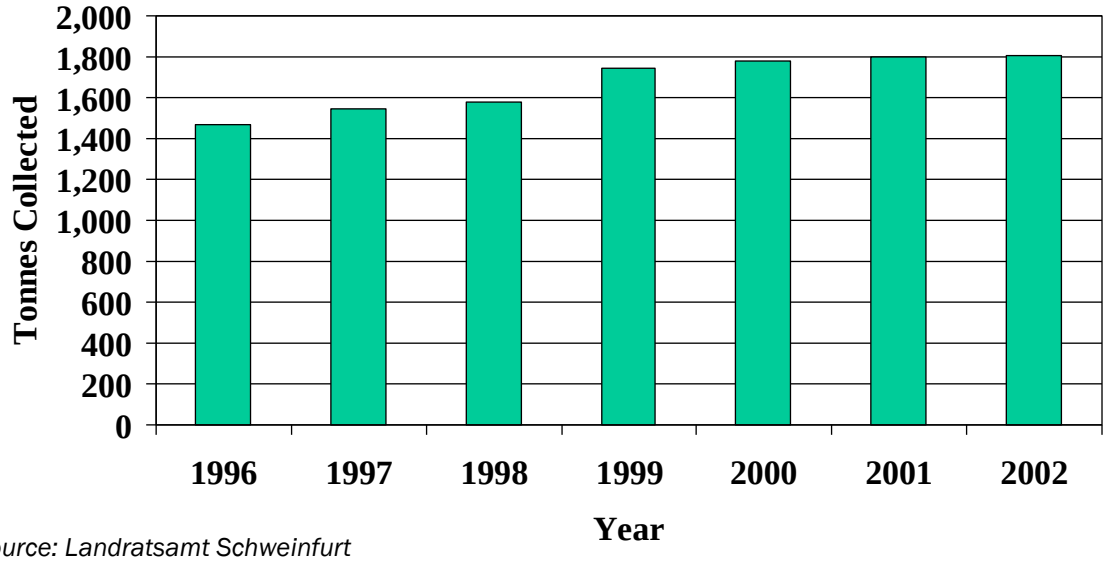



*Source: Landratsamt Schweinfurt*

Separate Collections (Bring Schemes)

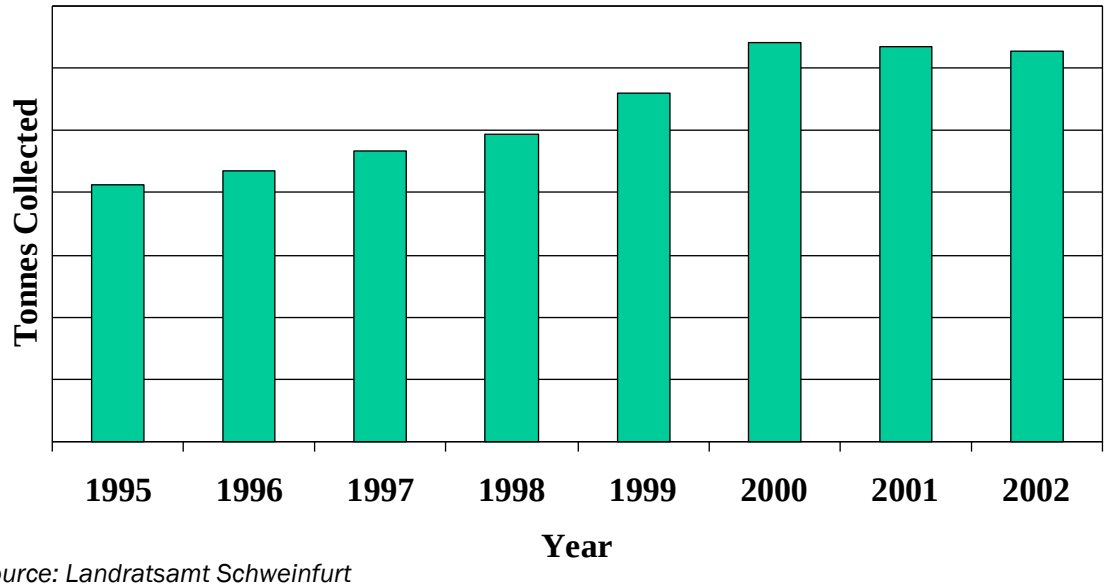
The county operates a network of 160 mini-recycling centres (bring sites) at which, typically, glass (colour separated), cans and plastics, paper and card, and textiles are collected. Paper collected separately by non-government organisations and through the bring sites increased by 400 tonnes (see Figure 7 and Figure 8).

Figure 7: Paper Collections through NGO Activities




*Source: Landratsamt Schweinfurt*

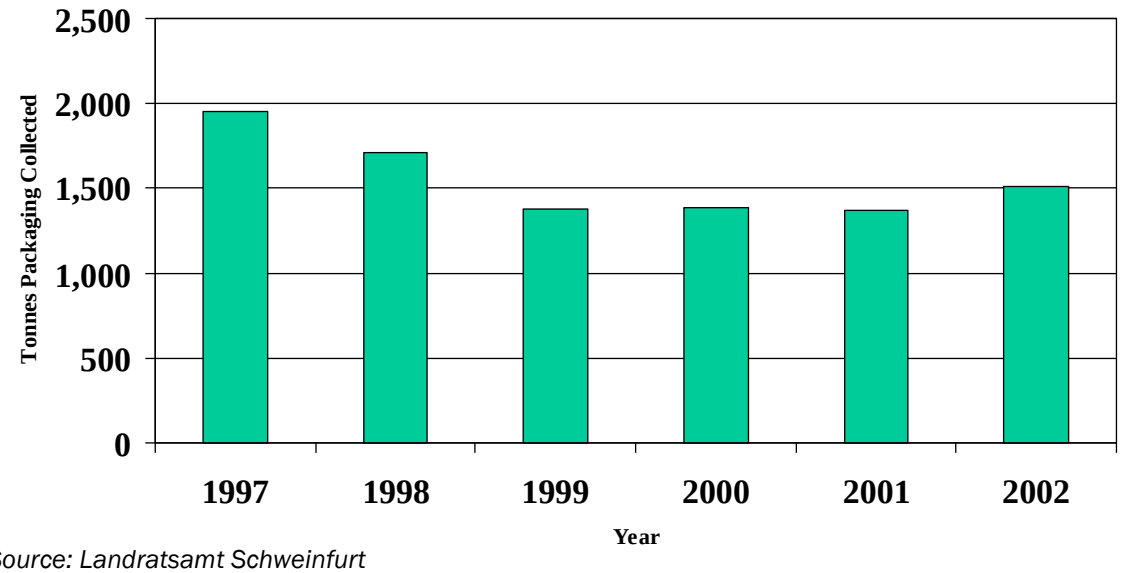
Figure 8: Paper Collection through Bring Schemes




*Source: Landratsamt Schweinfurt*

The amount of DSD waste collected changed very little (see Figure 9), initially falling, though this is believed to be due in part to ongoing changes in the nature of packaging placed on the market.

Figure 9: Packaging Waste Collected (doorstep system)

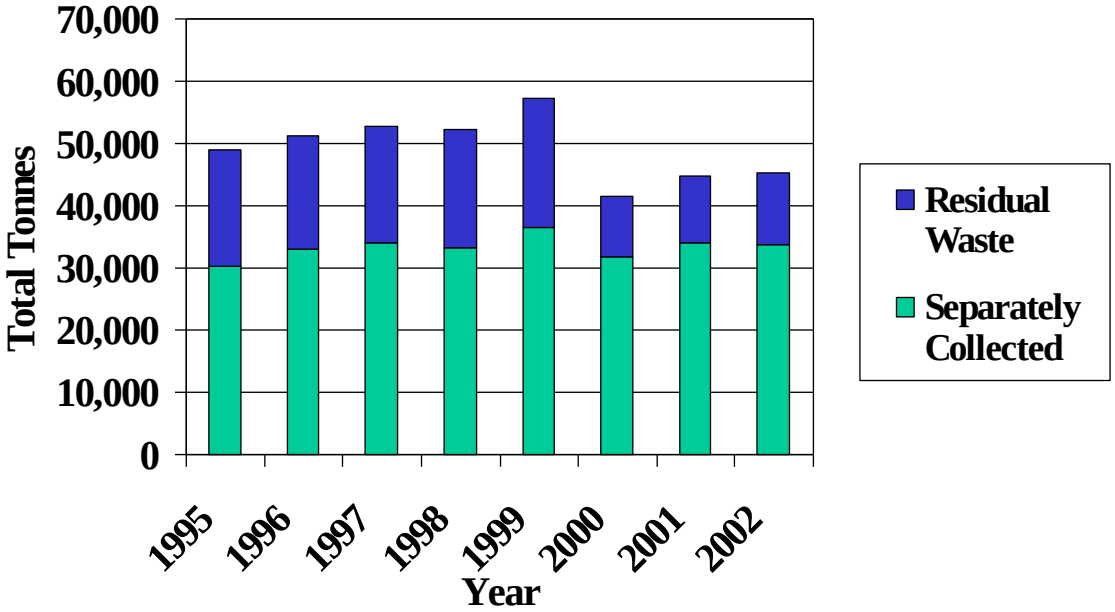



*Source: Landratsamt Schweinfurt*

## Net Effects

The net effect of these changes is shown in Figure 10. Between 1999 and 2000, total waste collected fell by 28%. However, this includes the expectations-related effects (in which 'clean-outs' occurred prior to scheme introduction). Taking this into account, the reduction was from 52,000 tonnes or so to 45,000 tonnes, a reduction of 13%.

Figure 10: Net Effect on Waste Management System




*Source: Landratsamt Schweinfurt*

Residual waste fell from a pre-scheme average of 165kg to a post-scheme average of 92kg per inhabitant, a reduction of 46%. The pre- and post-scheme average recycling rates shifted from 64% to 76%. This is a truly outstanding performance by any comparator.

The most recent figures for Landkreis Schweinfurt indicate that this performance has been sustained over time (see Table 6). Residual waste has increased marginally from 92kg per inhabitant to 97kg per inhabitant. The recycling rate has increased slightly from 76% to 78%.

Table 6: Performance of Landkreis Schweinfurt, 2010

| Waste Type | Quantity (kg/inh/yr) |
|---|---|
| Paper, cardboard | |
| Glass containers | 26.31 |
| Lightweight packaging | 35.86 |
| Scrap Metal | 5.19 |
| White goods | 0.61 |
| Biowaste | 63.97 |
| Green Waste | 96.24 |
| Wood | 22.50 |
| | **339.49** |
| household waste including commercial waste | 76.07 |
| bulky waste | 20.52 |
| Problem waste from households | 0.31 |
| **TOTAL RESIDUAL** | **96.90** |

*Source: Abfallberatung Unterfranken*

#### 6.1.2.7 Explaining the Changes

After the scheme was implemented, a second analysis of residual waste was carried out to try to understand what the effects of the system had been. Refuse had fallen to marginally more than 50% of what was there previously, but the organic fraction had also fallen significantly in proportionate terms from 33% to 8% of residual waste. This implies a reduction of around 29% of the original residual waste fraction due to changes in the way in which the organic fraction was being treated by households. Of course, such analyses cannot be relied upon to provide a completely accurate picture of the changes in quantitative terms, but they suggest a major change in how this fraction was treated. A survey of households was carried out to find out the main reason for the change, and it is believed that increased activity in respect of home composting is the principal explanation for the change.

The paper fraction, previously 12% or so of residual waste, fell only slightly in proportionate terms to 11% of residual waste. Yet, bearing in mind the reduction in absolute quantities, the reduction suggests that approximately 6% of the reduction in

residual waste was due to changes in householders' handling of the paper and card fraction.

The above line of thinking led the County to believe that it could readily explain around 35% of the drop in residual waste, but that the remaining reduction demanded closer examination. The County considered the possibilities for legitimate and illegitimate changes in behaviour which could explain the 'unexplained' reduction in residual waste.

As regards legitimate routes, the following were considered:

1. A reduction in the amount of inert building / DIY waste generated;

2. A change in the use of nappies from disposables to re-usables. Statistics and compositional analysis suggested a reduction from 9.4kg/inhabitant before the scheme to 7kg/inhabitant after implementation; and

3. Consumer choices. Some evidence suggested that consumers were changing consumption habits to reduce the quantity of packaging and / or waste generated.

As regards illegitimate changes, the following were considered:

1. Burning of waste by households. Regarding this issue, the County sought information from chimney sweeps. They do not think is an issue, and there is evidence to suggest that if burning did occur in gardens, that neighbours would tend to report such activity;

2. It was suggested that some householders might take their waste to their place of work. Though this is considered a possibility, there was no evidence to suggest it was, or is, an explanatory factor. The County is inclined not to believe that this is a major issue given the approbation that might follow from peers;

3. Some waste might be flushed into sewers (down toilets), but the County made checks with water companies and there was no evidence to suggest a change in the nature or quantity of sewage; and

4. Fly-tipping could have taken place. There appear to be two critical places where fly-tipping occurs. One is at the mini recycling centres, the other is at litter bins. There was no evidence of disposal of waste in fields and forests and so forth. The evidence that the County gathered through its monitoring suggested that clearly fly-tipping happens, but the degree to which there has been a significant increase is not clear. There has been an increase in the number of cases prosecuted. The attitude of the County was that this is something which happens anyway, it seems that it might have occurred more frequently in the early days of implementation than at present, and that with the support of politicians, the clean-up of bring sites will occur more often so as not to attract others to follow suit. It is not possible to quantify the material cleaned up. On the one hand, there was no pre-

scheme data, and on the other, once cleaned up, the waste sometimes enters the 'formal' system without data being separately recorded.

The upshot of the above discussion is that it is not clear how one should account, exactly, for around one sixth of the reduction in residual waste achieved. However, the degree to which this is related to undesirable activities is believed to be small, though clearly not zero.

#### 6.1.2.8 Summary

This is a system whose performance is outstanding. An already high-performing system was made more so by the application of an intelligently designed DVR charging scheme. The rigour with which the system was contemplated, prepared, implemented, and then monitored demonstrated a high level of commitment to the cause of pursuing a sustainable waste management strategy through appropriately incentivising households.

The use of the hybrid approach to charging is especially interesting. Weight-based systems, if not designed in this manner, might lead to high set out rates of bins with small quantities of waste. Here, the inclusion of both weight- and frequency-based charges acts to incentivise low set out rates of refuse bins, reducing the costs of the collection service. Citizens have clearly responded to this incentive (so that savings in the quantity of residual waste set out for collection translate into genuine savings in the collection system). The study clearly points the way towards 'tailor-made' incentive schemes to achieve specific objectives.

## 6.2 Evidence of Impacts

In this section, we consider the effects on waste prevention, and the evidence describing the parameters which can be considered to affect waste generation.

The evidence in respect of the impact of charging systems on waste prevention is highly variable, with the strength of the association potentially varying with the nature of the charging system (see Section 6.2.2) as well as the nature of the recycling system itself. Price also plays a role (see Section 6.2.4). Depending on scheme types, and charge levels, the *quantity of waste collected* can fall by 10% and sometimes more, as with the above case studies. Our review of literature suggests that greater reductions tend to be achieved where the system in existence prior to charging included free garden waste collections, and where the charging system introduces charges for garden waste. This is because, we believe, this incentivises additional home composting / reduced generation of waste in the first place.

Studies sometimes do not explicitly state the effect in terms of waste prevention. Frequently, the headline figure reported is the *reduction in the quantity of material sent to landfill or incineration*. It is not unusual for this figure to be in excess of 30%. However, it is clear that this does not necessarily represent 'waste prevention', but represents a combination of waste prevention and additional recycling.

The results of a study carried out by Skumatz, which sought to estimate source reduction effects of charging schemes through two approaches, are shown in Table 7.[86] These effects are not as great as one finds in many cases where charging schemes are implemented in Europe. Specifically, many weight-based and frequency-based schemes report much higher reductions in waste collected. Skumatz reported that no full-scale weight-based programmes were operating in the USA. Bin-volume or tagged sack schemes were in widespread operation (for example Seattle, San Francisco, etc.). Hence, the US figures are not necessarily representative of EU experiences.

Table 7: Source Reduction Estimates from Variable-rate Waste Disposal Programmes from Two Estimation Methods

| | Community Comparison Method | Time Series Method |
|---|---|---|
| Total effect of variable-rate programme | 16 % | 17.3% |
| Minus recycling effect for variable rates | 5-6% | 6.9% |
| Minus yard-waste effect for variable rates | 4-5% | 4.6% |
| **Leaves estimate of source-reduction effect attributable to variable rate programs** | **5-7% from source reduction** | **5.8% from source reduction** |

*Source: SERA (2000) Measuring Source Reduction: Pay as you Throw / Variable Rates as an Example, Seattle: Skumatz Economic Research Associates.*

Proietti cites a study conducted for the VROM (The Netherlands Ministry of Housing, Spatial Planning and the Environment) by KPMG in 1995-1996.[87] This reported a 12 to 30% reduction in *household refuse* owing to DIFTAR schemes, including:

- 6 to 8% due to improved sorting by householders;
- 3 to 10% due to 'unintended activity'; and
- *3 to 12% due to genuine prevention measures (calculated).*

These figures are reasonably well aligned with those from Skumatz above.

Not all studies report considerable quantities of waste 'being prevented', though the majority report some effect. The case of Comuni dei Navigli indicated little by way of

[86] SERA (2000) Measuring Source Reduction: Pay as you Throw / Variable Rates as an Example, Seattle: Skumatz Economic Research Associates.

[87] Stefano Proietti (2000) The Application of local Taxes and Fees for the Collection of Household Waste: Local Authority Jurisdiction and Practice in Europe, Report for the Association of Cities for Recycling, Brussels: ACR.

reduction in waste quantities.[88] This was a sack-based system where the approach to collection of organic wastes focused on the collection of kitchen waste. Given that in most studies, a key contribution to waste prevention appears to come through the increased uptake and intensity of home composting (see comment above), the likelihood that relatively little garden waste was being collected before the charging system was implemented suggests that the 'opportunity' for waste prevention through this avenue would have been reduced.

Most case studies are based upon 'before and after' assessment as though waste quantities are static. Evidently, what ought to happen is that comparisons should be made against a counterfactual. This type of approach is surprisingly rare in case studies, though some reports make comparisons between areas with, and areas without charging systems in place over a given period of time. [89]

Even so, there are some key caveats which need to be borne in mind when looking at much of the literature:

- First of all, and probably most importantly, many studies appear to concentrate (it is not always made clear) on door-to-door collections of waste and do not take a 'whole system' view (e.g. how the quantity of waste taken to CA sites changes in the wake of the charging system's implementation); and
- Second, there is ongoing discussion as to the underlying cause of the 'disappearing waste'. Not all studies seek to explain the changes witnessed.

The second of these is considered below. Here, we concentrate on some of the impacts associated with charging and their links to the wider waste management system.

### 6.2.1 The Effect of Charging Understood in a System Context

In many countries, different parts of the waste collection infrastructure play a more or less important role. In particular, the scope of kerbside collections in countries such as Denmark tends to be relatively narrow (sometimes, just paper and card and glass are collected at kerbside). In the Netherlands, the scope tends to include biowaste (and prior to the introduction of charging systems, garden waste has typically been collected free of charge), though it does not always include any packaging fractions. In other countries, notably Belgium, Austria and Germany, the scope of kerbside systems is generally more comprehensive. Bring sites tend to be more important as a means of recycling in countries such as Denmark, Netherlands, Sweden and Norway. Of course, there is local variation in the scope of kerbside collection schemes in most countries, but this is less true in countries / regions where:

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[88] Eunomia (2003) *To Charge or Not to Charge?* Final report to IWM (EB).

[89] Bischof, R., M. Chardonnens, M. Hugi, M. Textor, D. Lehmann, W. Siebert and K. Ammon (2003) La Taxe au Sac Vue par la Population et les Communes, Cahier de L'Environment 357, Berne: OFEPF; Arnold, Olivier, Damien George, Rachel Bauudry, Thomas Gaudin, Eve Toleedano d'Amorce (2005) Causes et Effets du passage de la TEOM a la REOM, Final Report, Ministere de L'Ecologie et du Developpement Durable, August 2005.

- Ordinances are in place for the collection of biowaste (Austrian, Germany, Catalunya (Spain), Flanders (Belgium), and 'voluntarily' in the Netherlands);
- Producer responsibility measures require materials organisation to fund, more or less completely, the kerbside collection of various materials, notably packaging materials (examples of 'full financial responsibility' being Germany, Austria, Belgium); or
- There are stipulations as to which materials should be collected and with which frequency (most notably, Flanders in Belgium).

In these cases, local variation is somewhat reduced.

Similarly, in different countries, civic amenity sites (or containerparks) play a more or less important role. The significance of this is that as charges are put in place on refuse collection, so there will be *own-price* effects (the demand for refuse collection will fall), but also *cross-price* effects (people may use the kerbside refuse service less, but use other services more). The literature has concentrated on cross-price effects vis-a-vis collections for recyclables and compostables *collected from the kerbside*. It would be equally helpful to examine cross price effects with respect to:

- The use of bring sites;
- The use of bulky waste services;
- The use of litter bins as a means of disposal;
- The use of civic amenity (CA) site (and non-CA site bring) recycling services; and
- The use of CA site services for refuse collection.

Our observation as to why this has not, generally, been carried out is that those seeking to understand price-responsiveness frequently have tended to see this as an academic exercise (an exercise in the application of econometrics) rather than one in which the nature of service provision might play a major role in determining outcomes. The quality of data, and of model specification, is likely to be critical to the results of any study. Practitioners refer, time and again, to the importance of having in place a quality, convenient collection service for a wide range of recyclables in order for charging systems to deliver the best outcomes.

The lack of clarity in some studies as to exactly what is being measured certainly has implications for the results (as they are often quoted) concerning charging schemes. It seems possible that, ironically, in schemes with *kerbside* recycling and composting schemes of *narrow* scope, the stated effects on total waste quantities may be exaggerated because more material is likely to be re-routed from the doorstep collection system (for example, to the bring systems or to containerparks, or to illegal activities). If there are limited opportunities for segregation of materials presented by the kerbside collection, waste may be squeezed in other directions. This would be especially true where:

- The recycling / composting service provision is more heavily oriented towards the provision of bring sites and CA sites;
- CA sites do not charge for the delivery of refuse-type waste; and

- The marginal cost of collections is high (for example, the costs of pick-ups per bin are relatively high).

A good example of a study suffering from this shortcoming appears to be that of Tønning for the Danish EPA.[90] It may also be true that the results of VROM study cited by Proietti appear as they do because of the nature of the collection systems involved.[91] It is difficult to know from the presentation of data whether studies suffer from the same shortcomings.

Some studies do seek to account for changes in waste flows when charging systems are introduced.[92] Indeed, work in Switzerland for the OFEFP (Office fédéral de l'environnement, des forêts et du paysage, or Federal Office for the Environment, Forests and Countryside) makes the point, regarding the reduction in residual waste quantities:

> *It is worth pointing out that in some municipalities, since the pay-per-sack scheme was introduced, some small traders deliver combustible waste materials directly to incineration plants used for treating household refuse. As a result, these quantities do not appear any more in the statistics of the municipality.*[93]

In other words, some of the 'waste prevention' which is observed is likely to be related to waste from small traders being re-routed into channels where, presumably, it should have been in the first place. It is well known that trade waste leaks into municipal waste.

A study by Eunomia in 2003 sought to rectify the situation characterised by an absence of thorough system studies of charging schemes.[94] This highlighted significant effects in a scheme in the Treviso 2 area of Italy, where a reduction of around 12% was achieved through a frequency based scheme. Effects were also observed in Gent though here, it was difficult to disentangle the effects of charging from other, ongoing changes. In Nijmegen (a sack-based scheme), quantities did not change significantly. In this case, the recycling infrastructure was poor, and it may be

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[90] Tønning, Kathe (2000) Fordele og ulemper ved gebyrdifferentierede indsamlingssystemer for husholdningsaffald, Teknologisk Institut, Miljoprojekt 576, Report for the Danish EPA.

[91] Stefano Proietti (2000) The Application of local Taxes and Fees for the Collection of Household Waste: Local Authority Jurisdiction and Practice in Europe, Report for the Association of Cities for Recycling, Brussels: ACR.

[92] OVAM (1999) The Effect Of Household Waste Taxes And Retributions On The Amount Of Household Waste Offered, February 1999; Arnold, Olivier, Damien George, Rachel Bauudry, Thomas Gaudin, Eve Toleedano d'Amorce (2005) Causes et Effets du passage de la TEOM a la REOM, Final Report, Ministere de L'Ecologie et du Developpement Durable, August 2005; Gellynck, Xavier and Peter Verhelst (2005) Onderzoek naar de gemeentelijke huisvuilbelasting- en retributiesystemen inclusief voor KMO's en zelfstandige ondernemers in Vlaanderen op 1 januari 2003, Report to OVAM, March 2005.

[93] Bischof, R., M. Chardonnens, M. Hugi, M. Textor, D. Lehmann, W. Siebert and K. Ammon (2003) La Taxe au Sac Vue par la Population et les Communes, Cahier de L'Environment 357, Berne: OFEPF

[94] Eunomia (2003) To Charge or Not to Charge? Final report to IWM (EB).

that households merely crammed their sacks more than previously. In Fingal, in Ireland, no clear information existed, but once again, half of households had no kerbside recycling scheme available to them, and the scheme led to many protests concerning the system.

In France, one study examined the differences between 4 municipalities without incentive based schemes, and six who had implemented them.[95] Generally, rates of recycling were higher and residual waste per inhabitant was on the decline in those areas with incentive based systems in place (see Table 8).

The weakest impact appears to have been in the volume based bin system. The final column is a telling one, showing changes in the proportion of waste managed through bulky waste and CA-site type locations. The study suggests that in several cases, the infrastructure in this regard was improved over the period, drawing more materials through this route. More analysis would be required to see whether total waste quantities were actually increasing or falling under the schemes.

Bischof et al compare residual waste generation in areas with and without sack-based systems in place.[96] Figure 11 and Figure 12 below show the rate of reduction following the introduction of a sack-based system in those areas where charges are in place (to the year 2001), the rate of increase between 1997 and 2001 in those areas without sack based schemes, and the total quantity of residual waste collected.

For those in areas with pay-per-sack schemes, the reduction has been of the order 20-40%, with an average figure of 30%. Following the system's implementation, growth tends to settle at around 0.5-0.8%, in line with those without sack-based charges. In areas without sack-based schemes, growth has been between 2%-13%, averaging 6%. The study notes, however, particular circumstances affecting Lausanne and Essertines in the period.

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[95] Arnold, Olivier, Damien George, Rachel Bauudry, Thomas Gaudin, Eve Toleedano d'Amorce (2005) Causes et Effets du passage de la TEOM a la REOM, Final Report, Ministere de L'Ecologie et du Developpement Durable, August 2005.

[96] Bischof, R., M. Chardonnens, M. Hugi, M. Textor, D. Lehmann, W. Siebert and K. Ammon (2003) La Taxe au Sac Vue par la Population et les Communes, Cahier de L'Environment 357, Berne: OFEPF

Table 8: Performance of Schemes in France With and Without Charging Schemes in Place

| | Area | Date of Implementation of Charging Scheme | Effect on Residual waste | Effect on Recycling | Date of Introduction of Separate Collection | Bulky Waste and CA |
|---|---|---|---|---|---|---|
| No Incentive to Hhlds | Communauté de Communes du Rougier de Camares | 2003<br>per inhabitant | - 5%/yr in 2 yrs<br>525 --> 471 kg/inh | + 13%/yr since 2003<br>70 --> 90 kg/inh | 2003 | 2002-2003 : stable<br>2004 : + 24% |
| | Communauté de Communes du Pays Châtillonnais | 2003<br>per inhabitant | + 3%/yr in 2 yrs<br>549 --> 588 kg/inh[a] | + 5%/yr in 2 yrs<br>109 --> 121 kg/inh * | 2002 | 2002-2003 : stable<br>2004 : x 12 |
| | SMC du Haut Val-de-Sèvres | 2002 : TEOM<br>2003 : return to REOM<br>per inhabitant | - 1%/yr in 2 yrs<br>265 --> 259 kg/inh | + 4%/yr in 2 yrs<br>70 --> 76 kg/inh | 2003 | 1999-2000 : + 22%<br>2000-2004 : - 2%/yr |
| | SMICTOM d'Alsace Centrale | 2003<br>per inhabitant | over 7 yrs : +/- 1%<br>average : 280 kg/inh | + 2%/yr over 7 yrs<br>134 --> 151 kg/inh<br>(but population not known with precision) | 1994 | 1997-2004 : + 17%/yr |
| With Incentive to Hhlds | SICTOM Loir et Sarthe | 2004 (trial)<br>number of lifts | Absence of data<br>after 1 yr, 273 kg/inh | 2003-2004 : + 31%<br>81 --> 106 kg/inh | 1980s | 2003-2004 : +7% |
| | Communauté de Communes de la Vallée de Kaysersberg | 1997<br>volume | 1997-2004 : -2%/an<br>381 --> 332 kg/inh | 1997-2003 : + 8%/inh<br>87 --> 139 kg/inh | Before 1996 | 1997-2001 : + 69%/yr<br>2001-2003 : + 3%/yr |
| | SM de Montaigu-Rocheservière | 2001<br>volume and number of lifts | 1999-2004 : -12%/yr<br>244 --> 126 kg/inh | 1999-2004 : + 11%/yr<br>49 --> 81 kg/inh | 1999 | 1997-2001 : + 20%/yr<br>2001-2004 : + 1% |
| | Ville de Besançon | 1999<br>volume and frequency | 2000-2003 : - 15%/yr<br>334 --> 207 kg/inh | 2000-2003 : + 9%/yr<br>57 --> 74 kg/inh | 1999-2005 | Unknown |
| | Communauté de Communes du Pays de Villefagnan | 2001<br>sack | 2000-2001 : -64% ;<br>2001-2003 : + 6%/yr<br>313 --> 113 kg/inh | 2001-2004 : + 5%/an<br>81 --> 93 kg/inh | 2001 | Opened in July 2001<br>2002-2003 : + 26% |
| | Communauté de Communes de Ribeauvillé | 2002<br>weight | 2001-2004 : -14%/yr<br>297 --> 186 kg/inh | 2001-2004 : +17%/yr<br>186 --> 297 kg/inh | 1990s | Unknown |

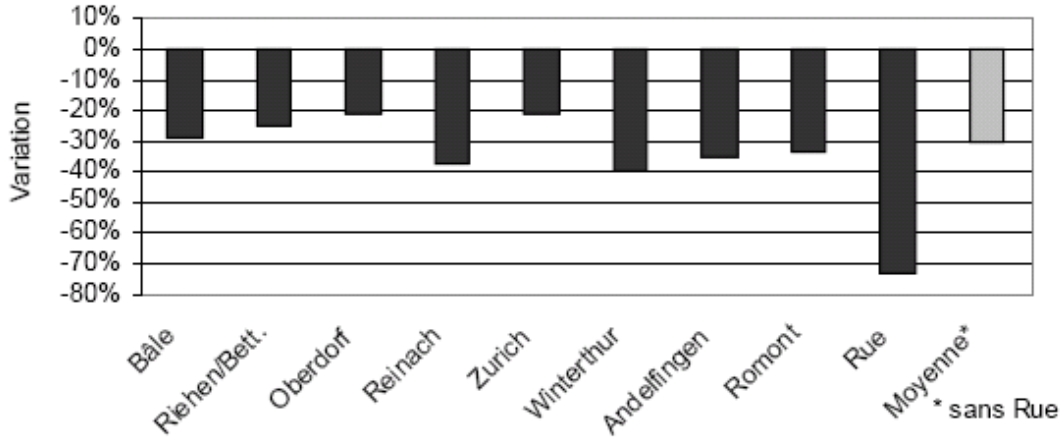
*Note:*

*[a] Excludes glass*

*Method of calculation: the reference point for the calculation of the evolution of tonnages was the year preceding the implementation of the charge, as far as possible*

*Source: Arnold et al (2005)*

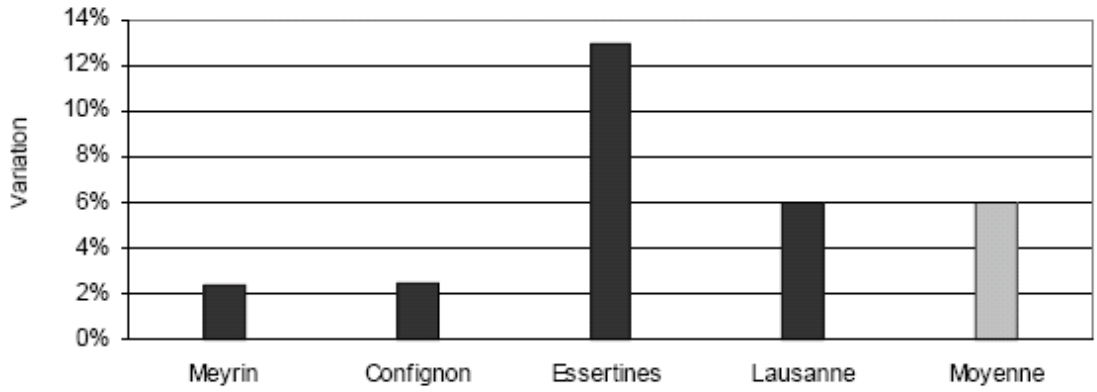
Figure 11: Reduction in Residual Waste in Swiss Schemes Using Pay-per-sack




*Bischof et al (2003)*

*Note: 'Moyenne' = average – the asterisk 'sans Rue' implies an average excluding the case of Rue (deemed to be an exceptional case)*

Figure 12: Increase in Residual Waste in Swiss Schemes Without Pay-per-sack




*Bischof et al (2003)*

*Note: Moyenne = Average*

The work of OVAM has also been consistently 'holistic'. Work for OVAM involved regression analysis on 306 municipalities with what appears to be good quality data (including collections from containerparks, etc.).[97] The aim was to understand the effect of certain variables on residual waste per inhabitant. The form of the equation was as follows:

[97] Gellynck, Xavier and Peter Verhelst (2005) Onderzoek naar de gemeentelijke huisvuilbelasting- en retributiesystemen inclusief voor KMO's en zelfstandige ondernemers in Vlaanderen op 1 januari 2003, Report to OVAM, March 2005.

$$Y = 131.456 + 1.822\ X_2 - 0.396\ X_5 - 0.159\ X_6 + 26.157\ X_9 - 20.984\ X_{11}$$

Where Y = residual waste per inhabitant (expressed in kg/inhabitant)

$X_2$ = income per inhabitant (expressed in '000 Euros per annum)

$X_5$ = the average cost per household per year of residual waste collection (in Euro per year, for the average family in the municipality)

$X_6$ = the proportion of total costs financed by direct costs of residual waste collection and treatment (%) (this refers to the fact that the cost of waste management in Flanders is composed of direct (to the citizen) and indirect (taxes) means of financing);

$X_9$ = the frequency of collection of residual waste from the doorstep (1 = weekly); and

$X_{11}$ = a dummy variable (whether or not there is a collection for so-called GFT waste, i.e. vegetable, fruit and grass / soft prunings).

The suggestion is that for every additional Euro paid for residual waste, quantities fall by around 400g per inhabitant. The average payment for residual waste in Flanders is around €60-70 per household per year, giving a net reduction of the order 24-28 kg per inhabitant per year (with additional effects potentially attributable to the proportion of costs covered through direct means). The Flemish experience is significant in that the country has set a target for residual waste to be no more than 150kg per inhabitant across the whole region. The above equation does not give evidence of waste prevention per se. It does, however, give some indication of the combined effect of recycling and waste prevention at the level of the individual. It does so taking into account a range of other variables.

On balance, one might say the literature offers support for the hypothesis that charging systems encourage waste prevention, but that the quality of evidence is somewhat variable, so that the accuracy of the reported effects is not what it might be. It would be reasonable to assume, however, that reported effects in schemes where the scope and convenience of the kerbside collection service is extremely good – as it is in Flanders, for example – will tend to give a more accurate picture than in those where the collection service is less convenient, and where it is clear that the focus is on waste collected from the doorstep only. Evidently, the nature of the marginal incentive may be important. A key determinant of this is the nature of the charging system, and this is explored below.

### 6.2.2 Influence of Type of Charging System on Waste Prevention Effect

A study was undertaken for VROM by KPMG in 2001.[98] The principal objectives of the study were to understand the fate of the materials diverted from the residual waste stream, in particular, to understand the degree to which the reduction in refuse

[98] KPMG Bureau voor Economische Argumentatie (2001) Gedragseffecten van Tariefdifferentiatie. The Hague: KPMG.

collected was due to a) 'positive' changes (in respect of genuine waste reduction) and b) 'negative' changes (in respect of evasive activities / illegal disposal.

Questionnaires were used to develop an understanding of households' behaviour, though the study team recognized that such questionnaires were unlikely to be reliable (since people would not voluntarily declare themselves to be acting illegally). Generally, it was concluded that both types of behavioural change –positive and negative – would occur, but it was impossible to generalize about the relative proportions of waste reduction which occurred through 'legal' and 'illegal' routes. Not only were there huge differences between different towns (and the difference between rural and urban municipalities was a feature), but it was also clear that systems functioned best where thorough provision was made for convenient source separation.[99]

3 types of each of 4 different systems were examined, the 4 types being:

1. Volume based bin;
2. Volume and frequency based bin;
3. Bag based; and
4. Weight based.

Comparative results across the system types were reported as shown in Figure 13.[100] These suggest that in terms of waste prevention, the order of ranking should be:

1. Weight based.
2. Bag based / Volume and frequency based bin; and
3. Volume based bin.

However, the study and its outcomes could be questioned on the basis that:

1. The municipalities using different systems may not be comparable in the manner suggested (there may be other factors compounding the analysis); and
2. Possibly more significantly, it has not been possible for us to clarify whether this study addresses only door-to-door collections (see above discussion).

Even so, the ranking is plausible to the extent that it is aligned with the nature of the marginal incentive.

The work by Tønning for the Danish EPA also suggests that weight-based systems deliver the strongest effect (see Table 9).[101] We know that this study does not include

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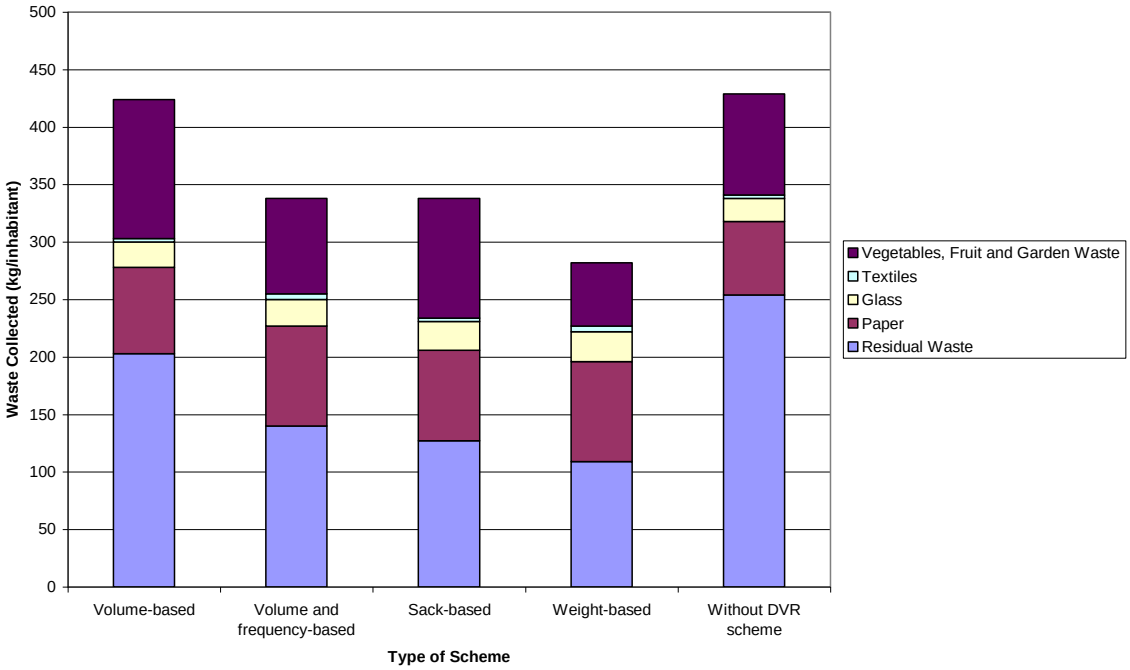
[99] Personal communication with S. van Weele, KPMG Netherlands.

[100] AOO (2001) Afval Informatief, Informatiebulletin, 06, Juin 2001.

'whole system' wastes, however, so it could not be entirely ruled out that what is happening is that waste is simply moving through different routes in the system. Other details of the study – for example, the rates of home composting quoted – seem to imply that it is unlikely that this could account for all of the differences across schemes.

