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1 EXECUTIVE SUMMARY

This report presents the findings of a follow-up study to the study to support the review of the Waste Thematic Strategy, which was completed in October 2010 and published on the DG Environment website in January 2011[1]. Within that study, a limited amount of research was undertaken on the use of economic instruments (EIs) to promote improved waste management, and the current study expands on that limited research. The objective of the present study is to analyse the relationship between the performances of the waste management systems of the EU Member States (MS) and their use of EIs. On the basis of this analysis, the opportunity of moving towards a European common approach for the use of EIs in relation to waste management is explored. The study aims to provide supporting information and analysis for the European Commission in the preparation of the follow up of the report published in January 2011 on the Thematic Strategy on the Prevention and Recycling of Waste.

Economic instruments addressed by the study

Rather than attempting to cover the whole landscape of EIs being used in the waste sector in the EU-27, a limited set of EIs have been addressed to enable more substantive research to be undertaken. The EIs being investigated were chosen after discussions with the Commission, and are amongst those where it was anticipated that waste management impacts could be most clearly seen and attributed to the use of EIs. The following EIs were studied:

1. Charges for waste disposal and treatment:
    a. Landfill taxes and fees (and restrictions/bans to provide context for the charges);
    b. Incineration taxes and fees (and restrictions/bans to provide context for the charges);
2. Pay-as-you-throw (PAYT) schemes; and
3. Producer responsibility schemes for specific waste streams (notably packaging, WEEE, ELV and batteries).

Landfill taxes

The study makes a distinction between landfill taxes (a levy charged by a public authority for the disposal of waste) and gate fees (a charge set by the operator of the landfill for the provision of the service). The sum of the tax and the gate fee represents the total charge for the disposal of waste in a landfill.

Eighteen MS currently have landfill taxes in place for the disposal of non-hazardous municipal waste sent to legal landfills (this will rise to 19 MS when a planned tax is introduced in LT in 2012). The level of taxation ranges very widely, from €3 per tonne in BG to up to €107.49 per tonne in NL. According to the data found during the study, the total typical charge for landfill (i.e. the tax plus the

[1] http://ec.europa.eu/environment/waste/strategy.htm

middle of the range of gate fees) to landfill one tonne of municipal waste in the EU ranges from €17.50 in LT to up to €155.50 in SE.

The analysis suggests that there is a relationship between higher landfill taxes (and higher total landfill charges) and lower percentages of municipal waste being sent to landfill. Three broad groups of MS emerge:

1. MS with high total charges for landfill and low percentages of municipal waste landfilled (AT, BE, DE, DK, LU, NL, SE);
2. MS with mid- to high-range total charges and mid-range percentages landfilled (FI, FR, IE, IT, SI, UK); and
3. MS with low total charges and high percentages landfilled (BG, CZ, GR, HU, LT, LV, PL, PT, RO, SK, CY, EE, ES). All except the last three of these MS have total landfill charges of less than €40 and are landfilling more than 60% of their municipal waste.

The MS in group 1 all have some form of landfill restriction in place for unsorted or untreated municipal waste; several of the MS in group 2 also have landfill restrictions in place for unsorted or untreated municipal waste. It is reasonable to believe that in addition to the taxes and total charges, these restrictions also have an influence on forcing landfill rates down to low levels.

A fairly clear and linear correlation was observed between the total landfill charge and the percentage of municipal waste recycled and composted in the MS. The MS that charge more for landfilling show a higher percentage of MSW recycled and composted. Evidently, other policies (including those to promote recycling, to encourage prevention, extended producer responsibility schemes and PAYT schemes) also influence recycling and composting rates, but it appears reasonable to state that in addition to simply reducing the amount of waste sent to landfill, higher landfill charges tend to push waste towards recycling and composting, therefore moving waste treatment up the waste hierarchy. It appears that MS are much more likely to meet a 50% recycling target once landfill charges (or the cost of the cheapest disposal option) approach €100 per tonne.

It was found that there is a clear trend for landfill tax rates to increase over time (although in some cases they may remain constant for an extended period). A strategy for MS looking to make greater progress in waste management is to increase tax rates over time above the rate of inflation. For 11 MS where adequate time series data have been found, however, not all show a strong correlation between increasing tax rates and decreasing amounts of municipal waste sent to landfill. Strong apparent correlations are observable in AT, SE and UK, although the effective landfilling bans in AT and SE have also had a strong impact on reducing landfill rates. Weaker apparent correlations are observed in three other MS (EE, FI, NL), whilst in a further five MS there is no distinguishable correlation (DK, FR, IE, LV, PL).

Data was gathered for a smaller number of MS on the landfill tax rates for inert waste (including construction and demolition (C&D) waste). In the majority of MS where data was collected on the cost of both municipal and inert/C&D waste, the cost of landfilling inert/C&D waste appears lower than that of municipal waste (this is not the case in DK where no distinction is made between the two types of waste for tax purposes, nor in EE and LV where inert/C&D waste is subject to a higher landfill tax than municipal waste). The two MS with the highest landfill taxes for inert/C&D waste (DK and NL) do demonstrate the highest levels of recycling of such waste, but there is a wide range

of recycling performance amongst the other MS which makes it difficult to infer any significant correlation between the rate of landfill tax and the recycling of C&D waste.

Case studies conducted on three MS (UK, AT, DE) suggest that there are different routes to reducing the landfilling of waste. It appears difficult to 'eliminate landfilling' through a tax alone. The effect of taxes has tended to be a shift away from landfilling towards material recovery. It appears that taxes have not been set at rates which give certainty to enable a complete switch away from landfilling. The principal effect of bans appears to have been to reduce the landfilling of waste and increase the amount of waste incinerated or sent to MBT. There may also have been an effect on recycling and waste prevention in the German case where the difference in price between landfill and incineration had not already been partially closed by the imposition of a landfill tax. There are some concerns regarding the effect of bans on the capacity of non-landfill residual waste treatment. In DE in particular, the ban has led to increases in capacity beyond what is required. Bans remove flexibility from the system; whatever the faults of landfill, it is more flexible to changes in throughput than treatments such as incineration and MBT.

Incineration taxes

The study makes a distinction between incineration taxes (a levy charged by a public authority for the disposal of waste) and gate fees (a charge set by the operator of the incinerator for the provision of the service). The sum of the tax and the gate fee represents the total charge for the disposal of waste in an incinerator.