Figure 13: Quantities of Separated Waste and Refuse by Charge System Type, 1999




*Source: AOO (2001)*

Table 9: Performance of Different Waste Systems in Denmark

| | Residual Waste (kg/inhab) | Paper and Card | Glass | Total | (Increase in) Paper Captures | (Increase in) Glass Captures |
|---|---|---|---|---|---|---|
| Weight-based | 325 | 108 | 38 | 471 | 71% | 87% |
| *Reference* | 729 | 67 | 34 | 830 | 41% | 77% |
| *Difference* | -404 | 41 | 4 | -359 | (+30%) | (+10%) |
| *Difference in %* | -55% | 61% | 12% | -43% | (+73%) | (+13%) |
| Volume-based | 552 | 104 | 40 | 696 | 61% | 89% |
| *Reference* | 660 | 76 | 30 | 766 | 44% | 67% |
| *Difference* | -108 | 28 | 10 | -70 | 17% | 22% |
| *Difference in %* | -16% | 37% | 33% | -9% | (+39%) | (+33%) |

*Source: Tønning (2000)*

[101] Tønning, Kathe (2000) Fordele og ulemper ved gebyrdifferentierede indsamlingssystemer for husholdningsaffald, Teknologisk Institut, Miljoprojekt 576, Report for the Danish EPA.

Another Dutch study looked at data from the Netherlands Waste Management Council (AOO) for 1998, 1999 and 2000 to estimate the effects of different charging schemes.[102] The study suggested that:

- Weight-based schemes reduce total waste by 38%;
- Sack-based schemes with compostable waste also charged reduce total waste by 36%;
- Where compostable waste is not charged for, the reduction in total waste is 14% (the difference in the two is reflected mainly in the quantity of material collected separately from the kerbside);
- The frequency based system delivers a reduction in total waste of 21%; and
- The volume based bin system delivers a reduction in total waste of 6%.

This suggests that it is not only the approach to charging for residual waste, but also the approach to charging of garden waste that is important. The results are probably also indicative of the Netherlands experience where garden waste had previously been collected free of charge. As Linderhof notes:

> *Except for the Tobit specification, the price elasticities of compostable waste are higher than the non-recyclable waste elasticities. The reason is that in the case of compostable waste there is an alternative of home composting. Such an alternative is not present in the case of non-recyclable waste.*

Such changes might not be expected where no such collection has ever been in place (as in the Comuni dei Navigli case).[103] This indicates a possible element of 'path-dependence' in the measured effectiveness of charging systems in respect of waste prevention.

The suggestion from the review is that weight-based systems can deliver more waste prevention. Other systems – where the charge structure imparts incentives at the margin for prevention – also fare well. However, schemes where households subscribe to a given bin size with no frequency based charge have little or no incentive to reduce waste beyond the capacity of the bin to which they have subscribed. It is not surprising, therefore, that such schemes do not show strong evidence of waste prevention.

Also important in determining the scale of waste prevention is the way garden waste is dealt with (especially in authorities outside dense urban areas). Because of the characteristics of garden waste – it tends to increase in quantity when collected free of charge, but it can be dealt with reasonably comfortably in the home where collections are not free - the effects of charging (or not) for garden waste have to be considered in the context of the pre-existing collection infrastructure (was the collection free of charge before, or not?).

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[102] Dijkgraaf, E., and Gradus, R. (2003) Cost Savings of Unit-Based Pricing of Household waste, the case of the Netherlands. Rotterdam: OCFEB

[103] Eunomia (2003) To Charge or Not to Charge? Final report to IWM (EB).

### 6.2.3 Quantifying Prevention Effects

There is a considerable amount of information concerning the price-responsiveness of households and communities to charging systems. The majority highlights price responsive behaviour, with the responses being weakest in the cases where the systems are based upon volume only. Only one study has estimated the elasticity for a frequency-based scheme.

Some meta-studies of charging schemes have been carried out (in the Netherlands, Flanders, and North America, for example), and these have sought to understand the effects of different levels of charge on the propensity to recycle, and on total waste quantities. Econometric studies seek to do this through measuring price elasticities. Some of the estimates from the literature are given in Table 10. We have grouped studies according to the type of scheme used (with the exception of those looking at various scheme types). It can be seen that there is a tendency for the elasticity (for refuse quantities) to increase moving through volume-based, to sack-based to weight-based schemes. This ranking is reinforced in the work of Dijkgraaf and Gradus (see Table 11).[104]

Work by Bischof et al included a linear regression on the reduction in *residual waste* (i.e. including the effects of recycling and waste prevention) as a function of the price of a sack.[105] This revealed '*an observed price elasticity of 80% (coefficient of determination = 0.62)*' (see Figure 14). This is clearly a small sample from which to draw a relationship, but the suggestion is of some relation between price and residual waste quantities. Gellynck and Verhelst have a much greater sample size to work with.[106] Their plot, again for residual waste, is shown in Figure 15.

The plot for the Walloon Region shown in Figure 16 highlights the change in residual waste quantities over time in the municipalities of the region. It shows these figures for municipalities with '*pas d'incitation*' (no incentive charge), '*incitation volume*' (a volume based charge) and '*incitation poids*' (weight based charges). The residual waste quantities are lowest for those with weight-based schemes, slightly higher in municipalities with volume-based schemes, and highest for schemes with no incentive in place. Clearly, this does not control for all variables, yet the graphic is clearly suggestive. The graphic also shows the progressive take up of different scheme types, with fewer and fewer municipalities using no incentive-based charge (around 5% in 2003 as opposed to more than 60% in 1997).

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[104] Dijkgraaf, E., and Gradus, R. (2003) Cost Savings of Unit-Based Pricing of Household waste, the case of the Netherlands. Rotterdam: OCFEB

[105] Bischof, R., M. Chardonnens, M. Hugi, M. Textor, D. Lehmann, W. Siebert and K. Ammon (2003) La Taxe au Sac Vue par la Population et les Communes, Cahier de L'Environment 357, Berne: OFEPF

[106] Gellynck, Xavier and Peter Verhelst (2005) Onderzoek naar de gemeentelijke huisvuilbelasting- en retributiesystemen inclusief voor KMO's en zelfstandige ondernemers in Vlaanderen op 1 januari 2003, Report to OVAM, March 2005.

Table 10: Empirical Estimates of the Effect of Unit-pricing

| Study | Data | Change in Refuse | Change in Recycling |
|---|---|---|---|
| Wertz (1976) | Volume based - Compares subscription program in San Francisco with flat fees imposed by "all urban areas" | ε= -0.15 | |
| Jenkins (1993) | Volume-based - Panel of 14 cities (10 with user fees) over 1980-88 | ε = -0.12 | |
| Hong et al. (1993) | Volume based - 1990 survey of 4,306 households in and around Portland, Oregon. | No significant impact | Unspecified positive relationship |
| Strathman et al. (1995) | Volume-based - Seven year (1984-1991) time series in Portland, OR | ε = -0.11 | |
| Van Houtven and Morris (1999) | Volume (household survey data) | ε = -0.10 | |
| Van Houtven and Morris (1999) | Volume (municipality data) | < 0 | |
| Reschovsky and Stone (1994) | Sack-based - 1992 mail survey of 1,422 households in and around Ithaca, NY. | | No significant impact |
| Fullerton and Kinnaman (1996) | Sack-based - Two-period panel of 75 households in 1992 | ε = -0.076 (weight)<br>ε = -0.226 (volume) | Cross-price elasticity is 0.073 |
| Podolsky and Spiegel (1998) | Sack-based - 1992 cross-section of 159 municipalities in NJ, 12 with unit-pricing | ε = -0.39 | |
| Kinnaman and Fullerton (1997) | Sack-based - 1991 cross-section of 959 towns across the U. S., 114 with unit-pricing | ε = -0.19<br>ε = -0.28 | ε = 0.23<br>ε = 0.22 |
| Van Houtven and Morris (1999) | Sack-based (household survey data) | ε = -0.26 | |
| Van Houtven and Morris (1999) | Sack-based (municipality data) | ε = -0.19 | |
| Hong (1999) | Sack-based – National data from Korean volume based waste fee | ε = -0.15 | |
| Linderhof et al (2001) | Weight-based – compostable waste (Oostzaan, Netherlands) | ε = -1.39 | |
| Linderhof et al (2001) | Weight-based - residual waste (Oostzaan, Netherlands) | ε = -0.34 | |
| Miranda et al. (1994) | Various - Panel of 21 cities over 18 months beginning in 1990 | 17%-74% reduction in garbage | Average increase of 128% |
| Callan and Thomas (1997) | Various (bag, tag, volume) - 1994 cross-section of 324 towns in MA, 55 with unit-pricing programs | | ε = -0.07 |
| Seguino et al. (1995) | 1993-1994 cross section of 60 towns in Maine, 29 with unit-pricing | 56% decrease | |

Table 11: Estimated Price Elasticities Under Different Charging Schemes

| | Price | Total | Unsorted | Compostable | Recyclable |
|---|---|---|---|---|---|
| Standard model | | | | | |
| *Weight* | 4.39 | -0.47 | -0.67 | -0.92 | 0.16 |
| *Bag, refuse and compostable* | 2.02 | -0.43 | -0.66 | -0.97 | 0.25 |
| *Bag, refuse* | 2.15 | -0.14 | -0.71 | 0.29 | 0.14 |
| *Frequency* | 3.91 | -0.22 | -0.28 | -0.40 | 0.08 |
| *Volume* | 1.94 | -0.06 | -0.12 | 0.01 | 0.01 |
| Model with environmental activism | | | | | |
| *Weight* | 4.39 | -0.40 | -0.53 | -0.81 | 0.12 |
| *Bag, refuse and compostable* | 2.02 | -0.36 | -0.51 | -0.85 | 0.20 |
| *Bag, refuse* | 2.15 | -0.07 | -0.58 | 0.40 | 0.09 |
| *Frequency* | 3.91 | -0.16 | -0.16 | -0.31 | 0.04 |
| *Volume* | 1.94 | -0.00 | 0.01 | 0.09 | -0.03 |

Figure 14: Plot of Reduction in Residual Waste per Inhabitant (y) against Sack Price (x) for Swiss Schemes

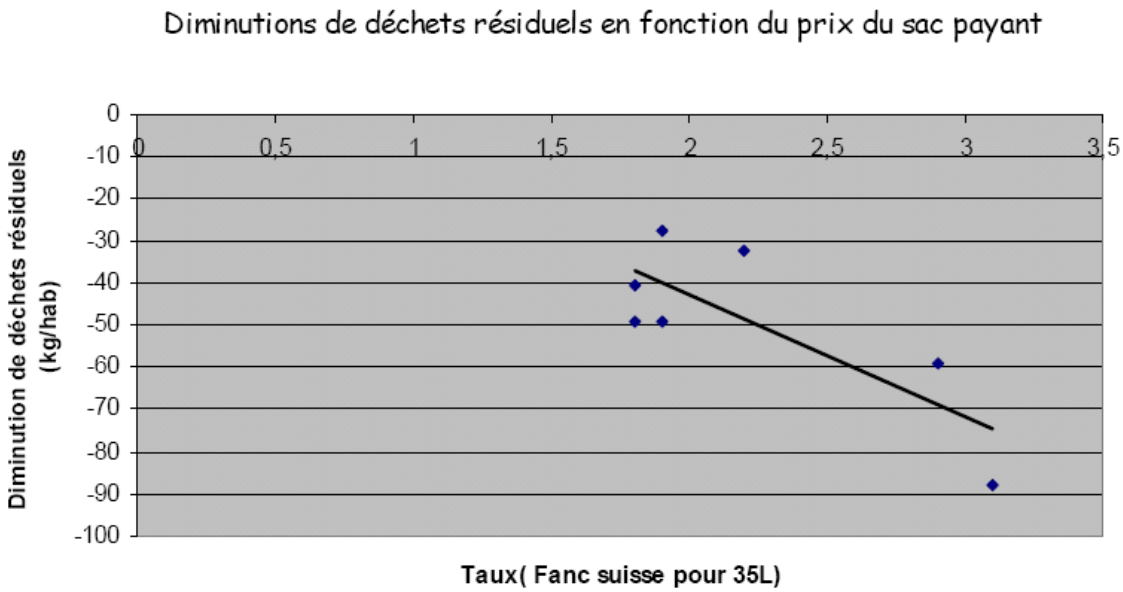



Figure 15: Plot of Residual Waste per Inhabitant (y) Against Sack Price (x) for Flemish Schemes

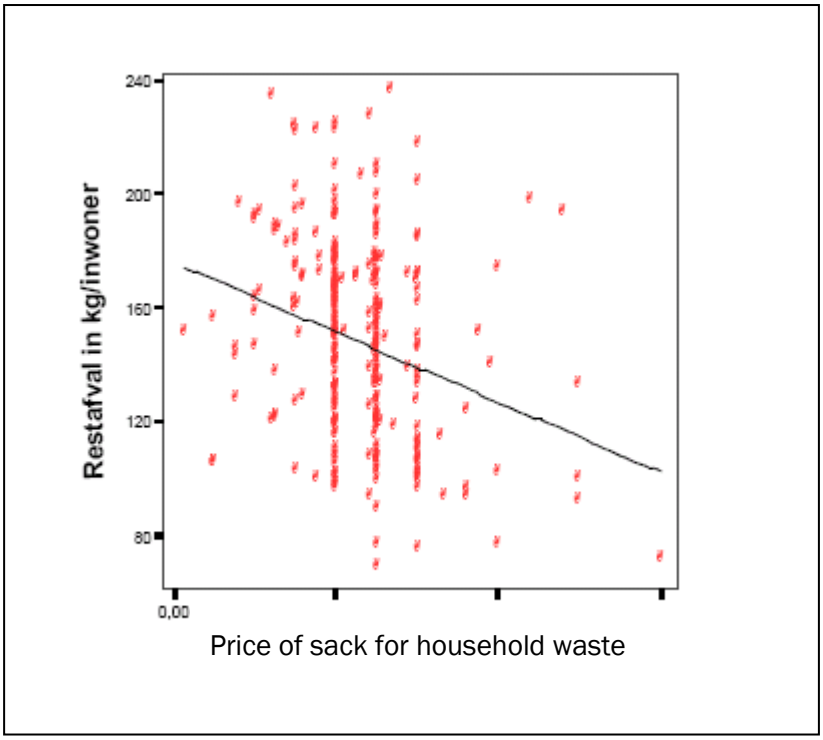
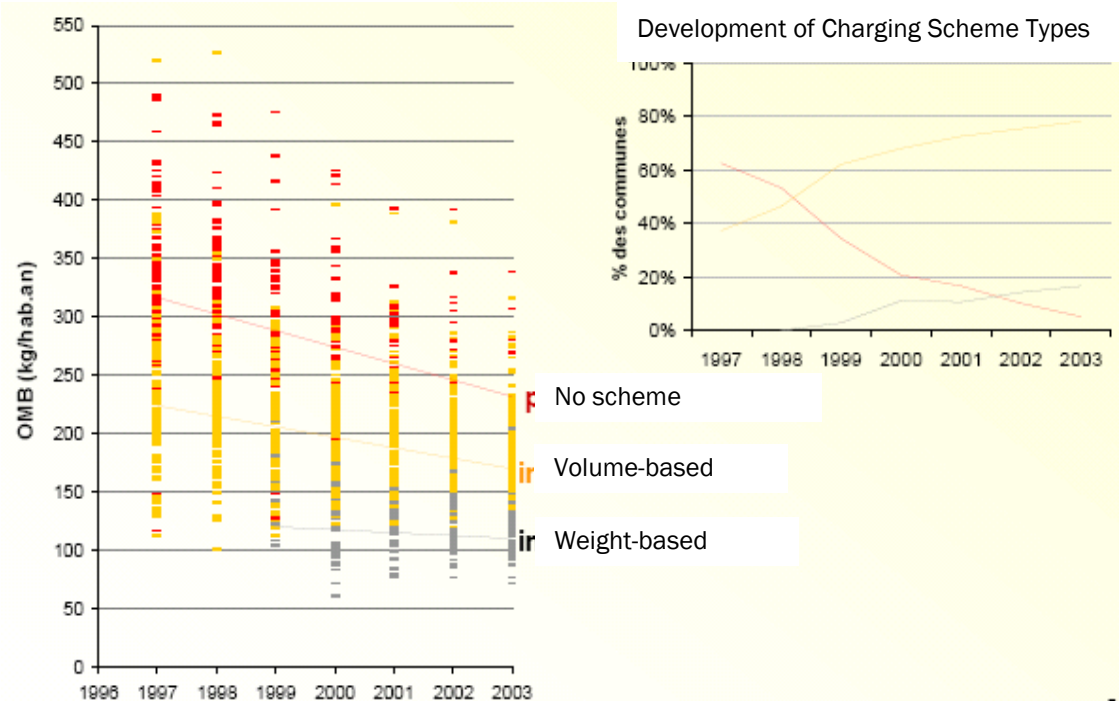


Figure 16: Residual Waste per Inhabitant Related to Charge Scheme




*Note: OMB (kg/hab.an) = Residual Waste per Inhabitant per Year*

SERA estimated that a $1 increase in rate differentials for 30 gallons of service (approx €1 per 140 l) increases recycling by 0.3%.[107] A $4 (approx €4) differential led to an increase in recycling of approximately 3%.

In summary, there is a considerable amount of information concerning the price-responsiveness of households and communities to charging systems. The majority highlights price responsive behaviour, with the responses being weakest in the cases where the systems are based upon volume only. Sack based (including composting) and weight-based schemes appear to give the strongest effects, although relatively few studies have sought to estimate price elasticities for weight-based schemes (reflecting the fact that much of the work on estimating elasticities is North American in origin, where 'subscription volume'- and sack-based schemes predominate) and only Dijkgraaf and Gradus have estimated the elasticity for a frequency-based scheme.[108]

### 6.2.4 The Nature of the Response to Price

In previous Eunomia work modelling local authority responses, we started with the view that it might be possible, with reference to the charge level, to model changes in:

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[107] SERA (2000) Measuring Source Reduction: Pay as you Throw / Variable Rates as an Example, Seattle: Skumatz Economic Research Associates.

[108] Dijkgraaf, E., and Gradus, R. (2003) Cost Savings of Unit-Based Pricing of Household waste, the case of the Netherlands. Rotterdam: OCFEB

- Total waste;
- Waste separated for recycling; and
- Waste separated for composting / digestion.

Given that a set of elasticities existed for different schemes from the work by Dijkgraaf and Gradus, we started by using these.[109] That work gives elasticity figures, alongside a prevailing price level (around which the elasticity was found to be relevant).

The price level was, for different schemes (weight-based, sack-based, frequency-based and volume-based) converted to an 'equivalent' charge expressed in terms of US$ per 35 gallons. The assumptions used in the conversion are set out in a footnote:

> *In the estimations we use the tariffs charged each time a can is emptied for the frequency system. For the volume system, we use the marginal weekly increase in the collection fee if a household subscribes to a larger can. To make comparisons between systems possible, the reported tariffs in Tables 1 and 5 are in real (2000) US dollars (using the GDP deflator) per 30 gal (114 l) of unsorted waste. Tariffs per mass unit are transformed to tariffs per volume unit using a regularly reported maximum weight of 0.76 kg/gal (3.79 l).*

There are difficulties in seeking to generalize across systems to derive elasticities. Furthermore, it is clearly not the case that the price incentives, converted in this way, are comparable. What matters is the way the marginal incentive operates on a particular household. Indeed, if the conversions were equivalent in all ways, one would expect similar elasticities at similar price levels across different charging systems. Dijkgraaf and Gradus produce quite different elasticities across the different systems (which reflect the way in which the incentive operates *at the margin*).

Also interesting is the conversion factor used. Weight based schemes are assigned a volume based charge on the basis of maximum densities. It might be expected that in bag, volume- and frequency-based schemes, residents would be more likely to compact refuse. There is no such incentive where schemes charge solely on the basis of weight. This would raise the question as to which density to use for the conversion (the high density likely in volume- / sack-based schemes or the lower density likely in weight-based schemes). If one uses a lower-end density figure, the actual 'volume based charge' equivalent for the weight-based charging system is likely to be half to two-thirds the value used by Dijkgraaf and Gradus.

Taking Dijkgraaf and Gradus' results at face value results in a situation in which the price-response curves for the different systems cross over at lower levels of the incentive. This is possible, but it seems somewhat unlikely in our view. It suggests, for example, that weight based schemes are effective at high charge rates, but relatively ineffective at low ones. Indeed, at low (but non-zero) charge rates, the curve suggests production of well in excess of 100% of the original amount of waste. At the

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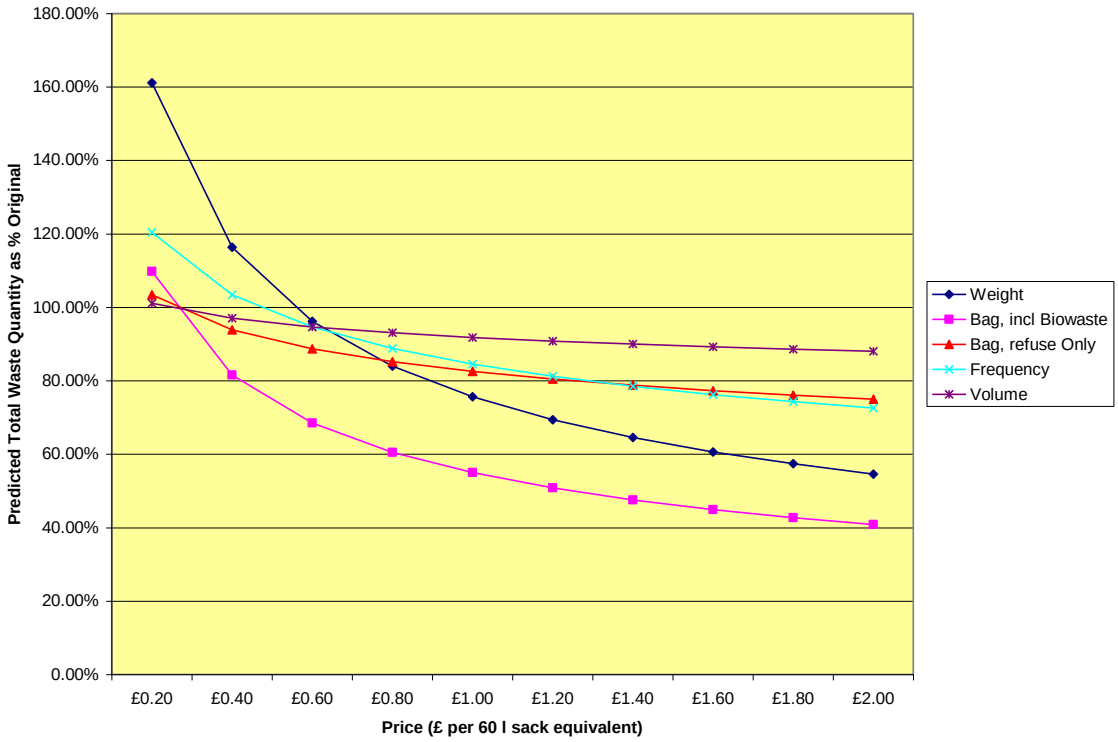
[109] Dijkgraaf, E., and Gradus, R. (2003) Cost Savings of Unit-Based Pricing of Household waste, the case of the Netherlands. Rotterdam: OCFEB

equivalent charge rate, the curve suggests sack-based systems which include biowaste in the charges will be achieving a reduction in total waste of around 30% (see Figure 17).

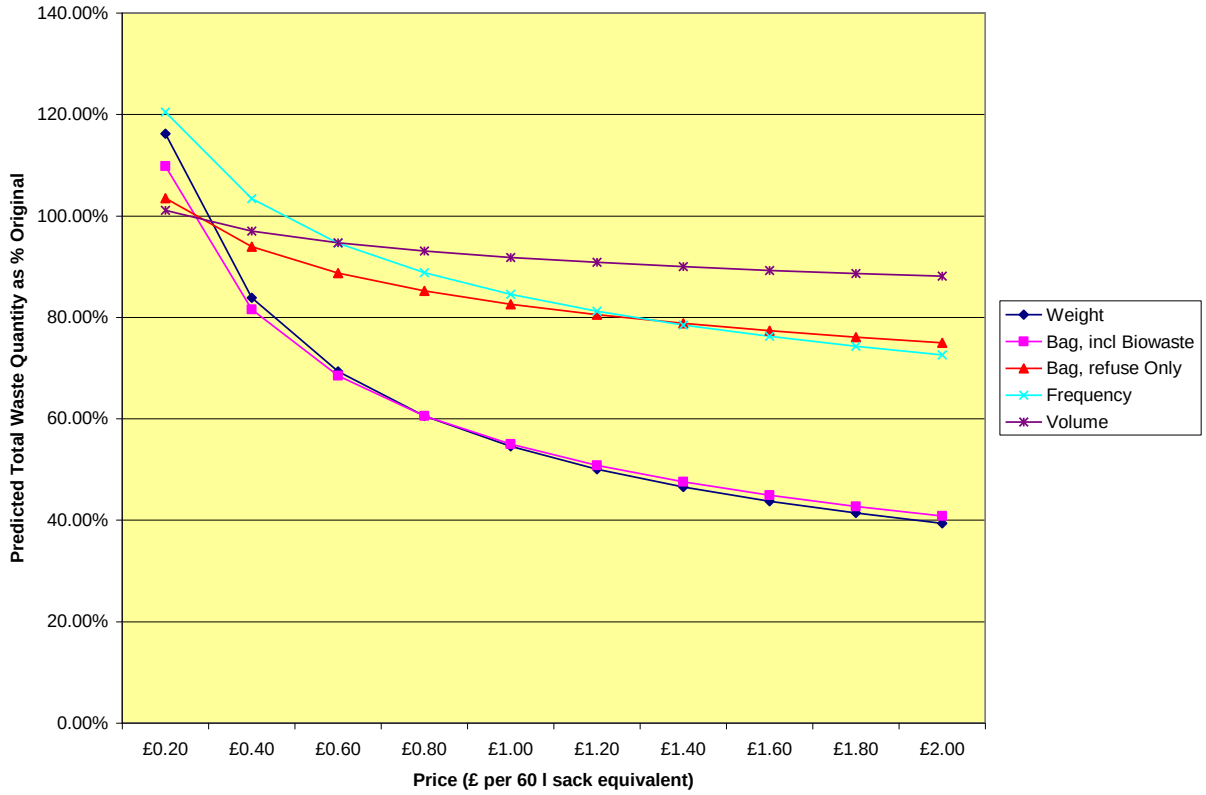
Figure 17: Plot of Waste Generation Against Charge Level




One possibility, therefore, is that the density conversion may be incorrect, based around too high a density for mixed waste. More generally, this highlights the potential problems associated with the use of elasticities, familiar as they are to economists, when considering changes in price level from zero to non-zero values. To the extent that elasticities may be well behaved for relatively marginal changes in price, they are less likely to be so for non-marginal changes. It seems likely, therefore, that even if elasticities can be estimated around specified price levels, they are unlikely to be constant across all price ranges. By way of example, the above Figure indicates a massive reduction of waste in the systems charging for sacks of refuse and biowaste. These look less plausible at higher charge levels suggesting that elasticities probably change (the price response is weaker) as the charge level increases.

These potential limitations aside, the situation may be such that the density conversion factor being used in the Dijkgraaf and Gradus study may be too high. To see how this affects the shape of the curves, we have plotted below the same price response curves using a lower density figure (of collected residual waste in the weight-based scheme) to convert from weight-based to volume based charges. For a density of 0.43 kg/gall (equivalent to 0.11 kg/l), the curves no longer cross. This is shown in Figure 18.

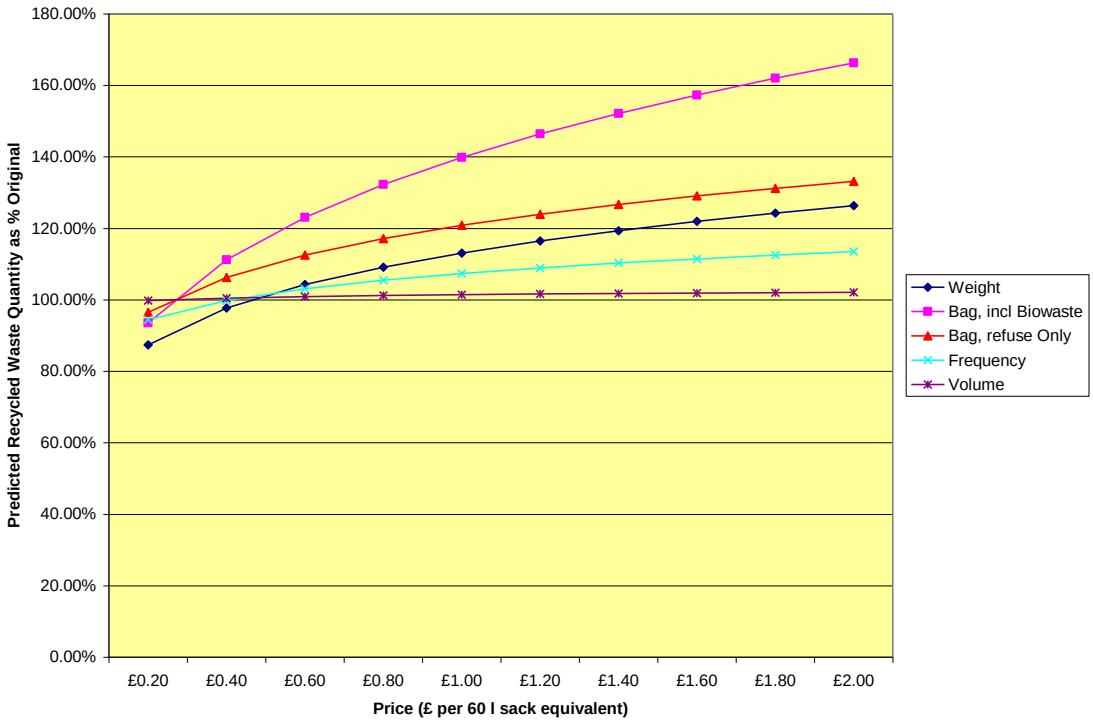
Figure 18: Plot of Waste Generation Against Charge Level, Using Lower Density for Weight-based Charging



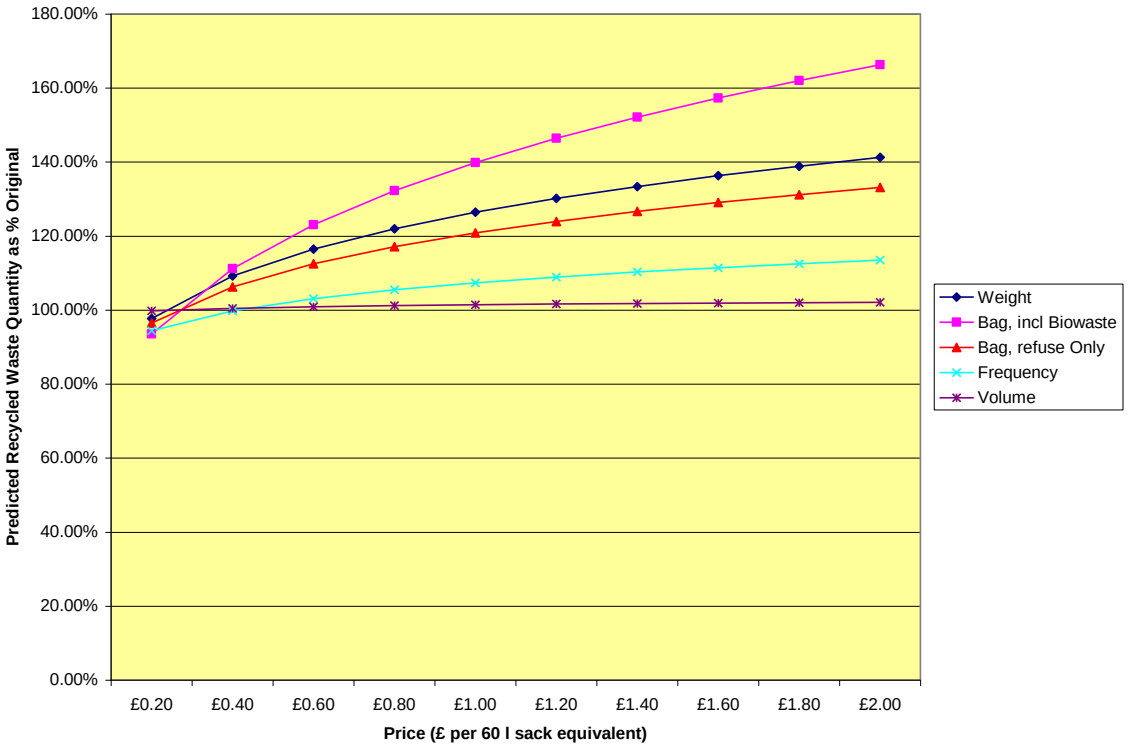

The same approach is taken below for the quantity of material being recycled (expressed as a percentage of the tonnage recycled before charging). Again, the curve for the weight-based scheme is counter-intuitive, indicating a quantity below the pre-charging scheme level before the charge rate has fallen to zero. The picture improves when the lower density is used.

Once again, the higher charge levels suggest what may be implausible increases in the quantity of material being recycled. Evidently, if one starts with an overall capture rate in excess of 60%, the 80% increase in this level is not possible for the targeted materials, suggesting a declining elasticity with price level.

## Figure 19: Plot of Waste Recycling Against Charge Level

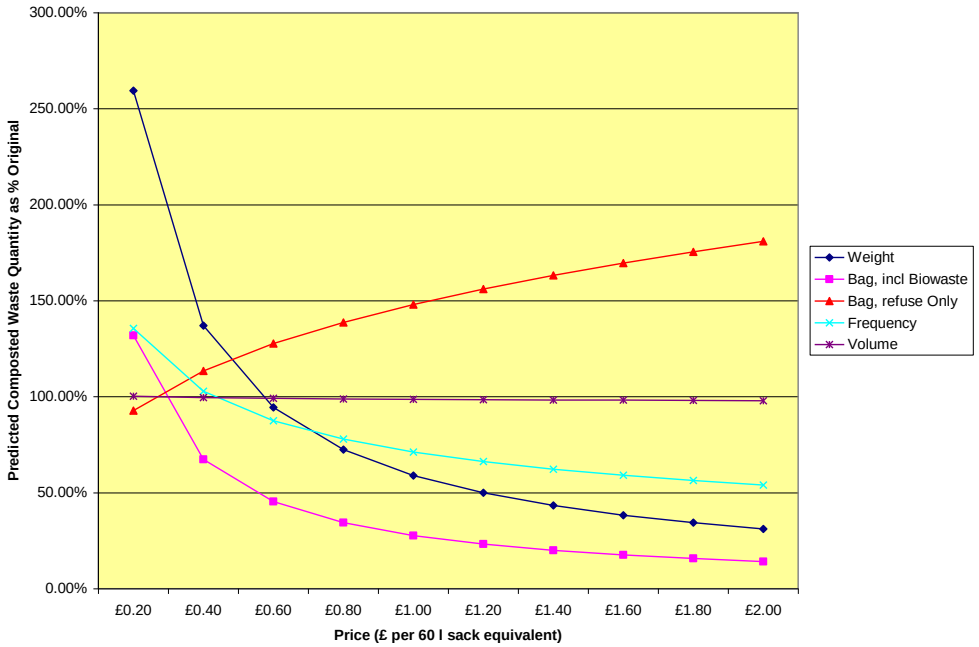



## Figure 20: Plot of Waste Recycling Against Charge Level, Using Lower Density for Weight-based Charging



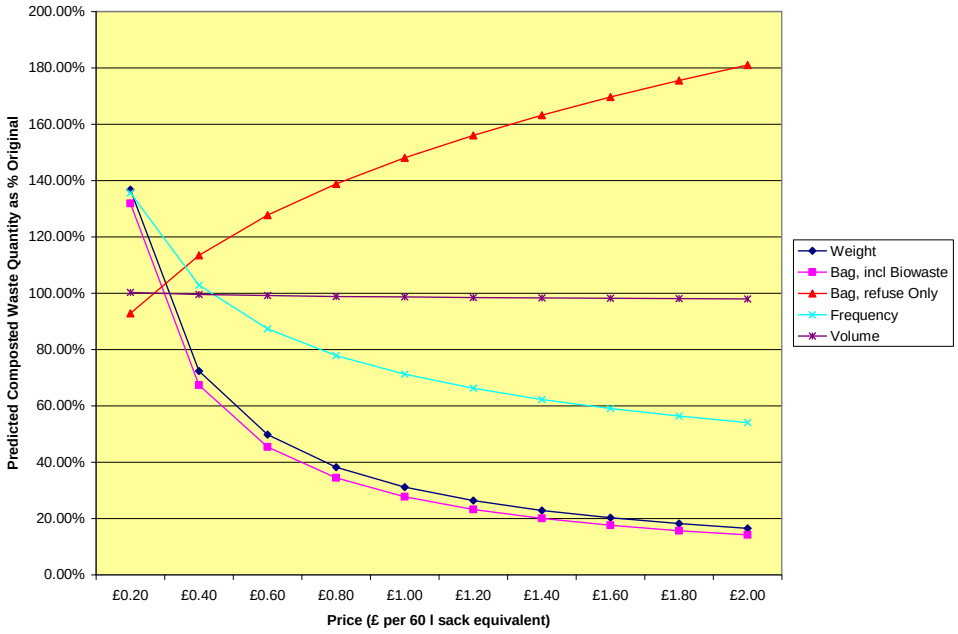
Finally, the plots for waste being composted against charge level are shown in Figure 21 and Figure 22.

Figure 21: Plot of Waste Composted Against Charge Level




Once again, with the density factor used by Dijkgraaf and Gradus, the curve for weight based charging shows counter-intuitive results, with reductions in composted waste at higher charge rates, but increases at lower rates. The revised density factor again acts to remove this counter-intuitive outcome.

Figure 22: Plot of Waste Composted Against Charge Level, Using Lower Density for Weight-based Charging



This discussion serves to highlight some of the issues associated with seeking to apply – attractive as it may seem – a simple constant elasticity approach. One can say:

- The constant elasticity approach is only likely to be valid around a specified price level only; and
- The nature of the responses ought to take into account the starting position of the schemes being considered.

### 6.2.5 Where Does the Prevented Waste Go?

Much of the literature does not seriously ask this question, the focus being on the 'headline numbers'. The 'before and after' fate of materials is generally poorly understood. This does not mean that municipalities are not interested in this issue. Rather, it probably means that many who study such systems are less familiar with the fact that if constraints are placed on one part of the waste management system, then material has a habit of flowing onto other parts of the system.

In the case study of Schweinfurt (see above), the director of the service was particularly keen to understand where the waste had gone.[110] Even after exploring all of the following, one-sixth of the reduction in residual waste remained unexplained:

1. Quantities of recycled material, before and after;
2. Composition of residual waste before and after;
3. Rates of home composting, before and after;
4. Possible causes of a reduction in the amount of inert building / DIY waste generated;
5. Change in the use of nappies from disposables to re-usables;
6. Consumer choices;
7. Burning of waste by households (regarding this issue, the County sought information from chimney sweeps).
8. Householders taking their waste to their place of work;
9. Waste being flushed into sewers (down toilets); and
10. Fly-tipping.

Clearly, this is a complex issue, and measuring any one of these in a manner which gives confidence is not straightforward.

The Treviso 2 area, where waste was reduced by more than 10%, estimates that there are three avenues through which waste quantities were reduced following charging:

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[110] Eunomia (2003) To Charge or Not to Charge? Final report to IWM (EB).

- Increases in home composting;
- Movement of commercial waste into the right avenues; and
- Genuine waste prevention.[111]

Another issue which may be important in some contexts is that of waste tourism. In a study for the OECD, in the case of the Spanish municipality of Torrelles de Llobregat, it was estimated that the phenomenon might account for 15% of the observed reduction in waste quantities.[112] The municipality would have been particularly likely to give rise to this type of abuse given a) its small size; and b) the fact that no other municipality in Catalunya (or Spain) was implementing such charges. Other things being equal, one would expect problems of waste tourism to be greater in such situations (the opportunity is greater and the benefit is maximised).

Linderhof et al report a study by the city of Oostzaan, which estimates that about 4–5% of waste is taken to surrounding municipalities (which is approximately 13–17% of the reduction in waste prompted by the introduction of the weight-based pricing system in the area).[113] Dijkgraaf and Gradus, on the other hand, carry out a modeling exercise to estimate the impact of variable charging on waste in surrounding communities.[114] Few of the coefficients were found to be of significance. The largest effect reported was for weight based systems, which the study suggested could lead to a 0.6% increase in the quantity of collected refuse in neighbouring communities.

One of the possible fallacies, which Eunomia highlighted in work for the OECD, was the assumption that if the amount of waste being collected after the scheme was less than amount collected before, all the reduction must be explicable through reference to actions of *householders*.[115] On the basis of experience with 'municipal' waste in the England, it is quite clear that the actions of commercial waste producers (and of waste collectors willing to collect such material in exchange for payment) have a significant impact on the quantity of collected waste. Charging schemes have the effect of in some (not all) cases of:

- Making the marginal cost of the collection of commercial waste non-zero. This can induce commercial waste producers to use the proper channels; and
- Making it more difficult for operatives to take back-handers in exchange for picking up commercial loads. Charging makes the link between waste collected and revenue received more transparent. This is especially true of

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[111] Personal communication, Paolo Conto, Consorzio Intercommunale Priula.

[112] Hogg, D. (2006) Impact of Unit-based Waste Collection Charges ENV/EPOC/WGWPR(2005)10/FINAL, Paris: OECD

[113] Linderhof, V., P. Kooreman, M. Allers and D. Wiersma (2001) Weight-based pricing in the collection of household waste: the Oostzaan case, Resource and Energy Economics 23, 359–371.

[114] Dijkgraaf, E., and Gradus, R. (2003) Cost Savings of Unit-Based Pricing of Household waste, the case of the Netherlands. Rotterdam: OCFEB

[115] Hogg, D. (2006) Impact of Unit-based Waste Collection Charges ENV/EPOC/WGWPR(2005)10/FINAL, Paris: OECD

schemes using chipped bins, in which failure to pre-pay for collection leads to a situation where the bin concerned cannot be emptied using the loading mechanism (it is, effectively, electronically rejected).

Apart from this shift of commercial waste into the correct channels (the extent of which is not known), the key waste prevention measure which is observed is home composting. This happens most of all where:

- (For obvious reasons) charges are levied on biowaste sacks / bins as well as refuse sacks / bins. This is becoming a widespread phenomenon in Germany, Austria and Belgium; or
- The only biowaste collection is for kitchen waste, and where support is offered for home composting (this occurs in Italy).

Beyond this, it is believed that some reduction in quantities of packaging being used may be achieved. However, there is a clear suggestion that this may depend on the nature of the charging scheme. If recycling is 'zero cost' to the householder (or included in a flat rate element of a charging system) then it may well be the case that collected quantities increase, especially if a collection system, for example, collects glass but not plastics (in which case, purchasing decisions at the margin might shift in favour of the material which is collected free of charge for recycling).

Although the answer to the question posed at the start of the section is not always answered well, equally, it should not be expected that a ready explanation would be forthcoming in all cases for the changes likely to be set in train by a given charging scheme. What is clear is that more is now known about these schemes and about the possible avenues through which 'waste prevention' actually occurs. Some waste managers with responsibility for specific schemes have evidently taken the matter more seriously than others.

## 6.3 Environmental Impacts

Relatively few studies have explicitly sought to understand the environmental effects of charging schemes. Dijkgraaf and Gradus undertook a cost-benefit analysis of DVR charging systems.[116] In the study, they note:

> *From a welfare point of view a number of effects are important with respect to the evaluation of unit-based pricing systems:*
>
> 1. *The change in collection costs due to the effect on the collected quantity.*
> 2. *The change in treatment costs due to the effect on the collected quantity.*
> 3. *The change in administrative costs due to the introduction and maintenance of the unit-based pricing system.*
> 4. *The social costs of extra illegal dumping due to the introduction of unit-based pricing system.*

[116] Dijkgraaf and Gradus (2003) and (2004) (The former paper is somewhat more extensive in its treatment.)

*[...] As we are interested in the welfare effects of the different systems not only the out of pocket costs (private costs) are important, but also the effects on the environment of collection and treatment.*

The system examined was based entirely on bring sites where the collection of dry recyclables (GPT, or glass, paper and textiles) was concerned. A door-to-door collection for refuse and for VFG (vegetable, fruit and garden) waste was in place.

The results of the analysis are shown in Table 12. One assumes that the presentation is based upon costs per household (the study is not absolutely clear on this).

Table 12: Costs Under Flat Rate Fees and Changes in Costs Under Different Charging Schemes (€ per household)

| Costs | Flat rate (level) | Weight (change) | Bag (change) | Vol. and Freq. (change) | Volume (change) |
|---|---|---|---|---|---|
| Total private costs | 45 | -14 | -12 | -6 | +1 |
| Total environmental costs | -12 | -6 | -6 | -3 | -0 |
| Total social costs | 33 | -20 | -18 | -9 | +0 |

*Source: E. Dijkgraaf and R. H. J. M. Gradus (2003) Cost Savings of Unit-based Pricing of Household Waste: The Case of the Netherlands, Research Memorandum 0209, OCFEB, Erasmus University, Rotterdam.*

Some of the observations regarding net costs and benefits are important:

> *It should be noticed that the weight-based and the bag-based system decrease the amount of solid waste with large environmental costs substantially and increase the amount of recyclable waste with high environmental benefits also substantially. From the view of private costs the weight-based system performs better than the other systems. The reason for this is the higher savings on collected waste. Therefore, in social terms the weight-based system performs slightly better than the bag-based system.*[117]

In a subsequent paper, the authors appear to have revised their views somewhat. They conclude:

> *We find that the weight- and bag-based pricing systems perform far better than the frequency- and volume-based pricing systems. The bag-based system seems to be the best option, as its effects are comparable to those of the weight-based system and yet its administrative costs are far lower.*[118]

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[117] E. Dijkgraaf and R. H. J. M. Gradus (2003) Cost Savings of Unit-based Pricing of Household Waste: The Case of the Netherlands, *Research Memorandum 0209*, OCFEB, Erasmus University, Rotterdam.

[118] E. Dijkgraaf and R. H. J. M. Gradus (2004) Cost Savings in Unit-based Pricing of Household Waste: The Case of The Netherlands, *Resource and Energy Economics*, Vol.26 (2004) 353-71.

The principle reason for this appears to be appreciation of the administrative costs of running the schemes. The later study suggests these are €6.86 per inhabitant for a weight-based scheme and €3.18 for a bag-based scheme.[119]

There was no explicit attempt in the assessment of costs and benefits to understand the possible implications of illegal dumping, which the earlier study of Dijgraaf and Graadus suggests may be an important element in the overall analysis of social costs and benefits. Instead, the authors assess the effects of varying the social costs of illegal dumping on the net benefits of the DVR charging system. Effectively, they demonstrate where the social costs would have to lie in order for the DVR charging system to 'break even' in terms of costs and benefits. They conclude:

> *It seems reasonable to assume that the social valuation of illegal dumping is above the level of 180 euro per ton. However, the shadow price of illegal dumping should be raised almost four times for the flat rate to perform better than the weight-based system. Therefore, from a social point of view there seems room for further implementation of weight-based or bag-based pricing systems. If the shadow price of illegal dumping is approximately 750 euro, social costs are equal for both systems. In this case the bag and volume and frequency system still produce social benefits. However, when the shadow price is more than 1072 euro, which can be interpreted as extreme dislike of society for illegal dumping, the flat rate system is preferred above all other systems.*[120]

In their later work, they suggest that although more investigation is merited, and whilst preventative measures and sanctions should be considered, dumping does not appear to be the primary explanation behind waste reduction under charging schemes:

> *In general, the high population density of The Netherlands would suggest a low level of illegal dumping compared with other countries. This is confirmed by the lack of clear anecdotal evidence despite the large number of municipalities with unit-based pricing. However, as the main disadvantage of unit-based pricing systems is the potential effect on illegal dumping, it seems worthwhile investigating an effective monitoring and fining system and the conditions under which such a system would work.....*
>
> *... Given the high population density of The Netherlands and the lack of anecdotal evidence, it seems implausible that a large part of the reduction in unsorted waste is due to illegal dumping.*[121]

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[119] These figures are taken from a report by VROM (1997) *Ervaringen met tariefdifferentiatie en huishoudelijk afval* ("Experience with differentiated tariffs and domestic waste"). Ministry of Environmental Affairs, Den Haag.

[120] E. Dijkgraaf and R. H. J. M. Gradus (2003) Cost Savings of Unit-based Pricing of Household Waste: The Case of the Netherlands, *Research Memorandum 0209*, OCFEB, Erasmus University, Rotterdam.

[121] E. Dijkgraaf and R. H. J. M. Gradus (2004) Cost Savings in Unit-based Pricing of Household Waste: The Case of The Netherlands, *Resource and Energy Economics*, Vol.26 (2004) 353-71.

Eunomia also made an attempt to model the costs and benefits of different unit pricing systems.[122] This was part of larger body of work seeking to understand the merits or otherwise of introducing such systems into the UK, where the legislation currently mitigates against the use of such systems. As such, the modelling of costs and benefits was based upon estimates of the potential impact of schemes operating under UK conditions, reflecting research carried out in other countries. All of the modelling assumed the presence of kerbside collections of dry recyclables and compostable materials.

The results derived from the study are shown below. Table 13 shows that where disposal costs are low, and where 30% of UK households are involved in DVR schemes, for a net cost of just under €21 million, benefits of €163-462 million are generated. This is associated with an increase in the national recycling rate from 32% to 37%, and a reduction in residual waste of 7% from the pre-scheme level. The implied benefit cost ratio is between 8:1 and 22:1. At higher disposal costs, representing the situation the UK will reach in 2012, or before, as a consequence of increase in landfill tax, the benefit cost ratio can become negative (since the net costs fall below zero). In this case, the net costs are negative, whilst generating the same benefits of €462 million are generated. The benefit cost ratio loses much of its meaning because of this net reduction in costs.

Table 13: Summary Costs and Benefits of Implementing DVR Schemes in the UK

| Scenario | Net Financial Costs (€mn) | Net Environmental Benefits (€mn) | |
|---|---|---|---|
| | | Low | High |
| 30% Coverage, Disposal Costs €44/tonne | €20.71 | €163 | €462 |
| 30% Coverage, Disposal Costs €72/tonne | -€30.10 | €163 | €462 |
| 70% Coverage, Disposal Costs €72/tonne | -€63.71 | €374 | €1,048 |

*Source: Eunomia (2003) Waste Collection: To Charge or Not to Charge? A Final Report to IWM (EB).*

The study also modelled situations where 70% of UK households were covered by such schemes. At 70% coverage, the benefits were estimated to be in the range €374-1,048 million. Residual waste requiring disposal falls by almost 6 million tonnes, or 16% of the total quantity of waste.

As with the study by Dijkgraaf and Gradus, therefore, the study indicated significant net benefits associated with unit pricing systems. However, as mentioned above, the study took no account of the potential private and external costs of illegal dumping. Nor was any attempt made to account for additional time spent by householders in

[122] Eunomia (2003) *Waste Collection: To Charge or Not to Charge?* A Final Report to IWM (EB).

sorting activity. The study noted the somewhat exploratory nature of the analysis undertaken:

> *Of course, the modelling as carried out here is somewhat speculative. It makes certain assumptions concerning behavioural change which might not be borne out in practice. Indeed, as we have suggested elsewhere, responses are strongly conditioned by the ability of householders to respond in terms of source separation and opportunities for waste reduction. However, these figures are indicative of what is achieved in systems examined in this report.*

The study also showed that the net private costs of implementing charging systems fall as the costs of treating or disposing of residual waste (i.e., the unit costs for sending waste to a residual waste treatment facility) rise. This is because one of the effects of charging systems tends to be to increase the proportion materials collected as dry recyclables and compostables. Consequently, if there is no source reduction, the costs of the waste management system will increase if the cost of collecting materials for recycling and composting is greater than the costs of collecting material and subsequently disposing of it. Where disposal costs are higher, there are more likely to be net savings to the municipality from collecting less of the waste as residual waste and more as recyclables / compostables.

Indeed, assuming a given response associated with a charging system of a given private cost, one could postulate the existence of a 'threshold' at which the net private costs turn negative. This is important since it suggests that the internalisation of negative externalities of disposal, or regulatory activity which makes disposal technologies more expensive, will tend to make a local authority more positively disposed to implementing kerbside collection schemes, and to implementing variable charging systems. As the study noted:

> *Where DVR schemes are concerned, higher disposal costs are not so much necessary as desirable, since they accentuate the benefits of the avoided disposal costs occasioned by the increases in source separation and the source reduction driven by the charging scheme. At a £35 landfill tax, the DVR schemes begin to look much more cost-effective. The higher avoided disposal costs make the logic of such systems even more compelling.*

Finally, the study played strongly on the desirability of introducing DVR schemes against the backdrop of quality and convenient kerbside collection services. These are generally thought to be, if not necessary, then important in limiting the degree to which illegal evasion of the charges occurs through dumping etc.

Similar work, reported below, was undertaken in a report for the OECD.[123]

### 6.3.1 Torrelles de Llobregat

The most significant outcomes were a reduction in the quantity of collected residual waste by 38%, and an increase in separately collected materials from 33% to 89%

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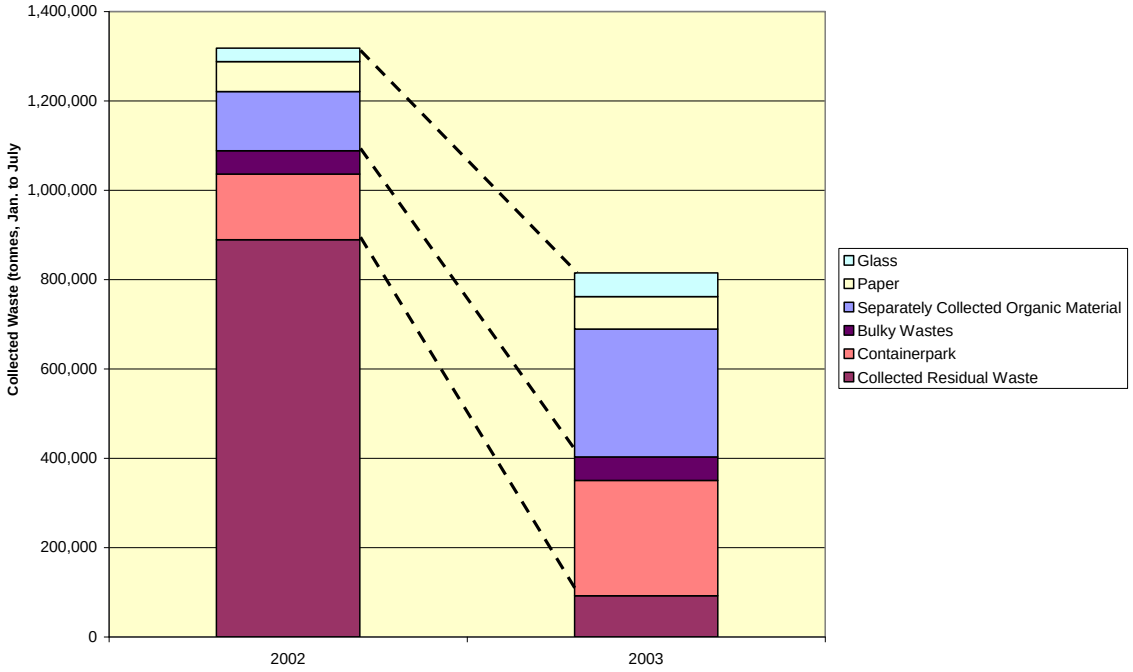
[123] Hogg, D. (2006) Impact of Unit-based Waste Collection Charges ENV/EPOC/WGWPR(2005)10/FINAL, Paris: OECD

including the bulky and containerpark fractions, or from 17% to 51% including only organic materials, paper and glass. Figure 23 illustrates the change in the collected fractions graphically.

Figure 23: Changes in Collected Quantities and Quantities Recycled

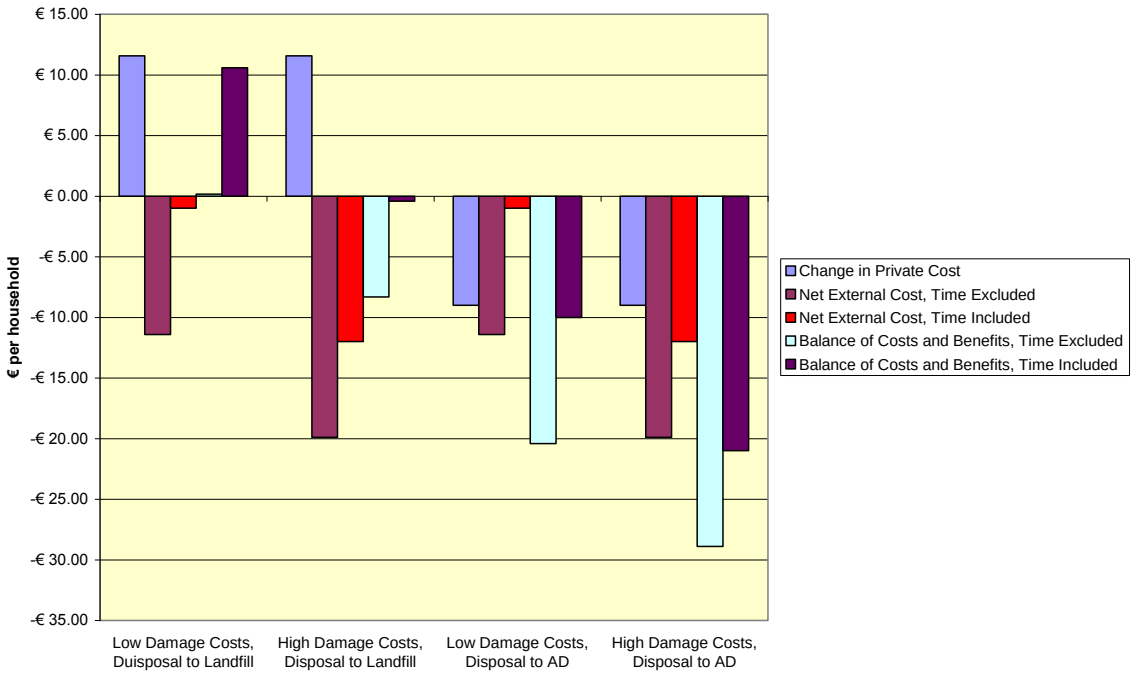



The case suggests the following:

- Net private costs of €11.58 per hh if the avoided disposal is landfill or -€9 per hhld if the avoided disposal is a non-landfill treatment; and
- Net external benefits of between €11- €20 depending upon whether high or low unit damage costs are used. This excludes external costs associated with illegal disposal. These fall (based on a range of assumptions) by around €8-10 per household once one accounts for time spent by households in making greater use of the containerpark.

The possible permutations for the balance of costs and benefits are shown in Figure 24. This shows that the balance of costs and benefits is positive (i.e. there is a net cost) only in the case where damage costs are low, and the avoided disposal route is lower cost landfill.

Figure 24: Balance of Costs and Benefits of DVR Scheme, Torrelles de Llobregat




Even in the cases where the costs of time are considered, the balance of costs and benefits is negative (there are net benefits). In the best scenario, the benefits exceed €20 per household.

It should be noted that these costs and benefits are not attributable solely to the charging scheme. They are attributable to a combination of a change in service and the introduction of the charging scheme. We have not tried to separate these out in this particular case, but perhaps the important thing is the fact that the combined scheme potentially delivers significant net social benefits, notably where the avoided form of disposal is not landfill.

These costs and benefits do not account for:

- The potential disbenefits associated with illegal disposal (other than those concerned with its disposal); and
- Changes in associated transport externalities. These could be assumed to be broadly internalised in the private costs of transport.

On balance, therefore, whilst negative social costs are possible under the scheme, the scheme appears more likely to offer net benefits.

The assumptions made concerning the value of time spent in engaging in additional recycling activity are potentially important, and where the private costs of residual waste treatment are low, these could even be decisive in an analysis of costs and benefits.

Important issues which the scheme appears to have raised relate to movement of waste into other routes. This may be a particular problem in small jurisdictions where a charging system is applied with no other schemes operated nearby. Effectively, the perimeter of such an area relative to the total area is large. All households are likely

to be close to the perimeter, and they may find it easy to move waste across administrative boundaries.

The increase in visitors to the containerpark is also of interest. This can probably be explained best by reference to the fact that the scheme effectively charged on the basis of volume, and some of the low-density / high-volume materials such as plastic bottles and cans could not be recycled through the door-to-door collection service. Different schemes – notably, those with greater materials coverage – are likely to give rise to smaller numbers of additional movements. It is worth stating also that, to the extent that movement of materials onto the containerpark is made more likely through charging schemes, any additional costs attributable to these movements are likely to be significantly lower where the density of sites is high (so that journey distances / times are kept down), and where the sites are located in places which are close to areas which are frequently visited by citizens (so that the number of 'dedicated' journeys is minimised).

### 6.3.2 Schweinfurt

This scheme (described in Section 6.1.2) gave rise to a reduction in costs of the order €6 per hhld. This includes the costs of monitoring and enforcement of fly-tipping, of which there has been some increase. The costs to the municipality do not include the costs of collecting packaging materials since these are borne by the DSD system. However, in this case, the packaging collections have not increased significantly other than at bring sites which are the least expensive service for contractors to run. Consequently, the costs of provision of this service probably changed relatively little as a consequence of the scheme.

The benefits are potentially considerable, and probably no less than €8 per tonne. The net social benefits, therefore, appear to be no less than €14 per household before accounting for illegal activity.

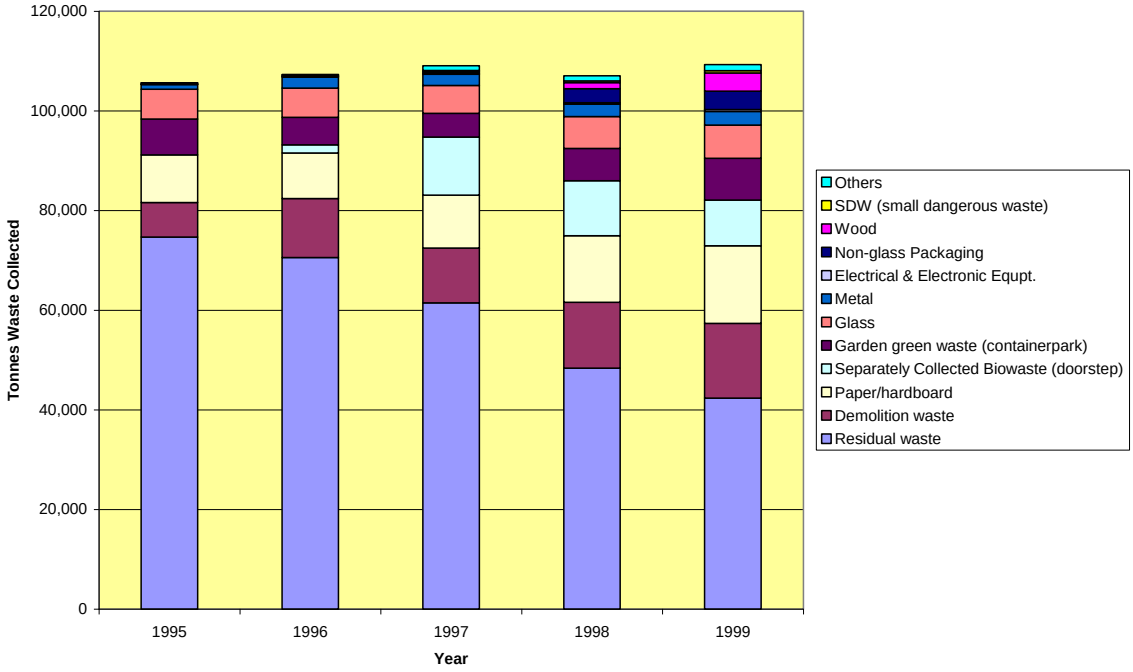
No estimate of additional time for sorting waste for recycling has been included. In this particular case, the principal increase in the quantity of material being recycled relates to the paper fraction (see above). This does not require additional washing and the dense network of bring sites makes it far less likely that households make significant additional journeys to take materials for recycling.

In this particular case, as described in Section 6.1.2.7, the municipality has made considerable effort to understand the exact nature of the waste reduction, including the extent of illegal dumping. Our view is that illegal dumping is unlikely to be the source of the reduction and that other factors – efforts in waste reduction and re-use, changes in consumption patterns, and, possibly, a move of commercial waste away from the municipal stream – are likely to have been important.

### 6.3.3 Gent and Destlebergen

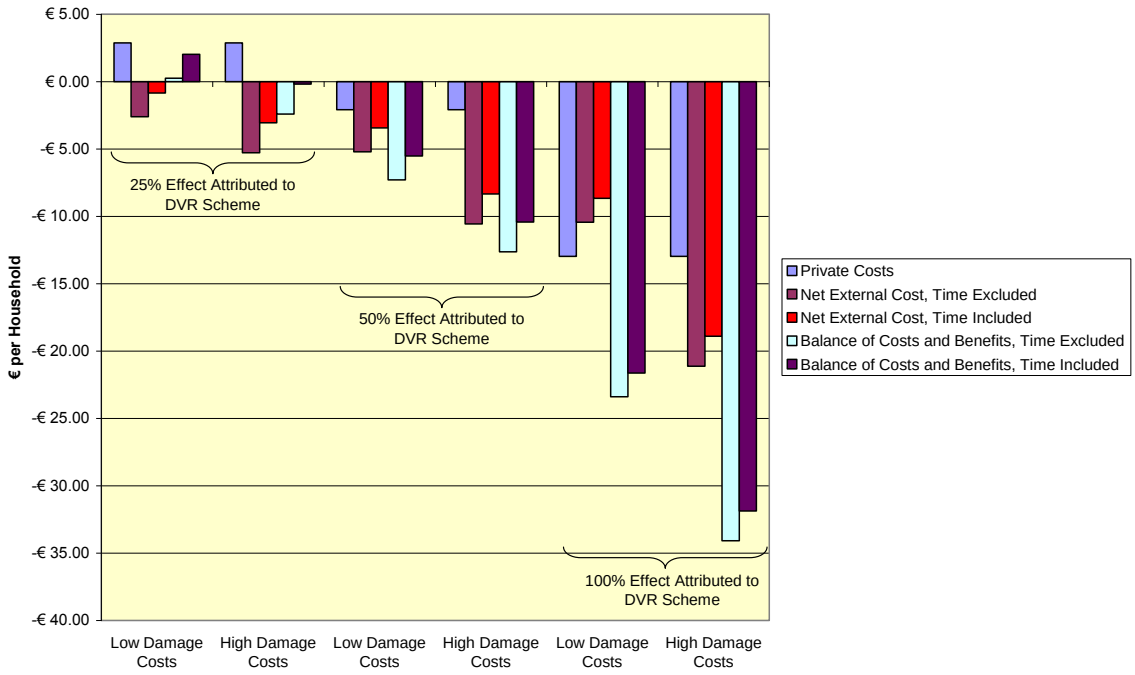
If one compares the period before 1998 and the period after, one sees a continuation of the improvements made in separate collection (see Figure 25), a continuous increase in waste delivered to bring sites, and – in the year 1998 – a slight constraint on overall growth of waste arisings (though this is barely discernible, and could not be assumed to be statistically significant) (again, see Figure 25).

Figure 25: Evolution of System Performance in IVAGO Area




The potential permutations of costs and benefits are shown in Figure 26 below. Because the scheme was introduced at the same time in a period where other changes were being made, these are shown for cases where 25%, 50% and 100% of the effects of the DVR scheme and the enhanced service are attributed to the DVR scheme. The net costs clearly depend upon the attribution. Attributing all the change to the charging scheme seems unrealistic and would lead to benefits (cost reductions) far greater than in the other schemes examined even though the effect on waste prevention is barely existent in this case. Even where 25% of the effects of the change from 1997-1999 are attributed to the scheme, however, net benefits (cost reductions) lie between €3 and €5 per household.

Figure 26: Balance of Costs and Benefits of DVR Scheme, Gent




As with the case of Torelles de Llobregat, there are few permutations where the system imposes net social costs. These are where a) a relatively small effect is attributed to the DVR scheme, and b) where the damage costs used are low (so benefits of avoided disposal and recycling are smaller). Again as in the Torelles de Llobregat, the situation appears slightly worse where the costs of time are taken into account. However, in the case of Gent, including these has a smaller effect, and even in the worst case, the net social costs are of the order €2.00 per household.

If, on the other hand, the DVR scheme is attributed with a more significant proportion of the change occurring between 1997 and 1999, then benefits may be as high as around €10 per household (50% effect attributable) rising to €21-34 if all the change occurring in the period is attributed to the DVR scheme.

On balance, therefore, and based upon the effects of DVR schemes in similar situations elsewhere, it seems likely that the DVR charging scheme will have contributed to the generation of net social benefits. Indeed, perhaps the more important observation is that, *as part of a package*, the DVR scheme contributed (however significantly) to the generation of net social benefits of the order €20-30 per household.

## 6.4 Key Observations Regarding Effectiveness

The above review of charging systems presents some valuable lessons regarding the implementation of DVR schemes, and the situations in which they are likely to work best. Charging tends to work best where:

- The marginal benefit of avoided residual waste treatment / disposal is high. Charging systems will be more likely to ensure financial savings where the

costs of landfilling / incineration are high, by which we mean, of the order €80 at least;

- Separate collection (of biowastes and recyclable materials) includes a wide range of materials, and is convenient (typically kerbside collected rather than through 'bring' systems) – this tends to limit the likelihood of illegal disposal / contamination of separately collected waste streams;
- Charge levels are set with a flat rate fixed fee supplemented by variable fees so as a) to ensure problems of revenue instability do not arise and b) to ensure variable rates are not so high they give rise to more compelling incentives to fly-tip;
- Charges are placed on residual waste taken to civic amenity sites as well as at the kerbside (so that waste does not simply move from one management route to another);
- Municipalities should be vigilant in the days shortly after the scheme's introduction so as to make sure that fly-tipping and illegal disposal are clearly shown to be unacceptable; and
- Charges are levied – albeit at different rates - on all waste streams, including recycling – this fully integrated approach is likely to deliver the strongest incentive for waste prevention.

Political leadership – nationally (regarding national policy) and locally (in respect of local implementation) - is important. In some countries, national or regional policy sets a clear and structured agenda for local waste charging. The case of Landkreis Schweinfurt was one where the local political leader, recognising the potential for opposition to the proposed changes, made sure the system was first trialled in his own neighbourhood. At the other end of the spectrum, despite significant investigation into the impact of charging for waste in the UK showing the benefits it would bring, this policy remains a political hot potato, with the coalition Government stripping away legislation which allowed local authorities to charge for waste.

Minimum standards for a quality, convenient collection service including a wide range of recyclables is desirable in order for charging systems to deliver the best outcomes. Eunomia described a charging scheme in Fingal, Ireland, where half of households had no kerbside recycling scheme available to them. The scheme led to many protests concerning the system. Another key point highlighted by Irish experience is that it is more difficult to operate DVR systems without problems where the waste collection system is a completely open market. The much more favourable circumstance – and the more common one in Europe – is to have all households 'linked to' the collection system, and with some of the costs of the service supported through (obligatory) local taxation.

# 7.0 Product Taxes / Fees / Charges

In Eunomia's accompanying inventory of economic instruments this sub-category makes up a significant proportion of the total examples which are listed. However, as stated above, we will focus here only on taxes/fees/charges relating to single-use items, such as plastic carrier bags and disposable cutlery. These types of taxes and charges aim to change consumer behaviour by acting to incentivise a switch to more sustainable alternatives. In addition, they may act as a means of internalising the environmental/social externalities of a particular product.

Due to differences in data reporting and analysis, it is not possible to compare the impacts of the different instruments applied in the various countries across the world. However, the evidence for waste prevention in this sub-category, especially with regard to taxes on plastic bags, is fairly well established and will be briefly discussed here.

The impact of Ireland's tax on plastic carrier bags has been assessed in some detail, and as such it has been identified as a good case study to demonstrate how an environmental tax can be used to assist in waste prevention. This case study will also function to highlight some of the key issues and problems associated with product taxes and suggestions that have been made to address some of the difficulties.

## 7.1 Case Study: Irish Plastic Bag Levy

### 7.1.1 Background

The Irish plastic bag levy was introduced in March 2002 under the Waste Management (Environmental Levy) (Plastic Bag) Regulations 2001. Initially, the tax was set at €0.15 per plastic bag, with exemptions for smaller plastic bags that met specific conditions and used to store non-packaged goods such as dairy products, fruit and vegetables, nuts, confectionary, hot or cold cooked food and ice –these are known as levy-free bags (reusable plastic bags are also exempt as long as the charge for the bag exceeds €0.70).[124] The tax is passed directly to consumers at the point of sale, and has thus been reported to provide a clearer, more consistent message than

[124] According to the Department of the Environment, Community and Local Government 'Bags not exceeding 225mm in width (exclusive of any gussets), by 345mm in depth (inclusive of any gussets), by 450mm in length, (inclusive of any handles) have been marketed as "Levy Free Bags". The regulations, however, do not provide for "Levy Free Bags". The Plastic Bag Levy applies on all plastic bags, even if marketed as "Levy Free Bags", when used in circumstances not exempted by the regulations'. See: Department of the Environment, Community and Local Government (2007) Plastic Bags, Date Accessed: 19 September 2011, www.environ.ie/en/Environment/Waste/PlasticBags/.

systems where retailers are responsible for the levy (such as in Denmark and South Africa).[125,126]

It has been reported that this policy has been very effective and has *'proved so popular with the Irish public that it would be politically damaging to remove it'*.[127] The tax was implemented to *'change consumers' behaviour to reduce the presence of plastic bags in the rural landscape, and to increase public awareness of littering'.* Revenues from the tax are paid into an Environmental Fund which is administered by the Department of Environment, Heritage and Local Government. The fund is used to cover administrative costs (3% of total revenues) and support a wide range of environmental programmes. The costs of implementation are reported to be very low because bookkeeping and reporting has been integrated with VAT returns. [128]

The levy is not a Pigouvian tax, in that the rate of the tax was not devised with the intention of internalising the marginal external costs. Instead, the Irish Government's intention was to set a rate of tax which would act to change consumer behaviour. As such, the initial rate of tax was set at six times consumers' average maximum willingness to pay for the purchase of plastic bags.[129] This ensured that there was a marked decrease in the use of plastic bags in the short term, a trend which has been reversed slightly over the years. The per capita usage of plastic bags decreased from an estimated 328 to 21 plastic bags per capita per annum after the introduction of the tax. However, the results of the 2006 census indicated that plastic bag usage had risen to 32 bags per capita over the course of 2006. Consequently the levy was increased to €0.22 on 1st July 2007 under Plastic Bag (Amendment) (No. 2) Regulations of 2007.[130]

An evaluation of the impact of the levy on householders and retail sector was undertaken by Convery et al.[131] The authors interviewed seven leaders in the retail

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[125] Dikgang, J. Leiman, A. and Visser, M. (2010) *Analysis of the Plastic-Bag Levy in South Africa*, Policy Paper No. 18, Environmental Policy Research Unit, School of Economics, University of Cape Town, July 2010, www.econrsa.org/papers/p_papers/pp18.pdf

[126] Plastic Bag: Friend or Foe? (no date given) *Market Based Examples*, Date Accessed: 20 September 2011, www.plasticbageconomics.com/index.php?option=com_content&task=view&id=26&Itemid=40

[127] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

[128] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

[129] Ibid.

[130] Department of the Environment, Community and Local Government (2007) *Plastic Bags Levy to be Increased to 22c from 1 July 2007*, Press Release: 21/02/2007, Date Accessed: 19 September 2011, www.environ.ie/en/Environment/Waste/PlasticBags/News/MainBody,3199,en.htm

[131] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

sector and conducted random telephone interviews with consumers, the results were as follows:

- Retailers found the effects of the tax on their well-being neutral or positive, closely related to the fact that the additional costs of implementation were generally less than the savings resulting from not having to purchase plastic bags. Implementation costs were low because book-keeping was integrated with VAT returns; and
- Overall, consumers were very much in favour of the levy. While the levy had caused them some expense, through either paying the levy or buying long-life bags, virtually all respondents responded that they felt the impact on the environment was positive, producing a noticeable reduction in plastic bags 'in the environment'.

### 7.1.2 Waste Prevention Impacts

A 2008 regulatory impact assessment of Ireland's plastic bag levy reported that:

> *'...whilst the preliminary data show the recent levy increase to 22 cent has reduced per capita usage the current actual level of approximately 30 bags per person remains considerably higher than the 2002 post-levy levels of approximately 22 bags per person. Thus the impact of the initial levy has not been sustained in terms of the reduction in per capita usage of plastic bags. This sustained increase in demand since 2002 is believed to be attributable in part to the decline in the real value of the initial 15 cent levy.'*.[132]

Prior to the implementation of the levy some 1.3 billion plastic bags were given away free of charge every year, this fell by over 90% in the first five months after the introduction of the tax in the spring of 2002.[133] However, as described above, the consumption of plastic bags soon began to rise and by 2007 had reached close to 150 million units (Figure 27).[134] From Figure 27 it is also evident that the revenue generated from the tax increased proportionately. There are no subsequent studies which have looked at how the 2007 increase in the levy affected demand; however, it is expected that there would have been a decrease, which would have been further fuelled by the steep downturn in the Irish economy in recent years.

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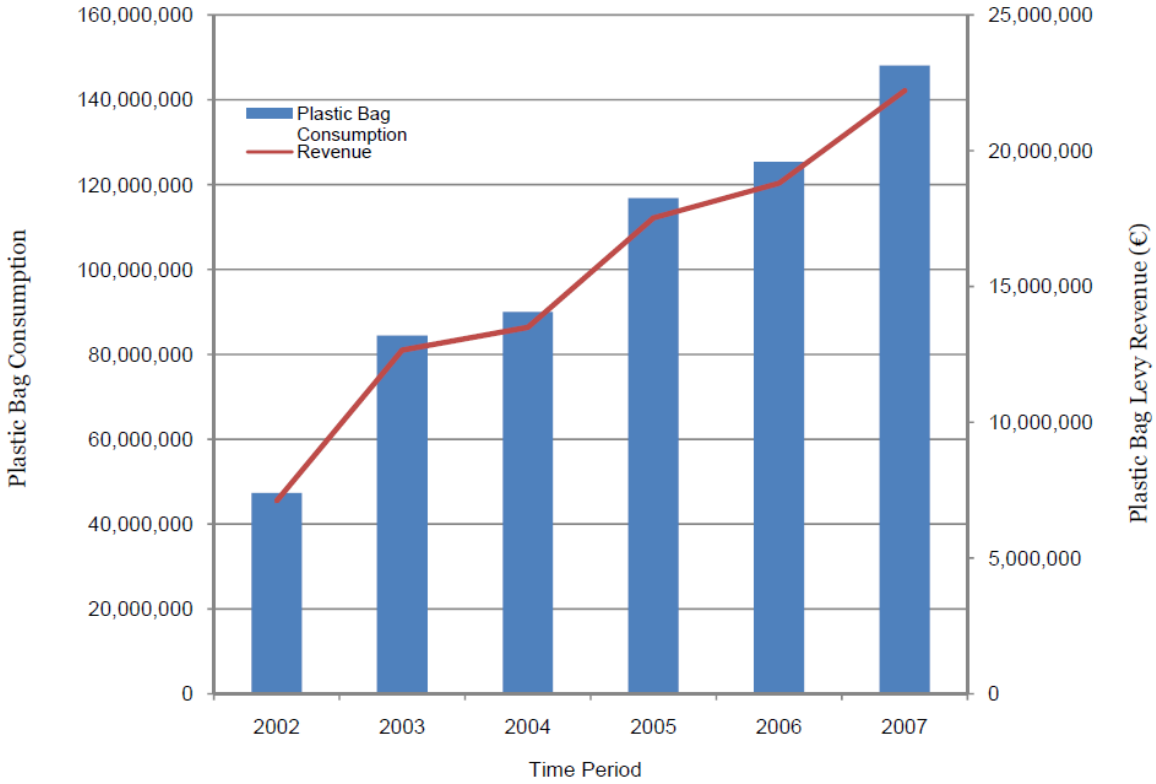
[132] AP EnvEcon Limited (2008) *Regulatory Impact Analysis on Proposed Legislation to Increase Levies on Plastic Shopping Bags and Certain Waste Facilities*, Report for the Department of Environment, Heritage and Local Government, November 2008, www.environ.ie/en/Legislation/Environment/Waste/WasteManagement/FileDownLoad,21599,en.pdf

[133] McDonnell, S. (2003) An Enquiry into the Use of Taxation on Environmentally Harmful Products: A Case Study of the Irish Plastic Bag Levy, UCD, Unpublished Thesis.

[134] AP EnvEcon Limited (2008) *Regulatory Impact Analysis on Proposed Legislation to Increase Levies on Plastic Shopping Bags and Certain Waste Facilities*, Report for the Department of Environment, Heritage and Local Government, November 2008, www.environ.ie/en/Legislation/Environment/Waste/WasteManagement/FileDownLoad,21599,en.pdf

Figure 27: Unadjusted Plastic Bag Consumption and Associated Revenue




*Source: Department of Environment, Heritage and Local Government, 2008*

In addition to reductions in the demand for plastic bags the Environment Fund has also been used to sponsor waste prevention initiatives. In 2007, of the €57,725,535 generated in income from the Environmental Fund (€22,577,535 of which was generated from the plastic bag levy) the following was spent on waste prevention initiatives:

- Waste prevention and the National Market Development Group, €1,518,213
- Producer responsibility initiatives, €905,793
- Environmental awareness programmes, €1,615,332

The above figures illustrate that a large number of environmental awareness campaigns and waste prevention programmes have been put in place in local authorities, due to the monies available from the environmental fund. Therefore, this levy has also indirectly influenced waste prevention initiatives in Ireland.

### 7.1.3 Environmental Impacts

The levy applies only to single-use plastic bags and as a result it has been suggested that since the introduction of the levy paper shopping bags are more prevalent (though it was not possible to find data on the consumption of paper bags before or after the introduction of the tax, although it is expected that usage has increased). Surveys have indicated, however, that up to 90% of shoppers used long-life bags in 2003, compared with 36% in 1999, which suggests that the switch to paper bags has

been a far from universal switch, and that there has been a discernible switch to long-life bags.[135]

Fehily Timoney *et al* carried out an ex ante study on the impact of the tax on the plastic bag industry. [136] This study included a life cycle assessment (LCA) of plastic and paper bags using a weighting system, based on how far away each impact was from a sustainable target level. Using this approach, plastic bags were shown to have a lower total impact score of 7.9 compared to paper bags with a score of 8.9.[137] The higher impact of paper bags has been confirmed in a more recent LCA published by the Environment Agency in England.[138] In this study it was found that a paper bag would have to be used three times before its global warming potential would match that of a HDPE plastic bag being used only once (Table 14). The researchers found that HDPE plastic bags were frequently reused, either as bin liners or for subsequent shopping excursions, and in such instances paper and cotton reusable bags would have to be used a significant number of times before their higher global warming potential had been offset. For example, if plastic bags were used as bin liners 40.3% of the time (a survey found this this to be the average usage rate in England) a paper bag would have to be used four times to match the global warming impact.