Only 6 MS were found to have incineration taxes in place for the disposal of municipal waste (NL has an incineration tax that is currently €0; CZ is considering introduction of an incineration tax; SE introduced a tax in 2006 that was abolished in 2010). The level of taxation ranges very widely, from as little as €2.40 per tonne in FR to €44 per tonne in DK. According to the data found during the study, the total typical charge for incineration (i.e. the tax plus the middle of the range of gate fees) of one tonne of municipal waste in the EU ranges from €46 in the CZ to €174 in DE.

Due to a lack of time series data on the change in the level of incineration taxes, it has not been possible to analyse the impact of the rate of incineration tax on the amount of MSW treated by incineration. There is a broad overall trend that higher incineration charges are generally associated with higher percentages of municipal waste being recycled and composted, indicating that higher incineration charges may help to push waste treatment up the waste hierarchy.

All MS that have incineration taxes also have landfill taxes, and in every case the landfill tax is higher than the incineration tax. In MS where information has been found for the total typical charge for both landfill and incineration, the total charges applied to incineration appear higher in 8 MS (AT, BE (Wallonia), CZ, DE, ES, FR, IT, PL). For AT and DE, where there are effectively bans on landfilling MSW-type waste, this is of little concern; however, it suggests that landfilling may preferable in purely economic terms in the other MS. The total charges applied to landfill are higher in 6 MS (BE (Flanders), DK, LU, NL, SI, UK).

Pay-as-you-throw (PAYT) systems

The study has found that 17 MS employ PAYT systems for municipal waste. Many other MS do charge households for waste collection/disposal, but through flat charges or municipal taxes rather

than variable charging. It appears that only three MS (AT, FI, IE) have PAYT schemes in place in all municipalities. In terms of the type of schemes in place (the basis for the variable charge within the PAYT schemes), 16 MS use volume-based schemes, 15 use frequency-based schemes, nine use weight-based schemes, and six use sack-based schemes (NB several MS use a mixture of different types of schemes).

The study identified specific examples of PAYT systems in the majority of MS using PAYT. There is a broad range of both the basis for charging and the amounts charged. Although the schemes are not necessarily directly comparable, the following indicates the variety of approaches:

- **Fixed annual fees per household** (as an element of a PAYT scheme) range from €40 (Miravet and Rasquera Municipalities, Tarragona region, Spain) to up to €2,415 (for a large 1,100l bin in Stuttgart, Germany);
- **Fees for the purchase of mandatory refuse bags** for residual waste range from €0.65 for a 17 litre bag (Argentona Municipality, Barcelona region, Spain) to €5.50 for a 70l bag (for bags over and above standard volume collected, Stuttgart, Germany);
- **Fees per emptying of a bin** (120l or 140l only, for comparison purposes) range from €0.50 (in the context of a scheme combining volume and frequency elements, Ribeauvillé, France) to €4.20 (north Helsinki, Finland); and
- **Fees per kg** range from €0.17 (Slovakia) to €0.36 (Sweden).

Attempts have been made to estimate the coverage (in terms of population or number of municipalities covered) of PAYT systems in the MS that have them. This varies widely, from a very small proportion in ES and the UK, to over 20% of municipalities in NL, 40% of the population in LU, and up to nationwide coverage in AT, FI and IE.

As PAYT systems vary so widely across the EU, the study looked at example schemes in a small number of MS (AT, DE, FI, IE, IT) with well-established PAYT schemes. These examples illustrate the variety in approaches and the resulting impacts of the schemes.

In AT, increased PAYT fees may have had a small dampening effect on waste generation, but the effect is somewhat limited and is not the only factor influencing waste generation. In FI, residents who compost waste at home realise large savings over those who separate their compostable waste for separate collection, and over those who do not separate compostable waste. In DE, a weight-based PAYT scheme in the County of Aschaffenburg observed a significant reduction in household waste generation in the first year of its introduction (from around 22,000 tonnes to around 12,000 tonnes) and waste generation appears to have since stabilised at around 8,000 tonnes. No increase was observed in the illegal dumping of waste. In IE, weight-based PAYT systems appear to have brought about greater reductions in household waste generation (49% after the first year of the system) than tag-based systems (around 23% over the same period). In IT, PAYT systems have had mixed results, although the most successful demonstrate some impressive results. In the Province of Trevisio, the amount of waste sorted by households (and therefore the amount of waste recycled) increased by 12.2% following the introduction of PAYT.

Packaging producer responsibility schemes

Producer fee schemes for packaging have been identified in 24 MS (the UK uses a system of tradable credits for packaging). DK, HU and NL apply taxation systems and deposit-refund systems which

cannot be defined as 'pure' producer responsibility systems. These schemes essentially oblige packaging producers to financially support (to varying degrees) the implementation of recycling schemes for packaging waste. The main focus of this study has been on producer fee schemes.

The data obtained for producer fee schemes show huge ranges of fees per tonne of packaging material placed on the market in the MS. Fees charged for paper range from €8.37 in RO to €175 in DE; fees for glass range from €4.80 in FR to €260.93 in LT; fees for wood range from €0.40 in FI to €80 in PL; fees for aluminium range from €7.26 in RO to €573.10 in NL; fees for steel range from €3 in FI to €282.18 in SE; and fees for plastic range from €20.54 in RO to €1,296 in DE.

The efficiency and effectiveness of the schemes also depends on the proportion of costs of collection, sorting and recycling of waste packaging that are actually covered by producers' contributions. Research under this study suggests that the producer fee schemes in only three MS (AT, BE and DE) cover the full costs to local authorities/waste collection authorities of these activities (this will also be the case in FI after implementation of the new waste law in 2013). In other MS it is unclear whether there is full cost coverage by producers. All three of the MS whose producer fee schemes cover full costs had met the Packaging Directive's targets by 2008 (the only other MS to have met the target was ES), suggesting perhaps that the inclusion of the full cost of packaging waste collection and treatment in the producer fee scheme played a role in meeting those targets.

Attempts were made to link the fees paid into producer fee schemes with the packaging recovery/recycling performance in the MS. However, no conclusive patterns were observed in this regard; some 'cheap' schemes demonstrate high levels of recovery/recycling (notably BE and LU) and some 'expensive' schemes demonstrate low levels of recovery/recycling (notably EE and PL). As the BE scheme covers 100% of the costs of collection and recycling of packaging, this suggests considerable inefficiency in a number of the existing schemes in other MS.

WEEE producer responsibility schemes

Producer responsibility schemes for WEEE have been identified in 25 MS. However, comparable data on costs paid into the schemes proved difficult to find; the way in which the costs are determined varies between MS, and in several cases cost information is only available to the schemes' members.

Despite the lack of data on the actual amounts charged to producers, the vast majority of WEEE schemes were found to charge fees based on the amount placed on the market by producers (either per unit, or per kg or tonne). This ensures that producers pay in proportion to their market share of EEE sold in the MS. In IE, fees are based on the turnover of EEE by the company and the length of time for which the company has been placing EEE on the Irish market. Fees paid in DE vary according to the contracts negotiated with waste management firms. Research indicates that the fees are translated to 'visible fees' for consumers (i.e. the fee paid to the scheme is displayed to the consumer at the point of sale) in BE, RO and SK (although the fee will no longer be displayed to the consumer in RO after February 2013).

The efficiency and effectiveness of schemes depends on the proportion of costs of collection, recycling and recovery of WEEE that are covered by producers' contributions. This study found that schemes in eight MS (AT, BE, CY, CZ, DK, IE, LV, PL) cover the full costs of these activities, or the full costs once WEEE has been delivered to specific collection points, through producers' contributions. In FI, producers pay for 100% of the costs for management of business-to-consumer WEEE; for

business-to-business WEEE, producers and end users may agree to share costs. In other MS it is unclear whether producers' contributions cover full costs.

In terms of waste management performances for WEEE, the average collection rate (for the 25 MS for which data were available, minus BG which appears to import large quantities of WEEE and would therefore skew the figures) was 31.5% by weight of amounts put on the market; an increase from 23% in 2006. It is likely that considerably more than this is collected but not reported, and that a substantial part of this undergoes sub-standard treatment in the EU or is illegally exported. Where WEEE is collected separately it is widely recycled: for 23 MS where recycling rates can be calculated, the average recycling rate was 75.8%.

Attempts were made to link the fees paid into WEEE producer responsibility schemes with the WEEE recovery/recycling performance in the MS. Based on the only three MS for which comparable data were available, it appears that where contributions for large appliances are made per item, a higher cost of contribution is associated with higher recovery and recycling rates. For schemes where contributions for large appliances are based on a cost per kg, there does not appear to be any relationship between the cost of contributions and recovery and recycling rates. For schemes for small appliances, there does not appear to be any relationship between the cost of contributions and the rates of recovery and recycling, whether the contributions are per item or per kg. This analysis is however based on a very limited data set therefore cannot be taken as an accurate overall representation of the cost and effectiveness of WEEE schemes. The Swedish El-Kretsen scheme has been identified as a particular success story.

ELV producer responsibility schemes

ELV producer responsibility schemes have been identified in 24 MS. All ELV producer responsibility schemes specify that ELV must be taken back at no cost to the final owner of the vehicle (in several cases unless the vehicle has been modified or is no longer intact). Several MS (BE, BG, DK, GR, LV, LT, NL, PT, UK) have organisations or 'eco-organisms' that coordinate the take back and recovery of ELV on behalf of producers. Only limited information has been found on financial contributions to producer responsibility schemes for ELV, and these are not directly comparable between MS.

In CZ, the person who registers a used vehicle makes a contribution based on the EURO standard of the vehicle. In DK, the landfiller of shredder residues and the car owner both make financial contributions. The producer/importer pays a financial contribution in FI (a one-off joining fee and a small fee per car sold), NL (a fee per new car registration, ultimately passed on to the consumer), PT (a fee based on the number of vehicles sold) and SK (a contribution per kg of vehicle to a recycling fund).

In terms of waste management performances for ELV, by 2008 22 MS had met or exceeded the 2006 reuse and recycling target in the ELV Directive (only CY, FR, IE, PL had not; data was not available for MT). Eight MS (BE, EE, DE, GR, LV, LT, SK, SI) had already met or exceeded the 2015 reuse and recycling target. Also by 2008, 18 MS had met or exceeded the 2006 reuse and recovery target (CY, DK, FI, FR, HU, IE, PL, UK had not; data was not available for MT). AT was alone in already having met or exceeded the 2015 reuse and recovery target.

Based on the lack of comparable data collected within the study on producer responsibility schemes for ELV, conclusions have not been drawn on the performance of MS in relation to collection, recovery and recycling/reuse of ELV and the impacts of ELV producer responsibility schemes.

Batteries producer responsibility schemes

The study identified producer responsibility schemes for batteries in 24 MS. Comparable data on the amount charged to producers, however, proved extremely difficult to find.

The schemes charge fees to producers based on the amount of batteries placed on the market, either per kg, per battery or according to market share. The MS determine the cost based on the type of battery, and the classification of batteries varies from country to country, e.g. consumer/vehicle/industrial battery in AT and LT; the chemical content of the battery (e.g. lead-acid/nickel-cadmium/alkaline/zinc carbon/lithium/button/lithium ion) in LV and PT; and size or weight of battery in CY and SK.

As the dates for the main targets in the Batteries Directive have not yet arrived, data has not yet been gathered on the collection and recycling rates for batteries. Based on this and also the lack of data collected within the study on producer responsibility schemes, it was not possible to draw conclusions on the performance of MS in relation to collection and recycling of batteries and the impacts of batteries producer responsibility schemes.

Producer responsibility schemes dealing with other waste streams

Although not the focus of the study, attempts were made to identify producer responsibility schemes dealing with other materials, to provide a somewhat more complete picture of the application of producer responsibility in the EU-27 (although it is worth noting that the information included in the study is incomplete). The additional materials subject to producer responsibility schemes range from the collection of tyres and paper/card (frequent) to photochemicals (rare). Of these 'other' materials covered, producer responsibility schemes were most commonly found for the recovery of tyres (15 MS), paper/card (nine MS), oils (including mineral, motor, edible and lubricating oils) (seven MS), and the collection and disposal of old and unused medicines (three MS).

Modelling

A modelling exercise was undertaken within the study to investigate what might be the effect in quantitative and qualitative terms of implementing new (sets of) EIs in the waste sector. A baseline scenario was developed to model MSW generation, which suggested that average municipal waste generation per capita would increase from 446kg to 532kg per year. Only seven MS are anticipated to show absolute decoupling of MSW generation from household consumption.

Two scenarios were developed:

- **Scenario A:** All MS reach a level of landfill tax of at least €40 per tonne, leading to landfill diversion and the distribution of waste to recycling, composting and incineration in line with the actual distribution across these treatment methods;
- **Scenario B:** A variation on Scenario A, including an increased landfill tax and a mixture of other EIs (of which PAYT and extended producer responsibility may be the most important.