Table 14: The Amount of Primary Use Required to Take Reusable Bags Below the Global Warming Potential of HDPE Bags with and Without Secondary Use

| Type of carrier | HDPE bag (no secondary use) | HDPE bag (40.3% reused as bin liners) | HDPE bag (100% reused as bin liners) | HDPE bag (used three times) |
|---|---|---|---|---|
| Paper bag | 3 | 4 | 7 | 9 |
| LDPE bag | 4 | 5 | 9 | 12 |
| Non-woven PP bag | 11 | 14 | 26 | 33 |
| Cotton bag | 131 | 173 | 327 | 393 |
| *Source: Environment Agency, 2011* | | | | |

[135] Department of the Environment, Community and Local Government (2007) *Plastic Bags Levy to be Increased to 22c from 1 July 2007*, Press Release: 21/02/2007, Date Accessed: 19 September 2011, www.environ.ie/en/Environment/Waste/PlasticBags/News/MainBody,3199,en.htm

[136] Fehily, Timoney & Company (1999) *Consultancy Study on Plastic Bags*, Report prepared for the Department of Environment and Local Government, Dublin.

[137] Ibid.

[138] Environment Agency (2011) Life-Cycle Assessment of Supermarket Carrier Bags, http://www.environment-agency.gov.uk/research/library/publications/129364.aspx

One way of viewing this is that LCAs play an important role in highlighting some of the potentially contradictory factors of such taxes and the importance of incorporating data on the overall environmental impact of the various available options. In this case it might be perceived that a narrow focus on litter or waste prevention may in fact exacerbate the environmental impact of a particular activity. Indeed an environmental group in Ireland has called for all single use bags, most notably paper bags, to be included in the tax system.[139]

However, criticism could also be levelled at the LCA approach, which arguably places too much emphasis on greenhouse gas impacts to the exclusion of other, less well understood impacts. One of the key issues as far as plastic bags are concerned is the downstream impact of plastic bags as land-based and marine litter. In the terrestrial environment, plastic bags are one of the more visible forms of litter, and presumably, for this reason, contribute much to litter-related disamenity. Indeed, the early discussion around a levy in South Africa was given impetus by the environment minister's encounters with plastic bag litter in otherwise pristine environments. Plastics dominate marine litter and represent a significant threat to the marine environment due to their abundance, longevity in the marine environment and their ability to travel vast distances.[140] Despite representing only 10% of all waste produced, plastics account for between 50-80% of marine litter and this is not expected to decline for the foreseeable future (particularly as plastics do not degrade quickly).[141] Of all plastics, it is, arguably, single use plastic bags that have the greatest impact. Data taken from the International Bottom Trawl Survey and the Clean Seas Environmental Monitoring Programme indicate that plastic bags make up 40% of all marine litter in the waters of the North East Atlantic. The French research institute IFREMER has also found that in the Bay of Biscay most of the waste items found on the seabed were plastic (92%) and of those 94% were plastic bags. [142]

It is thus essential that as far as possible, an holistic view be taken when setting up taxes on products, one in which *all* the environmental impacts of the various options are quantified and accounted for, not just those associated with emissions under

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[139] Friends of the Irish Environment (2010) *Call for Ireland to Extend Levy to all Single-use Bags*, Date Published: December 2010, Date Accessed: 19 September 2011, www.friendsoftheirishenvironment.net/index.php?do=friendswork&action=view&id=878

[140] KIMO (2010) Economic Impacts of Marine Litter, Kommunernes Internationale Miljøorganisation Local Authorities International Environmental Organisation, September 2010, available at http://www.kimointernational.org/Portals/0/Files/Marine%20Litter/Economic%20Impacts%20of%20Marine%20Litter%20Low%20Res.pdf

[141] Th ompson, R.C., Swan, S.H., Moore, C.J. and vom Saal, F.S. (2009a) Our Plastic Age. Philosophical Transactions of the Royal Society B: Biological Sciences 364(1526): 1969-2166; Barnes, D.K.A., Galgani, F., Thompson, R.C. and Barlaz, M. (2009) Accumulation and fragmentation of plastic debris in global environments. Philosophical Transactions of the Royal Society B: Biological Sciences 364(1526): 1985-1998; Thompson, R.C., Moore, C.J., vom Saal, F.S., and Swan, S.H. (2009b) Plastics, the environment and human health: current consensus and future trends. Philosophical Transactions of the Royal Society B: Biological Sciences 364(1526): 2153-2166.

[142] Seas at Risk (2011) Commission Consults on Binning Plastic Bags, available at http://www.seas-at-risk.org/news_n2.php?page=408

assumptions that the materials are all well-managed. The Carbon Based Packaging Tax introduced in the Netherlands in 2008 has been one of the first systems which has attempted to base the levy on the relative impact of different packaging materials. However, the tax, which considers the life cycle impact of packaging materials based on greenhouse gas emissions and is applied using a relevant metric for each material, does not, as per the discussion above, necessarily address all impacts.[143,144] The Danish packaging tax considered a wider range of poillutants, but not litter, but the Danish system also includes a deposit refund system for beverage packaging which tends to reduce littering of these items.

### 7.1.4 Litter Impacts

The main objective of Ireland's plastic bag tax has been to reduce quantities of litter. In this regard the tax has had a marked effect and again Convery et al report that:

> *'A combined project by Irish Business Against Litter and An Taisce (National Trust of Ireland) produced a number of litter surveys. These have found that between January 2002 and April 2003 the number of "clear" areas (i.e. areas in which there is no evidence of plastic bag litter) has increased by 21%, while the number of areas without "traces" has increased by 56%.*[145] *These numbers are remarkably high given the long lasting nature of plastic bags in the environment. A different source, the National Litter Pollution Monitoring System notes that plastic bag litter accounted for 5% of national litter composition before the introduction of the levy. In 2002, this number fell to 0.32%, in 2003 to 0.25% and to 0.22% in 2004'.*[146]

This rate has remained more or less constant since this time, as is shown in Figure 28 below.[147] It is worth noting that the Department of Environment, Heritage and Local Government estimated the figure of 5% in their first Annual National Litter Pollution Monitoring Systems Annual Report (May 2003). Consequently, one cannot be certain that the decline in litter quantities has been as dramatic as the figure

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[143] CE Delf (2007*) Environmental Indices for the Dutch Packaging Tax*, November 2007, www.cedelft.eu/publicatie/environmental_indices_for_the_dutch_packaging_tax/724?PHPSESSID=f138219238c72e8038a0a5694354af1d

[144] CE Delf (2010) *The Environmental Impact of the Dutch Packaging Tax*, August 2010, www.cedelft.eu/publicatie/the_environmental_impact_of_the_dutch_packaging_tax/1102?PHPSESSID=0e0760e789da090aec15fb6e48a0d3c9

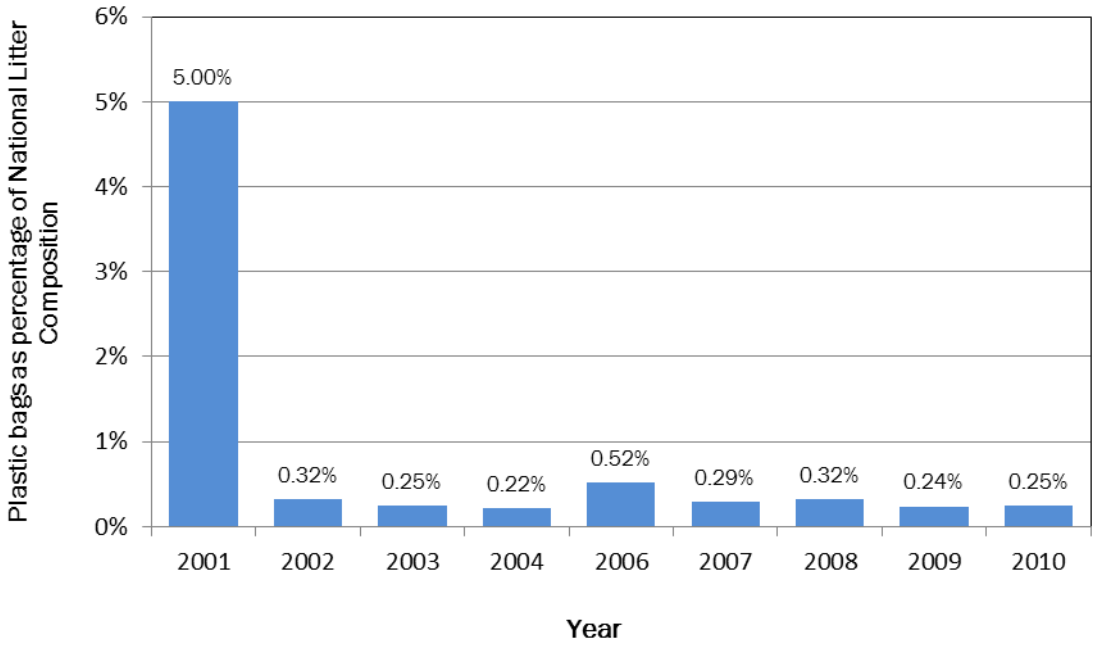
[145] "Traces" of litter is defined as up to five items over a linear distance of 1 m.

[146] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

[147] Litter Monitoring Body and TOBIN Consulting Engineers (2011) *The National Litter Pollution Monitoring System – Litter Monitoring Body: System Results 2010*, Report for the Department of the Environment, Heritage and Local Government, April 2011, www.litter.ie/system_survey_results/index.shtml

would appear to suggest. However, the public commonly believes that the amount of plastic bag litter has decreased substantially since the introduction of the tax.[148]

Figure 28: Plastic Bags as a Percentage of Ireland's National Litter Composition

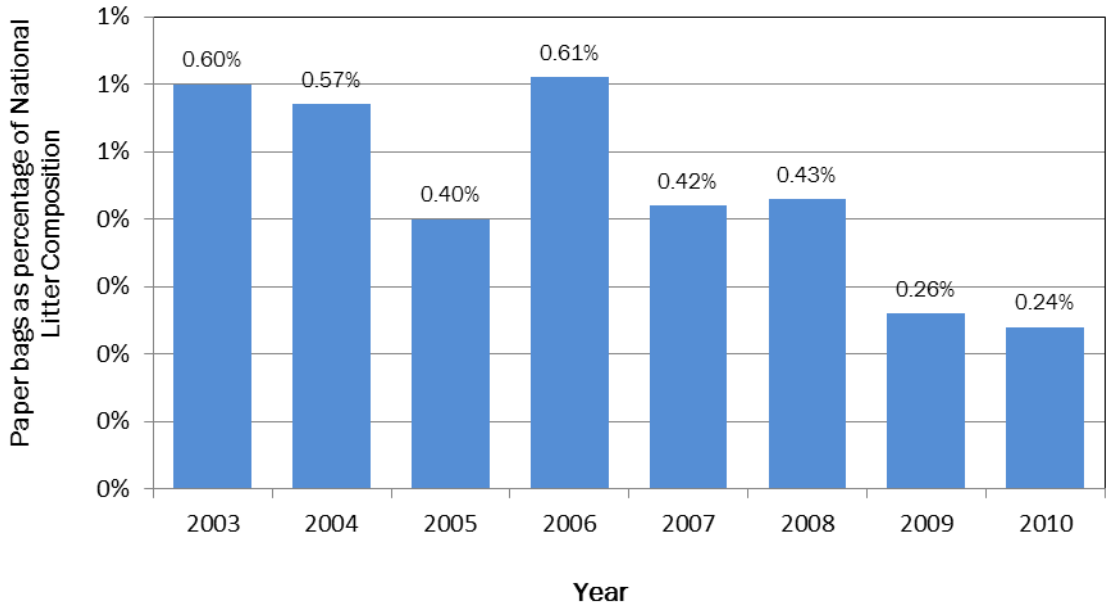



*Source: Litter Monitoring Body, Annual System Results, 2011*

Despite possible increases in the use of single-use paper bags it appears as if their presence as litter has decreased over recent years (Figure 29). This may further corroborate the results of the survey mentioned above, which suggests that the tax has largely caused people to shift towards the use of reusable bags, rather than paper bag substitutes.

[148] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

Figure 29: Paper bags as a Percentage of the National Litter Composition*




*Source: Litter Monitoring Body, Annual System Results *Note: The figures provided for 2007 to 2010 appear under the sub-category 'bags' and it is unclear if this refers exclusively to paper bags – in earlier years this category appears to have been reported as 'paper bags'*

## 7.2 Evidence of Impacts

Taxes on products, even addictive products such as cigarettes and alcohol, have been widely shown to curb consumption.[149,150] It is thus unsurprising that the case study above provides a clear example that the taxation of single-use plastic bags leads to reduced consumption. Further proof comes from Belgium, where under the "pic-nic tax" wholesalers are liable to pay a tax on various single-use items.[151] It has been reported that the tax on disposable plastic bags – set at €3.00 per kg – has had a marked impact in terms of reducing their use over recent years (decrease of 80% between 2003 and 2007; during the same period, the sale of reusable bags rose from 4.5million units in 2004 to 25.4million in 2007). However, it is also reported that despite the fact that *'the retail prices of disposable kitchen utensils, food wrap*

[149] Hu, T. W. and Mao, Z. (2001) Effects of Cigarette tax on Cigarette Consumption and the Chines Economy, *Tabaco Control*, Vol. 11, pp.105-105

[150] Chaloupka, F. J., Grossman, M. and Saffer, H. (2002) The Effects of Price on Alcohol Consumption and Alcohol-Related Problems, National Institute on Alcohol Abuse and Alcoholism, August 2002, http://pubs.niaaa.nih.gov/publications/arh26-1/22-34.htm

[151] These include the following taxes: €3.00 per kg of non-biodegradable disposable plastic carrier bags; €2.70 per kg for plastic food wrapping (product price increase of approximately 70%); €4.50 per kg for aluminium foil (product price increase of approximately 100%); €3.60 per kg of disposable kitchen utensils.

*and aluminium foil have gone up substantially, the impact on consumption has been less marked'*.[152]

As a means of comparison Table 15 summarises the impacts of the plastic bag levies introduced in Belgium, Italy, Ireland and South Africa. From this Table it is evident that levy's on plastic bags have had a marked, if not always long-lasting, effect on demand. It might be supposed that households may have 'a stock' of plastic bags wich they use for various purposes (bin liners etc.). It may be that consumption increases as this stock is drawn down.

Table 15: Examples of Taxes on Plastic Carrier Bags and Their Impact on Consumption

| Rate of Tax | Consumption trends | Impacts on litter |
|---|---|---|
| Belgium, April 2007[1,2] | | |
| €3.00 per kg of plastic bags (1 to 10 cents per bag, depending on weight) | Reduction in sales of 80% between 2003 and 2009 | n/a |
| Ireland, March 2002[3] | | |
| Initially €0.15, but raised to €0.22 per plastic bag in July 2007 | Consumption decreased from 328 bags per capita prior to the levy, to 21 the year after (this increased to 30 units per capita prior to the price increase in 2007) | Plastic bag litter reduced from 5% (estimated figure) in 2001 to 0.25% in 2010 |
| Italy, 2002[4] | | |
| Initially €0.13, but raised to €0.20 per plastic bag in 2007 | Use of plastic bags decreased from 1.3 billion prior to the tax to 20 million units the year after (consumption then began to increase to 140 million units per annum) | n/a |
| South Africa, May 2003[5] | | |
| Initially ZAR 0.46 (€0.04) for standard 24L bags, but subsequently decreased as retailers have absorbed the costs (retailers are liable for the tax) | For high-income earners consumption of plastic bags per ZAR 1,000 worth of shopping (€92 on 22 September 2011) has decreased by approximately 57% and for low-income earners the reduction has been approximately 50%. There was an initial sharp drop in demand, but | According to the cited paper, no pre or post levy data exists on litter levels in South Africa |

[152]Bruxelles Environment (2010) *Mapping Report on Waste Prevention Practices in Territories within EU27 - Pre-Waste: Improve the Effectiveness of Waste Prevention Policies in EU Territories*, October 2010, http://www.bruxellesenvironnement.be/uploadedFiles/Contenu_du_site/Professionnels/Formations_et_s%C3%A9minaires/Conf%C3%A9rence_Pre-waste_2011_(actes)/p3-%20prewaste-mapping-report.pdf

| Rate of Tax | Consumption trends | Impacts on litter |
|---|---|---|
| | this was soon reversed | |

*Notes:*

*1. Pre-Waste (2011) Good Practice in Waste Prevention, International Pre-Waste Workshop, March 2011, http://www.bruxellesenvironnement.be/uploadedFiles/Contenu_du_site/Professionnels/Formations_et_s%C3%A9minaires/Conf%C3%A9rence_Pre-waste_2011_%28actes%29/p2-posters-good-practices.pdf*

*2. Bruxelles Environment (2010) Mapping Report on Waste Prevention Practices in Territories within EU27 - Pre-Waste: Improve the Effectiveness of Waste Prevention Policies in EU Territories, October 2010, http://www.bruxellesenvironnement.be/uploadedFiles/Contenu_du_site/Professionnels/Formations_et_s%C3%A9minaires/Conf%C3%A9rence_Pre-waste_2011_(actes)/p3-%20prewaste-mapping-report.pdf*

*3. The full impacts of this levy are covered in the case study described in the preceding section*

*4. Friends of the Irish Environment (2010) Call for Ireland to Extend Levy to all Single-use Bags, Date Published: 30 December 2010, Date Accessed: 19 September 2011, www.friendsoftheirishenvironment.net/index.php?do=friendswork&action=view&id=878*

*5. Dikgang, J. Leiman, A. and Visser, M. (2010) Analysis of the Plastic-Bag Levy in South Africa, Policy Paper No. 18, Environmental Policy Research Unit, School of Economics, University of Cape Town, July 2010, www.econrsa.org/papers/p_papers/pp18.pdf*

## 7.3 Key Observations Regarding Effectiveness

In 2008 AP EnvEcon Limited reviewed Ireland's plastic bag levy and concluded that in order to be effective it ne`eded to be more flexible. Greater flexibility reduces the need to continually revisit primary legislation, and can more easily account for changing economic and consumer environments. The authors of the review suggested that each year the tax should be allowed to increase with inflation (measure by the Consumer Price Index), and that on top of this there should be the option to increase the levy by up to 10% of the base level for that year.[153] It would seem that flexibility in the levy structure of any eco-tax on disposable products would be desirable, especially at a time of much economic uncertainty and where rates of inflation and consumer spending are likely to fluctuate substantially over coming years.

The literature suggests that levies on plastic bags tend to be more successful when they are passed directly to consumers. In Denmark and South Africa, for example, the retailer is expected to cover the tax and not obliged to pass on the full costs to the consumer. This can have the effect of reducing the financial incentive for consumers

[153] AP EnvEcon Limited (2008) *Regulatory Impact Analysis on Proposed Legislation to Increase Levies on Plastic Shopping Bags and Certain Waste Facilities*, November 2008, www.environ.ie/en/Legislation/Environment/Waste/WasteManagement/FileDownLoad,21599,en.pdf

to change their behaviour and 'hide' the tax from public view. In order to create public awareness and achieve the greatest degree of behaviour change Ireland applies its tax at the point of sale and advertised the tax widely before it was implemented. Indeed, Convery et al. note that when introducing taxes on single-use products it is frequently necessary to undertake a publicity campaign to clearly demonstrate the reasons and rationale behind the tax. This was undertaken in Ireland and, according to these authors, it helped to improve the initial acceptance and effectiveness of the tax.[154]

In addition, the introduction of direct and variable rate charging at the household level, would, at the margin, support the financial case for reusable rather than disposable products, such as bags and cutlery. Such a charging scheme, placing the incentive directly at the household level would enhance the effectiveness of measures to reduce consumption of non-recyclable items.

Given the above, it would seem that the following approaches may be necessary when implementing taxes on single-use disposable products (amongst others):

- Apply taxes to items where alternatives are clearly available (this is likely to ensure a reasonable response to the tax);
- Continual review of the tax to ensure that its effectiveness is not being eroded over time (e.g. through inflation);
- Ensure the tax is designed with sufficient inbuilt flexibility to adapt to changing economic conditions;
- Prior to introducing the tax, develop an effective communication campaign to advertise the rationale behind the tax. In this respect, there should be a clear rationale for the tax; and
- Albeit that this is desirable rather than necessary, it is helpful to be introducing such measures against the backdrop of a DVR charging for household waste. This can help strengthen the response to price changes occasioned by the tax.

[154] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

# 8.0 Subsidies for Products

In the arena of waste prevention at the household level it was found that the most common subsidies included those for home composting schemes and reusable nappies. The former are well established in many countries across the world and a significant body of evidence exists which demonstrates the success of such initiatives. Although fairly widely practiced, financial support for the purchase of reusable nappies and laundry services has not been as widely studied. Given that home composting schemes are well established in many areas, Bruxelles Environment expressed an interest in examining the use of subsidies to promote the uptake of reusable nappies in particular.

In an effort to reduce waste arisings, municipalities in a number of countries have begun subsidising the cost of reusable nappies (subsidies for both home composting and nappies are typically implemented at the local level and vary from one municipality to the next). The UK based charity, Go Real, has reported that approximately 3 billion disposable nappies, equalling 690,000 tonnes, are sent to landfill every year.[155] The charity states that this is as much as 4% of the household waste stream; however, this appears to be an overestimate as it has been reported by the Department of Environment, Food and Rural Affairs (DEFRA) that in 2006/7 sanitary products (including nappies) accounted for 2.51% (712,015 tonnes) of the total municipal waste stream, or 3.54% of waste collected directly from households (662,259 tonnes).[156] In Vienna disposable nappies have been reported to account for 4.3% of the residual waste stream (13.3 kg per inhabitant per year).[157] These are significant quantities and have spurred many municipalities to provide incentive/behaviour change schemes to curb the use of disposable nappies.

Through a variety of schemes a large number of local authorities across the UK have begun providing subsidies for reusable nappies. Although there appears to be a paucity of data on the impacts of these schemes, the UK provides an interesting case study, with work by West Sussex County Council being cited as a particular example of good performance.

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[155] Go Real (2010) *Reducing Disposable Nappy Waste: Review of Waste Policies - Call to Evidence*, October 2010, www.goreal.org.uk/media/documents/Call_to_Evidence_v2_041010.pdf

[156] Resource Futures (2009) *Municipal Waste Composition: A Review of Municipal Waste Component Analyses*, Report for the Department for Environment, Food and Rural Affairs, 31 March 2009, http://randd.defra.gov.uk/Default.aspx?Menu=Menu&Module=More&Location=None&Completed=0&ProjectID=15133

[157] This figure was for 2003; source: Salhofer, S., Obersteiner, G., Schneider, F. and Lebersorger, S. (2008) Potentials for the Prevention of Municipal Solid Waste, *Waste Management*, Vol. 28, pp. 245–259

## 8.1 Case Study

The Go Real campaign is a UK wide charity, reportedly working with over 60 local authorities, that aims to promote the use of non-disposable nappies. Sometimes using local authority support, the charity often sets up subsidised schemes to provide households with nappies and regular laundry services.[158] The Women's Environmental Network has also worked extensively to promote the use of reusable nappies across the UK and has done much to push the agenda over the last two decades.[159] Other local schemes have also been set up, such as the Real Nappies for London programme, which represents a number of London based authorities.[160] Some of the authorities represented by this programme offer a subsidy of £54.15 (€63.27 at 20 September 2011) per baby for the purchase of reusable nappies. There are many other examples across the UK, some of which have been listed in Table 16. Despite their widespread use, no research appears to have been conducted that has sought to understand the significance of various factors on the rate of conversion of users from disposable to real nappies.

Table 16: Some Examples of Local Authorities Providing Subsidies for Reusable Nappies in the UK

| Authority | Subsidy |
|---|---|
| Gwynedd County Council | €35 voucher |
| Bedfordshire Real Nappy Cash Back Scheme | €29 when spending €58 or more on nappies or laundry service |
| Bracknell Forest Borough Council | €35 for nappies or laundry service |
| Buckinghamshire County Council | €35 for nappies or laundry service |
| Camarthenshire County Council | Free nappy trial for residents |
| Cheshire County Council | €29 for reusable nappies or a one month free laundry service |
| Cumbria County Council | €35 when spending €52 or more, or €52 off a €78 spend for twins |
| Derbyshire County Council and Derby City Council | €23 for nappies when spending more than €58 |
| Dorset County Council | €35 for nappies |
| Essex County Council | Council provides subsidised sample packs for €11.50 each (valued at €29) |
| Hertfordshire | €46 for the purchase of nappies or laundry services |
| Lancashire County Council | €52 for nappies (€80.50 for two or more children) |
| Leeds City Council | €23 when spending more than €58 on nappies |
| Leicester County Council | €35 for nappies or laundry services |

[158] Subscription costs for the Go Real campaign vary depending on the birth rate within the authority: £350 to £1000 per annum for enhanced membership, excel VAT. See: Go Real (2011) *Home Page*, Date Accessed: 20 September 2011, www.goreal.org.uk

[159] Women's Environmental Network (2011) *Home Page*, Date Accessed: 20 September 2011, www.wen.org.uk

[160] Real Nappies for London (2011) *Home Page*, Date Accessed: 20 September 2011, www.realnappiesforlondon.org.uk/index.php

| Authority | Subsidy |
|---|---|
| Liverpool City County Council | €58 for nappies or a one month trial at a laundry service |
| Milton Keynes | €35 for nappies |
| Norfolk Council | €35 when spending €52 or more on nappies or laundry services |
| Northampton County Council | €29 when spending over €69 on nappies |
| Pembrokeshire | €35 when spending €58 on nappies, or €69 when spending €173 |
| Perth & Kinross | €58 for nappies, with trial kit also available to residents at €11.50 |
| Plymouth | Parents can access a free trail kit for one month |
| Sheffield | €11.50 when spending €35 or more on nappies |
| Trafford Council | Free laundry service for one month |
| Wakefield Council | €46 for nappies |
| Wiltshire and Swindon | €35 when spending €69 or more on nappies |
| *Source: BabyKind (2011) Real Nappy Incentive Schemes, Date Accessed: 20 September 2011, www.babykind.co.uk/incentiveschemes.htm* | |

Many local authorities in the UK provide subsidies in order to reduce their disposal costs, which have been increasing steadily over many years (the country's landfill tax is set to rise – at €9.20 per annum – from the current €64.4 per tonne to €92 in 2014/15). Due to the rising cost of landfilling many of these subsidies can effectively be cost neutral to the authority since they contribute to a reduction in the costs of disposal of disposable nappies. Indeed, the Go Real campaign claims that *'local authorities would need to achieve less than a 5% conversion rate in their area to payback the membership fee.'* The charity reported in 2010 that if only 10% of UK households with new-born children were converted to using reusable nappies the savings on disposal costs would amount to €6.2 million per annum.[161]

A 2008 LCA conducted by the Environment Agency (EA) assumed that over the first two and a half years of a baby's life it will use on average 4.16 nappies per day (figure based on EA survey). With an average nappy weight of 38.6g (2006 average) this adds up to a total of 146.5kg worth of discarded nappies in the first two and a half years of a baby's life.[162] This weight excludes that of the excreta (the above study assumed 727kg for the first two and a half years), and once collection and disposal costs are taken into account it is easy to see that waste prevention subsides may readily pay for themselves.

In a 2007 publication Husaini et al. report on the outcomes of a subsidised nappy scheme initiated in West Sussex at the end of the 1990's. In their comparison of

[161] Go Real (2010) *Reducing Disposable Nappy Waste: Review of Waste Policies - Call to Evidence*, October 2010, www.goreal.org.uk/media/documents/Call_to_Evidence_v2_041010.pdf

[162] Environment Agency (2008) *An Updated Lifecycle Assessment Study for Disposable and Reusable Nappies*, October 2008, http://randd.defra.gov.uk/Document.aspx?Document=WR0705_7589_FRP.pdf

economic instruments it was deemed to be an effective instrument, with high applicability for implementation across England.[163] The authors provide the following details:

> *'The West Sussex real nappy scheme provides free laundry services for families using the scheme. The results of the scheme show that 500 families participated in the scheme between 1999 and 2000 and this saved the production of 800 tonnes of disposable nappies as well as cost savings of €32,000 (~£20,000) for the local authority. The waste reduction per family that used the scheme translated to 1,600 kg per family per year and the incentive provided amounted to €48 per family per year'.*

In this study it was stated that participating households reduced their waste arisings by 1,600kg over the year. Given the weight of reusable nappies and excreta used in the LCA outlined above – the sum of which is said to total 874kg over the first two and a half years of a baby's life – this would appear to be an overestimate. It is thus likely that the waste prevention figure quoted here was not solely the result of a reduction in the use of disposable nappies, but the result of a combined or coordinated waste prevention scheme in the area (e.g. awareness campaigns, general publicity, change in local authority collection services etc.).

West Sussex County Council currently estimates that 70,000 disposable nappies are discarded daily by its residents.[164] The council has continued the subsidy and at present offers parents with new-borns free starter packs worth £100, plus £30 for the purchase of additional cotton nappies. The starter packs are delivered in two instalments: the first for the initial four months of life and the second for babies over four months. Since the inception of the subsidy programme in 1999 the council reports that it diverted approximately 5,189 tonnes of nappies from landfill and has saved over £500,000 (€575,000) in avoided landfill charges (an annual average of £41,666). The scheme is said to cost £20,000 (€23,000) per annum to operate (5% of the Council's waste prevention budget) and as such the scheme has saved the Council close to £22,000 (€25,300) per annum (on average).[165]

## 8.2 Evidence of Impacts

### 8.2.1 Waste Prevention Impacts

As stated above, Eunomia was unable to identify any studies – either in the UK or abroad – which have sought to quantify the relationship between the level of subsidy

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[163] Husaini, I. G., Garg, A., Kim, K. H., Marchant, J., Pollard, S. J. T. and Smith, R. (2007) European Household Waste Management Schemes: Their Effectiveness and Applicability in England, *Resources, Conservation and Recycling*, Vol.51, No.1, pp.248-263.

[164] West Sussex County Council (2011) *West Sussex Waste Prevention Advisors – West Sussex Initiatives*, Date Accessed: 20 September 2011, www.westsussexwpa.org.uk/west_sussex_initiatives

[165] Prices based on exchange rate on 20 September 2011; source: West Sussex County Council (2011) *West Sussex County Council – Agenda Item No. 5(a) – Questions*, Question 2, Date Published: 22 July 2011, www2.westsussex.gov.uk/ds/cttee/cc/cc220711i5a.pdf

and the corresponding uptake of reusable nappies. It is likely that conversion to using reusable nappies will be influenced by a number of factors, such as: local socioeconomic conditions; extent of advertising and supporting behaviour change campaign; value of subsidy; ease with which subsidy can be accessed; and presence of local nappy laundry services etc. These influencing factors would make it hard to compare regions, especially where the subsidies in themselves are administered in different ways. Despite the apparent lack of data, the above case study provides a clear indication that potentially significant cost savings and reductions in waste arisings can be achieved if families start using reusable nappies.

Other sources have reported on countries like Italy, where nurseries and households wishing to use reusable nappies – in the regions of Emillia Romagna, Parma, and Colorno – can receive free educational programme and subsidies from their local municipality. Under this programme families with new-borns can receive a voucher worth €50 towards purchasing a reusable nappy kit valued at €120. Nurseries who have signed up to the scheme are now exclusively using reusable nappies. It is reported that 31% of new-borns and 25% of 0 to 36 month old babies in participating municipalities are now using these nappies since the programme started (there do not appear to be data on the actual quantities of waste diverted).[166]

Sharp *et al.* have reported that if 10–20% of households started using reusable nappies it may be possible to achieve overall reductions in residual waste arisings of approximately 0.5–1%.[167] In Vienna it has been estimated that a 10 to 20% conversion to the use of reusable nappies could potentially reduce household waste arisings by 2kg per household per year. The authors, however, point out that conversion rates have, in practice, been much lower than this, and this is despite the significant subsidy on offer (€100 subsidy when purchasing a basic starter pack worth €250).[168]

The UK's Waste & Resources Action Programme (WRAP) has developed an online Waste Prevention Toolkit which can be used to determine the likely diversion rates for municipalities who are able to encourage families to use reusable nappies (a downloadable excel spreadsheet). The impacts are dependent on the number of households which start using real nappies (this is assessed through monitoring the redemption of vouchers in outlets, attendance at training sessions, or pre and post campaign surveys of households with new-borns). For example, in an authority with

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[166] Bruxelles Environment (2010) Mapping Report on Waste Prevention Practices in Territories within EU27 - Pre-Waste: Improve the Effectiveness of Waste Prevention Policies in EU Territories, October 2010, http://www.bruxellesenvironnement.be/uploadedFiles/Contenu_du_site/Professionnels/Formations_et_s%C3%A9minaires/Conf%C3%A9rence_Pre-waste_2011_(actes)/p3-%20prewaste-mapping-report.pdf; 7

[167] Sharp, V., Giorgi, S. and Wilson, D. C. (2010) Delivery and Impact of Household Waste Prevention Intervention Campaigns (at the Local Level), *Waste Management & Research*, Vol.28, 256-268.

[168] Salhofer, S., Obersteiner, G., Schneider, F. and Lebersorger, S. (2008) Potentials for the Prevention of Municipal Solid Waste, *Waste Management*, Vol. 28, pp. 245–259

100,000 households, 230,000 inhabitants, and a birth rate of 5,000 babies a year, a 5% conversion to reusable nappies could divert an estimated 90 tonnes of waste every year.[169]

Given the above it is evident that there is much evidence to suggest that subsidies and behaviour change campaigns do lead to reductions in the use of disposable nappies. However, it is not clear how the value of the subsidy drives conversion. Research finds that there are many behavioural barriers to using reusable nappies – for example, inconvenience, time for laundering, smell, storage issues etc. – and thus it is possible that many of the current subsidies on offer are insufficient to overcome these barriers.[170] It would be expected that higher subsides would naturally lead to higher uptake, but further research would have to be undertaken to determine this. In addition there appears to be no research available on the extent to which those who receive the subsidy continue to use reusable nappies. It is conceivable that a proportion will revert to use of disposables over time, but no figures have been reported in relation to this.

### 8.2.2 Environmental Impacts

The English and Welsh Environment Agency's LCA introduced above, found that the use of the average disposable nappy (available in 2006) would result in the average global warming impact of 550kg per child over the first two and a half years of life. This was better than that calculated for reusable nappies, which for the same time period produced 570kg of greenhouse warming equivalents. The authors stress, however, that the impacts of reusable nappies are highly dependent on the way in which they are laundered:

> *'Washing the nappies in fuller loads or line-drying them outdoors all the time (ignoring UK climatic conditions for the purposes of illustration) was found to reduce this figure by 16 per cent. Combining three of the beneficial scenarios (washing nappies in a fuller load, outdoor line drying all of the time, and reusing nappies on a second child) would lower the global warming impact by 40 per cent from the baseline scenario'*.

The authors go on to say:

> *'In contrast, the study indicated that if a consumer tumble-dried all their reusable nappies, it would produce a global warming impact 43 per cent higher than the baseline scenario. Similarly, washing nappies at 90°C instead of at 60°C would increase global warming impact by 31 per cent over the baseline. Combining these two energy intensive scenarios would increase the global warming impact by 75 per cent over the baseline*

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[169] It is necessary to register online before accessing the toolkit; see: WRAP (2011) *Waste Prevention Toolkit*, Date Accessed: 20 September 2011, www.wrap.org.uk/applications/waste_prevention_toolkit/restricted.rm

[170] Brook Lyndhurst Ltd (2009) *Household Waste Prevention Evidence Review* , Report for the Department for Environment, Food and Rural Affairs, October 2009, http://randd.defra.gov.uk/Document.aspx?Document=WR1204_8365_FRP.pdf

*scenario, or some 420kg of carbon dioxide equivalent over the two and a half years.'*

It is thus clear that the impact of reusable nappies can be both better and worse than disposable nappies. Such ambiguity makes for difficult decision making and has led some to question the practice of subsidising reusable nappies. In order to ensure that reusable nappies are used in a sustainable fashion it may be necessary for municipalities to provide advice on how best to launder their nappies. For example, it has been suggested that the best outcomes can be achieve by:

- Line drying nappies outside whenever possible;
- Tumble drying as little as possible;
- When replacing appliances, choosing more energy efficient appliances (A+ rated machines are preferred);
- Not washing above 60°C;
- Washing fuller loads; and
- Reusing nappies on other children.

## 8.3 Key Observations Regarding Effectiveness

As alluded to above, subsidies aimed at waste prevention only really become financially viable when the cost of disposal or treatment is raised to a level which will incentivise local authorities to encourage their residents to reduce their waste arisings. Thus, the presence of a tax on landfill or incineration can help create the necessary incentives to actively seek to promote waste prevention through household subsidies. Landfill bans may also have a similar effect on the costs of disposal if alternatives to landfill are more expensive than landfill itself. However, the effect of bans is less 'certain', not least since they can lead to over-capacity in alternative treatments, and a decline in prices for residual waste management.

It was also suggested that there are a number of behavioural barriers to using reusable nappies and that the subsidy would have to provide a sufficient incentive to overcome this. Obviously, as the cost of waste disposal/treatment increases it may become more feasible to offer higher subsidies; unless, of course, other more effective waste prevention measures become viable.

In addition, the introduction of direct and variable rate charging at the household level, would, at the margin, support the financial case for reusable rather than disposable nappies. This would also provide an on-going inducement for those who have taken the (one-off) subsidy to continue to use the reusable nappies.

Given the above, the following actions are to be recommended when considering providing a subsidy for reusable nappies:

- Although not controlled locally, the presence of a landfill/incineration tax (or other means to make the costs of managing residual waste more costly) will make the provision of waste prevention subsidies more economically justifiable to the local authority;

- The subsidy needs to be provided with supporting information / training / support to assist with optimal usage and to provide details on how to ensure that the best environmental outcome is achieved when using reusable nappies; and
- As with taxes on disposable products, a desirable rather than a necessary feature is the use of a DVR charging for household waste. This can help strengthen the response to the subsidy.

# 9.0 Deposit-Refund Systems for Beverage Containers

Deposit-refund systems (DRSs) are a particular form of product tax/recycling subsidy and have been defined as follows:

> *'A deposit-refund system is the surcharge on the price of potentially polluting products. When pollution is avoided by returning the products or their residuals, a refund of the surcharge is granted.' OECD, Glossary of Statistical Terms.* [171]

A DRS encourages the return of the materials into an organised reuse, recycling or treatment / disposal process. The producers typically finance the process through the payment of an administration fee on each container. Drinks containers are the most common target of DRSs, though economic theory suggests the schemes could be applicable to hazardous materials and other waste streams, subject to transaction costs being minimised.

It seems fair to say that within the EU, the emphasis of deposit refunds has shifted somewhat from a desire to encourage re-use of containers to a measure to increase recycling. Other jurisdictions have also consciously employed DRSs to reduce littering. Given this, it seems clear that not all DRSs are 'waste prevention' measures per se. Indeed, even in countries, such as Germany, where the law has sought to maintain / increase the market share of multi-trip packaging, this has proved difficult, especially given the requirement to respect EU Single Market rules which seek to ensure that national producers are not treated more favourably than those in other countries (unless there are very clear reasons for doing so).

The first case study looks at Germany, and the extent to which the system has achieved its objectives in terms of re-use. We then discuss where DRSs have been used more widely, before highlighting their environmental effects. Before considering the case studies, we first review the economic rationale for DRSs.

## 9.1 Economic Rationale for Deposit Refund Schemes

In such schemes, also known as 'bottle bill' programmes (in North America), consumers pay a deposit (tax) on a container at the time of purchase. This should, in theory, be set at the extra social cost of improper disposal over the net recycling cost (assuming there is already an Advance Disposal Fee (ADF) on the manufacturer equal to the net recycling cost). This means that if the product is improperly disposed of, that individual pays the external cost of improper disposal by foregoing the refund, which would typically be set equal to the initial deposit.

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[171] OECD (2001) *Glossary of Statistical Terms: Deposit-Refund System*, Date Accessed: 28 June 2011, http://stats.oecd.org/glossary/detail.asp?ID=594

This incentive is considered particularly appropriate for items with hazardous contents where it is important to manage them in the best way, and to discourage their illegal dumping. Likewise for products where the temptation to litter is high, or the resulting litter is considered to create significant disamenity impacts, deposits may be a suitable mechanism.

The distinction between deposit refund schemes on the one hand, and ADFs coupled with a household recycling refund on the other, is in the re-collection of the product at the end of its useful life. Mandatory deposit schemes involve a separate collection path, rather than being collected as part of the municipal recycling system.

Several theoretical studies have argued that a deposit/refund is the best policy in the presence of illegal disposal.[172] Palmer et al model paper, glass, plastic, aluminium, and steel. They find a substantial difference in the intervention levels necessary to achieve reductions in disposal with the various policies. A $45/ton deposit /refund would reduce disposal by 10%. Alternatively, the government could obtain a comparable reduction using an ADF of $85/ton or a recycling subsidy of $98/ton.