These instruments favour separated collection and recycling, and Scenario B assumes that recycling and composting are the final destination of all wastes diverted from landfill.

The results of the modelling exercise were as follows.

Under **Scenario A**, the amount of MSW diverted from landfill in 2025 in EU-27 is 43 Mt compared to the baseline scenario or 19 Mt compared to the 2008 quantities. Of the total of 291.5 Mt MSW generated and collected in 2025, 29.4% goes to landfill, 23.2% to incineration, 12.6% to paper cardboard recycling, 2.6% to plastic recycling, 5.3% to glass recycling, 1.7% to metals recycling, 16.8% to other (or non specified) recycling options, 7.5% to composting, 0.5% to home composting, and 0.3% to anaerobic digestion.

The aggregate environmental impacts of **Scenario A** can be summarised as follows:

- Total reduction of CO2 emissions: 60,732,006 CO_2e, or 48,123,286 CO_2e if the carbon sink effects of landfilling and composting are taken into account;
- Total avoided natural resource depletion: 251,074 t Sb eq; and
- Total avoided fossil resource depletion: 11,401,382 toe.

Under **Scenario B**, the amount of MSW diverted from landfill in 2025 in EU-27 is 43 Mt compared to the baseline scenario or 19 Mt compared to the 2008 quantities, as in scenario A. The difference is to be found in the way in which the diverted waste will be treated. Of the total of 291.5 Mt MSW generated and collected in 2025, 29.5% goes to landfill, 19.5% to incineration, 13.7% to paper cardboard recycling, 2.8% to plastic recycling, 5.9% to glass recycling, 1.8% to metals recycling, 17.9% to other (or non specified) recycling options, 7.9% to composting, 0.5% to home composting, and 0.4% to anaerobic digestion.

The aggregate environmental impacts of **Scenario B** can be summarised as follows:

- Total reduction of CO2 emissions: 63,929,509 CO_2e, or 51,597,749 CO_2e if the carbon sink effects of landfilling and composting are taken into account;
- Total avoided natural resource depletion: 337,833 t Sb eq; and
- Total avoided fossil resource depletion: 15,346,929 toe.

Policy options

Based on the work undertaken in the study, a selection of potential policy options have been suggested for consideration by the Commission, to promote an 'optimal' use of EIs by the MS.

Some general principles could be applied to the implementation of the various policy options, most notably:

- Allowing some **flexibility for MS** to implement EIs in the most appropriate way for their own particular conditions (i.e. respect of the subsidiarity principle);
- Ensuring an **appropriate balance between regulatory instruments (e.g. targets, technical standards, bans) and EIs**;
- Considering carefully what should be done with **revenues generated from EIs**;
- Providing a **clear policy framework for the foreseeable future** within which the waste management industry can operate, to allow rational investment in infrastructure;

- **Fully taking into account the economics of the waste management sector**, allowing the development of EIs to rest on rational cost analysis;
- A full understanding by MS of the **external costs and benefits of various waste management options**; and
- **Requiring better reporting** by MS on waste generally and on the use of EIs specifically.

Three main policy options have been identified based on the work undertaken within the study:

Option 1 – Setting a minimum level of landfill tax to be applied in all MS

There appears to be a good case for MS to implement landfill taxes with the aim of increasing the rate of recovery of materials in line with the waste hierarchy. Whilst recommending the same minimum taxation level for all MS is not appropriate, a common method for calculating a minimum tax level could be developed, and taxes could be more strongly encouraged in the worst performing MS. For MS that have bans that effectively prohibit the landfilling of untreated MSW, it may not be necessary to impose a tax.

Means of executing this policy option could include a revision of the Landfill Directive, a Decision addressed to the MS, or, if a non-legislative approach is preferred (although this would be more difficult to enforce), guidance issued to the MS.

Option 2 – setting criteria/producing guidance for the design of producer responsibility schemes

The most successful producer responsibility schemes appear to share some common features: a common, fully private body that is created, run, owned and supported by the obligated producers; requiring producers to fully fund the collection and recycling scheme; and high targets. Many other design features may influence the success of EPR schemes, and future research could be conducted into these: targeting both households and consumption of products 'away from home'; incentivising eco-design; greater application of individual producer responsibility (IPR); communication and best practice exchange between producers and between all relevant actors; basing variable fees on defined criteria; setting targets to better differentiate between materials; and appropriate involvement of governments, municipalities and waste management operators.

Means of executing this policy option could include integration of aspects of the option within relevant waste Directives (the Packaging, WEEE, ELV, Batteries and/or Waste Framework Directives), a Decision (or Decisions) addressed to the MS, or guidance on the design of EPR schemes if a non-legislative approach is preferred.

Option 3 – Encouraging the use of charging that ensures waste generators face incentives in line with the waste hierarchy

In principle, all producers of waste should face some incentive to reduce waste generation and to make use of cost-effective recycling services. Landfill taxes will increase the range of recycling services that can be offered cost-effectively (see Policy Option 1 above), and the costs of providing some services may be fully supported through producer responsibility schemes (see Policy Option 2 above). The same could be said for incineration taxes.

MS are required, under the Waste Framework Directive, to both implement the hierarchy in policy and law (Article 4) and to implement Waste Prevention Programmes (Article 29). Such programmes should encourage the implementation of PAYT, for example, recognising that it has a fundamental role to play in reducing household waste generation and increasing household recycling. Appropriately ambitious recycling targets may result in a natural evolution towards the implementation of PAYT schemes.

Guidance could usefully help to define how to design PAYT schemes so that they enhance the prospects for prevention and recycling, whilst ensuring charging structures recover costs. The infrastructure for recycling should be both comprehensive and convenient for the user. Charging systems should be structured to introduce incentives to reduce, continuously, the quantity of residual waste being generated. PAYT schemes should generally seek to charge: the highest variable fee for residual waste; a lower (but non-zero) fee for biowaste if garden waste is targeted by biowaste collection (to encourage home composting); a zero fee where only kitchen waste is targeted by biowaste collection; and a low or zero fee for collected dry recyclables. Schemes should also include a fixed fee element. With regards to waste prevention, weight-based systems are most successful, followed by volume and frequency-based/sack-based systems, and then volume-based systems (i.e. schemes where households simply choose a specific size of container). Care should be taken for PAYT and producer responsibility schemes to be complementary.

Several additional options are also included for consideration. These are not necessarily based on the work undertaken within the current study, but based on the experience of the study team and comments received from stakeholders.

Option 4 – Primary materials taxes

Taxes on primary materials may act to reduce the consumption of materials, encourage re-use and preparation for re-use, and increase the use of recycled materials. In theory, this could be an attractive option for improving the sustainability of materials management. In reality, obstacles confront the use of such an instrument: trade in materials and products both within EU countries, and between the EU and the rest of the world; difficulties in achieving the unanimity required for EU taxation; and the risk of promoting switching between materials placing some materials at a disadvantage. In practice, therefore, this may be a difficult policy option to execute. It is therefore suggested that the other options below be considered as more practical and achievable alternatives.

Option 5 – Levy on excess residual household waste

The quantity of *residual* household waste per inhabitant is a good indicator for the effectiveness of sustainable consumption, waste prevention and waste recycling measures, and would likely even out the level of performance between MS with high consumption levels and recycling rates, and those with lower consumption levels but lower recycling rates. A levy could be applied for the generation of residual waste above an average level (expressed in terms of kg per inhabitant) and refunds provided to those whose residual waste is below the average level. This would generate a dynamic incentive for reducing residual waste per inhabitant. Such a scheme could be introduced for municipalities at MS level, or on an EU-wide basis.

Option 6 – Deposit refunds for hazardous materials/materials containing valuable materials

Deposit refunds are potentially useful where a waste material or product can give rise to problems if poorly managed (e.g. hazardous materials, materials of potentially high value (including critical materials) and materials with a propensity to arise as litter). One interesting application of this could be the collection of small WEEE items, encouraging higher rates of take-back and therefore enabling the retention of the material value, and avoiding improper management of the waste.

Option 7 – Refundable compliance bonds

Refunded compliance bonds may have a useful role to play in increasing recycling rates for C&D wastes (in conjunction with landfill taxes and taxes on primary aggregates). Contractors would pay a variable financial sum to the relevant regulatory land-use planning authority, related to the size of the project, plus a small administrative fee. The sum would be retained as a bond to ensure that a specified recycling rate was met, and all of the variable element of the bond would be returned on demonstrating achievement of the desired recycling rate. A proportion of the bond only would be refunded where partial compliance with the desired target was achieved.

Option 8 – Product Taxes and Charges

Product taxes are likely to reduce waste when they are applied to products for which there are clear substitutes which lead to lower levels of waste generation, such as disposable items or plastic bags. Taxes on such items may discourage their consumption, encourage the use of non-disposable alternatives, and contribute to a reduction in waste generation.

Option 9 – Subsidies for Waste Prevention Activities

Subsidies for activities that promote waste prevention may be justified where they lead to behaviours that generate financial savings to the public purse and/or environmental benefits which exceed the level of subsidy being granted. Existing examples address mainly home composting and the use of reusable nappies, but there may be other relevant target waste streams which may justify support to facilitate households in switching to lower-waste products or services.

Option 10 - Variable VAT Rates

Within the scope of the existing VAT Directive (2006/112/EC), MS have some discretion to support particular activities through reduced VAT levels. The repair of white goods (large household appliances) is a category of service which seems to comply with the spirit of the Directive. Another potential use of variable VAT rates might be to lower VAT where longer warranties are offered by manufacturers. Other possible applications could be for products and services that are intrinsically less waste-intensive than close substitutes, and the provision of services that can replace the purchase of products. It should however be recognised that other approaches such as product taxes may be more cost-effective, and potentially more 'visible'.

Option 11 – Recycling credits

Recycling could be given increased support to reward the contribution made to reducing greenhouse gas emissions. Establishing the principle within the EU ETS (Emission Trading Scheme) of recognising the benefit of recycling relative to use of virgin materials within the could be an important first step to wider recognition of recycling within a future design of the ETS.

CONTENTS

TABLES AND FIGURES

ABBREVIATIONS AND ACRONYMS

Abbreviations

ALSAG - Act on the Remediation of Contaminated Sites

B2B - Business-to-business

B2C - Business-to-consumer

BTA - Border tax adjustment

C&D - Construction and demolition (waste)

C&I - Commercial and industrial (waste)

CHP - Combined heat and power

CO_2 - Carbon dioxide

DKK - Danish Krone

EEE - Electronic and electrical equipment

EI / EIs - Economic instruments

ELV - End-of-life vehicles

EPR - Extended producer responsibility

EU - European Union

EU15 - EU Member States prior to the 2004 enlargement

EU ETS - EU Emission Trading Scheme

GC - Green certificates

GDP - Gross domestic product

GHGs - Greenhouse gases

IPR - Individual producer responsibility

IT - Information technology

kg - Kilogram

kW - Kilowatt

kWH - Kilowatt hour

l - Litre

LAC - Local authority collected

LASs - Landfill Allowance Schemes

LATS - Landfill Allowances and Trading Scheme

LTL - Lithuanian litas

LVL - Latvian lats

MBI / MBIs - Market-based instruments

MBO - Environmental policy agreement (Flanders, Belgium)

MBT - Mechanical biological treatment

MS - Member States

MSW - Municipal solid waste

Mt - Million tonnes

MW - Megawatt

MWh - Megawatt hour

PAYT - Pay-as-you-throw

PBU - Pay-by-use

PET - Polyethylene terephthalate

PRO - Producer Responsibility Organisation

PVC - Polyvinyl chloride

R&D - Research and development

RES - Renewable energy sources

t - Tonne

t CO_2e - Tonnes of CO_2 equivalent

TOC - Total Organic Content (also defined as Total Organic Carbon)

VAT - Value added tax

WEEE - Waste electrical and electronic equipment

WFD - EU Waste Framework Directive

WIP - Waste Implementation Programme (Defra, UK)

Organisations (acronyms)

ACEA - European Automobile Manufacturers' Association

ATO - Office of Optimal Territorial Scope (Italy)

BREW - Business Resource Efficiency and Waste programme (UK)

CEPI - Confederation of European Paper Industries

CEWEP - Confederation of European Waste-to-Energy Plants

CLEPA - European Association of Automotive Suppliers

DRS - Danish Dansk Retursystem

DPA - Danish Producer Responsibility

Defra - Department for Environment, Food and Rural Affairs (UK)

EAR - Electrical and Electronic Equipment Register (Germany)

EBRA - European Battery Recycling Association

EEA - European Environment Agency

EPA - Environmental Protection Agency (Denmark, Ireland)

EPBA - European Portable Battery Association

FEAD - European Federation of Waste Management and Environmental Services

HMRC - HM Revenue & Customs (UK)

IPC - Interregional Packaging Commission (Belgium)

LASU - Local Authority Support Unit (UK)

OECD - Organisation for Economic Cooperation and Development

OVAM - Public Waste Agency of Flanders

TAC - Technical Adaptation Committee

WRAP - Waste and Resources Action Programme (UK)

WTO - World Trade Organization

2 INTRODUCTION

2.1 INTRODUCTION TO AND OBJECTIVES OF THE STUDY

This report presents the activities and analysis undertaken within ENV.G.4/FRA/2008/0112 – Use of Economic Instruments and Waste Management Performances.