A key point is that the deposit/refund creates incentives for both recycling and source reduction, whereas an ADF or a recycling subsidy takes advantage of only recycling or source reduction in isolation.[173] However, it is important to note that the theoretical studies, in abstracting from the real world situation, have not taken into account all the potential costs of administering such schemes. In fact they are not precisely modelling existing 'bottle bill' programmes, but rather a more generalised version of deposits and refunds, applied 'upstream' on manufacturers and recyclers.

Palmer and Walls accept that in practice there could be significant administrative costs associated with refunding deposits, which could reduce the efficiency of the approach.[174] This issue is discussed by Palmer et al with numerical estimates of the effects of administrative costs on the overall efficiency of deposit refunds relative to product taxes and recycling subsidies.[175] Viewing their results alongside empirical evidence from Ackerman et al., they suggest that administrative costs may be of the

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[172] T. Dinan (1993) Economic Efficiency Effects of Alternative Policies for Reducing Waste Disposal, *Journal of Environmental Economics and Management* 25: 242–56; D. Fullerton and T. C. Kinnemann (1995), Garbage Recycling and Illicit Burning or Dumping, *Journal of Environmental Economics and Management*, 29 (1); Peter S. Menell (1990) Beyond the Throwaway Society: An Incentive Approach to Regulating Municipal Solid Waste, *Ecology Law Quarterly*, vol. 17, pp. 655-739; Hilary Sigman (1995) A Comparison of Public Policies for Lead Recycling, *RAND Journal of Economics*, vol. 26, no. 3 (Autumn), pp. 452-478.

[173] K. Palmer, H. Sigman and M. Walls (1997) The Cost of Reducing Municipal Solid Waste, *Journal of Environmental Economics and Management* 33, 128-50.

[174] Palmer and Walls (1997) Optimal Policies for Solid Waste Disposal Taxes, Subsidies and Standards. *Journal of Public Economics* 65(8): 193-205.

[175] K. Palmer, H. Sigman and M. Walls (1997) The Cost of Reducing Municipal Solid Waste, *Journal of Environmental Economics and Management* 33, 128-50.

same order as the cost savings from using a deposit/refund.[176] Due to such considerations, Palmer et al., Fullerton and Kinnaman, and Palmer and Walls all argue that deposit refunds should be imposed upstream on producers rather than on final consumers to minimize administration and transaction costs.[177]

## 9.2 Case Studies

### 9.2.1 Germany

Mandatory deposits were introduced in Germany in January 2003 for non-refillable containers (cans, glass and plastic bottles) for water, beer and carbonated soft drinks. Prior to this, only refillable containers were subject to a deposit, in such cases implemented by manufacturers rather than through legislation. The level of the deposit for refillables was also set by the manufacturers.In December 2004 the European Court of Justice confirmed that the compulsory deposit scheme is, in principle, compatible with EU law.[178] This decision paved the way for new provisions. On 28th of May 2005 the 3rd amendment of the Packaging Ordinance came into force.[179] It simplifies the deposit on cans (regulated in Art. 9 of the Packaging Ordinance).

The new provisions were implemented in two steps. The first step came into force on the 28th of May 2005: There is only one standard deposit of 25 euro cents for all sizes of containers ranging from 0.1 litres to 3 litres. Since May 2006, the third amendment to the German Packaging Ordinance has been in force – with an extended scope of deposits being levied on non-returnable beverage packages and the simultaneous launch of a nationwide standardised clearing system. The deposit was extended to all one-way drinks packaging which is not "ecologically advantageous" and covers the following beverages (see Article 9, paragraph 2 of the Packaging Ordinance):

- Beer (including alcohol-free beer) and mixed drinks containing beer,
- Mineral water, spring waters, table waters and remedial waters,
- Carbonated and non-carbonated soft drinks (specifically lemonades, including cola drinks, fizzy drinks, bitter drinks and ice-tea). Fruit juices, fruit nectars, vegetable juices, vegetable nectars, drinks with a minimum of 50 per cent milk or other milk-derived products, dietetic drinks within the meaning of Article 1

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[176] Frank Ackerman, Dmitri Cavander, John Stutz, and Brian Zuckerman (1995) *Preliminary Analysis: The Costs and Benefits of Bottle Bills*, Draft report to U.S. EPA/Office of Solid Waste and Emergency Response, Boston, Mass.: Tellus Institute.

[177] D. Fullerton and T. C. Kinnemann (1995), Garbage Recycling and Illicit Burning or Dumping, *Journal of Environmental Economics and Management*, 29 (1); Palmer et al (1997); Palmer and Walls (1997).

[178] ECJ from 14.12.2004, C-463/01 (Mineralwässer); ECJ from 14.12.2004, C-309/02 (Radlberger).

[179] Ordinance of 21 August 1998 (*Federal Law Gazette* I p. 2379); as amended by Article 1 Third Amending Ordinance of 24 May 2005 (*Federal Law Gazette* I of 27 May 2005, p. 1407); last amendment by Fifth Ordinance of 2 April 2008.

(1) of the Ordinance on Dietetic Foodstuffs (Diätverordnung) and mixes of such drinks shall not be soft drinks within the meaning of sentence 1.

- Mixed alcoholic drinks
  - produced using
    - products which are subject to spirits tax under Article 130 (1) of the Federal Spirits Monopoly Act (Branntweinmonopolgesetz) or fermentation alcohol made from beer, wine or wine-like products, including in processed form, which has been processed using technology which no longer meets the requirements for good manufacturing practice and containing less than 15 per cent alcohol or
  - containing less than 50 per cent wine or wine-like products, including in processed form.

Consequently, wine, milk and fruit-juices, for example, continue to be exempted from the non-returnables deposit, as do packages that under the German Packaging Ordinance are classified as 'ecologically advantageous'. In Article 3, paragraph 4, the Packaging Ordinance legally defines the following drinks packaging as "ecologically advantageous":

- Drinks carton packaging (brick packs, gable-top cartons);
- Drinks packaging in the form of polyethylene bags; and
- Stand-up bags.

Furthermore so-called "individual solutions" (major discounters such as Aldi, Lidl and Plus) were discontinued. Under the "individual solutions", discounters only had to take back one-way drinks packaging sold by their own sales chain. Since 2006, stores that sell drinks cans, glass or plastic bottles are obliged to take back packaging from all drinks manufacturers. Empty one-way bottles and cans can be returned to any outlet where one-way packaging is sold. This regulation has effectively promoted the development of a uniform nationwide return system and this is now well-established, generating high return rates.

#### 9.2.1.1 Reason for the Policy

The main reason for the introduction of the deposit on cans in Germany was the failure to achieve the targeted reuse rate (72 % of the packaging of beverages had to be reusable) over a number of years.

As a first step towards the current situation, the necessary regulation (Art. 9 Para 2 of the Packaging Ordinance from 1998) came into force and the deposit on cans was introduced. With this legal regulation in place, the deposit on cans now became independent of compliance with the target reuse rate. Furthermore it was the objective of the former German government to promote reusable drinks packaging as a way of implementing a deepening of producer responsibility and at the same time, strengthening the promotion of the saving of natural resources by waste prevention.

The following timetable outlines the key dates associated with the policy's introduction:

- **1991**: The Packaging Ordinance came into force (the ordinance included a target reuse rate of 72 % for the promotion (and protection) of reusable beverage packaging as environmentally advantageous packaging; the Ordinance stated that if the rate at which beverages were filled in reusable packages fell below 72%, a deposit would become compulsory for several types of one-way packaging). Today the Ordinance aims to increase, to at least 80 per cent, the share of beverages sold in reusable drinks packaging and "ecologically advantageous one-way drinks packaging";
- **1998**: amendment of the Packaging Ordinance;
- **1997-2002**: repeated non-achievement of the reuse rate;
- **April 2002**: assessment of costs for the introduction of the deposit on cans;
- **January 2003/October 2003**: introduction of the one-way deposit in two steps;
- **May 2005**: The Ordinance was adopted by the German Parliament;
- **May 2005**: In order to simplify the regulations on one-way drink packaging, a new Packaging Ordinance came into force (third amendment of the packaging ordinance);
- **January 2006**: The fourth amendment of the Packaging Ordinance came into force (implementation of the Directive 2004/12/EU – packaging and waste packages);
- **May 2006**: The deposit became compulsory for non-carbonated soft drinks and alcohol-mixed drinks (alcopops);
- **January 2009**: The fifth amendment of the Packaging Ordinance entered into force, but this did not alter the regulations for one-way drink packaging.

#### 9.2.1.2 Effects of the Policy

The government's ambitious target was to increase the market share of beverages sold in reusable drinks packaging and ecologically advantageous one-way drinks packaging to at least 80 per cent. Instead, the market share of reusable drinks packaging has continuously decreased.

For a short period after the introduction of the one-way deposit policy the weighted average market share of beverages in returnable bottles increased, from 56.2 % in 2002 to 63.6% in 2003.

But this was only a temporary rise as from 2003 onwards, the proportion of reusable drinks packaging has been falling. For all drinks the market share of reusable packaging fell from 71.7% (1991) to 44.3% (2009).[180] Figure 30 shows the changes

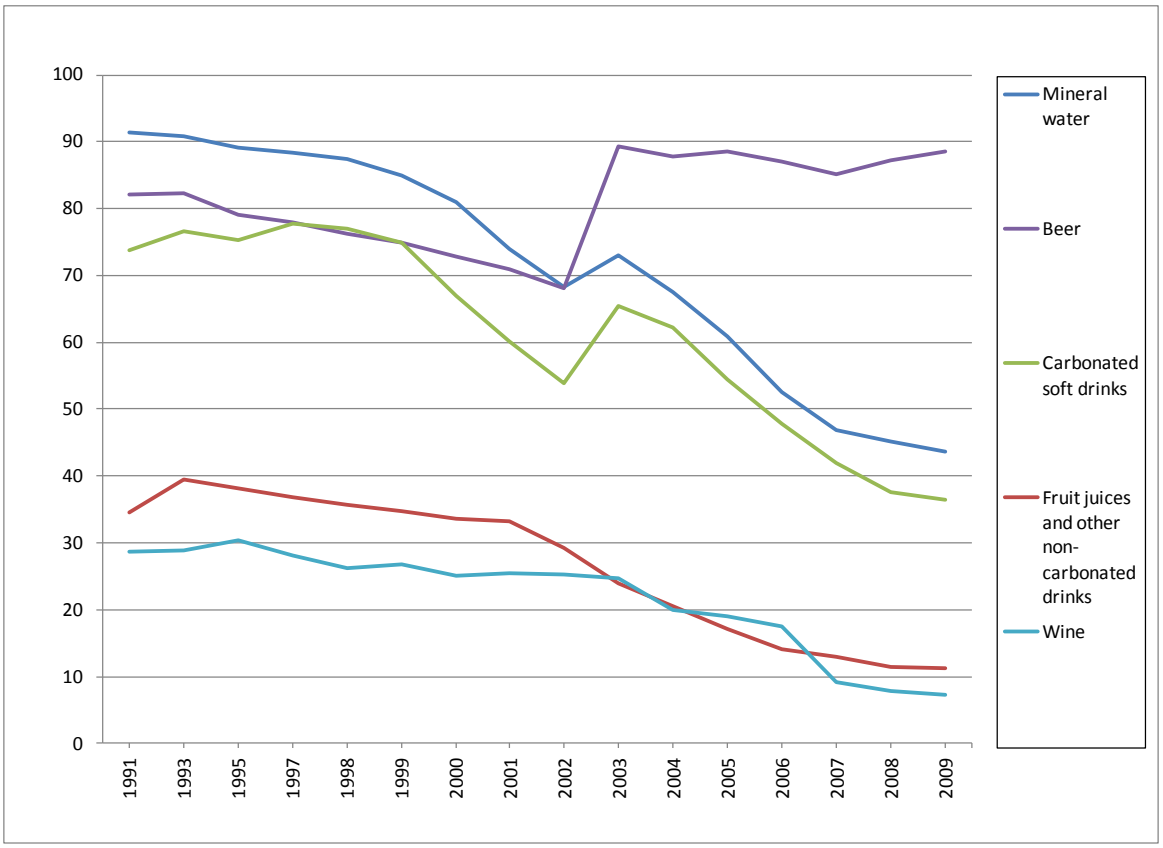
[180] Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (2011) Share of Reusable Packaging in Drinks Consumption by Type of Drink from 1991 to 2009, available at http://www.bmu.de/files/english/pdf/application/pdf/mehrweganteil_zeitverlauf_en.pdf (accessed December 2011).

for the various types of drink. From 1 January 2003 the compulsory deposit for disposable drinks packaging was applied to:

- mineral water
- carbonated soft drinks; and
- beer.

From Figure 30 it can be seen that for these three types there was an immediate increase in the share of reusable packaging, but for both mineral water and carbonated soft drinks the decline continued in subsequent years. For beer, however, the rate of reusable packaging has remained at, or just below the 89% reached in 2003.

Figure 30: Share of Reusable Packaging in Drinks Consumption by Type of Drink (1991 to 2009)




*Source: Gesellschaft für Verpackungsmarktforschung mbH (GVM) 2011*

From 1 May 2006 the compulsory deposit also applied to non-carbonated soft drinks (but excluding fruit juices). The rate of decline in reusables for this type, as shown in Figure 30, can be seen to level out from 2006, whereas for wine, which remained exempt from the deposit, the rate of decline in the share of reusables from 2006 actually increased.

Thus it would appear that the introduction of the deposit has had a waste prevention effect. For mineral water and carbonated soft drinks it has arguably deferred the decline by one or two years, if not actually slowed the year to year rate of decline

since 2003. However, in the case of beer, the use of refillables has, since 2003, been at a higher rate than it was in 1991 (approximately 88% compared with 82%, and well above the low of 68% seen in 2002).[181] It is difficult to estimate the exact extent of the waste prevention effect since this ought to be considered against the counterfactual situation in which no deposit was in place. In this counterfactual scenario, the market share of refillables might have dropped much more swiftly and steeply.The extent of the 'avoided decline' is clearly dependent on what one believes the most plausible counterfactual to be, and this is not entirely straightforward to understand (to what extent would use of refillables have continued without the deposit system?).

Given the trends shown in Figure 30, it is interesting to consider the potential combination of deposits and taxes on one-way containers and refillables. A packaging tax which was set at a higher level for one-way containers would, intuitively, provide (further) support for reusables.

A remarkable slowdown of sale-figures for cans, from 7.2 billion cans per year to just under 3 billion cans in 2004 was observed.[182] Discounters switched from selling alcohol-free drinks and beer in cans to glass-bottles (for beer) or PET-bottles (for alcohol-free drinks). The switch for sparkling drinks (like beer or soft drinks) has gone hand in hand with new PET-technologies (nano-particular layers for PET-bottles) which prevent $CO_2$ loss from the product. One-way PET-bottles, with a market share of 63%, are the most favoured packaging type followed by reusable PET-bottles with 13.7% and glass-bottles with 13.5%.[183]

The move away from metal beverage cans was due to the obligation on retailers to take back packaging made of the same material and shape, even if they had been sold by competitors. If a retail store paid out a deposit for a can it hadn't sold it meant a 0.25 €Cent loss. As a consequence every retail chain created individually shaped packaging in order to be obliged only to take back their own products. It is not easy to create an individually shaped metal beverage can. Therefore, all German retailers took deposit cans off their shelves and no longer listed them.

Comparing the market share of ecologically-unfriendly one-way drinking packaging with ecologically-friendly one-way and reusable containers shows a clear shift in favour of the first group between 2004 to 2006. As Figure 31 shows, ecologically-unfriendly one-way packaging (yellow) rose from 28.9% to 40.3%, ecologically-friendly one-way packaging (blue) remained at about 4.2% and reusable packaging (green) fell from 66.3% to 55.5%.

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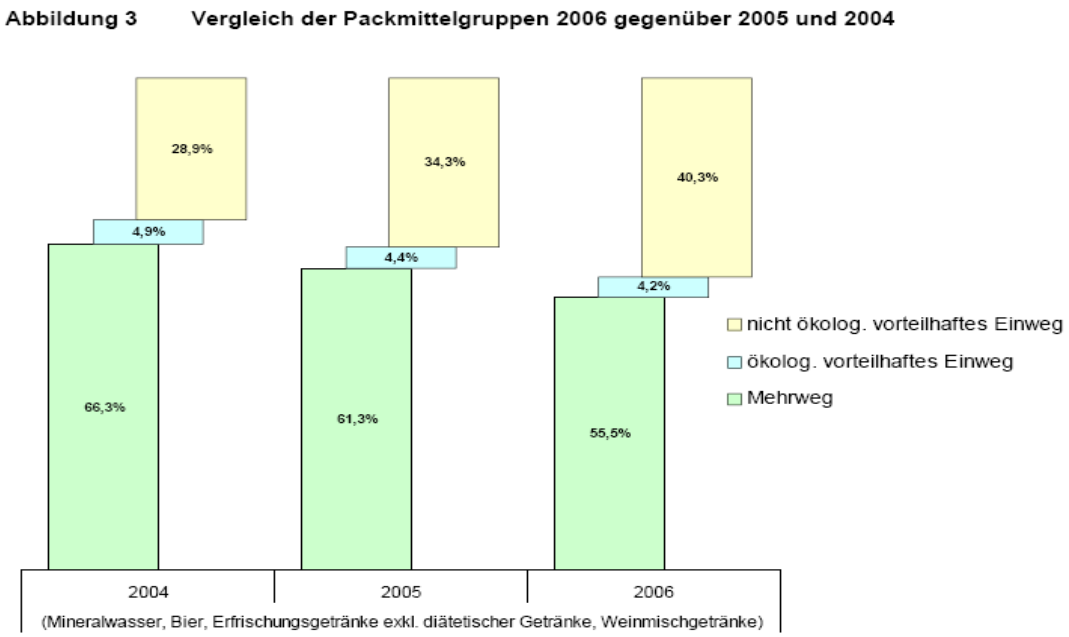
[181] Federal Ministry for the Environment, Nature Conservation and Nuclear Safety (2011) Share of Reusable Packaging in Drinks Consumption by Type of Drink from 1991 to 2009, available at http://www.bmu.de/files/english/pdf/application/pdf/mehrweganteil_zeitverlauf_en.pdf (accessed December 2011).

[182] EUWID from 14.02.2006, p. 2.

[183] EUWID from 19.08.2008, p.6.

Figure 31: Reusable vs. One-way Containers




*Source: German Environmental Agency (Umweltbundesamt) 2006, Texte 15/08*

*Note: ecologically-unfriendly one-way packaging = yellow*
*ecologically-friendly one-way packaging = blue*
*reusable packaging = green*

### 9.2.1.3 Litter

The aim of the one-way deposit for drinking bottles to reduce the littering of the landscape with drinking packages has been achieved, according to the Federal Environment Ministry.[184] Furthermore it can be observed, anecdotally, across the countryside that due to the refunds available, these products are either not thrown away, or are collected by people to receive the deposit. The compulsory deposit is a step towards turning people away from their "throw-away mentality". The deposit on cans and one-way bottles has reduced the littering of streets, public places and landscapes with drinking bottles.

### 9.2.1.4 Lessons Learned

The political and the public responses to the policy have been quite different. The beverage industry and the retail industry (especially the discounter markets Aldi, Lidl and Plus) resisted the planned introduction of the deposit on cans. But there was acceptance from the environmental federations and the reusable packaging industry. Also the German States (Länder) were divided into supporters and opponents, so that an extensive political debate was held between the federal government and the Upper House of the parliament (the representation of the Länder in the legislative procedure).

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[184] EUWID from 19.08.2008, p. 6.

The supporters of the deposits referred to the fact that during the utilization of one-way packages, there is a greater environmental impact. More resources continue to be used if one-way packaging is used instead of reusable packaging. The opponents of the deposit take the view that the recovery of one-way beverage packing is usually possible; very high recycling rates can be reached and thus can be seen as equivalent to re-usable packaging.

The subject of the "Dosenpfand" was of great significance for the German public. The Federal Ministry of the Environment commissioned a survey in June and October of 2003, questioning 2,000 citizens in order to understand their position on the mandatory deposit system. 75 % of the respondents stated that they are in favour of the deposit duty. 48 % of the interviewees stated that they have had mostly good experiences with the nationwide return system. Around 70 % however were not satisfied with the way in which the retail markets were applying the system.

According to an inquiry by the Bielefelder market research institute on behalf of the working group packaging and environment, the new deposit regulation, which became effective on 1st of October 2003, was labelled a failure. The institute had asked 600 households via telephone between 2nd and 6th of October. According to the study 78 % of the German households are of the opinion that the new system of one-way packaging did not improve the situation in relation to the pre-existing system. 52 % pronounced that they were in favour of an abolishment of the deposit obligation. 26 % of respondents criticised some shops for refusing to take back packaging or to pay back the deposit. 36% feel the return of empty cans and bottles is annoying. 28 % complained about delays at the till. 51% buy fewer or no cans, or one-way packages, because of the deposits.[185]

Over several years the German Council of Environmental Advisors ("Sachverständigenrat für Umweltfragen, SRU") reached the conclusion that the deposit system for one-way containers should be withdrawn. The SRU regard the deposit as ecologically ineffective and economically inefficient.[186] Under the current legal climate, after the amendment of the packaging ordinance, the SRU sees further possibilities to encourage reusable systems to meet environmental and economic objectives.[187]

The facts show that in terms of prevention, the major impact has been a material switch from cans to (reusable) glass for beer. Also, probably facilitated by parallel technological change, there has been a change to PET bottles, which now account for 76% of the drinks container market, which is split into 63% one-way and 13% reusable. Even so, and although the scheme has not prevented a progression of the packaging used away from returnables, the market share of returnables remains reasonably high.

The key success and failure factors of the policy for one-way deposits are:

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[185] EUWID, No. 43 from October 2003, p.8.

[186] Council of Environmental Advisors (SRU), Environmental Report 2002 ("Umweltgutachten"), p. 411.

[187] SRU, Environmental Report 2004, p. 350.

- Standardised deposit regulation (no differentiation by content of beverage);
- A standardised level of deposit (25 euro cents); and
- A nationwide deposit-return system.

No variation in the implementation of the Ordinance across Germany was identified.

### 9.2.2 Additional Cases

There are a number of other jurisdictions where deposit refund schemes have been applied. Table 17 shows a list of these. In the following sub-sections, we give a flavour of some of the schemes which have been implemented in different countries.

#### 9.2.2.1 Denmark

The "Danish Bottle Bill" was passed in 2002, and set up the Dansk Retursystem A/S to implement and administer the deposit-refund system for beverage containers. The following deposits apply:

- Beverage containers up to 1 litre: €0.13;
- Plastic containers 0.5 litre: €0.27; and
- Beverage containers > 1 litre: €0.40.

When a producer or importer sells beverages to retailers in Denmark, the deposit is added to the price of containers.

## Table 17: Experience with Deposit Refund Schemes in Other Countries / States

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| Austria | Law to make deposit regulatory | 1992 | PET bottles (non-refillables excluded) | 30% PET<br>60% Cans | $0.40 | | Government | www.BottleBill.org |
| Belgium Ecotaxes Act of 1993 | Containers taxed $0.52 per litre unless they have deposit. | 1993 | Beer, soda and soft drinks containers | | $0.12 <50cl<br>$0.24 >50cl | | Government | www.BottleBill.org |
| Croatia | Deposit-return plus 'incentive fee' to be paid by producer if 50% refill isn't met (5% paid still, if target is met). | 2005 | Glass, PET and metal containers for beer, soft drinks, water, wine and spirits. | | | | Government | EUROPEN Report 2007 |
| Denmark | Packaging Law. All beer and soft drinks must be sold in refillable bottles. Metal banned until 2002. Regulatory deposit for imported glass/plastic containers. Ecotax also. | 1989 (amended 1991) | Beer and soft drinks containers. Deposits on some wine and spirit bottles dependent on retailer. | 99.5 % (beer and soft drinks containers only) | $0.27 <99cl<br>$0.78>99cl<br>Tax $0.14-0.33 | | Government | www.BottleBill.org |
| Estonia | Deposit-return | 2004 | Beer, low alcohol drinks, carbonated/ non-carbonated soft drinks, water, juice, cider and perry. | | Glass 1.0 kroon (refill and NRB)<br>Metal and PET < 0.5 l 0.5 kroons PET>0.5l 1kroon | Retailers | Government | EUROPEN Report 2007 |
| Finland | Tax on beverage containers Exemption from tax only if part of refillable deposit scheme. | 1970s 1990 | One-way beer and soft drink containers | Glass bottles 99%<br>Cans 86% | Non-refillables $0.11<br>$0.45 for larger sizes<br>Tax<br>$0.24 beer $0.47 plastic $0.71 glass | 8,000 sites | Government | http://www.ymparisto.fi/download.asp?contentid=16253&lan=EN |
| Germany | Einwegpfand Deposit on one-way a standard amount, deposit on refillables manufacturer dependent, not legally specified, though tend to be similar. | 2003 | Not containers for wine, fruit juice or spirits | Quota-<br>Glass 90%<br>Alu. 90%<br>Plastic 80% | Glass refillable-<br>Beer € 0.08 Soft drink € 0.15 (many prices, not all listed) | | Manufacturers | http://www.bookrags.com/Container_deposit_legislation |
| Hungary | Tax linked to market share quotas. | 2005 | Beer, low-alcohol drinks, wine, mineral water, carbonated and non- | Quota-<br>Beer 67%<br>Low alcohol | | | | |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| | | | carbonated soft drinks. | 28% Wine 20% | | | | |
| Iceland | Tax on non-refillable containers. | 2008 | Non-refillable glass, steel, aluminium and plastic. | | | | | |
| Kiribati | Special Fund Act 2004 | 2004 | Aluminium cans and PET drinks bottles | | $0.05 ($0.04 returned) | Kaoki Mange operating centres. | | www.BottleBill.org |
| Malta Deposit Return System | Previous ban of non-glass beverage containers, lifted | | | | | | | |
| Mexico | Higher tax on non-refillable bottles and cans. | | | | | | | |
| Fed. States of Micronesia Kosrae Recycling Program | (Deposit-return) | 1991 (amended 2006) | Currently only aluminium cans, but glass and plastic expected to be added soon. | 20,000 cans per day | $0.06 ($0.05 back) | Kosrae Island Resource Management Authority (KIRMA) sites | | www.BottleBill.org |
| Netherlands | Agreement deposit | 1993 | Soft drinks and water in one-way and refillable glass and PET containers | Refillable glass 98% Refillable PET 99% | PET and glass: $0.16 <5l $0.72>5l | | Industry | www.BottleBill.org |
| Norway | Deposit on containers and tax dependent on return rate. Refillables only exempt if 95% return rate is achieved. Retailers (on site >25m$^2$) selling non-refillables, must also sell similar products in refillable. | 1994 | Most drinks excluding milk, vegetable juices and water | Wine/ spirits 60% Beer 98% Soft drinks 98% | $0.16 <5l $0.40 >5l (+Tax inversely proportional to return rate, but if above 95%, no tax) | Over 9000 establishments in the country, plus 3000 deposit machines where receipt is given | Tax is government driven, but recycling fee in place is retailer driven | www.BottleBill.org |
| Peru | Deposit on some bottles | | 620ml size beer bottles | | | | | |
| Portugal | Fillers must ensure quotas met Retailers must sell refillables for all non- | | Quotas- Beer 80% Wine (with certain exceptions) 65% Soft | | | | | |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| | refillables sold. | | drinks 30% | | | | | |
| South Africa | Deposit return system, voluntary i.e. manufacturer driven, not Government. | Around 1948 | Approx. 75% beer, 45% soft drinks and some wine and spirits bottles | | Between 8-15% of product cost (or 0.5-1% if wine/spirit) | | Manufacturer | |
| Spain | | | | Return overall 87% Reuse beer 57% | | | | http://www.cerveceros.org/ |
| Sweden | Law requires rate of 90% recycling of aluminium cans, or complete ban. Industry implemented deposit system to avoid this. PET introduced later as well. deposit | Deposit on one-way containers- 1984 for cans. 1994 for PET (refillables already in place) | Aluminium cans and PET law. Deposit now on most beverage containers. | Recovery rate of 80-90% on one way containers | Voluntary Cans $0.07 Refillable PET $0.56 One-way PET $0.14-0.24 | | Law government driven. Standard bottle and deposit brewer/bottler driven. | www.BottleBill.org http://en.wikipedia.org/wiki/Container_deposit_legislation |
| Switzerland | Deposits required on all refillable drinks containers except cans, which have a voluntary tax of $0.04. | 1990 | All above a certain weight (currently all!) | Refillable glass 95-98% Refillable PET 70% | Ref. glass $0.16<6l $0.40 >6l Ref and one-way PET $0.40>1.5l | | Government | www.BottleBill.org |
| South Australia | Container Deposit Legislation- deposit required on almost all drinks containers, with onus on manufacturer/ wholesaler to ensure convenient system in place for deposit of container/ refunds for customers. | 1975 (integrated into Environment Protection Act in 1993) | Most included except wine (unless in plastic bottle), milk, pure fruit juice or flavoured milk >1l. | 85% non-refillable glass 84% cans 74% PET | $0.10 if refillable to retailer (rare) $0.05 if refillable to collection depot (99.9% done this way) | Mostly collection depots, though some store refillables. | Government legislation with manufacturer/ wholesaler responsibility | www.BottleBill.org |
| Canada-Alberta | All containers sold in Alberta (including imports) must be registered | 1972 | All beverage containers regulatory except milk, which is under a voluntary | Glass (AB Beer) 96% Glass | $0.05 <1l $0.20 >1l Beer $0.10 | 215 independent depots and 78 | Initially government, until 1997 when | www.BottleBill.org |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| | through the Beverage Container Management Board (BCMB). | | scheme | (import beer) 92% Alu (beer) 89% Alu (soft) 79% Overall 78% | | retail outlets (for beer bottles and cans only) | it was turned over to private sector | |
| Canada-British Columbia | All containers must be refillable, and none collected can be landfilled or incinerated. Beer separate system, though still under legislation. | 1970 | All beverage containers except milk, soya milk, infant formulas, dietary or meal supplements, or other milk substitutes. | 81.3% | Non-alcoholic $0.05 <1l $0.10> 1l Alcoholic (not incl. beer) $0.10 < 1l $0.20 >1l Beer $1.2 per dozen | Depots or retailers (all retailers obliged to take back as much as they sell). Beer back to retailer. | Industry | www.BottleBill.org |
| Canada-Manitoba | Beverage producers given option of setting up deposit-return system, or adding a 2 cent per container levy. Only beer producers choose the former. | 1995 | Beer containers only | Refillable beer 95.5% Dom beer 74% Glass 34% Overall residential 31% | $0.10 | Retailer | Opportunity government driven, implementation producer driven | www.BottleBill.org |
| Canada-New Brunswick | Deposits paid on all containers (bar milk), but whilst full paid back on refillables, only half paid back on non-refillables. | 1992 (revised 1999) | All except milk | Refillable beer 96% Dom beer 75% Non-alcoholic 75% | <500ml $0.10 >500ml $0.20 | 89 depots around the province. | Industry | www.BottleBill.org |
| Canada-Newfoundland | Half-back system, with manufacturers prohibited from selling containers other than recyclable or refillable for selected products. Beer operated | 1997 | Beverage containers smaller than 5l, excluding milk, dietary supplements and medicine. | Refillable beer 95% Domestic beer 55% | Non-alcoholic $0.08 ($0.04 back) Alcoholic (excluding beer) $0.20 ($0.10 back) Beer varies -full refund | 'Green Depots' run as businesses. Beer returned to certain retail outlets. | Government, but brewers for beer system. | www.BottleBill.org |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| | separately, run by brewers. Only have to refund when customer buying (1 for 1), otherwise negotiable. | | | | when same number of beer bought as empties returned. | | | |
| Canada-Northwest Territory | Deposit-return system, with additional handling charges for different products/ materials in container. | 2005 | All beverage containers except milk. | Very new system, so no certain figures yet. Approx. 72% | Wine or spirit $0.25 Other $0.10 Plus additional $0.05-0.10 handling fee | 18 government depots or 26 community depots. | | www.BottleBill.org |
| Canada-Nova Scotia | Half-back deposit system. Full refund on refillables, half on non-refillables. | | All beverage containers except milk. | Refillable beer 96% Dom. beer 70% | Non-alcoholic $0.10 Alc. refillable <1l $0.10 >1l $0.20 Alc. non-refillable <500ml $0.10 >500ml $0.20 | 83 province-wide depots. | RRFB-Resource Recovery Fund Board Government and industry. | www.BottleBill.org |
| Canada-Ontario | Deposit-return system on alcoholic drinks containers only. Use of 'Industry Standard Bottle'. | | Alcoholic drinks containers | Refillable 'industry standard bottles' beer 97% | Containers up to 630ml, or metal containers up to 1l $0.10 Over those sizes $0.20 | Beer store only | Brewers | www.BottleBill.org |
| Canada-Prince Edward Island | Non-refillable drinks containers for beer or soft drinks banned since 1977. Wine may have half-back system in place. | 1977 ban, 1984 deposit | Soft drinks and alcoholic drinks. Wine may be included. | Refillable beer 96% Wine/ spirit 59% Soft 98% | Non-al <500ml $0.15 500ml-1l $0.30 >1l $0.70 Alc. $1.20 per dozen, or ).07 each | Mainly retailers (inc. supermarkets and convenience stores), also 15 depots | | www.BottleBill.org |
| Canada-Quebec | Return-to-retail deposit system, with industry required to fund kerbside collection for containers not part of the system. | | All beer and soft drinks containers (not juice, water and iced tea) | Refillable beer 98% Dom. beer 76% | Soft drinks and beer cans $0.05 Beer bottles $0.10 Beer bottles and soft drinks >450ml $0.20 | Retailers (including depanneurs - small convenience stores not usually included in Canada). | | www.BottleBill.org |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| Canada-Saskatchewan | Deposit-return system plus environmental handling charge (EHC) for non-refillable containers, for recycling, and beer bottle deposit system for refillables. | 1973- Litter Control Regulations (unclear, appears the deposit system introduced to this in 1998) | All beverage containers apart from milk (under voluntary system). | Refillable beer 92%<br>Dom. beer cans 95%<br>Alu. cans 95%<br>Glass 83%<br>Overall 86% | Deposits vary widely for diff. materials and sizes<br>Non-ref. glass $0.40-1.00<br>Metal cans $0.10-0.20 | Beer bottles can only receive full refund if returned to 10 specific sites, but can be returned for less at retailers. Other returns at 71 SARCON site | Government | www.BottleBill.org<br>http://www.sarcsarcan.ca/sarcan/faqs.php |
| Canada-Yukon | No kerbside collection. Deposit-return system, with 'recycling club' for children offering 'prizes' as well as refund if certain numbers reached. Refillables not charged recycling fund fee, all others are. | 1998 | All beverage containers except milk. | Refillable bottles 103%<br>Non-refill. bottles 113% (?)<br>Liquor containers <200ml 99%<br>1L 90%<br>>1L 79% (includes refillables) | D=deposit, R=refund<br>Liquor ref. D=$0.10 R=$0.10<br>Liquor non <500ml D=$0.15 R=$0.10<br>>500ml D=$0.35 R=$0.25 | 22 depots or four Liquor Commission outlets | Government | www.BottleBill.org |
| USA-California | California Beverage Container Recycling and Litter Reduction Act Deposit-return system on non-refillable containers | 1987 (Expanded 2000 to include all non-carbonated and non-alcoholic drinks excluding milk.) | Non-refillable drinks containers, inc. beer, spirits, carbonated, fruit drinks and some vegetable juices. Not milk. | Alu 73%<br>Glass 58%<br>PET 46%<br>HDPE 51%<br>Overall 61% | Under 24oz $0.05<br>Over 24oz $0.10 | Redemption centres (not retailers) | | www.BottleBill.org |
| USA-Connecticut | Beverage Container Deposit and Redemption Law | 1980 | Beer, malt, soft drinks and mineral water. | Not recorded. In 2004 | $0.05 | Redemption centres, or retailers (but | | www.BottleBill.org<br>http://www.cga.ct.gov/2005/rpt/2005-R- |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| | Deposit-return system. | | | CRI estimated recycling rate to be similar to Massachus etts of 69% | | only for brands /products they sell). | | 0836.htm |
| USA-Delaware | Beverage Container Legislation Deposit-return system | 1982 Wholesale 1983 Retail | All non-aluminium beer, malt, carbonated, mineral water and soda water containers less than 2 quarts (approx. 1.9L). | Not recorded. | $0.05 | Retail stores, but only for brands they sell. | | www.BottleBill.org |
| USA-Hawaii | Deposit Beverage Container Law Deposit-return system | 2002 | All beverage containers excluding milk and dairy derived products, except tea and coffee or liquor containers. | 72% for 2008 | $0.05 | Redemption centres or retailers (if not within 2miles of red. centre in highly pop. areas, or if under 5,000sq ft of retail space | Government | www.BottleBill.org |
| USA-Iowa | Beverage Container Deposit Law Deposit-return system. Deposit containers banned from landfill in 1990. | 1979 | Beer, soft drinks, soda water, mineral water, wine, liquor and wine coolers. | 93% | Not less than $0.05 | Redemption centres or retailers (who can refuse if they have an agreement with former). | | www.BottleBill.org |
| USA- Maine Maine | Refillable Beverage Container Law Deposit-return system | 1978 | Beer, soft drink, wine cooler, mineral water. Expanded to include wine, liquor, water and non-alcoholic drinks in 1989. | Not recorded. | Wine and liquor $0.15 Other $0.05 | Redemption centres or retailers (who can refuse if they have an agreement with former). | | www.BottleBill.org |
| USA-Massachus etts | Beverage Container Recovery Law Deposit-return system | 1983 | Beer, soft drinks and carbonated water. | 69% | $0.05 | Any retail establishment that sells the container. | | www.BottleBill.org |

| Country | System | Year of Intro | Containers Covered | Capture Rate* | Deposit | Redemption Site | Driver | Reference |
|---|---|---|---|---|---|---|---|---|
| USA-Michigan | Michigan Beverage Container Act<br>Deposit-return system | 1978 | Beer, soft drinks, carbonated and mineral water. Wine coolers and canned cocktails in 1988. | 97% | $0.10 | Retail stores | | wwiw.BottleBill.org |
| USA- New York | New York State Refillable Container Law<br>Deposit-return system | 1983 | Beer and other malt drinks, carbonated soft drinks, wine coolers, mineral and soda waters. | Soft drink 62%<br>Beer 77%<br>Wine coolers 65%<br>Overall 70% | Minimum of $0.05 | Retail stores and redemption centres. | | www.BottleBill.org |
| USA-Oregon | The Beverage Container Act<br>Deposit-return system<br>Only US deposit law with no handling fee. | 1972 | Beer, malt, carbonated soft drinks, mineral and soda water and (as of 2009) water and flavoured water. Bottles and cans under 3L | Overall 84% | Standardized refill bottles $0.02<br>Non-standardized and non-refillable $0.05 | Retail stores. | | www.BottleBill.org |
| USA-Vermont | Beverage Container Law<br>Deposit-return system | 1973 | Beer, soft drinks, malt, soda and mineral water, mixed wine and liquor (added 1987). | Overall 90-95% | Liquor above 50ml $0.15<br>Other $0.05 | Retail stores and redemption centres. | | www.BottleBill.org |

*Source: Oakdene Hollins (2008) Refillable Glass Beverage Container Systems in the UK, Report for WRAP, 26 June 2008.*

*The report notes that capture rate includes containers returned for recycling as well as refilling. Separate figures were not so readily available. Unless specifically listed as something else, the monetary unit is American dollars. The only exception is Canada where the Canadian dollar is used.*

*Percentages given for US capture rates are taken from various sources, often telephone conversations by the Bottle Bill researchers. For more detailed references see www.BottleBill.org*

The total deposit collected by the producers and importers is paid to Dansk Retursystem. That deposit cost is passed onto the consumer by the retailer and then the consumer returns the container either manually or via a reverse vending machine to the retailer. The containers are then transported to Dansk Retursystem's collection centres where they are registered and counted and on the basis of this the company pays the refund back to the retailers.

Essentially, even prior to the Danish Bottle Bill, deposit refund systems had been in place for decades for refillable containers. Until 2002, domestically produced beer and soft drinks were *only* sold in reusable glass, and plastic bottles with a deposit, whilst sales in cans were prohibited. In 2002, under pressure from the EU, the law was changed, allowing all beer and carbonated soft drinks to be sold in one-way packaging, including metal cans. It was this change that prompted the introduction of the Bottle Bill. The deposit system is also linked to the Danish packaging tax.

#### 9.2.2.2 Sweden

Deposit refund schemes are in place in Sweden for glass, PET and aluminium drinks containers. Both refillable and one-way containers are managed under the system. A number of materials-specialist companies manage the scheme, for example, Ab Svenska Returpack-Pet, Svenska Returglas 50-Cl Ab and Ab Svenska Returpack. An interesting characteristic of the Swedish DRS is that it is not a scheme implemented by government, but rather it is industry-led, and was effectively developed as a response to the Government's requirement that the industry achieved a high recycling rate for one-way aluminium beverage cans.

Deposits are paid as follows:[188]

- Cans: €0.04;
- 0.33 L glass bottle: €0.05;
- 0.50 L glass bottle: €0.08;
- ≤ 1 L non-refillable PET bottle: €0.09; and
- > 1 L non-refillable PET bottle: €0.18.

In one small Swedish municipality, a deposit-refund system has been put into place for batteries. A very small (€0.03) deposit is levied on each battery sold, and this is redeemed when the battery is returned to one of the shops participating in the scheme. The estimated result is that approximately 80% of batteries are returned. The cost of the system is covered by the municipal budget.