This is a follow-up study to the study to support the review of the Waste Thematic Strategy, which was completed in October 2010 and published on the DG Environment website in January 2011. Within that study, a limited amount of research was undertaken on the use of economic instruments (EIs) to promote improved waste management, and the current study was requested to expand significantly on that limited research.

The objective of the present study is to analyse the relationship between the performances of the waste management systems of the Member States (MS) – with a particular focus on recycling and prevention – and the use of EIs. On the basis of this analysis, the opportunity of moving towards a European common approach for the use of EIs in relation to waste management is analysed. The study aims to provide supporting information and analysis for the European Commission in the preparation of the follow up of the report published in January 2011 on the Thematic Strategy on the Prevention and Recycling of Waste[2].

A limited set of EIs have been addressed by the study in order to provide greater focus, enabling more substantive research to be undertaken into a defined set of EIs rather than attempting to cover the whole landscape of EIs being used in the waste sector in the EU-27. The EIs being investigated were chosen after discussions with the Commission, and are amongst those where it was anticipated that waste management impacts could be most clearly seen and attributed to the use of EIs.

The following EIs were studied:

1. Charges for waste disposal and treatment:
    a. Landfill taxes and fees (and bans to provide context for the charges);
    b. Incineration taxes and fees (and bans to provide context for the charges);
2. Pay-as-you-throw (PAYT) schemes; and
3. Producer responsibility schemes for specific waste streams including packaging, WEEE, ELV, batteries and paper/cardboard.

[2] http://ec.europa.eu/environment/waste/strategy.htm

The last of these might not be considered an EI *per se*, but we have taken the view that since most MS make use of measures that imply that producers pay for some or all of the cost of collecting and recycling the targeted waste stream, such 'producer fee schemes' have the character of an EI.

2.2 OVERVIEW OF METHODOLOGICAL APPROACHES

The Commission contracted the consortium (led by IEEP and including BIO IS, Ecologic, Arcadis, Umweltbundesamt and Eunomia), to undertake five specific tasks. These are outlined below, alongside the key methodological tools employed. Methodological information is also included within the analytical sections of this report to aid understanding of the outcomes and the conclusions reached.

The main research mechanisms employed were data collation (including input from stakeholders and experts) and literature review. This was completed in a structured manner as explained below.

A series of factsheets were prepared on the EU-27, examining their use of EIs in the waste sector. These formed a core information resource for use throughout the project (Task 1). In order to help focus the efforts of the consortium, 18 MS were subjected only to a broad analysis of their use of EIs related to waste management (Task 1.1), whilst nine MS (Austria, Belgium, Denmark, Estonia, France, Germany, Netherlands, Slovenia, and the UK) were the subject of more substantial research (Task 1.3) due to their successful waste management performance, or their interesting or successful approaches to the use of EIs. Information was also gathered on the waste management performances of the MS (Task 1.2). The data collected has been compiled into a series of tables and charts presented throughout the report.

The analysis was further supplemented by the preparation of a series of case studies (Task 2) on three MS (Belgium, Germany and Slovenia) and a series of specific EIs (landfill taxes and restrictions, PAYT schemes and producer responsibility schemes for packaging, WEEE and ELV), in an attempt to identify their success factors.

These data collation activities were supported by modelling efforts (Task 3), to assess the potential future impacts of the use of EIs, with a focus on landfill taxes (the EI for which the most data was found).

Attempts were then made to draw together conclusions and a set of potential policy options (Task 4) for the Commission to consider in order to encourage the wider use of EIs in the waste sector, in particular by those countries with poorer waste management performance.

Stakeholder and expert consultation was carried out throughout Tasks 1 and 2, and a stakeholder event was organised towards the end of the study under Task 5.

2.3 MS WASTE MANAGEMENT PERFORMANCES

The figures in this section attempt to provide an overview of the waste management performance of MS, in order to provide some context for the study. The main focus is on municipal solid waste (MSW) as this is the waste stream for which the most complete data sets are available (although it must be noted that the definitions of municipal waste can still vary between MS). All figures on municipal waste are based on Eurostat statistics. The data from this section is used in the attempts to compare the effectiveness of the EIs identified throughout the study.

2.3.1 MUNICIPAL WASTE

The Waste Framework Directive (2008/98/EC) sets the following targets:

- The separate collection of paper, metal, plastic and glass by 2015;
- At least 50% of paper, plastics, metal and glass from households and similar origins prepared for reuse or recycled by 2020; and
- At least 70% of non-hazardous construction and demolition (C&D) waste reused, recycled or undergoing material recovery by 2020.

The Landfill Directive (1999/31/EC) sets the following targets for the reduction of biodegradable municipal waste going to landfill:

- A reduction to 75% of 1995 levels by 2006;
- A reduction to 50% of 1995 levels by 2009; and
- A reduction to 35% of 1995 levels by 2016.

Those MS sending more than 80% of their municipal waste to landfill in 1995 are permitted to postpone attainment of the above targets by up to four years.

Figure 1 and Figure 2 summarise the amount of municipal waste generated in the MS in 1995, 2002 and 2009. They demonstrate that only six MS (Bulgaria, Estonia, Germany, Hungary, Lithuania and Slovenia) reported generating less municipal waste (both in absolute terms and in kg per capita) in 2009 than in 1995. The reasons for this are likely to be mixed; this study details evidence in Germany of the implementation of a number of successful economic instruments (notably the landfill ban combined with PAYT schemes and successful producer responsibility schemes) which may well have made a contribution to the reduction in municipal waste generated. However, for some of the newer MS the decline in waste generation may instead be attributable to factors other than the use of EIs, including the ongoing implementation of the EU waste *acquis* following their accession to the EU, and potentially also changes in data reporting and definitions of municipal waste.

Figure 1 Municipal waste generated, 1995-2002-2009 (1,000 tonnes)

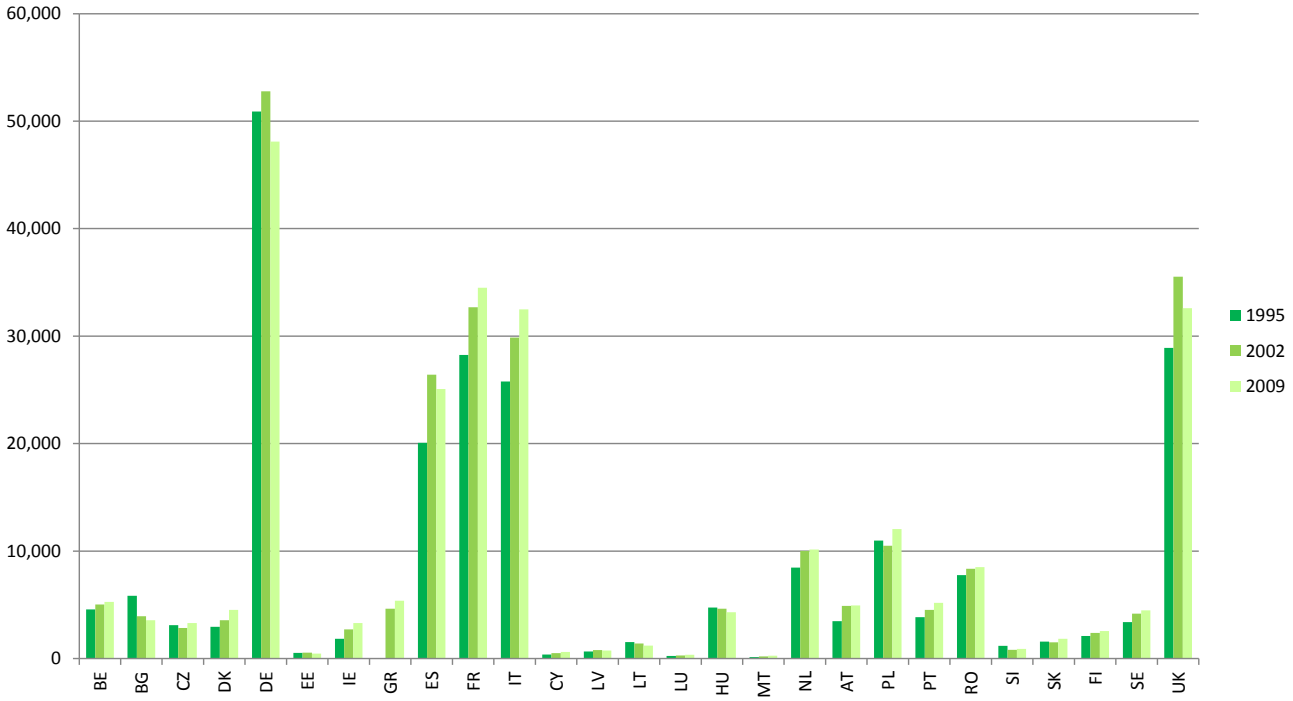


Figure 2 Municipal waste generated, EU 27, 1995-2002-2009 (kg/capita)

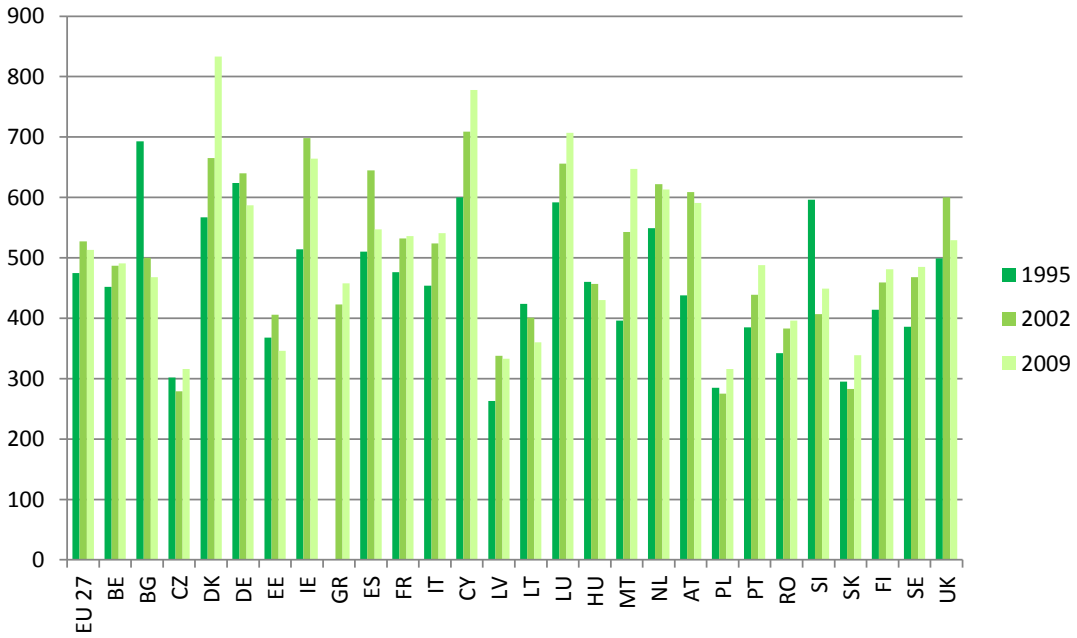


Figure 3 and Figure 4 show the amount of municipal waste sent to landfill in the MS in 1995, 2002 and 2009. Eighteen MS landfilled less (in absolute terms) in 2009 than in 1995; 22 MS landfilled less in kg per capita in 2009 than in 2005. Particularly notable cases of reduction in the rate of landfilling include Germany (which saw a major decrease in landfilling to almost zero following the

implementation of its landfill ban), Austria (which again saw a decrease to almost zero following the implementation of a landfill ban), and the UK (where a combination of recycling targets, the landfill allowance schemes and constant and planned increases in the landfill tax have had a significant impact in terms of reducing the amount of waste sent to landfill, even if the amount of waste landfilled per capita remains relatively high in comparison with several of the EU15).