#### 9.2.2.3 Taiwan

Taiwan has used a deposit refund system for PET bottles since 1989. Manufacturers and importers pay fees into a recycling management fund, and the end-consumers are given a refund when they return the bottle to a designated collection site. The

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[188] Sveriges Brygerier, accessed 2009, http://www.sverigesbryggerier.se/eng/1-emballage/1-index.html

fees are set by a Recycling Fund Management Board, on a level per kilogram plus the per bottle refund amount. For example, in 1998, PET bottles attracted a fee of NT$13.01 per kg (US$0.39 at January 1998 rates) for single material containers, plus a refund amount of NT$0.70 (US$0.02 at January 1998 rates) per bottle.

Recycling rates started from a low base but were 41% in 1991. By 1992, the PET recycling rate had jumped to 80%.[189] The scheme then ran into problems with manufacturers' not registering with the scheme, under-reporting production, and not paying fees so that the fund ran into a deficit and had to reduce the refund from around US$0.06 to around US$0.02.[190] However, recycling rates continue to be high, reaching 100% and remaining there (which does however, suggest that there continues to be some under-reporting of production). In 2007, over 97,000 tonnes were recycled.

#### 9.2.2.4 Other Countries

Many other countries in Europe, as well as countries in Asia and the Americas use the deposit refund system for beverage containers.

Eleven US states have a legal deposit on bottles and cans for beer and soft drinks. Six states also have deposits on mineral water containers. Deposits are made mandatory through the State's 'bottle bills' and range from 2.5¢ to 15¢ per container. According to the US Container Recycling Institute, beverage container recycling rates are far higher (72% on average by weight) in states with bottle bills than those without (28% recycling rate).[191] In most states, retailers are required to take back containers that are in their product line, even if the product was not purchased through their store. In some states (e.g. Maine and California) retailers are exempt where they are near to redemption (civic amenity) sites.

Deposit refund arrangements for containers are also in place in most Canadian provinces.[192] Such schemes raise awareness and help to change consumer attitudes and behaviour. They have also been shown (in the USA) to have a low direct cost per tonne of material recovered although the administrative costs can be high if the scheme is not well designed and implemented. In the USA the states with bottle bills also have approximately three times the national average market share for refillable beverage containers, i.e. around 15%, rather than around 5%.[193]

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[189] D. O'Connor (1996), *Applying Economic Instruments In Developing Countries: From Theory To Implementation,* Report for the OECD Development Centre, May 1996 *web.idrc.ca/uploads/user-S/10536145810ACF2AE.pdf*

[190] Asia-Pacific Environmental Innovation Strategies (2004) *Deposit-Refund Systems for PET bottles in Taiwan,* Report for Taiwan Environmental Protection Administration, December 2004, www.iges.or.jp/APEIS/RISPO/inventory/db/pdf/0133.pdf

[191] http://www.bottlebill.org

[192] Eunomia (2002), *Maximising Recycling Rates, Tackling Residuals,* Report for Friends of the Earth, http://www.foe.co.uk/resource/reports/maximising_recycling_rates_report.pdf

[193] Robert C. Anderson (2000) *The United States Experience with Economic Incentives for Protecting the Environment*, Report for US EPA, Nov 2004.

All the provinces of Canada operate some deposit-refund systems for drinks containers. The beer industry organises collection and refilling of its refillable bottles and has a return rate of 90-99%. For other drinks containers, industry associations fund the kerbside recycling programs run by municipalities. They use a number of different mechanisms to raise fees:

- *Container recycling fee*. This reflects the net cost of recycling, and therefore depends on the material and the recovery rate. These range from zero to $0.08;
- *Environmental handling charge*. $0.03 to $0.07 per unit, depending on size and material. The provincial government collect the funds to operate the program and retain any surplus;
- *Beverage container levy*. $0.02 per unit, used to finance 80% of kerbside collection;
- *Half-back*. Represents half of the deposit paid on a non-refillable container.

In systems other than the half-back (in which only half the deposit is refunded to the consumer on return), the whole deposit – between $0.05 and $0.40 – is refunded on return of the container.

There is also some application of the concept in relation to other waste streams. Austria has deposit-refund schemes for light bulbs/tubes, batteries and refrigeration equipment. The US State of Rhode Island requires a US$5 deposit on all replacement vehicle tyres. The deposit is refunded if the customer returns the old tyres to the point of sale within 14 days of purchasing new tyres. It is also used for pesticide containers in some US states.

The OECD noted in 2001 that among middle-income countries, South Korea has one of the most extensive deposit systems in terms of items covered. Under a 1991 amendment to its Solid Waste Management Act, South Korea introduced a comprehensive deposit program in 1992. The products affected by the system include packaging, batteries, tyres, oil, televisions, air conditioners and washing machines. Producers and importers of the listed products pay the deposits into a "Special Account for Environmental Improvement" and receive refunds as they collect and treat the resulting post-consumer waste. The products covered and the size of the deposit were modified in 1993 and again in 1996. The largest deposit applied to large tyres and amounted to about $0.40. The deposit on paper, metal, glass and plastic packaging was a fraction of a US cent per container.[194]

## 9.3 Environmental Effects

Deposit-refund systems are reported, in the literature, to have a range of possible environmental benefits. The key ones mentioned in the literature are:

- Increasing the use of / reducing the extent of decline in the use of refillables;

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[194] Robert C. Anderson (2004) *The United States Experience with Economic Incentives for Protecting the Environment*, Report for US EPA, Nov 2004.

- Increasing the recycling of containers covered by deposits (for refill or recycling);
- Reducing the extent of littering; and
- Avoiding harmful chemicals being mobilised in the environment (usually not in beverage schemes, e.g. lead acid batteries, or pesticides).

In addition, the logistics may or may not be more efficient. These issues are examined in turn below.

### 9.3.1 Maintaining Use of Refillables

Deposit-refund schemes have been used to encourage the refillable container market, particularly for beer bottles, in Canada and some states of the USA. As suggested above, where distances to transport the bottles are relatively short this probably has a positive environmental benefit. Refillable PET bottles are increasingly an alternative to the heavier, traditional refillable glass, particularly for the large volume containers. Not all deposit schemes are designed with the intention of shoring up the market share of refillables, and where they are, they are not always well designed. The question we are interested in this Section is whether this is an environmentally justifiable goal.

A study in 2002 reviewed the outcomes of 11 different LCAs comparing refillable against one-way packaging (including glass).[195] The study suggested that in terms of five types of air pollutants, the use of refillable containers was, on balance, beneficial. Refillables were also found to generate less solid waste per unit volume of packaged beverage. Comparing refillables with one-way bottles, furthermore, revealed that refillables use less water and use less energy. However, cans compared favourably to refillable glass bottles in respect of water pollution and energy use. Conversely, a study summarising the results of seven LCA studies of refillable glass versus aluminium showed that all LCAs favoured refillable packaging with a 47-82% reduction in water use.

IFEU undertook an LCA on beer packaging in Germany and concluded that from the perspective of climate change, re-usable PET and glass bottles had the least environmental burden producing less than half the emissions (expressed in $CO_2$ equivalent) of the three one-way systems (see Figure 32).

Interestingly, the same organization – IFEU – undertook a separate analysis on refillable packaging on behalf of PETCORE (the PET recycling company).[196] This study suggested a more equivocal view. It highlighted the significance of assumptions regarding collection and recovery of used packaging, and distribution logistics.

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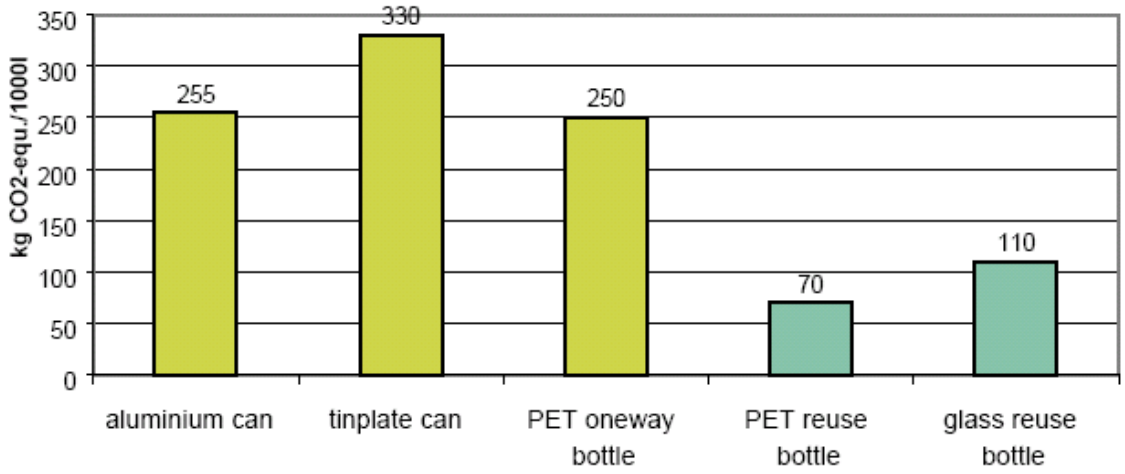
[195] Institute for Local Self-reliance (2002) *Environmental Benefits of Refillable Beverage Containers*, Washington: ILSR.

[196] IFEU (2004) *Okobilanz fur PET-Einwegsysteme unter Berucksichtigung der Sekundarprodukte*, Repoirt for PETCORE, August 2004.

Figure 32: Greenhouse Gas Impacts of Different Packaging Options




*Source: IFEU (2003) Relevant Packaging for Beer, LCA III, in Peter Lee, Paul Vaughan, Caroline Bartlett, Tracy Bhamra, Vicky Lofthouse and Rhoda Trimingham (2008) Refillable Glass Beverage Container Systems in the UK, report for WRAP, June 2008.*

The 2006 Beer Store Annual Report 2006 included LCAs provided by the Canadian Government. The Beer Store, which is responsible for 75% of all beer sold in Ontario, reported that on average, each glass bottle is washed and refilled 12-15 times before being recycled. It calculates that re-using refillable glass bottles an average 15 times results in an overall energy saving of 2.4 million GJ and the avoidance of nearly 160,000 tonnes of greenhouse gases.[197]

A study undertaken for the EC in 2005 included a review of LCA studies that compared refillables against one-way systems. The report found that no type of packaging was clearly always better or always worse for the environment, irrespective of the assumptions used. The report concluded that the findings of these studies were highly dependent on the product supply system, return rates, transport distances, control mechanisms, incentives such as deposits and electricity generation methods.[198] The relationship between capture rate and transport distance was reviewed in detail and Table 18 shows the findings from the analysis. The analysis suggested refillables (glass) would be beneficial when the transport distance is below 100km and the capture rate is high, and one-way containers would be beneficial when transport distance is high and capture rates are low. It suggested that refillable PET bottles may be the best option for short distribution distances (<193km), taking external costs into account. For greater distances than that (from 800 km), one-way

[197] *Beer Store Annual Report 2006/0793*.

[198] Ecolas – Pira (2005) *Study on the Implementation of Directive 94/62/EC on Packaging and Packaging Waste and Options to Strengthen Prevention and Re-use of Packaging,* Final Report to the European Commission, Report 03/07884.

PET with 80% recycling – became preferable to refillable containers.[199] These comments, regarding capture rates, are not unimportant since the policies in place may determine the likelihood of achieving the specified capture rate. Interestingly, few recycling schemes which make no resort to deposits will achieve an 80% capture of PET. This suggests that the answer to the question 'what is better for the environment?' is actually endogenous to the policies in place (in other words, which policies are in place will affect the answer). Other things being equal, because well designed DRSs tend to achieve high capture rates, the suggestion is that one of the key factors which would favour the use of returnables is indeed likely to be achieved under a DRS.

Table 18: Effects of Transport Distance and Capture Rate on Environmental Liability of Refillables v One-way Systems

| | | Transport Distance | | |
|---|---|---|---|---|
| | | Low (below 100km) | Medium (between 100km and 1,000km) | High (above 1,000km) |
| Capture Rate | High | Refillables | Inconclusive | Inconclusive |
| | Low | Inconclusive | Inconclusive | One-way |

*Source: Ecolas – Pira (2005) Study on the Implementation of Directive 94/62/EC on Packaging and Packaging Waste and Options to Strengthen Prevention and Re-use of Packaging, Final Report to the European Commission, Report 03/07884.*

In summarising the results of these studies, Lee et al state:[200]

> *The existing studies show that the environmental comparison of refillables versus one-way containers is not a simple one and that a number of factors need to be considered. Three key factors, cited in a number of the studies, are* ***transport distance, capture rate for reuse*** *and* ***collection rate for recycling.***

Another factor they do not mention is the number of uses per container, which is partly a function of capture rate, but also a function of the number of uses prior to breakage. This is usually a key factor in such studies (i.e. how many 'lives' does a container have before it enters the 'discard' phase).

The suggestion is that although the studies are not conclusive, the performance of DRSs is likely to shift matters in favour of the use of returnable packaging. The studies also suggest that some variables – such as transport distances – should be entitled to vary freely. Issues such as transport distance are, however, amenable to

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[199] RDC-Environment & Pira International (2003) *Evaluation of costs and benefits for the achievement of reuse and recycling targets for the different packaging materials in the frame of the packaging and packaging waste directive*, Report for the European Commission, March 2003

[200] Peter Lee, Paul Vaughan, Caroline Bartlett, Tracy Bhamra, Vicky Lofthouse and Rhoda Trimingham (2008) *Refillable Glass Beverage Container Systems in the UK*, report for WRAP, June 2008.

being influenced by policy. LCAs appear to treat this as an *exogenous* variable, yet clearly, it is not, and one might legitimately ask whether restriction of transport movements might not be a legitimate area for policy intervention. This important point was raised in the study by Ecolas-PIRA, where it was suggested that: '*This inevitably leads to debates on the proper balance' between Internal Market and environmental objectives, to which there is no simple answer.*'[201]

The actual outcomes of some deposit schemes are not always as intended. To the extent that the German scheme was designed to increase the market share of refillables, it has not obviously succeeded. Bevington notes:[202]

> *The mandatory 25 cent deposit on non-refillables is much higher than the commercially-determined deposits on refillables: 8 cents (glass) and 15 cents (PET). This was intended to encourage consumers to buy refillables, in order to protect the refill market. A perverse result of this has been that consumers who do not intend to return their empties buy refillables so they lose a smaller deposit. Thus the return rate for refillables has fallen.*
>
> *The deposit has failed to protect refillables, and their market share has fallen in all categories except beer since 2003 [...]The fact that beer is not commonly sold in PET, together with the conservative nature of the German beer market, explains why refill levels for beer have been maintained.*

This may also reflect the ease with which consumers can make use of the DSD system as an alternative route – free, to themselves, at the point of disposal – for ensuring an environmentally sound management of beverage containers, namely, recycling.

The above example does not necessarily imply that a scheme *could not be* designed where refillables were not encouraged. For example, if deposits were the same for refillable and non-refillable packaging, and a tax was placed on the non-refillables with a lower rate for refillables, then if households were also incentivised to reduce the use of recycling services through applying small charges on recycling services, this might do more to improve the incentive to use refillables rather than one-trip packaging. The combined effect of a primary material tax and a deposit – as is in place in Denmark – might reinforce the case for refillables.

### 9.3.2 Increasing Recycling

To the extent that re-use might not be the objective of a deposit-refund scheme, it becomes of interest to know whether deposit refund schemes can increase recycling.

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[201] Ecolas – Pira (2005) *Study on the Implementation of Directive 94/62/EC on Packaging and Packaging Waste and Options to Strengthen Prevention and Re-use of Packaging, Final Report to the European Commission*, Report 03/07884.

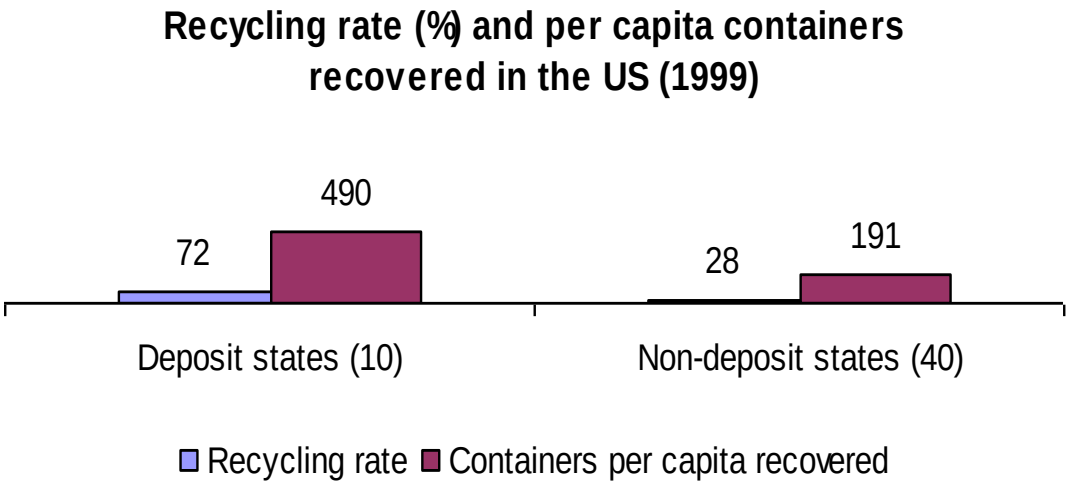
[202] G. Bevington (2008) *A Deposit and Refund Scheme in Ireland*, Report commissioned by Repak Ltd., September 2008.

#### 9.3.2.1 Effects of Deposit Schemes on Recycling Rates

Ideally, one has some indication of 'before' and 'after' performance, controlling for other variables. To some extent, this is made difficult by the absence of data of a usable form. Surprisingly few studies actually take this approach.

Some data allows for comparison of performance in areas with and without deposits, In the US, in 1999, the recycling performance of states with and without deposits in place is shown in Figure 33. The recycling rates, and the number of containers recovered per capita, were far higher in the deposit states.

Figure 33: Performance of US States with and Without Deposits, 1999




Also of interest is the performance of deposit schemes in the context of wider recycling systems. In Sweden, for example, the recycling rate for plastic packaging increased from 17% to 30% between 2003 and 2005 (44% in 2006). In the same period, recycling rates for PET under the deposit scheme were 77% to 82% (85% in 2007). This, in and of itself, might not prove much. The components of plastic packaging are many and varied, and PET bottles are readily recyclable. Perhaps more telling, however, is the performance in respect of metals. Metal packaging recycling rates were around 65% in 2004-2005, but the recycling rate for aluminium under the deposit system was 85% to 86% in the years 2002 to 2007. More impressive still is the return rate for glass bottles which is 99% on 33cl bottles and 90% on 50cl bottles. [203] In Denmark, return rates in 2007 were 84% for cans, 93% for plastic bottles and 91% for glass bottles.[204]

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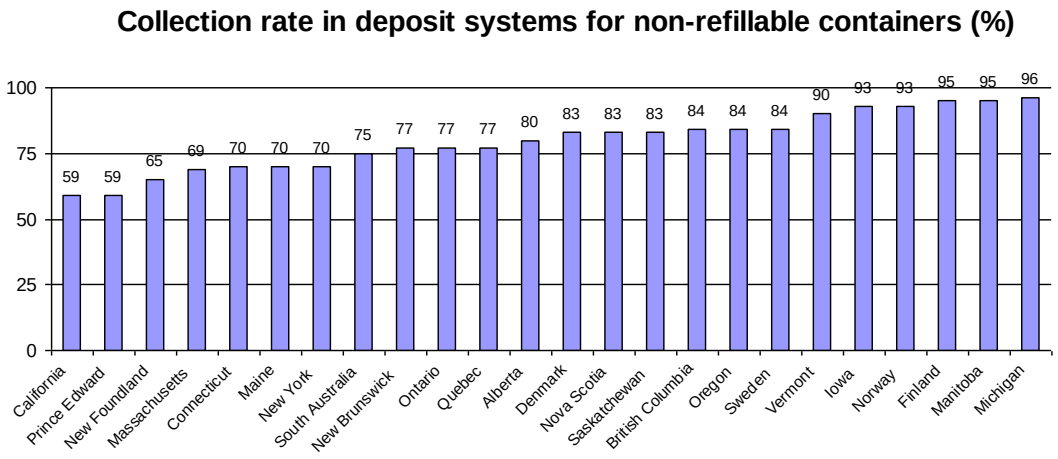
[203] http://www.sverigesbryggerier.se/eng/1-emballage/1-index.html, accessed January 2009.

[204] ERM (2008) *Review of Packaging Deposit Systems for the UK,* Report for DEFRA, December 2008, accessed from http://randd.defra.gov.uk/Document.aspx?Document=WR1203_7722_FRP.pdf

Similarly, in Germany, recycling rates in 2005 for plastics, tinplate, aluminium and glass were 50%, 85%, 76% and 79% respectively. The reported return rates under the deposit scheme are 95-98%, but these rates are not official.[205]

Figure 34 shows collection rates achieved in 2002 in deposit schemes. This shows that very few countries see low rates of return, with some jurisdictions achieving close to 100% return rates. As would be expected under economic theory, deposit schemes' return rates increase as the deposit increases, these leading to an enhanced incentive (see Figure 35). Figures for Denmark are shown in Figure 36.

Figure 34: Collection Rates for Non-refillable Containers in Deposit Systems, 2002

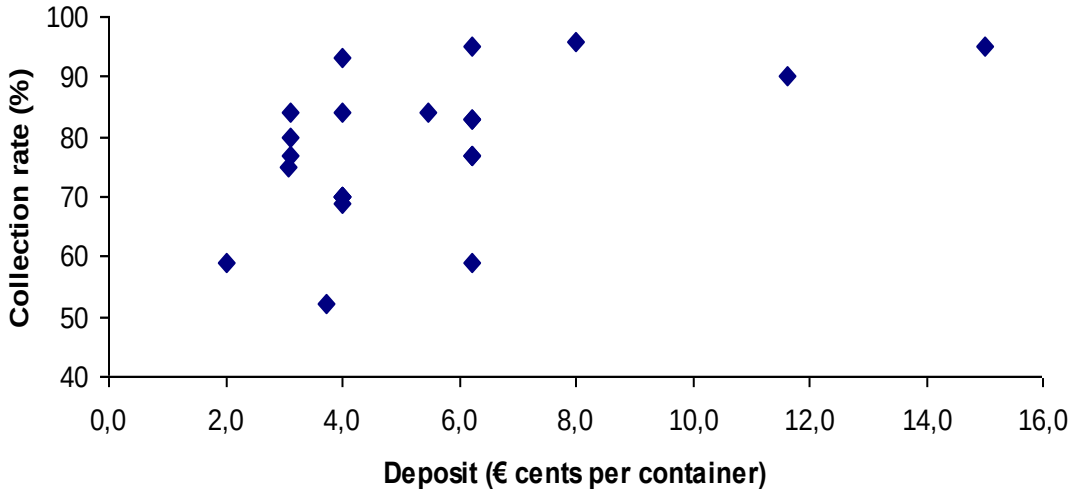



*Note: Figures based on data collected from system operators, data from 2002*

*Source: Wolfgang Ringel (2008) Introduction on Deposit, Scottish Government Litter Summit, Edinburgh 26th November 2008.*

[205] Wolfgang Ringel (2008) *The German Deposit System on One Way Beverage Packaging*, Presentation to the first Global Deposit Summit, Berlin 2008.

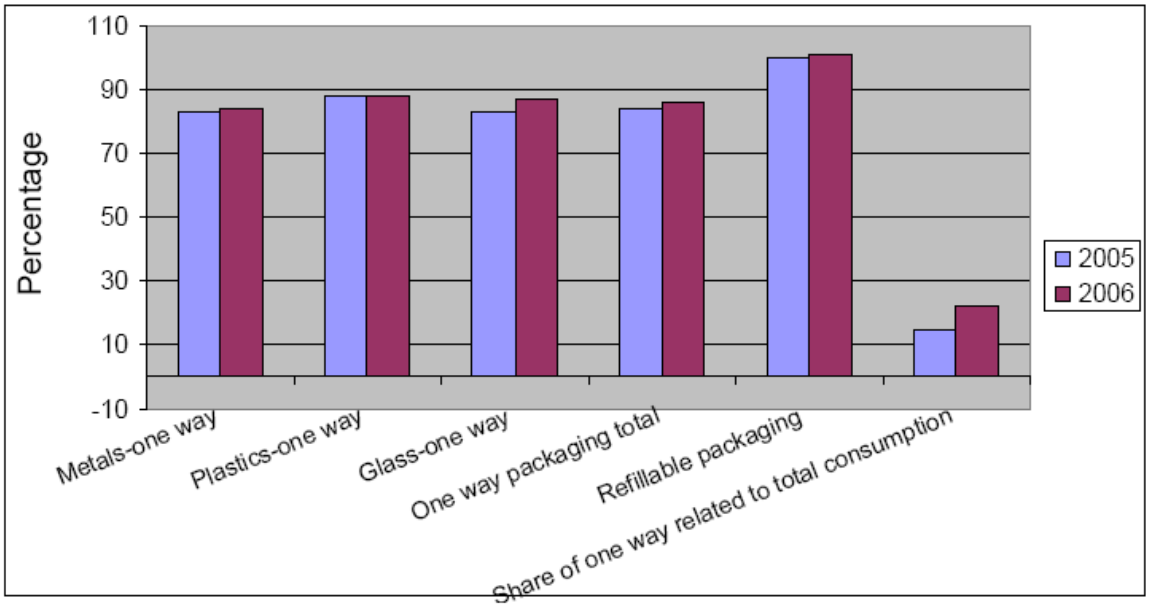
Figure 35: Relation Between Level of Deposit and Return Rate




*Source: System operators, Container Recycling Institute, Data from 1997-2002*

Taiwan differs from the European context, as the deposit refund scheme started without any other producer responsibility systems in place. The scheme therefore acted in isolation to increase recycling, first of PET and later with a number of other materials. As described earlier, Taiwan now claims a 100% PET recycling rate, using its deposit-refund system.

Figure 36: Return Percentages of One Way & Refillable Beverages




*Source: Christian Fischer (2008) Producer Responsibility Schemes Versus Deposits and Taxes- Danish Experiences, PRO Europe Congress, 15 May 2008*

It is common to hear it suggested that it is not the case that recycling rates are higher under deposit schemes. However, those who suggest this usually do so on the basis of reviewing recycling rates for *all* packaging. For example, EUROPEN, argues:[206]

> *There are no compensating benefits with regard to an overall improvement in recycling performance. The Perchards report showed that overall recycling rates in Member States with deposit systems are not higher than those of comparable EU countries where there are no special arrangements for beverage containers.*

The problem with this appears to be that the Perchards report looked at rates for *all* packaging in seeking to make this point. Deposits do not apply to *all* packaging (a point that is frequently made to downplay the potential impact of deposits on total recycling rates, often by the same organisations who seek to downplay the potential effectiveness of deposit refunds by using statistics which cover *all* packaging).

Perchards themselves state:[207]

> *It is certainly true that deposit systems for non-refillable beverage containers can achieve higher recycling rates for the beverage containers affected than when these containers are handled through general recycling systems. However European experience shows that* ***deposit systems do not achieve a higher recycling rate for all packaging of a given material, because beverage containers represent too small a proportion of the total tonnage of that packaging material.***
>
> *Drinks containers typically represent only about 10% of all packaging and the recycling rate for beverage containers in general recycling systems is likely to be higher than the recycling rate for all packaging of the same materials.*

They then allude to the performance of Belgium in respect of the recycling of *all* packaging even though this is clearly not a good comparator for reasons which the previous extract makes clear (the targeted materials – beverage containers – are a relatively small fraction of packaging). In particular, the largest fraction of the packaging stream is *always* paper and card, which is also an easy, and relatively low cost, material to recycle. Consequently, in most countries, the packaging recycling rate will be heavily influenced by capture of a material that is irrelevant to any sensible discussion regarding deposit refund schemes.

This is not to deny the possibility of high recycling rates of packaging being achieved without deposit refund schemes. Other EU countries have achieved impressive recycling performance without deposit-refund systems, such as Belgium. However, for some countries, the quality of the reported data would appear to warrant closer scrutiny. One other advantage of DRSs is that the quality of the data tends to be

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[206] EUROPEN (2007) *Economic Instruments in Packaging and Packaging Waste Policy*, Brussels: EUROPEN.

[207] G. Bevington (2008) *A Deposit and Refund Scheme in Ireland*, Report commissioned by Repak Ltd., September 2008.

rather good precisely because financial transactions are associated with both the sale and the return of the packaging.

In the UK, Alupro, the aluminium industry's trade body, says 98% of English households have kerbside collections of aluminium cans, but capture rates can be anywhere between 30% and 70%.[208] The 'cans-only' recycling rate is estimated to be 52% in 2008.[209] Therefore, even with a 'free to the consumer' system (in terms of marginal cost), and a very widespread coverage, the capture rate is still much less than is seen in the deposit-refund scheme countries. This may be partly a reflection of the fact that 35% of aluminium cans are consumed away from home, in the workplace, and at sports, leisure and travel locations, according to Alupro. However, such a waste stream is one for which deposit refund schemes may be well suited to dealing with, not least since such containers are less likely to arise as litter where deposits are in place.

#### 9.3.2.2 Price-responsiveness of Behaviour

One of the crucial elements in the deposit model is the setting of the deposit itself. Figure 37 shows the return rate as a function of the deposit. The deposit is converted from the local currency of the deposit refund system to Euros using OECD Purchasing Power Parities from 2008.[210] This gives a better estimate of the value of the deposit than simply using the prevailing exchange rate.

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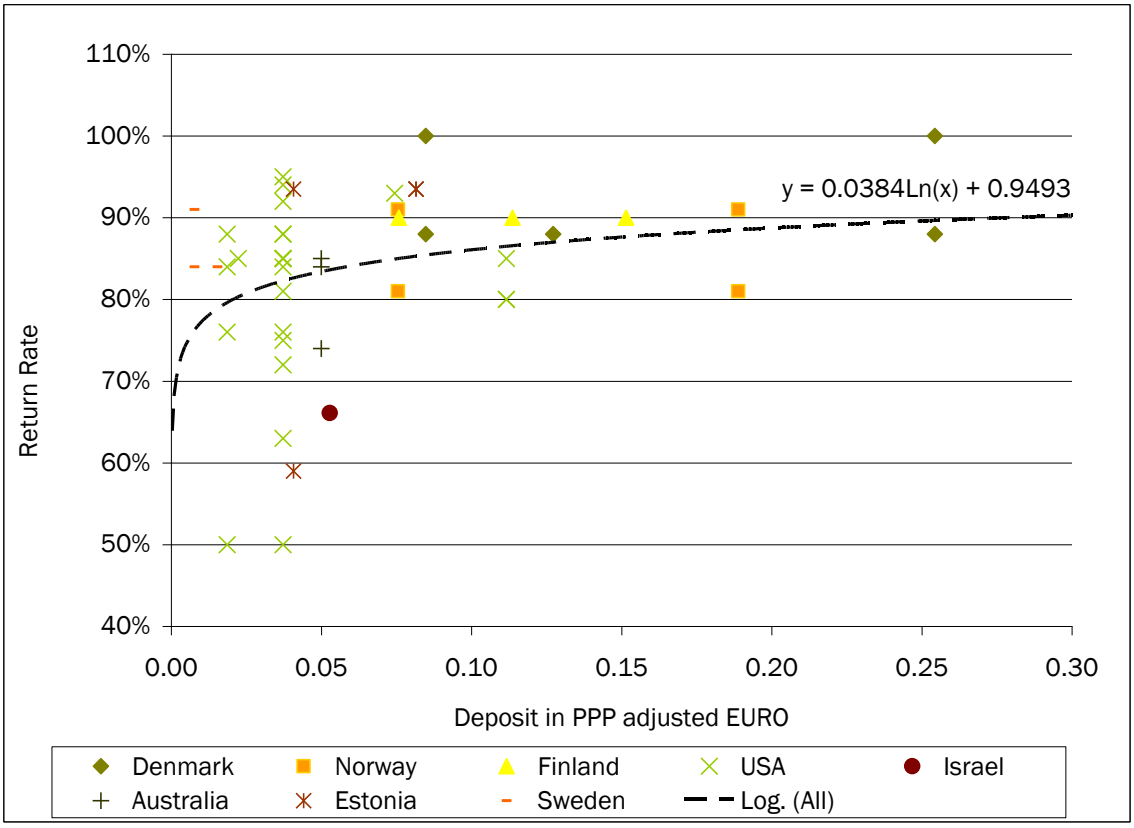
[208] Ends Report (2009) *Defra Report Rejects the case for Bottle Deposits*, January 2009 http://www.endsreport.com/index.cfm?action=report.article&articleID=20119&q=deposit%20refund&boolean_mode=all

[209] Alupro website, http://www.alupro.org.uk/facts%20and%20figures.htm, accessed May 2009.

[210] OECD (2010) Purchasing Power Parities (PPP), Accessed May 2010, http://www.oecd.org/department/0,3355,en_2649_34357_1_1_1_1_1,00.html

Figure 37: Return Rates as a Function of Deposits in PPP-Adjusted Spanish Euros




*Source: Eunomia*

### 9.3.3 Effects on Littering

There is evidence to suggest that deposit refund policies can reduce litter and even reduce the number of lacerations caused by glass in the environment.[211] Several one-way deposit systems were implemented with the clear objective of reducing littering (e.g. Sweden, British Columbia, California, Michigan and others). Hawaii is a more recent example of this trend. The potential for deposit systems to be effective in reducing littering has an intuitively plausible rationale - if the deposit is significant, then if the consumer does decide to litter, the possibility exists that someone else will pick up the container to redeem the deposit.

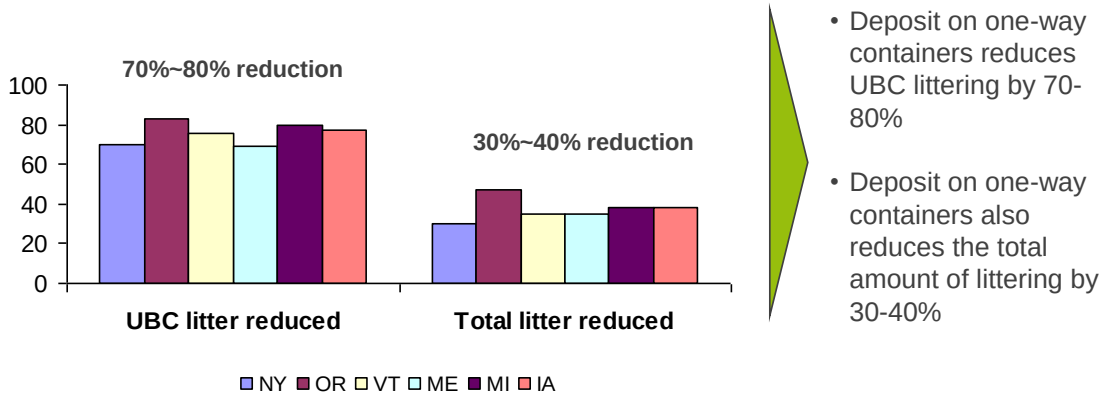
The Container Recycling Institute suggested significant reductions in littering following introduction of deposits in some US states (see Figure 38). The effects on used beverage containers (UBCs) and on total litter are shown as being between 70-80% and 30-40%, respectively. It must be said, however, that all studies of this nature suffer in terms of the lack of clarity about the metric used to measure the contribution

[211] M. Douglas Baker, MD, Sally E. Moore, and Paul H. Wise, MD, PhD, MPH, "The Impact of 'Bottle Bill' Legislation on the Incidence of Lacerations in Childhood", *American Journal of Public Health*, October 1986.

of beverage containers to total litter. It is not clear what the most relevant indicator should be (counts, volume, hazardousness, etc.) partly because no systematic studies have been carried out, to our knowledge, to understand the contribution of different attributes of litter to the disamenity experienced by those who experience litter. There is also the matter of cost to be considered since clean up of litter costs money. The effect of litter reduction on costs is considered below.

Figure 38: Reduction in Littering in US States Linked to Deposit Schemes

Reduction of littering in 6 US states after the introduction of container deposit systems.




Source: Container Recycling Institute, USA

Where supposed counter-arguments to the 'litter reduction' effect are put forward, these very rarely challenge the likely reality of this effect. Indeed, the counter-arguments tend to adopt the view that this effect is not significant because beverage containers constitute only a small proportion of litter. Even if one accepts the argument that this might be true, implicit in the counter-argument appears to be an assumption that if litter 'is there', then the amount of it is not a matter of any importance, or more specifically, that the reduction in the quantity of beverage packaging in litter is of no significance. Yet none of the literature actually offers any evidence to support this implied claim. The validity of the implied claim is also affected by the nature of the assumption (as highlighted above) concerning the metric used to measure 'litter'. What the right metric might be has not, as discussed above, been given adequate consideration by either advocates, or detractors, of the effects of deposit schemes.

In Ireland, Perchards argue that:[212]

> *The National Litter Survey for 2006 indicates that drinks containers (excluding cartons) represent 5.36% of total litter, with all packaging representing 13% of litter. This indicates that* ***a deposit could reduce the incidence of drinks containers***

[212] G. Bevington (2008) *A Deposit and Refund Scheme in Ireland*, Report commissioned by Repak Ltd., September 2008.

***in packaging, but it would have little impact on total litter****. Other litter surveys undertaken around the world have reached the same conclusion. Through Repak, Irish industry is already helping to combat litter, and it is unlikely that a deposit would result in significant cost savings for Irish local authorities on litter abatement activities.*

A problem with this analysis is that it assumes that the relevant indicator regarding litter is the measure used in the Litter Survey. This actually measures 'counts' of litter.

It could be argued that the disamenity effect of litter is a function more of its volume, and possibly, its potential to persist, than the number of items (i.e. the counts). In this respect, it is worth reporting that the 2007 National Litter Survey reports that the two most prominent items in litter in terms of counts are cigarette related litter, accounting for 46.7% of counts, and food related litter, accounting for 28% of counts. The majority of the two combined– though certainly requiring clean-up – are less visible cigarette ends and chewing gum. Chewing gum clearly has the potential to cause nuisance in its own particular way. After these categories, the component with the highest number of counts is packaging items, at 11.75%, of which around 5.64% - roughly half – were beverage containers.[213]

Given the relative insignificance – in volume terms – of chewing gum and cigarette ends, it might reasonably be considered that beverage containers could actually constitute a significant proportion of litter when considered in volume terms. As a result, one might argue that they are not as insignificant – in terms of their contribution to the disamenity associated with litter – as the count number would suggest if, as seems not entirely implausible, some of the litter-related disamenity experienced by communities relates to litter's visibility.

Furthermore, in terms of value of materials in the litter stream, their contribution may also be significant. It is notable that the contribution of beverage containers to the litter count associated with packaging is around 50%, even though in weight terms, such containers account for only 10% or so of packaging in the waste stream. In other words, where packaging is concerned, beverage containers appear to figure in a disproportionately significant manner within litter. Hence, whilst many argue that deposits only address a fraction of all packaging, their effect on litter may address a form of packaging which contributes disproportionately to the problem of litter.

In jurisdictions such as Hawaii, where the prevalence of beverage containers in litter has been a motivation for the scheme, the problem also extends to pollution of the sea. One report from the State of Hawaii shows how beverage containers have changed in terms of their prevalence in litter (debris) over time.[214] The data are shown in Table 19 and Table 20.

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[213] TOBIN (2008) *Litter Monitoring Body: System Results 2007*, The National Litter Pollution Monitoring System Survey, Report for DoEHLG, June 2008, http://www.environ.ie/en/Environment/Waste/LitterPollution/NationalLitterMonitoringSystem/PublicationsDocuments/FileDownLoad,18616,en.pdf

[214] State Of Hawaii Department Of Health (2008) *Pursuant To Sections 342g-102.5(H), 342g-114.5(B), And 342g-123, Hawaii Revised Statutes, Requiring The Department Of Health To Give A Report On The*

Table 19: Number of Debris Found During Cleanup

| Beverage Container Type | 2003 | 2004 | 2005 | 2006 | 2007 |
|---|---|---|---|---|---|
| Glass Bottles | 7,687 | 11,362 | 7,194 | 5,759 | 5,008 |
| Plastic Bottles | 5,246 | 5,215 | 3,824 | 4,799 | 2,965 |
| Metal Cans | 4,946 | 6,894 | 3,518 | 3,959 | 2,932 |
| **Total** | **17,879** | **23,471** | **14,430** | **14,517** | **10,905** |

*Source: State Of Hawaii Department Of Health (2008) Pursuant To Sections 342g-102.5(H), 342g-114.5(B), And 342g-123, Hawaii Revised Statutes, Requiring The Department Of Health To Give A Report On The Activities Of The Deposit Beverage Container Program, Report To The Twenty-Fifth Legislature State Of Hawaii 2009, November 2008*

Table 20: Percentage of Total Debris Collected During Cleanup

| Beverage Bottles & Cans | 2003 | 2004 | 2005 | 2006 | 2007 |
|---|---|---|---|---|---|
| **Glass, Metal, & Plastic** | 15.9% | 14.5% | 12.3% | 8.7% | 6.7% |

*Source: State Of Hawaii Department Of Health (2008) Pursuant To Sections 342g-102.5(H), 342g-114.5(B), And 342g-123, Hawaii Revised Statutes, Requiring The Department Of Health To Give A Report On The Activities Of The Deposit Beverage Container Program, Report To The Twenty-Fifth Legislature State Of Hawaii 2009, November 2008*

The report notes:

> *While there appears to be a downward trend in the number of bottles and cans found at beaches, beverage containers, along with associated caps and lids, continue to be a large portion of beach litter. This is why it is important to continue to place a deposit on beverage containers to decrease the temptation to litter and increase the incentive to recycle.*

A somewhat interesting feature of the Hawaii data is that they show that the problem is not simply land related. Indeed, beverage containers appear to be (relatively) more problematic in underwater cleanups (see Table 21).

Regarding plastics in particular, a UNEP report notes the prevalence of plastic bottles, caps and bags among the key forms of marine litter giving rise to increasingly serious problems at sea. Evidently, in the marine environment, it is the longevity and potential harm caused by plastics in the marine environment that makes them of particular concern.[215]

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*Activities Of The Deposit Beverage Container Program*, Report To The Twenty-Fifth Legislature State Of Hawaii 2009, November 2008.

[215] Ljubomir Jeftic, Seba Sheavly, and Ellik Adler (2009) *Marine Litter: A Global Challenge*, Report for UNEP, April 2009,

Table 21: Top 5 Debris Items Collected During the 2007 Cleanup

| Land Cleanups Only | Number of Debris Items | Percent of Total Collected |
|---|---|---|
| 1. Cigarettes & Filters | 72,053 | 44.7% |
| 2. Caps & Lids | 21,210 | 13.1% |
| 3. Food Wrappers and Containers | 16,554 | 10.3% |
| 4. Beverage Containers *(glass, metal, plastic)* | 10,505 | 6.5% |
| 5. Cups, Plates, and Utensils | 7,331 | 4.5% |
| **Underwater Cleanups Only** | **Number of Debris Items** | **Percent of Total Collected** |
| 1. Fishing Line | 1081 | 54% |
| 2. Beverage Containers *(glass, metal)* | 393 | 19.6% |
| 3. Cigarettes, Filters, & Cigar Tips | 248 | 12.3% |
| 4. Food Wrappers and Containers | 55 | 2.7% |
| 5. Caps & Lids | 39 | 1.9% |

*Source: State Of Hawaii Department Of Health (2008) Pursuant To Sections 342g-102.5(H), 342g-114.5(B), And 342g-123, Hawaii Revised Statutes, Requiring The Department Of Health To Give A Report On The Activities Of The Deposit Beverage Container Program, Report To The Twenty-Fifth Legislature State Of Hawaii 2009, November 2008*

Interesting evidence of the effects of deposits on littering comes from Denmark. In Denmark, there is a prominent cross-border trade in alcohol owing to the differences in excise duties between the countries. The Danish Society for Nature Conservation is the largest nature conservation and environmental organisation in Denmark. With the support of 140,000 members, they work to protect nature and the environment, and each year conduct litter clean-up campaigns. What is most intriguing about these campaigns is the proportion of littered cans which do not carry a deposit, because they are imported from Germany from areas specifically exempted from the German deposit system. A short summary of the main results concerning beverage cans since 2008 from the "Clean Up Denmark" campaigns is given below:

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http://www.unep.org/regionalseas/marinelitter/publications/docs/Marine_Litter_A_Global_Challenge.pdf

- 2008: 154,400 cans, of this only 7,800 with a paid Danish deposit;
- 2009: 153,000 cans, of this only 10,000 with a paid Danish deposit; and
- 2010: 197,000 cans, of this only 7,800 with a paid Danish deposit.

The data indicates that the vast majority of cans which are found in litter are those which bear no deposit. The suggestion appears to be that the deposit system has a significant bearing upon whether cans are littered or not. The Danish EPA notes that the majority of the machines receiving containers bearing the Danish deposit are also equipped to receive those which do not. Hence, the only differences between the German and Danish containers is that the Danish ones bear a deposit, which seems to act as a significant incentive to motivate return to the appropriate system. The absence of incentive in the case of German containers leads to greater littering,

A study undertaken in Australia suggested that deposit schemes were likely to be the most effective policy option for reducing litter amongst those considered for improving recycling:[216]

> *A national CDS [container deposit scheme] is expected to provide the greatest reduction in overall litter levels, with the potential to provide a 6% reduction in the total national litter count and a 19% reduction in the total national litter volume.*

Finally, there is another way in which removal of used beverage containers from litter could contribute to cleaner streets. To the extent that beverage containers are relatively voluminous items, then their removal from litter bins would leave more room for other waste. The CPRE's Litterbugs report reports that 91% of the public believe that increasing the number of bins is the most effective way of reducing litter.[217] An equivalent approach might be to free up space in existing bins. The report cites the New York bottle bill as reducing container litter by 70-80%. Clean-up costs, as well as landfill costs, were reduced. The scheme enjoys solid public support (84% of voters in 2004) and so has been extended in 2009 to cover non-carbonated drinks, which make up 27% of beverage sales.

## 9.4 Evasion and Enforcement

Deposit refund systems are effectively self-policing to a large extent, as there is an incentive to return the used product. However, in Denmark, one type of fraud that has been tried has been to copy the product code onto a non-returnable empty bottle (e.g. brought from Germany) so that the reverse vending machine (RVM) will accept the bottle and pay out the refund. Dansk Retursystem A/S has developed systems to prevent such fraud in Denmark.[218] Firstly, for refillable (more rigid) bottles the RVM

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[216] BDA Group (2009) *Beverage Container Investigation*, Report for the EPHC beverage Container Working Group, March 2009.

[217] A. Lewis, P. Turton and T. Sweetman (2009) *Litterbugs How to Deal with the Problem of Littering*, Report for CPRE, March 2009.

[218] Personal Communication with Anker Andersen a/s, 11 September 2008.

matches the shape with the product code before paying, and secondly, for one-way packaging, a special ink has been developed for the product code so that it is not possible to copy onto another (non-refundable) bottle.

The German system was particularly at risk of fraud given following conditions:

- The high deposit (25 Euro cents);
- Nine bordering countries;
- The anonymous environment of the reverse vending machine; and
- Not necessarily any differentiation of shape or weight between deposit and non-deposit containers.

Therefore, in addition to a bar code, a 'DPG' sticker is added and the combination of the two is used to recognise a valid deposit container.

## 9.5 Key Observations Regarding Effectiveness

DRS schemes are used, increasingly, to promote recycling as much as they seek to promote re-use. Indeed, the principle objective might be seen to be a high capture of the material being targeted. One of the issues facing Member States that seek to encourage the use of refillables is that such a scheme needs to respect the principles of the Single Market. Where schemes seek to promote use of refillables, then it seems likely that such schemes will be more open to legal challenges from fillers seeking to exploit economies of scale across borders in the Single Market. It is not impossible to promote the use of refillables, but it is not straightforward either. Evidently, such issues could be overcome were it to be decided that an EU-wide scheme was desirable, but this appears to be a remote possibility at present for political reasons.

Lessons from international implementations include:

- Deposit refund policies can encourage both reuse, and as such waste prevention, and recycling. They can incentivise this through encouraging the use of refillable container systems. The German system started by having lower deposits on refillables than on one-way containers. This does not appear to have been a sensible policy so we would recommend the same deposit on all packaging (both to ensure high return rates of all packaging, and to ensure that those not intending to return packaging are indifferent to the type of packaging used);
- Care has to be taken in designing deposit refund schemes in Europe such that they are proportionate in their effect, and do not effectively become trade barriers, or obstacles to the free movement of goods, in ways which are disproportionate relative to the environmental outcome being sought;
- The German experience has been that a mandatory deposit has not halted the overall decline in refillable and environmentally favourable containers. However, in the case of beer, the introduction of the deposit on disposable drinks packaging would appear to have had a waste prevention effect, with the proportion of reusable packaging for this type of drink increasing from 68% in 2002 to 89% in 2003. This level has fluctuated slightly but has broadly been

maintained, with figures for 2009 showing reusables accounting for 88.5%of packaging. The effect of this policy should not be considered, however, only in the 'absolute' sense. One has to consider also what might have happened in the absence of the policy. In other words, the policy might not have increased market shares of refillables markedly, but relative to the situation which might have developed in the absence of deposits, the effect may have been much more profound;

- Rates of take back can be extremely high, but as expected, this return rate is sensitive to the deposit rate;
- The quality of material captured under a deposit refund scheme tends to be high relative to what is obtained though kerbside recycling schemes, especially for PET.;
- There must be sufficient collection points to ensure a convenient end-consumer experience;
- Deposit refund schemes can reduce the prevalence of litter;
- Issues associated with cross-border purchases need to be considered in scheme design. Issues of interoperability of DRS / packaging systems can arise at borders, especially where excise duties for alcohol (or other drinks) are very different on the two sides of the same border;
- An appropriate labelling scheme or other system may be to deal with free riders; and
- Refillable containers may lead to greater distances being travelled for returning bottles, so *requiring* refillables to be used may lead to claims that the market is being structured in such a way as to place imported goods at a disadvantage.

One sensitivity that any state system in Europe needs to take into consideration is the issue of free trade. The OECD reports that deposit-refund systems will create barriers to trade:[219]

- If the initial deposits are high compared to the value of the goods;
- If foreign producers see that the costs of participating in a co-operative retrieval and recycling scheme are out of proportion to their market share;
- If non-refillable containers are an important condition for the competitiveness of imports;
- If they are applied only to certain types of containers or packaging which are primarily used for imported products; or
- If they are applied in a fashion which is discriminatory or which unduly favours domestic products.

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[219] OECD (1993) *Applying Economic Instruments to Packaging Waste: Practical Issues for Product Charges and Deposit Refund Systems*, Paris: OECD.

As well as governmental resolve to force industry to accept the extra costs that deposit refund schemes will entail, sufficient infrastructural provision is necessary to cope with increased / altered collection and reprocessing requirements. As stated above, this has also been a key concern within Europe:[220]

Another pre-requisite, partly because of some of the complaints which deposit refunds have drawn in respect of their implementation, is the political will to take the policy through to implementation. Schemes will require co-operation of various parties, including the public, so that engagement with the relevant parties is necessary.

[220] European Commission (2009) *Communication from the Commission on Beverage packaging, deposit systems and the free movement of goods*, C(2009) 3447 final Brussels 8th May 2009.

# 10.0 Packaging Taxes/Fees/Charges

There are two main instruments of interest under this heading:

1. The first are packaging taxes, which seek, usually, to internalise the environmental costs of packaging consumption within the price of packaging; and
2. Packaging fees, which arise as a result of the operation of producer responsibility schemes, which are effectively market determined, and which are usually linked both to the nature of targets set under producer responsibility, and the extent to which producers are made financially responsible for meeting these targets.

In principle, both could be assumed to have some form of effect on waste prevention.

The principle examples of packaging taxes in the EU are those of the Netherlands and those of Denmark. The case of the Dutch tax is a relatively recent one, and the tax is based upon the climate change impacts of the packaging concerned. It was introduced in January 2008, but was subsequently simplified in August 2008 to facilitate wider compliance and make planning payments easier. Focus has also been shifted from companies that specialize in packaging or that undertake packaging activities to those that supply packaging materials. The tax finances a Waste Fund, which is to be used to assist in the provision, at municipality level, of a separate collection of plastic packaging material from households. This means that the packaging tax both provides a financial incentive to reduce packaging waste generation, but also funds increased plastic packaging waste recycling by improving collection.

The Danish packaging tax covers beverage containers and other packaging. For beverage containers, rates are set according to the volume of the container, the nature of the material, and according to whether the container is refillable or not. For other packaging the tax is based upon weight and by material used, with the tax rate for each packaging material being based on the results of a cost-benefit analysis related to a life-cycle assessment. Taxes are also applied to non-reusable paper and plastic carrier bags with handles, single-use tableware and vending cups, and on specified PVC film packaging. The tax was reported to have cut plastic bag usage by 66%. Before the levy was introduced the Danes used over 700 million plastic bags each year, this was drastically reduced to 300 million (approx. 55% reduction) with the levy, but it increased to 450 million (approx. 35% reduction) in 2007.

In this section, we concentrate on the effects of producer responsibility schemes, recognising that packaging taxes are likely to influence, at the margin, decisions regarding packaging use, and the choice of packaging material.

## 10.1 Packaging Fees under Producer Responsibility

All EU Member States are required to transpose the Packaging Waste Directive into law.[221] The method of transposition differs from country to country.

Producer responsibility makes the companies that bring packaging to the market responsible for its collection and recycling, though the nature and extent of this responsibility varies from one country to another. All packaging, including retail packaging, such as plastic or paper bags and fast food wrappers can be included in the policy's scope. Occasionally, the policy also aims explicitly to enhance *both* prevention and recycling of waste packaging.

The companies who typically have some form of obligation under the policy may include those that:

- Package products;
- Sell packaged products;
- Import packaged products (including raw materials and parts);
- Bring packaged products into the market under their brand name;
- Provide customers with a carrier bag, other bag or box; or
- Produce carrier bags, other bags or boxes.

As discussed above, some countries make use of packaging taxes, notably, Denmark and the Netherlands. In most other EU Member States, one or more producer responsibility organisations have been established through which funds are directed from packaging producers to the organisations charged with recycling and recovery. The fee companies pay to the producer responsibility organisation is usually linked to the amount and type of packaging they introduce to the market and may also be linked to the costs of collecting, sorting and reprocessing the collected material, though this depends upon detailed implementation. The fee setting exercise is typically carried out annually and reflects the cost of the operation and takes into account the price received for the recyclates. In principle, the better quality and higher the market value of the recyclates, the cheaper the overall fee ought to become, reflecting the level of revenue generated by material sales.

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[221] Council Directive 94/62/EC on packaging and packaging waste (OJ No. L 365, 31.12.1994, p. 10) as amended by:

- Council Regulation (EC) No 1882/2003 adapting to Council Decision 1999/468/EC the provisions relating to committees which assist the Commission in the exercise of its implementing powers laid down in instruments subject to the procedure referred to in Article 251 of the EC Treaty (OJ No. L 284, 31.10.2003, p. 1.);
- Council Directive 2004/12/EC amending Directive 94/62/EC on packaging and packaging waste (OJ No. L 47, 18.2.2004, p. 26.); and
- Council Directive 2005/20/EC amending Directive 94/62/EC on packaging and packaging waste (OJ No. L 70, 16.3.2005, p.17).

Another key factor influencing the magnitude of the levies paid, and the level of achievement of the policy, is the level of the targets set as part of the policy.

From a waste prevention perspective, there are three ways in which producer responsibility for packaging can aid prevention:

1) Reduce the volume and weight of packaging used (e.g. light weighting and reducing excessive packaging);
2) Increase the share of reusable packaging in the packaging market (e.g. refillable beverage cans); and
3) Reduce the toxicity of materials used in packaging (for example, heavy metals in glass).

In respect of points 1) and 2) above, the ability of schemes to act as incentives for waste prevention is likely to be related to the nature of the economic incentives imparted by the scheme. In principle, the greater the proportion of the costs of recycling and recovery which are covered by the scheme, the greater will be the fees that producers will need to pay. In principle, this ought to give a greater incentive to prevent waste in the first place. Alternatively, if countries implement packaging taxes, then there will be a direct incentive to reduce consumption of packaging.

The following Sections review the way in which some different Member States implement producer responsibility with a specific emphasis on the implications for what producers are required to pay in order to discharge their obligations under the schemes concerned.

### 10.1.1 Spain

In Spain, household packaging recycling is the responsibility of two organisations, Ecovidrio and Ecoembes. Ecovidrio is the scheme for glass recycling, and Ecoembes is the scheme for recycling paper, plastics and metals.

Local authorities are responsible for putting in place infrastructure for the collection of packaging. This is based almost exclusively on the use of bring systems. Ecovidrio apparently offers no support to local authorities for the collection of glass, but Ecoembes does make payments to local authorities to support the collection of light packaging and cardboard. The basis for the payments is a calculation made by Ecoembes which is used as the basis for the financial support.

Importantly, there is some discussion as to how effective the existing scheme is. Household have little or no incentive to use the scheme since there are very few PAYT systems in Spain. The scheme, based as it is on bring recycling, is not especially convenient for residents. It is also difficult to prevent contamination of the recyclables. This means it is likely that the recycling rates are relatively low and that much of the target material is still to be found in residual waste. To the extent that producers pay a levy based upon the quantity of packaging material placed on the market, then the fact that only a small fraction is recycled implies that the payment per unit of packaging placed on the market does not have to be especially large to cover the costs of the scheme. It is difficult to know what fraction of the costs of the collection of packaging are covered by the payments to local authorities.

### 10.1.2 Ireland

Overall waste management strategy is developed by the Department of the Environment Heritage and Local Government (DoEHLG). Within the area of packaging waste, the DoEHLG are also the government body, which approves the application of Compliance Schemes/Compliance Bodies, of which to date only one has been approved in Ireland, namely Repak.

Repak, as the operator of the only approved compliance scheme, is responsible for the administration and demonstration of compliance for participating companies, through funding from packaging levies. Repak, sets a schedule of payments to incentivise the collection of packaging. These payments vary by material, and across the commercial and household packaging streams.

The Repak payments are not as significant in the commercial sector as they are in the household sector. The majority of packaging recycling has been associated with non-household waste, and here, the effect of rising landfill costs and improving prices for secondary materials are likely to have been at least as significant as the payments in driving performance.

As in Spain, there is no requirement for producers to pay for the management of the packaging that remains in residual waste. Also as in Spain, there is no real evidence as to what proportion of the costs of collecting, sorting and recycling packaging is covered by the incentives provided by the payments. Repak estimated this to be between 68-92% in 2009 but the basis for this was not clear.

### 10.1.3 Belgium

In Belgium the European Packaging Directive has been translated as a take-back obligation, with producers being responsible for meeting the targets. It is up to each producer to decide how to meet the targets, whether that is via their own collection, sorting and recycling systems or via an accredited organisation.

The implementation of the take-back obligation is realised through the establishment of an interregional secretariat (IVCIE), uniting the Flemish, Walloon and Brussels Regions of Belgium, and national industry bodies FOST-Plus and VAL-I-PAC. FOST-Plus organises the collection of household waste in collaboration with the intermunicipal utility companies that organise selective collection rounds for paper, and for the combined stream of plastic packaging, metal packaging and beverage cartons.

VAL-I-PAC organises mainly the data gathering on industrial packaging waste and creates incentives for companies to sort this kind of waste. The split between household and industrial waste is roughly 50/50.[222]

Companies are not obliged to join one or both organisations, but can instead opt to fulfil their take-back obligations individually. In this case they have to report directly to the IVCIE.

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[222] IVC activiteitenverslag (2007) *Annual report IVCIE*, Dutch.

Parties responsible for household packaging that join FOST-Plus sign an open ended agreement (which can be terminated each year) with FOST-Plus. They make a yearly declaration of the weight, type and quantity of packaging they put on the Belgian market. They then pay a fee depending on the material and the weight of that material placed onto the market. Simpler declaration systems are in place for smaller companies.[223]

Intermunicipalities (waste management services that cover a number of municipalities) are responsible for dealing with waste in their area and sign a standard five year agreement with FOST-Plus. They can either use their own collection infrastructure or contract it out to a waste management company (through a tender process managed by the intermunicipality and supervised by FOST-Plus). There is roughly a 50/50 split between those collecting themselves and those contracting it out. Intermunicipalities that choose not to collect the waste themselves consult the market using a standardised tender book, written by FOST-Plus. This ensures that the requirements and expectations of service delivery are equally met across Belgium. FOST-Plus and the intermunicipality make a joint decision as to which waste management company to use.[223]

Essential in the cooperation between FOST-Plus and the intermunicipal utility organisations is the idea that none of the costs for the collection of packaging waste should be borne by the public purse. Even if public infrastructure is used for the collection of the waste, the costs have to be paid by the producers of the packaging material through FOST-Plus. In order to guarantee this principle, the IVCIE ensures the application of realistic costs for the use of this infrastructure, e.g. for the use of civic amenity sites or public waste collection rounds.

In turn, as part of the agreement FOST-Plus provides the intermunicipalities with specifications for collection and sorting, including quality criteria. It also provides an administrative monitoring system (ProFost) which allows individual trucks to be monitored and the location of waste to be known at all times.[223]

The recyclers used are selected by FOST-Plus, again using a standardised tender book with detailed specifications. The selection of recyclers is supervised by a joint committee of the intermunicipalities, IVCIE and FOST-Plus. The price indexation is linked to the evolution of raw materials. The recycling process is verified by independent auditors.[223]

FOST-Plus therefore pays the intermunicipalities the full cost of collection and sorting as well as paying for communication with residents, follow up and quality-based bonuses, and the costs of managing the unrecycled packaging waste.

### 10.1.4 United Kingdom

The Producer Responsibility Obligations (Packaging Waste) Regulations 2005 are the UK Government's means of implementing the requirements of the Packaging Waste Directive. The current Regulations are a consolidated version of the original

[223] FOST-Plus (2007) *Annual report*, English version available.

regulations, which were amended a number of times. The regulations concentrate on important industrial and commercial sources (i.e. companies that handle more than 50 tonnes of packaging per annum and have a turnover of more than £2 million per annum) and apply the shared producer responsibility approach.

In the UK, considerable time was spent seeking to understand who, in the supply chain, should hold what level of obligation of packaging placed on the market. The obligation which a given enterprise must meet is a function of the quantity of packaging it handles, with the proportion of this quantity being affected by its position in the supply chain:

- Raw material manufacturer: 6%.
  Manufacturing of packaging raw materials. Example: Manufacturer of steel for baked beans cans.
- Converter: 9%.
  Manufacturing a recognised packaging item. Example: Manufacturer of the steel can for the baked beans.
- Packer/filler: 37%.
  Putting a product into packaging or applying packaging to a product. Example: the Company which fills the can with baked beans.
- Seller: 48%.
  Supplying the packaging to the end user of that packaging. Example: The supermarket which sells the baked bean can to the consumer. Additionally it may be noted that a wholesaler selling cans of beans bulked up into boxes would have an obligation for the boxes which are removed by the supermarket.
- Importer: rolled up obligation.
  Companies who directly import packaging, packed goods or packaging material are also obligated. The level of their obligation depends on the stage of the chain at which the packaging is brought into the UK.

Not all companies who handle packaging are obligated to recover and recycle packaging under the UK Regulation. 'Producers' are defined within the Regulation through Schedule 1. This sets thresholds for companies who are, or are not obligated to recover and recycled packaging under the Directive.

Under the packaging waste regulations, a producer is obliged to recover and recycle the packaging that is needed to discharge its obligation either through its own actions, or through joining a compliance scheme. In the latter case, which is followed by the vast majority of obligated entities, the compliance scheme then takes on the producer's obligation. There are currently 24 such compliance schemes in the UK.

A *de facto* system of tradable compliance credits helps drive compliance forward.[224] The nature of evidence which is generally used to demonstrate that the obligation of a company, or compliance scheme, has been discharged is the Packaging Recovery Note (and the Packaging Export Recovery Note). Accredited reprocessors are entitled

[224] This scheme was not initially designed explicitly as a tradable credit scheme.

to issue these when packaging waste is recycled or recovered, and they may sell them to obligated companies / compliance schemes. Essentially, if the market for 'evidence', in the form or PRNs / PERNs is tight (demand is strong relative to supply), then the value of PRNs / PERNs will be higher than in situations where it is well known that compliance is assured. It should be noted that PRNs / PERNs are material specific, and since the relevant targets are also, the way in which PRN / PERN prices move reflects the market for evidence of compliance for specific materials. It also reflects what happens 'at the margin'. In other words, if in a given year, compliance is certain to be secured, PRN/PERN values are very low. If, on the other hand, the market is tight, values increase, so that compliance costs are affected to a significant degree by what happens 'at the margin'. Equally, the overall costs of PRNs and PERNs to producers and compliance schemes bear limited relation to the overall costs of compliance, as represented by the costs of collecting, sorting and reprocessing packaging materials.

### 10.1.5 Summary

The above presentations highlight a range of circumstances in respect of the funding, by producers, of packaging recycling required under producer responsibility. The following examples can be discerned:

1. Full cost coverage – this approach, demonstrated by Belgium, ensures that producers' funding covers 100% of the costs incurred by local authorities in collecting packaging waste. Other examples are the situations in Austria and Germany. The Belgian system goes even further by demanding coverage of the costs of dealing with packaging found in residual waste too;
2. Partial cost coverage – the Irish and Spanish approaches typify this, albeit the mechanisms used are different. The schemes cover some, but not all, the costs of the local authority, and it seems reasonable to state that the level of cost coverage is not well known;
3. Limited cost coverage – the UK scheme effectively asks producers to pay for recycling 'at the margin'. There is no link between the costs of collection and recycling of packaging, and the amount that producers must pay to compliance schemes, in any given year.

Perchards sought to estimate the level of cost coverage by different schemes in 2005.[225] This data was updated in 2009 by Eunomia, and we present the latter below (see Table 22). This highlights the range in cost recovery across those EU-15 Member States which do not rely on taxes as the basis for their packaging policy. The level of cost coverage by schemes varies from around 5% to 100%.

This is reflected in part in the fees paid by packaging producers to producer responsibility organisations. However, a close look at these fees reveals very little by way of an obvious pattern. For example, the three countries with 100% cost coverage

[225] Perchards (2005) *Study On The Progress Of The Implementation And Impact Of Directive 94/62/Ec On The Functioning Of The Internal Market*, Final Report to the Europen Commission, May 2005.

– Austria, Belgium and Germany – charge radically different fees to their producers for each tonne of packaging material placed on the market. This is shown in Table 23, which shows these three countries at the top of the Table. Comparing the three countries, Belgian costs are far lower, suggesting that, notwithstanding the (probably) more favourable terrain over which packaging is collected, the Belgian scheme is relatively efficient.

Quite apart from highlighting potential inefficiencies in the collection schemes (the Belgian scheme delivers the highest reported rate of recycling of any EU Member State), this Table raises some interesting points for discussion. In principle, the greater the coverage of cost by the scheme, then if all systems were equally efficient, it would be expected that the costs to producers would be higher. This should give rise to stronger incentives for waste prevention. But the reality is different. Not all schemes are equally efficient. The more efficient is the scheme in terms of collection, the lower (other things being equal) will be the producer fee, so the waste prevention incentive will be weaker.

Table 22: Split of the Financial Burden of Collection and Sorting of Household Packaging

| Country | % of collection/sorting cost borne by recovery organisation | % of collection/ sorting cost borne by municipality / households | Comment |
|---|---|---|---|
| Belgium | 100 | 0 | Fost Plus organises and finances collection of glass (bring site); paper and card; plastic bottles and flasks, metallic packaging and drink cartons |
| Austria | 100 | 0 | ARA runs nationwide systems for the collection and recovery of all household and commercial packaging waste |
| Germany | 100 | 0 | Duales System Deutschland GmbH organises the collection, sorting and recycling of packaging waste in Germany with the support of 724 waste management partners |
| Luxembourg | unclear | unclear | Designed to cover all expenses of collecting, sorting and recycling packaging. Valorlux transfer funds per tonne to the municipalities based on tonnages handed over to registered recycling plants |
| Sweden | 95 | 5 | The producer responsibility organisation, REPA owns the national bring system, around 5,800 places. It contracts waste management companies for transportation and recycling. Note – beverage packaging handled through deposit refund scheme |
| UK | Estimated at 5% on average | Estimated at 95% on average | PRN revenues provide support for recycling market rather than direct support to municipalities. |
| Portugal | 65 | 35 | |
| Italy | 83 | 17 | The principle is that 100% of costs are covered by the PRO, but the estimated position is 83%[226] |
| France | 65 | 35 | Still deemed an accurate assessment in 2009[227] |
| Ireland | 68 - 92 | 8 - 32 | Repak estimates[228] |
| Spain | 65 | 35 | |
| Finland | 7 | 93 | Note that Finland also makes use of a deposit refund scheme for beverage packaging |
| Greece | 0 | 100 | |

*Source: Perchards (2005), and updates based on personal communications*

[226] Personal communication with Enzo Favoino of the Scuola Agraria del Parco di Monza, June 2009.

[227] Personal communication with Pascal Gislais, June 2009.

[228] Personal communication with Tony O'Sullivan, Repak, June 2009.

Table 23: Producer Responsibility Fees under Different Countries' Producer Responsibility Schemes (€/tonne of packaging on the market unless stated)

| Member State | Paper | Glass | Aluminium | Steel | Plastic | Wood |
|---|---|---|---|---|---|---|
| AT[2] | €120 | €71 | €450 | €270 | €670 | €14 |
| BE[2] | €17.60 | €18.40 | €137.90 | €37.60 | €199.40 | ? |
| DE[3] | €175 | €74 | ? | ? | €1,296 | ? |
| BG | €80 | €40 | €100 | €30 | €130 | €50 |
| CY[3] | €47.14 | €29.06 | €21.38 | €95.39 | €105.89 | ? |
| CZ[2] | €106.44 | €58.67 | €81.76 | €61.39 | €215.99 | €42.14 |
| DK | - | - | - | - | - | - |
| EE | €110 | €100 | €260 | €260 | €410 | €40 |
| FI | €23.50 | €10 | €21 | €3 | €21 | €0.40 |
| FR[1] | €163.30 | €4.80 | €60.60 | €30.20 | €237.80 | ? |
| GR | €52.50 | €10.90 | €8.80 | €21 | €66 | €9.50 |
| HU[3] | €32.20 | €18.50 | €24.40 | €12.50 | €77.40 | €25.50 |
| IE | €22.73 | €9.18 | €83.62 | €78.51 | €89.16 | €10.60 |
| IT | €22 | €17.82 | €52 | €31 | €140 | €8 |
| LV | €16 | €49 | €68 | €68 | €133 | €16 |
| LT | €59.22 | €260.93 | €112.82 | €112.82 | €310.68 | ? |
| LU[3] | €37.70 | €25.60 | €148.50 | €22.50 | €343.20 | €13.80 |
| MT[3] | ? | ? | ? | ? | ? | ? |
| NL | €64.10 | €45.60 | €573.10 | €112.60 | €355.40 | €22.80 |
| PL | €150 | €40 | €300 | ? | €600 | €80 |
| PT | €86.30 | €18.30 | €164.40 | €96 | €228.20 | ? |
| RO[3] | €8.37 | €10.49 | €7.26 | €7.26 | €20.54 | €4.58 |
| SE | €58.23 | ? | €282.18 | €282.18 | €153.39 | ? |
| SK[3] | €12.50 | €12.50 | €27.50 | €27.50 | €45 | ? |
| SI | €87 | €38 | €79 | €79 | €112 | €57 |
| ES | €68 | €0.0028 per unit + €0.0197 per kg | €102 | €85 | €377-472 | €21 |

Note that this assumes that producers themselves do benefit from their efforts to reduce packaging use. This seems a reasonable assumption given that most schemes appear to spread their funding requirement across all units of packaging placed on the market.

Sadly, there is relatively little evidence to support our hypotheses. On balance, however, the high fees in Germany would suggest that evidence of prevention / slow growth in packaging waste would be apparent. Two different studies claim positive effects of the German system on waste prevention.

Prognos AG calculated an 18 % decrease between 1991 and 2000 (1.6 Million tonnes per annum in 2000) of packaging material in Germany caused by the DSD system.[229] The figure was derived from a hypothetical calculation of the packaging

[229] *Assessment of Sustainability and the Perspectives of the DSD*, Prognos AG, commissioned by the Duales System Deutschland AG, Düsseldorf, June 2002.

material developments with, and without, the existence of the DSD. Prognos AG concluded that there had been a decoupling of the amount of packaging material from GDP growth in Germany.

The second study was carried out by the Öko-Institut commissioned by the DSD.[230] The results of the work showed a 4 % absolute decrease of packaging material in Germany over the period of time 1990 – 1999. The Öko-Institut pointed out, that this decline followed a long period of continuous increases in packaging material and packaging waste. Furthermore the Netherlands (without any system of fees) showed an increase of packaging material (15 – 20%) over the same time period. The Öko-Institut assumed that the relative effect (against the growth in the Netherlands) of the DSD on waste prevention could be as high as 25 %. However, the report suggested deeper investigations of waste prevention and the effects of DSD would be required to obtain a clearer picture. The task of isolating the effects of the packaging waste fees on waste prevention is not trivial because various other developments and influences (GDP growth, trends in the packaging sector such as usage of PET instead of glass, light weighting of packaging) must be taken into account.

We previously reviewed – in a tentative manner - evidence of packaging waste reduction by reviewing countries where fees were highest, and investigating the per capita packaging waste arisings. The picture for all packaging is shown in Figure 39. It is likely that this is somewhat affected by differing perceptions regarding reporting requirements as well as different methodologies for reporting of data. Whilst Belgium, Austria and Sweden have a smaller waste generation per capita than the EU15 average, Germany and Luxembourg do not (despite the high charges per tonne of packaging in Germany).[231] Other reports (see above), suggest that in an earlier period (from 1991 to 2000), there was a waste prevention impact. The packaging waste generation per capita also reflects the wealth and consumption habits of the country. Unfortunately the household/non-household split of packaging waste was not obtainable for all countries, but this would certainly help shed additional light on the situation. Another complicating factor might be the presence of re-use targets and deposit refund schemes in some countries.

Another indicator of the success, more generally, of packaging waste schemes is the quantity of residual packaging waste per inhabitant. This is shown in Figure 40, alongside the packaging waste per inhabitant. One interesting feature of this graphic, with countries ordered from left to right, in descending order of their residual waste per inhabitant, is the position of the countries with 100% coverage of costs. Two of these, Austria and Belgium, are to be found to the right hand end of the figure, indicating a low quantity of residual packaging waste per inhabitant. For the most part, they sit among countries with much lower per capita GDP, and much lower overall quantities of packaging waste per inhabitant. The same is true, though to a

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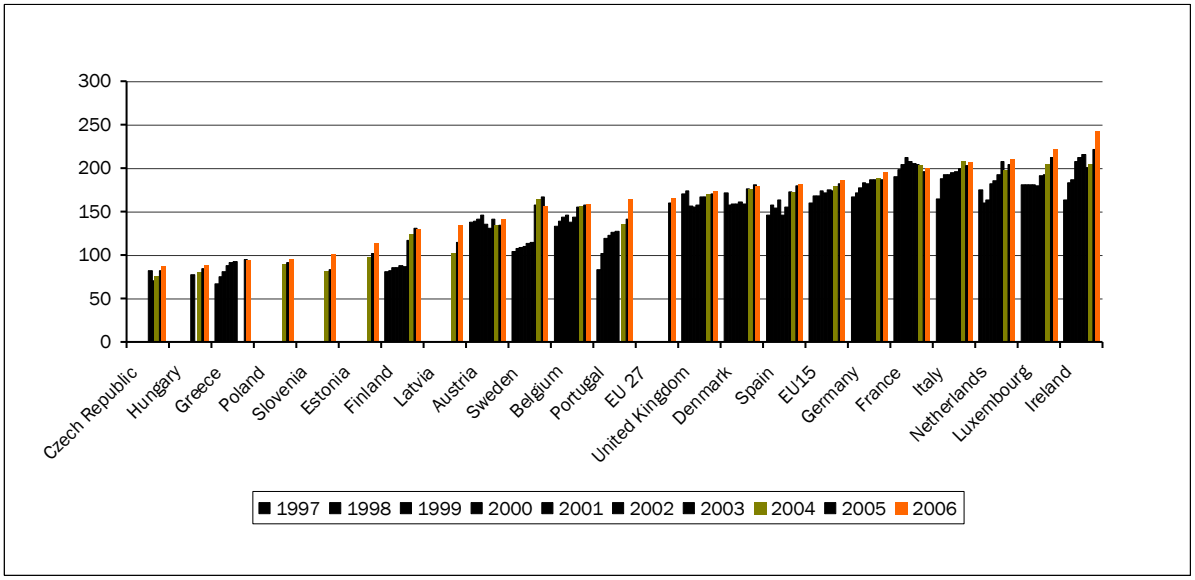
[230] *Advantage of the Green Dot for the Environment*, Öko-Institut, commissioned by the Duales System Deutschland AG, Düsseldorf, March 2002.

[231] European Environment Agency website, accessed January 2009, http://dataservice.eea.europa.eu/atlas/viewdata/viewpub.asp?id=2696

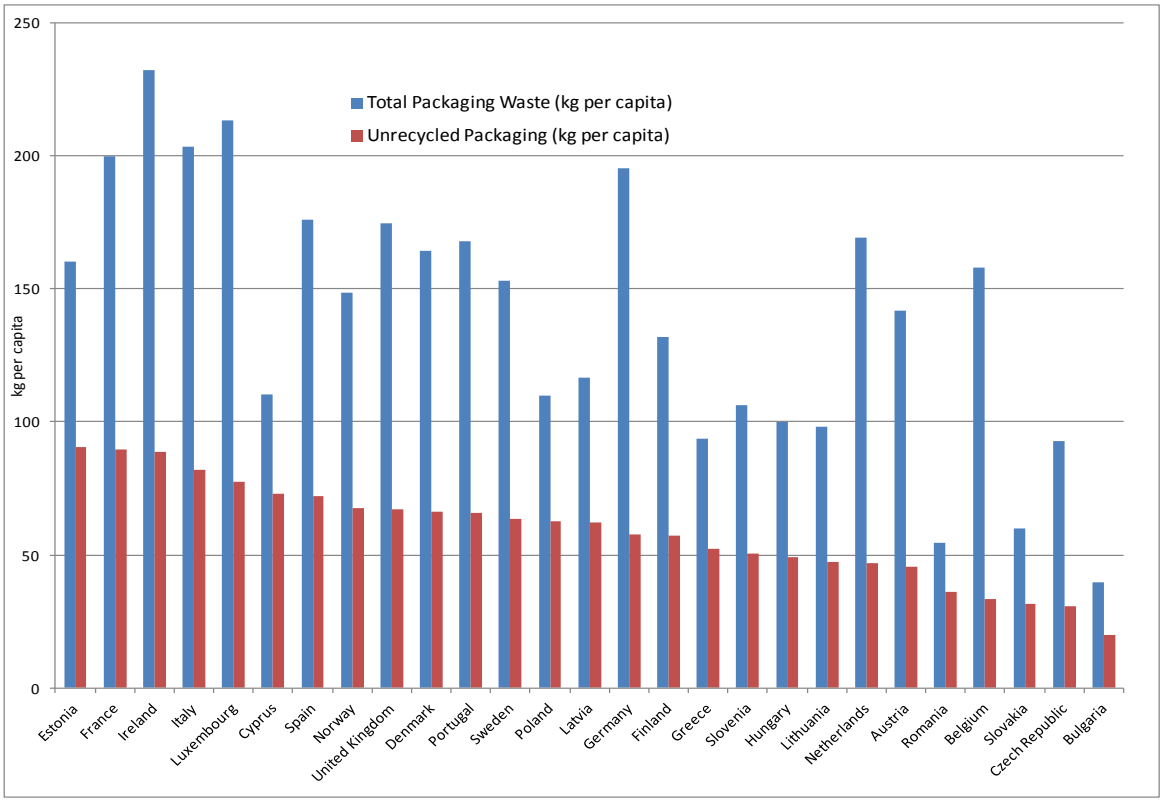
less dramatic extent, for Germany. The suggestion is that, whatever effect a full cost coverage scheme may have on waste prevention, the approach has the effect of reducing the quantity of residual packaging waste per inhabitant. This indicator reflects performance both in respect of waste prevention, re-use and recycling.

Figure 39: Packaging Waste Generation (kg per capita)




*Source: European Environment Agency, accessed January 2009*

Figure 40: Packaging Waste per Capita, and Unrecycled / Unrecovered Quantities




*Source: Eurostat,2008.*

## 10.2 Key Observations Regarding Effectiveness

If the policy is aimed towards household packaging, as for example in Germany, then the overall cost of recycling packaging tends to be higher, whereas if industry can choose the source of recycling to meet its targets (such as in the UK) then the cheapest sources will be addressed first. As recycling targets are increased then the cheaper – commercial and industrial sectors – will have to be supplemented by household recycling, and the higher cost of collection will increase the overall cost (which will then be passed onto the consumer).

To the extent that one seeks to ensure an efficient, yet environmentally effective scheme for dealing with packaging, then:

1. Targets need to be ambitious (so that not only the 'low hanging fruit' is captured);
2. Producers ought to cover the full costs of meeting the targets, as well as picking up any additional costs of dealing with that part of the packaging waste stream which remains in residual waste;
3. Producers ought to have some involvement in the design of collection schemes and their procurement (they should be involved at the procurement stage) so as to ensure that they are supporting cost-effective schemes;
4. Linked to the above point, it makes sense for the producers to carry the risks and rewards associated with material revenues;
5. For households, the capture for recycling of dry recyclables will be promoted by DVR schemes, but there may be some merit in still ensuring a non-zero variable rate charge for the recycling service to encourage waste prevention; and
6. There is no reason why schemes should not also be supported by packaging taxes. The previous recommendations merely allude to the desirability of a particular distribution of costs deemed necessary for achieving a given level of, typically, recycling performance. Someone has to pay for this, and in the UK system, the consequence of producers *not* paying much is that taxpayers pay the majority. The aims of packaging taxes might be more explicitly expressed in terms of reducing packaging consumption (or influencing the mix of materials used). There seem to be no reasons why taxes should not be used alongside the types of scheme being discussed.

Other policies which support producer responsibility are policies which make the costs of dealing with residual waste relatively high (landfill / incineration levies and bans). In these cases, irrespective of the distribution of costs, there will be incentives to increase recycling and to prevent waste so as to avoid the high costs of residual waste treatment / disposal.

Note that there are obviously other areas for consideration in the design of producer responsibility schemes, but here, we have focused on the aspects which appear most strongly related to waste prevention.

# 11.0 Variable VAT Charge

A European Commission Working Paper, accompanying the 2010 Green Paper on the future of VAT, acknowledges that there are strong arguments to the effect that integrating environmental aspects into taxation can encourage consumers and producers to switch to more environmentally-favourable products through the effect they exert on price structures. The document states that:

> *'The current VAT system does not recognise this phenomenon. Broadening the objectives of VAT by linking it to sustainable consumption would be in line with the recent Commission initiatives aiming at achieving a resource efficient economy which is one of the flagship initiatives of the EU 2020 Strategy. Changing price structure can help to shift demand towards less polluting and more resource- and energy-efficient products.* [232]

While apparently attractive, there is no clear evidence of impacts relating the use of variable VAT charges for waste prevention. This in part reflects the scarcity of examples of the instrument's use for this purpose. Accordingly this section has a focus on the *potential* application of variable rates of VAT to encourage waste prevention.

## 11.1 Case Studies

There are only a few examples of variable VAT rates being used to promote waste prevention. Details of these are provided below:

- The Belgian ecotax policy included, for a short period, a reduced rate of VAT for re-usable packaging. Initially, beverage packaging and paper and cardboard were exempt from the tax on the condition that strict recycling targets were met by industry. [233] In 2003, the exemption for disposable beverage packaging came to an end and the concept of an eco-premium was introduced to generate a sufficient price differential between reusable and non-reusable packaging. VAT on re-usable packaging was subject to VAT of 6%, whilst that for one-way packaging was set at 21%. In 2005, the VAT differential was abolished and since 2006, all beverage packaging (including re-usable packaging) has been the subject of a levy. The impacts of the ecotax on

[232] European Commission (2010) Commission Staff Working Document: Accompanying document to the Green Paper on the future of VAT – Towards a simpler, more robust and efficient VAT system, COM(2010) 695, available at http://ec.europa.eu/taxation_customs/resources/documents/common/consultations/tax/future_vat/sec%282010%291455_en.pdf

[233] Gewone wet van 16 juli 1993 tot vervollediging van de federale staatsstructuur, Boek III Milieutaksen.

consumer behaviour remain unclear, and to our knowledge there has been no specific evaluation of the effects of the VAT reduction; [234] and

- In Belgium, a reduced rate of VAT is applied to 'recycle shops', which provide employment to low-skilled unemployed people. There is, however, no evidence as to the waste prevention impacts of this reduction. Moreover, it could be seen to distort competition between 'recycle shops' and commercial sellers of second-hand goods.[235]

## 11.2 Description of Possible Applications and Impacts

Within the scope of the existing VAT Directive, there is a certain amount of discretion given to Member States to support particular activities through reduced levels of VAT. Article 98 of the VAT Directive (2006/112/EC) permits Member States to apply either one or two reduced rates of VAT to supplies of goods or services as set out in Annex III. [236] Item 15 in Annex III is:

> *'supply of goods and services by organisations recognised as being devoted to social wellbeing by Member States and engaged in welfare or social security work'.*

This could include charitable organisations involved in the repair of items for their owners. Such organisations also engage in preparation for reuse activities which are not, strictly speaking waste prevention, but facilitate re-use as a form of waste prevention.

The Directive also includes, in Annex 106, provisions for reduced rates for particular labour intensive services. [237] The services must meet the following conditions:

1. They must be labour-intensive;
2. They must largely be provided direct to final consumers; and
3. They must be mainly local and not likely to cause distortion of competition.

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[234] A. Heyerick, B. Mazijn and R. Doom (2003) Study in preparation of the evaluation of the federal environment-oriented production policy, Final report, Centrum voor duurzame ontwikkeling, University of Ghent, 2003.

[235] Institute for Environmental Studies (2009) *Economic Instruments and Waste Policies in the Netherlands: Inventory and Options for Extended Use*, Report for the Dutch Ministry of Housing, Physical Planning and the Environment, March 2009

[236] Council Directive 2006/112/EC of 28 November 2006 on the Common System of Value Added Tax, available at http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:2006:347:0001:0118:en:PDF

[237] This was originally a temporary provision until 31 December 2010, but was adopted on a permanent basis under Directive 2009/47/EC. See http://ec.europa.eu/taxation_customs/taxation/vat/key_documents/legislation_recently_adopted/index_en.htm

These reduced rates may be applied by Member States to services from no more than two of the categories set out in Annex IV of the Directive (or in exceptional cases, three categories). The list of categories in Annex IV includes:

1. Minor repairing of:
    a. Bicycles;
    b. Shoes and leather goods;
    c. Clothing and household linen (including mending and alteration);
2. Renovation and repairing of private dwellings, excluding materials which account for a significant part of the value of the service supplied; and
3. Window-cleaning and cleaning in private households.

What this seems to imply is that commercial organisations are limited in the extent to which they may benefit from being able to offer a lower rate of VAT on their services. Meanwhile, organisations focused on social wellbeing would appear to have a freer rein to offer reduced VAT over a wider range of services.

It is interesting to consider here the example of the commercial repair of domestic white goods, which can bring about a waste prevention effect to the extent that final disposal of the item in question is deferred. Such a service, provided by a commercial entity, is not included in the list at Annex IV. However, it would appear to meet the criteria outlined in Article 106:

1. It is labour intensive, with the bulk of the cost of repair typically relating to labour rather than parts;
2. The service is entirely provided direct to the final consumer; and
3. Domestic white goods repair is a market characterised by a large number of small firms operating in a distinct local area, and there would not appear to be any concerns about the distortion of competition.

While it is difficult to establish *ex ante* the extent to which such a reduction would lead to an increase in repair versus disposal, at the margin, it would, intuitively, have a positive impact.

The likely impacts of differential VAT rates for specific products with recognised environmental benefits, were considered in a 2008 study. This noted some key factors relating to the instrument's use including the following:[238]

- Reduced VAT rates are usually applied for non-environmental reasons, such as distributional concerns, 'public good' (or 'merit good') arguments, or employment generation. These may either coincide (e.g. public transport) or conflict (e.g. meat, domestic energy) with environmental motives;

[238] Institute for Environmental Studies (2008) *The Use of Differential VAT Rates to Promote Changes in Consumption and Innovation*, Report for the European Commission Directorate-General for Environment, June 2008, http://ec.europa.eu/environment/enveco/taxation/pdf/vat_final.pdf

- Application of reduced VAT rates requires a clear and unambiguous distinction between the qualifying 'green' products and their 'non-green' counterparts (e.g. on the basis of eco-labelling criteria);
- In competitive markets, VAT reduction is likely to be passed through fully to consumers, with incomplete pass-through (or none at all) if markets are not competitive or the reduction is seen as temporary;
- VAT differentiation (if passed through fully) may reduce the price of 'greener' products by some 10 to 15 percent. This may not always be enough to bridge the gap with the 'less green' alternative;
- Data on the impact of relative price changes on the demand for 'green' and 'less green' products (cross price elasticities) are scarce. Moreover, such elasticities should be applied with caution. Demand responses may differ between price increases and decreases, and between large and small price changes; and
- There is evidence for the existence of a 'signalling effect': fiscal incentives, if properly communicated, tend to have an impact on consumer demand beyond the purely financial advantage they confer.

Taking the example again of white goods, the increased emphasis on energy efficiency over recent years has lead to dramatic reductions in use-phase energy consumption. However, the marginal improvements in efficiency are diminishing over time (as the greatest gains have already been made). Therefore it may be increasingly appropriate to focus attention on extending the lifespan of an individual item, in order to delay as far as is possible, the point at which it is discarded, and thus defer the required manufacture of a replacement item.

In order to encourage manufacturers to offer an extended warranty, the rate of VAT could be inversely linked to the duration of the warranty. There might for instance be a threshold, whereby if free warranties are of five years duration or over, the VAT rate will be reduced from the standard rate (e.g. 15%) to the reduced rate (e.g. 5%). Notice could then be given in advance that from specific dates, the threshold period would be extended to six, and then seven years.

Such an approach would tally well with the key points noted in the 2008 study for the European Commission on differential VAT rates, namely:

- All things being equal it would promote employment-intensive white goods repair;
- There would be a clear and unambiguous distinction between those goods that qualify, and those that don't, based on the duration of their warranty;
- To the extent that the avoided VAT is passed through, the consumer will receive a further incentive (beyond the enticement of the extended warranty itself); and
- Even if the direct fiscal stimulus is small, the 'signalling effect' associated with the reduced level of VAT may serve to enhance consumer demand.

Other possible applications could be for products and services that are intrinsically less waste-intensive, especially in situations where there is a close substitute that is

'waste-intensive', which would remain under the higher rate of VAT. Examples might include:

- Non-disposable razors versus disposable razors;
- Rechargeable versus non-rechargeable batteries; and
- Real (reusable) nappies as opposed to disposable ones.

Similarly where the purchase of products is replaced with the provision of a service, these could be supported by a reduction in VAT. Relevant examples could be:

- Car-clubs / car sharing schemes, where the purchase of a car may be displaced by the provision of the service of 'mobility'. There are a number of wider environmental benefits of such schemes, beyond waste prevention, but the important effect of reducing demand for the primary manufacture of vehicles will serve to reduce future levels of discarded vehicles;
- The provision of flooring as a service, where the incentive for the service provider is to maximise the lifespan of the items supplied, rather than to maximise product sales. Intrinsically less waste intensive, reduced VAT would act as a further stimulus to adoption of this service.

Much as there is a lack of data on the waste prevention impacts of existing reductions in VAT, it is difficult to estimate the results of reducing rates for the examples given above. Bearing in mind that in most cases the maximum reduction would be from a rate of 15% to 5%, such a change would lead to an 8.7% reduction in the purchase price of the good or service which may not be a significant incentive to change. For each product or service it would depend upon the extent to which similar products/services are substitutable, and if so, knowing the own price-elasticity of demand for the product/service, and the cross-price elasticity of demand.

However, as noted above, the 'signalling effect' may well increase the attractiveness of the VAT-reduced products or services.

## 11.3 Key Observations Regarding Effectiveness

Continuing with the example of white goods, it could be expected that the waste prevention potential would be enhanced if the reduced rate of VAT on sales of items with a longer warranty period were supported by a lower rate of VAT on commercial repair of domestic white goods. This would then mean that:

- For consumers, beyond the period covered by the warranty, repair of the item would be relatively more cost-effective than if a reduced rate of VAT on repairs were not available; and
- For producers, the cost of labour that would be covered under the terms of their warranty would be relatively cheaper than would otherwise be the case, thus making the longer warranties more cost-effective for them than would otherwise be the case.

More generally, the effectiveness of VAT reductions seem likely to be greatest where:

1. There are obvious alternatives to the product or service where differentials are being applied (so that the VAT rate plays the role of a differential tax);

2. The lower rate or exemption is applied to something where the environmental rationale for doing so is very clear;
3. The services being exempted are involved in prevention activity, and the alternatives (e.g. no repair) are subject to high VAT rates; and
4. The costs of disposal are high (this is likely to enhance commercial interest in leasing-type arrangements);

In addition, the introduction of direct and variable rate charging at the household level, would, at the margin, support the financial case for repair of an item rather than disposal.

Such a charging scheme, by weight or volume, would also support the household level financial case for reusable nappies rather than disposables, as they can constitute a significant proportion of a family's residual waste. The use of reusable razors, by contrast, would not receive such support from DVR charging due to the small size and low weight of the disposable products, and the relative infrequency with which they are discarded compared to nappies.

However, it is important to bear in mind the possible limitations of reductions in VAT, and consider whether direct fiscal incentives might actually be more desirable. A European Commission Working Paper on the role of fiscal instruments in environmental policy makes a number of observations. These largely relate to reduced VAT for energy efficient products, but some of the points could equally apply to VAT reductions for items with extended warranties:[239]

- Subsidy schemes can be better targeted to specific groups. This may help to alleviate the free rider problem, namely the fact that the benefit of a reduced VAT rate also goes to consumers who would purchase a product with an extended warranty in any case. Thus the same target could be achieved more cost-effectively through targeted subsidies. This point also applies to those who would have paid for the item to be repaired even without the reduced rate;
- Direct fiscal incentives are likely to be more visible to consumers and thus may have a stronger signalling effect than reduced VAT rates;
- Direct fiscal incentives would not probably create the risk of distorting cross-border trade in the same way as reduced VAT rates, if they are targeted only to the residents of a country;
- Subsidies delivered at the checkout or as income tax credits to consumers are more certain to reach the consumer than reduced VAT which may not be entirely passed through to retail prices;

[239] DG Taxation and Customs Union (TAXUD) (2009) Working Paper No.19, The Role of Fiscal Instruments in Environmental Policy, available at http://ec.europa.eu/taxation_customs/resources/documents/taxation/gen_info/economic_analysis/tax_papers/taxation_paper_19.pdf (accessed December 2011)

Direct subsidies can be calibrated to the product characteristics. Some products need higher subsidies than others to motivate consumers and reduced VAT may not sufficiently bridge the upfront price gap (which is the most relevant market failure for VAT to tackle).On the other hand, compared to reduced VAT rates, the creation of a subsidy scheme can be administratively more cumbersome than the differentiation of rates in an existing tax regime (VAT) and thus may entail higher adminstrative costs. A final point made by the authors is that direct fiscal incentives, unlike reduced VAT rates, belong to the sole competence of the EU Member States and therefore their use remains inevitably dispersed if the Member States do not co-ordinate their action in this regard.[240] The corollary of this, of course, is that direct fiscal incentives can most likely be implemented more swiftly.

While it appears that there is some potential for using reduced rates of VAT to stimulate waste prevention, there is clearly a need for further research to understand how this could be applied to best effect. Modelling the likely waste prevention impacts in a range of situations, and comparing these against the likely impacts of other measures, such as direct fiscal incentives, would be a useful next step.

[240] DG Taxation and Customs Union (TAXUD) (2009) Working Paper No.19, The Role of Fiscal Instruments in Environmental Policy, available at http://ec.europa.eu/taxation_customs/resources/documents/taxation/gen_info/economic_analysis/tax_papers/taxation_paper_19.pdf (accessed December 2011)

# 12.0 Summary of Findings

In general terms, studies reviewed have often been unable to unequivocally attribute evidence of waste prevention effects, or the size of any effects, to the specific economic instruments under consideration. This is typically due either to methodological shortcomings, or to the fact that other factors confound the analysis, or because the analysis has simply not been undertaken.

Nonetheless, from our review, the following observations can be made:

- DVR Charging Schemes provide the most compelling examples of waste prevention effects, and can themselves provide strong support to other economic instruments for waste prevention. Weight based systems appear to lead to the strongest waste prevention effect, but frequency based schemes also function well. DVR charging schemes can be adopted at the local authority level, though some regions / countries take a lead by requiring, or encouraging, their use. In the UK, these schemes are actively discouraged at present, which is a questionable position given the requirement to enshrine the waste hierarchy in policy and law;
- Product Taxes/Fees/Charges, in the form of plastic bag levies that were evaluated for this study, have been shown to be effective. However, care needs to be taken to ensure that the value of the tax is adjusted from time to time to counter the effect of inflation. Such instruments would probably have to be implemented at the national, or EU-wide, level rather than at local authority or regional level;
- Subsidies for Products, in the form of subsidies for 'real' nappies, demonstrate wide variations in the level both of subsidy and uptake, and there appears to be no clearly established relationship to link the level of subsidy to the rate of uptake. Moreover, there is a lack of evidence on the rate at which participants revert to the use of disposables. Such schemes are typically undertaken at the local authority level, and the likelihood of waste prevention impacts, and the cost-effectiveness of any impacts, is likely to depend on local circumstances:
    - For example in a warm, dry area where people have large gardens, washing and drying nappies may be easier, and much cheaper, than in a cold, wet location where residents have little or no outdoor space. Accordingly where conditions are more suitable, all things being equal, the initial subsidy would not need to be so high to encourage uptake, and there would be a lower inclination to revert to the use of disposables;
    - The avoided cost of disposal to the household would also be a key factor in the initial decision to adopt reusable nappies, and importantly, in encouraging the continued use of reusables. Hence DVR charging would be an appropriate supporting measure;
- Deposit-refund systems for beverage containers, while promoting reuse in some cases, are more typically used to encourage high rates of return of good

quality material for recycling. The extent to which they can be said to have a waste prevention effect depends to a large extent on the effect attributed to them in respect of refillables. The effect of deposit refunds needs to be considered against what might have happened in the absence of the policy. In other words, the policy itself might not have increased market shares of refillables markedly, but relative to the situation which might have developed in the absence of deposits, the effect is likely to have been much more profound. The effect attributed to the policy depends heavily, therefore, on what one believes to be the most likely counterfactual (i.e. without deposit) scenario. . No detailed analysis appears to have been done to consider the market share of refillables in countries with and without DRSs. There would appear to be potential for deposit-refunds to lead to waste prevention where DRSs are used in conjunction with packaging taxes, though the promotion of refillables typically raises questions regarding the acceptability of the measure in light of commitments to a Single Market. There may be scope for prevention where DRSs are applied to other product sectors, e.g. EEE. DRSs for beverage packaging do lead to a reduction in litter;

- Packaging Tax/Fee/Charges under producer responsibility obligations have not typically been shown to have waste prevention impacts, albeit a well implemented scheme may lead to a reduction in residual waste arisings. There is variation between schemes in terms of the proportion of the costs of collection covered by producers. In principle, one would expect that the greater the coverage of cost by the scheme, the costs to producers would be higher (if all schemes were equally efficient). This should give rise to stronger incentives for waste prevention. However, the reality is different. Not all schemes are equally efficient. The more efficient the scheme in terms of collection, the lower (other things being equal) will be the producer fee, so the waste prevention incentive will be weaker; and
- Variable Rate VAT, while not widely used for waste prevention purposes would appear to have some potential for application in this area, in further encouraging the repair of goods rather than their replacement, and in incentivising products and services that are intrinsically less waste-intensive. DVR charging would further support the aims of this mechanism.

## 12.1 DVR Charging Schemes

The available evidence suggests that there is a waste prevention effect associated with DVR charging systems, with the strength of the association potentially varying with the nature of the charging system. Depending on scheme types, and charge levels, the quantity of waste collected can fall by 10% and sometimes more, as with a number of the case studies that have been presented. Sack based (including composting) and weight-based schemes appear to give the strongest effects, although relatively few studies have sought to estimate price elasticities for weight-based schemes.

Price also plays a role, with a number of studies estimating the price-responsiveness of households and communities to charging systems. The majority highlights price responsive behaviour, with the responses being weakest in the cases where the

systems are based upon volume only. Only one study has estimated the elasticity for a frequency-based scheme.

Greater reductions tend to be achieved where the system in existence prior to charging included free garden waste collections, and where the charging system introduces charges for garden waste. This is because, we believe, this incentivises additional home composting / reduced generation of waste in the first place.

Indeed home composting is the key waste prevention measure observed. This happens most of all where:

- Charges are levied on biowaste sacks / bins as well as refuse sacks / bins. This is becoming a widespread phenomenon in Germany, Austria and Belgium; or
- The only biowaste collection is for kitchen waste, and where support is offered for home composting (this occurs in Italy).

The waste prevention effects of DVR charging schemes can be greatest where:

- The marginal benefit of avoided residual waste treatment / disposal is high. Charging systems will be more likely to ensure financial savings where the costs of landfilling / incineration are high, by which we mean, of the order €80/tonne at least;
- Separate collection (of biowastes and recyclable materials) includes a wide range of materials, and is convenient (typically kerbside collected rather than through bring systems) – this tends to limit the likelihood of illegal disposal / contamination of separately collected waste streams;
- Charge levels are set with a flat rate fixed fee supplemented by variable fees so as a) to ensure problems of revenue instability do not arise and b) to ensure variable rates are not so high they give rise to more compelling incentives to fly-tip;
- Charges are placed on residual waste taken to civic amenity sites as well as at the kerbside (so that waste does not simply move from one management route to another); and
- Charges are levied – albeit at different rates - on all waste streams, including recycling – this fully integrated approach is likely to deliver the strongest incentive for waste prevention.

Local and national political leadership is also important in enabling proposed schemes, which may encounter hostility from a sceptical public, to be properly implemented. In some countries, national or regional policy sets a clear and structured agenda for local waste charging. By contrast, despite significant investigation into the impact of charging for waste in the UK showing the benefits it would bring, this policy remains a political hot potato, with the coalition Government stripping away legislation which allowed local authorities to charge for waste.

It is also difficult to operate DVR systems without problems where the waste collection system is a completely open market. The more favourable circumstance is to have all households 'linked to' the collection system, and with some of the costs of the service supported through (obligatory) local taxation.

## 12.2 Product Taxes / Fees / Charges

Our analysis of impacts focused on taxes/fees/charges relates to plastic carrier bags, drawing in particular on the experience of the Irish plastic bag levy. This levy can be considered a success in terms of waste prevention impacts, and also in reducing the level of litter. However, the initial reductions were not sustained, in part due to the effect of inflation eroding the extent to which the levy acted as a deterrent.

The Irish Government's intention was to set a rate of tax which would act to change consumer behaviour. As such, the initial rate of tax was set at six times consumers' average maximum willingness to pay for the purchase of plastic bags.[241] This ensured that there was a marked decrease in the use of plastic bags in the short term, a trend which has been reversed slightly over the years. The per capita usage of plastic bags decreased from an estimated 328 to 21 plastic bags per capita per annum after the introduction of the tax. However, the results of the 2006 census indicated that plastic bag usage had risen to 32 bags per capita over the course of 2006. Consequently the levy was increased to €0.22 on 1st July 2007.[242]

Further proof of effectiveness of such instruments comes from Belgium, where under the "pic-nic tax" wholesalers are liable to pay a tax on various single-use items.[243] It has been reported that the tax on disposable plastic bags – set at €3.00 per kg – has had a marked impact in terms of reducing their use over recent years (decrease of 80% between 2003 and 2007, during this time the sale of reusable bags rose from 4.5million units in 2004 to 25.4million in 2007). However, it is also reported that despite the fact that *'the retail prices of disposable kitchen utensils, food wrap and aluminium foil have gone up substantially, the impact on consumption has been less marked'*.[244]

From our review it would appear that the following supporting actions may be necessary, or supportive when implementing taxes on single-use disposable products:

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[241] Convery, F., McDonnell, S. and Ferreira, S. (2007) The Most Popular Tax in Europe? Lessons from the Irish Plastic Bags Levy, *Environmental and Resource Economics*, September 2007, Vol. 38, No. 1, pp. 1-11

[242] Department of the Environment, Community and Local Government (2007) *Plastic Bags Levy to be Increased to 22c from 1 July 2007*, Press Release: 21/02/2007, Date Accessed: 19 September 2011, www.environ.ie/en/Environment/Waste/PlasticBags/News/MainBody,3199,en.htm

[243] These include the following taxes: €3.00 per kg of non-biodegradable disposable plastic carrier bags; €2.70 per kg for plastic food wrapping (product price increase of approximately 70%); €4.50 per kg for aluminium foil (product price increase of approximately 100%); €3.60 per kg of disposable kitchen utensils.

[244]Bruxelles Environment (2010) *Mapping Report on Waste Prevention Practices in Territories within EU27 - Pre-Waste: Improve the Effectiveness of Waste Prevention Policies in EU Territories*, October 2010, http://www.bruxellesenvironnement.be/uploadedFiles/Contenu_du_site/Professionnels/Formations_et_s%C3%A9minaires/Conf%C3%A9rence_Pre-waste_2011_(actes)/p3-%20prewaste-mapping-report.pdf

- Apply taxes to items where alternatives are clearly available (this is likely to ensure a reasonable response to the tax);
- Continual review of the tax to ensure that its effectiveness is not being eroded over time (e.g. through inflation);
- Ensure the tax is designed with sufficient inbuilt flexibility to adapt to changing economic conditions;
- Prior to introducing the tax, develop an effective communication campaign to advertise the rationale behind the tax. In this respect, there should be a clear rationale for the tax; and
- Albeit that this is desirable rather than necessary, it is helpful to be introducing such measures against the backdrop of a DVR charging for household waste. This can help strengthen the response to price changes occasioned by the tax.

## 12.3 Subsidies for Products

For waste prevention at the household level it was found that the most common subsidies included those for home composting schemes and reusable nappies. The former are well established in many countries across the world and a significant body of evidence exists which demonstrates the success of such initiatives. Although fairly widely practiced, financial support for the purchase of reusable nappies and laundry services has not been as widely studied, and was thus considered further in this report.

Through a variety of schemes a large number of local authorities have begun providing subsidies for reusable nappies, although there appears to be a paucity of data on the impacts of these schemes. Moreover, the data that exists is subject to a number of uncertainties.

The first area of uncertainty is the extent of the avoided waste from used disposable nappies and excreta that can be attributed to the use of reusable nappies. The second is that there have been no studies linking the level of the subsidy with the rate of uptake. Research finds that there are many behavioural barriers to using reusable nappies – for example, inconvenience, time for laundering, smell, storage issues etc. – and thus it is possible that many of the current subsidies on offer are insufficient to overcome these barriers.[245] Thirdly, even when uptake may be high, there is little evidence on the length of time for which participants continue to use reusables. It is conceivable that a proportion will revert to use of disposables over time, but no figures have been reported in relation to this.

Thus the evidence suggests that both the waste prevention impacts and (related) cost-effectiveness of subsidies for reusable nappies may be rather context-specific, varying perhaps by local authorities. Circumstances relating to housing type and

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[245] Brook Lyndhurst Ltd (2009) *Household Waste Prevention Evidence Review* , Report for the Department for Environment, Food and Rural Affairs, October 2009, http://randd.defra.gov.uk/Document.aspx?Document=WR1204_8365_FRP.pdf

prevailing weather conditions could affect householders' propensity to adopt, and continue to use, reusable nappies.

As has previously been noted, subsidies aimed at waste prevention only really become financially viable when the cost of disposal or treatment is raised to a level which will incentivise local authorities to encourage their residents to reduce their waste arisings. Thus, the presence of a suitably priced landfill/incineration tax will help create the necessary incentives to actively seek to promote waste prevention through household subsidies.

In addition, the introduction of direct and variable rate charging at the household level, would, at the margin, support the financial case for reusable rather than disposable nappies. This would also provide an on-going inducement for those who have taken the (one-off) subsidy to continue to use the reusable nappies.

All this having been said, for reasons of economic efficiency, where the products concerned are not deemed necessities (some form of nappy clearly is in modern society), it seems preferable to seek to shift consumption patterns not through subsidies, but through differential taxation (on products / services).

## 12.4 Deposit-Refund Systems for Beverage Containers

DRS schemes are used, increasingly, to promote recycling as much as they seek to promote re-use. Indeed, the principle objective might be seen to be a high capture of the material being targeted. With a switch towards one-way rather than refillable beverage containers, there is evidence of DRSs bringing about a reduction in littering, and leading to high levels of recycling. However, the evidence on waste prevention is less clear cut. Data relating to the German DRS suggests that across the board, the introduction of the compulsory deposit on one-way packaging has slowed, or at least deferred the decline in reusables. The notable exception to this is for beer, where an increase in the market share of reusables was observed at the time that the deposit was introduced. This increased market share has been sustained in subsequent years. The effect of the might be considered to have been more profound against a counterfactual 'no deposit' scenario.

One of the issues facing Member States that seek to encourage the use of refillables is that such a scheme needs to respect the principles of the Single Market. Where schemes seek to promote use of refillables, then it seems likely that such schemes will be more open to legal challenges from fillers seeking to exploit economies of scale across borders in the Single Market. It is not impossible to promote the use of refillables, but it is not straightforward either. Evidently, such issues could be overcome were it to be decided that an EU-wide scheme was desirable, but this appears to be a remote possibility at present for political reasons.

Key lessons from our review include:

- The German system started by having lower deposits on refillables than on one-way containers. This does not appear to have been a sensible policy so we would recommend the same deposit on all packaging (both to ensure high return rates of all packaging, and to ensure that those not intending to return packaging are indifferent to the type of packaging used);

- Care has to be taken in designing deposit refund schemes in Europe such that they are proportionate in their effect, and do not effectively become trade barriers, or obstacles to the free movement of goods, in ways which are disproportionate relative to the environmental outcome being sought;
- The German experience has been that a mandatory deposit has not halted the decline in refillable and environmentally favourable containers. At the same time, refillable packaging retains a significant market share, particularly for the packaging of beer;
- There must be sufficient collection points to ensure a convenient end-consumer experience; and
- Deposit refund schemes can reduce the prevalence of litter;

There is clear potential for deposit refunds to have waste prevention effects, and not just on beverage containers. One could consider the example of Electrical and Electronic Equipment (EEE). With rapid technological progress, such items may be considered obsolete by their users even though they are still functioning perfectly well. Many such items end up being stored in cupboards, recycled, or sent for disposal. A deposit refund system could be designed to provide a sufficient incentive to encourage the return of such items to a retailer. Obviously some may head for recycling, and others for preparation for reuse, neither of which are waste prevention. However, a proportion may be suitable directly for reuse, constituting waste prevention.

## 12.5 Packaging Tax/Fee/Charge

Our review concentrated on the effects of producer responsibility schemes, where companies that bring packaging to the market are made responsible for its collection and recycling.

From a waste prevention perspective, there are three ways in which producer responsibility for packaging can aid prevention:

- Reduce the volume and weight of packaging used (e.g. light weighting and reducing excessive packaging);
- Increase the share of reusable packaging in the packaging market (e.g. refillable beverage cans); and
- Reduce the toxicity of materials used in packaging (for example, heavy metals in glass).

In respect of points 1) and 2) above, the ability of schemes to act as incentives for waste prevention is likely to be related to the nature of the economic incentives imparted by the scheme. In principle, the greater the proportion of the costs of recycling and recovery which are covered by the scheme, the greater will be the fees that producers will need to pay. In principle, this ought to give a greater incentive to prevent waste in the first place.

However, from the examples reviewed, there does not appear to be a relationship between the costs of collection, the fees levied on producers, the level of recycling and the level of packaging waste generated. For example, the three countries with

100% of the costs of collection and recycling passed on to producers, – Austria, Belgium and Germany – charge radically different fees to their producers for each tonne of packaging material placed on the market. Belgian costs are far lower, suggesting that, notwithstanding the (probably) more favourable terrain over which packaging is collected, Belgium's scheme is relatively efficient.

In principle, the greater the coverage of cost by the scheme, then if all systems were equally efficient, it would be expected that the costs to producers would be higher. This should give rise to stronger incentives for waste prevention. But the reality is different. Not all schemes are equally efficient. The more efficient the scheme in terms of collection, the lower (other things being equal) will be the producer fee, so the waste prevention incentive will be weaker.

Two different studies claim positive effects of the German system on waste prevention (i.e. of reducing its amount and its hazardousness).

Prognos AG calculated an 18 % decrease between 1991 and 2000 (1.6 Million tonnes per annum in 2000) of packaging material in Germany caused by the DSD system.[246] The figure was created by a hypothetical calculation of the packaging material developments with, and without, the existence of the DSD. The experts of the Prognos AG underscored the decoupling of the amount of packaging material from GDP growth in Germany.

The second study was carried out by the Öko-Institut commissioned by the DSD.[247] The results of the in depth work showed a 4 % absolute decrease of packaging material in Germany over the period of time 1990 – 1999. The Öko-Institut pointed out, that this decline followed a long period of continuous increases in packaging material and packaging waste. Furthermore the Netherlands (without a system like DSD) showed an increase of packaging material (15 – 20%) over the same time period. The Öko-Institut assumed that the relative effect (against the growth in the Netherlands) of the DSD on waste prevention could be as high as 25 %. However, the experts of the Öko-Institut suggested deeper investigations of waste prevention and the effects of DSD would be required to obtain a clearer picture.

The task of isolating the effects of the DSD, and indeed any other form of producer responsibility for packaging on waste prevention is not trivial because various other developments and influences (GDP growth, trends in the packaging sector such as usage of PET instead of glass, light weighting of packaging) must be taken into account.

Accordingly, there is little direct evidence of producer responsibility for packaging leading to waste prevention impacts, though a suggestion that higher fees, other things being equal, may enhance prospects for waste prevention.

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[246] *Assessment of Sustainability and the Perspectives of the DSD*, Prognos AG, commissioned by the Duales System Deutschland AG, Düsseldorf, June 2002.

[247] *Advantage of the Green Dot for the Environment*, Öko-Institut, commissioned by the Duales System Deutschland AG, Düsseldorf, March 2002.

## 12.6 Variable VAT

While intuitively attractive, there is no clear evidence of impacts relating to the use of variable VAT charges for waste prevention. This in part reflects the scarcity of examples of the instrument's use for this purpose. Accordingly for this report we focused on the *potential* application of variable rates of VAT to encourage waste prevention.

Within the scope of the existing VAT Directive, there is a certain amount of discretion given to Member States to support particular activities through reduced levels of VAT. However, the Directive seems to imply that commercial organisations are limited in the extent to which they may benefit from being able to offer a lower rate of VAT on their services. By contrast, organisations focused on social wellbeing would appear to have a freer rein to offer reduced VAT over a wider range of services.

The example of white goods repair is given as a category of service which would seem to comply with the spirit of the Directive on a number of points. While it is difficult to establish *ex ante* the extent to which a reduction in VAT on white goods repair would lead to an increase in repair versus disposal, at the margin, it would, intuitively, have a positive impact on waste prevention.

Continuing with the example of white goods, the increased emphasis on energy efficiency over recent years has lead to dramatic reductions in use-phase energy consumption. However, the marginal improvements in efficiency are diminishing over time (as the greatest gains have already been made). Therefore it is increasingly appropriate to focus attention on extending the lifespan of an individual item, in order to delay as far as is possible, the point at which it is discarded, and thus defer the required manufacture of a replacement item.

In order to encourage manufacturers to offer an extended warranty, the rate of VAT could be inversely linked to the duration of the warranty. There might for instance be a threshold, whereby if free warranties are of five years duration or over, the VAT rate will be reduced from the standard rate (e.g. 15%) to the reduced rate (e.g. 5%). Notice could then be given in advance that from specific dates, the threshold period would be extended to six, and then seven years.

As a supporting instrument, the introduction of direct and variable rate charging at the household level, would, at the margin, support the financial case for repair of an item rather than disposal.

Other possible applications could be for products and services that are intrinsically less waste-intensive, especially in situations where there is a close substitute that is 'waste-intensive', which would remain under the higher rate of VAT. Examples might include:

- Non-disposable razors versus disposable razors;
- Rechargeable versus non-rechargeable batteries; and
- Real (reusable) nappies as opposed to disposable ones.

Similarly where the purchase of products is replaced with the provision of a service, these could be supported by a reduction in VAT. Relevant examples could be:

- Car-clubs / car sharing schemes, where the purchase of a car may be displaced by the provision of the service of 'mobility'. There are a number of wider environmental benefits of such schemes, beyond waste prevention, but the important effect of reducing demand for the primary manufacture of vehicles will serve to reduce future levels of discarded vehicles;
- The provision of flooring as a service, where the incentive for the service provider is to maximise the lifespan of the items supplied, rather than to maximise product sales. Intrinsically less waste intensive, reduced VAT would act as a further stimulus to adoption of this service.

However, a note of caution must be sounded that even if reduced rates of VAT may be effective, for certain applications, other approaches such as direct fiscal incentives may be more cost-effective. Further research is required to better understand the situations where one instrument would be preferable to the other.

# 13.0 Recommendations

Based on the foregoing analysis, the following recommendations have been made as to the future application of economic instruments for waste prevention.

## 13.1 Effective Economic Instruments for Waste Prevention

Of the instrument types considered above, the evidence of prevention effects is strongest for DVR charging, and product taxes (e.g. plastic bag tax). The former is an economic instrument that can be implemented at the local level, while the latter would have to be applied at the national level.

While the evidence is less clear for subsidies for products (e.g. reusable nappies), and non-existent due to a lack of examples for variable rate VAT, these two instruments also hold the promise of waste prevention effects. Again, the former is an economic instrument that can be implemented at the local level, and the desirability, or otherwise of doing so would be very much dependent upon the specific context. The latter, however would have to be applied at the national level, and would potentially involve agreement at the European level.

Given limited resources, the question then arises as to the prioritisation of such measures. The most appropriate first step would be, where it doesn't yet exist, to introduce DVR charging, for the following reasons:

- This is an action that local authorities can take independently (albeit this is not permitted in most of the UK);
- It encourages waste prevention across a range of material/product types;
- Sufficient experience of DVR among EU Member States and beyond means that lessons learned can effectively inform the development of new schemes, to avoid potential pitfalls; and
- DVR charging provides a solid foundation for the establishment of further economic instruments for waste prevention, through creating a supportive financial rationale for household level waste prevention.

With DVR charging in place, there is a strong incentive for households to prevent waste, but this will affect purchasing/use habits in different ways depending upon the item in question. For example, use of single-use plastic bags, being very lightweight and low volume (when compressed), would not be significantly deterred by the existence of DVR charging. By contrast, there would be a stronger incentive to avoid bulky or heavy packaging. In such cases, especially where there is a wide range of associated negative impacts (e.g. litter, including marine litter impacts from plastic bags) there is a strong justification for imposing a tax to internalise these external costs.

Intuitively, imposing DVR charging and then a number of carefully targeted product taxes/fees/charges is a more cost-effective approach than introducing product taxes/fees/charges alone. To achieve the same waste prevention effect, the number

of products selected would have to be high, with the associated cost of understanding their negative impacts in order to justify the level at which the tax were set.

Following the implementation of DVR charging and product tax/fee/charges, further product specific waste prevention impacts could be achieved through the provision of subsidies for products (e.g. reusable nappies, on a local authority specific basis) and the implementation of variable VAT charges.

Being targeted at specific items, in terms of tonnages of waste prevented, these two instruments would again be expected to be less effective than DVR charging, but they could perform a useful role. For subsidies, these could meet an identified local need if, and where, the circumstances suggest they would bring about further waste prevention and would be cost-effective.

For reductions in VAT, the overall effect on price reduction, and subsequent changes in consumption (dependent upon demand elasticities) might be somewhat limited, but the existence of a 'signalling effect' could enhance the waste prevention effect. Moreover, there could be some interesting interactions through aligning incentives both for product durability and repair rather than disposal through the VAT system.

## 13.2 Supporting Measures

DVR charging clearly supports the other recommended economic instruments through providing a clear financial incentive at the household level to prevent waste. However, it is worth noting that there is potentially an interesting dynamic at work in the order in which such economic instruments are implemented. While we recommend the prior establishment of DVR charging, this may reduce the marginal effectiveness of subsequent instruments, for quite understandable reasons.

To take the example of subsidies for reusable nappies, in the absence of DVR charging, the subsidy may lead to a 10% reduction in disposable nappy use among relevant households compared with the baseline situation (with no subsidy). It is entirely conceivable, however, that the introduction of a DVR scheme on its own could lead to a 20% reduction in disposable nappy use. Subsequent application of a subsidy, on top of a DVR charging scheme might only lead to a further 5% reduction in disposable nappy use. Notwithstanding this reduced marginal level of uptake, the DVR charging scheme would provide an important incentive for households to continue to avoid disposable nappies.

However, diminishing marginal impacts may not always occur, and it may on occasion be the case that impacts are wholly additional, or indeed greater than would have been the case in the absence of DVR charging. Much depends upon the technical possibilities for preventing the targeted wastes.

Finally, it is worth emphasising that DVR charging itself, as well as being a desirable means of enhancing the incentives provided by waste prevention measures, is dependent to some degree on there being a reasonably high cost of residual waste management. This enhances the cost savings associated with more sustainable waste management. In this sense, although rarely seen as major contributors to waste prevention at the household level, instruments such as landfill taxes, incineration taxes and restrictions on landfilling can play an instrumental role in

enhancing household waste prevention. The effect of landfill taxes on prevention of industrial wastes, in particular, is more clearly evident.

## 13.3 Further Research

In order to obtain a better understanding of the waste prevention impacts of future economic instruments, we recommend that wherever possible, full *ex ante* impact assessments take place. These will establish a likely baseline situation (in the absence of the instrument, and compare this with the expected impact of the instrument. The existence of *ex ante* assessments increases the accuracy, and therefore value, of *ex post* assessments, which themselves will better inform the development of future policy instruments.

There remains scope for additional cross-sectional analysis to understand whether DRSs and packaging charges have an effect of waste prevention. We noted that the prevention effect of DRSs is often considered in terms of whether market shares for refillables are maintained or not, but the effect of a DRS ought to be measured relative to a counterfactual where no DRS is implemented. Relatively little analysis has been undertaken in these areas, and further analysis seems warranted. The same could be said of subsidies for nappies, where, as reported in the analysis, few attempts have been made to relate the price incentive to the level of take up. That having been said, as mentioned above, subsidies for waste prevention are likely to be less preferable than differential tax rates, for reasons of economic (and resource) efficiency.

For levies on disposable bags, some further analysis of the growing number of instruments in place – in Denmark, Ireland, Italy, for example – would be desirable to understand the extent to which plastic bags are being substituted by other bags.

The modelling of the expected impacts of reductions in VAT on goods and services that are inherently less waste intensive should be undertaken. Such modelling would be an important step towards an improved understanding of the true potential of these instruments.

Finally, there remains a lack of knowledge as to the benefits of some of the measures concerned. This is especially true where the issue of litter is concerned. The disamenity associated with litter is poorly understood, and until this gap in our knowledge is resolved, the appropriate level of effort justified in reducing litter will remain a matter for somewhat dogmatic debate.