Figure 3 Municipal waste landfilled, EU 27, 1995-2002-2009 (1,000 tonnes)

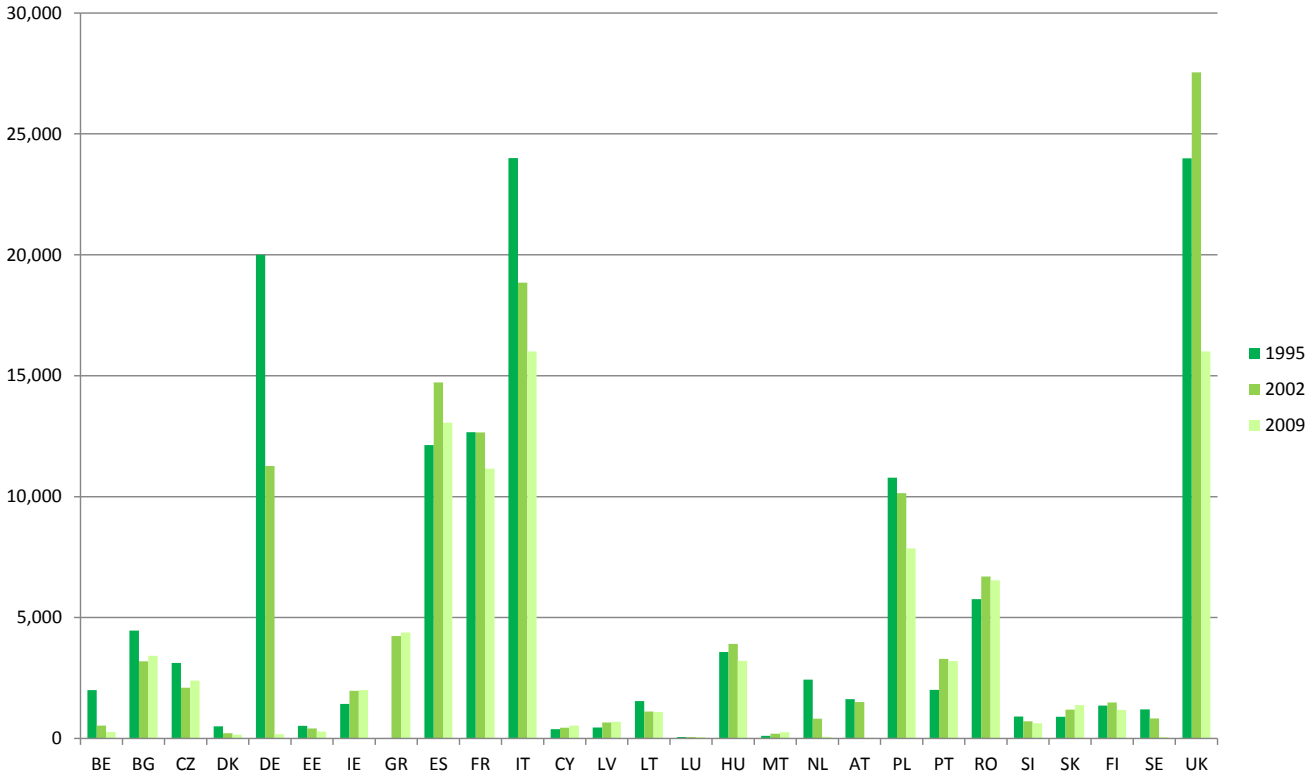


Figure 4 Municipal waste landfilled, EU 27, 1995-2002-2009 (kg/capita)

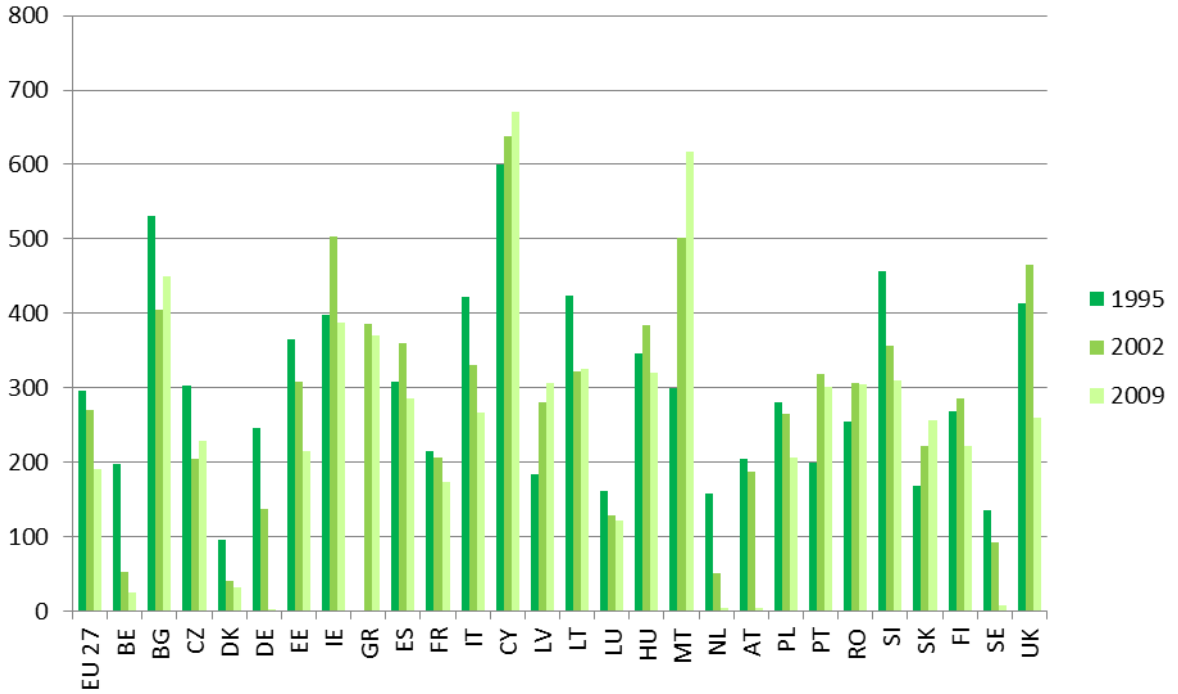


Figure 5 and Figure 6 show the amount of waste incinerated in the MS in 1995, 2002 and 2009. In all MS that have municipal waste incineration, the amount sent to incineration in absolute terms was higher in 2009 than in 1995. Luxembourg, however, reported a decrease in incineration in kg per capita, and the increases in Belgium and France in particular were marginal. These observations would seem to indicate that waste treatment might be moving away from landfill towards incineration (with energy recovery), as a result of the progressive implementation of policies (and perhaps also EIs) to this end.

Figure 5 Municipal waste incinerated, EU 27, 1995-2002-2009 (1,000 tonnes)

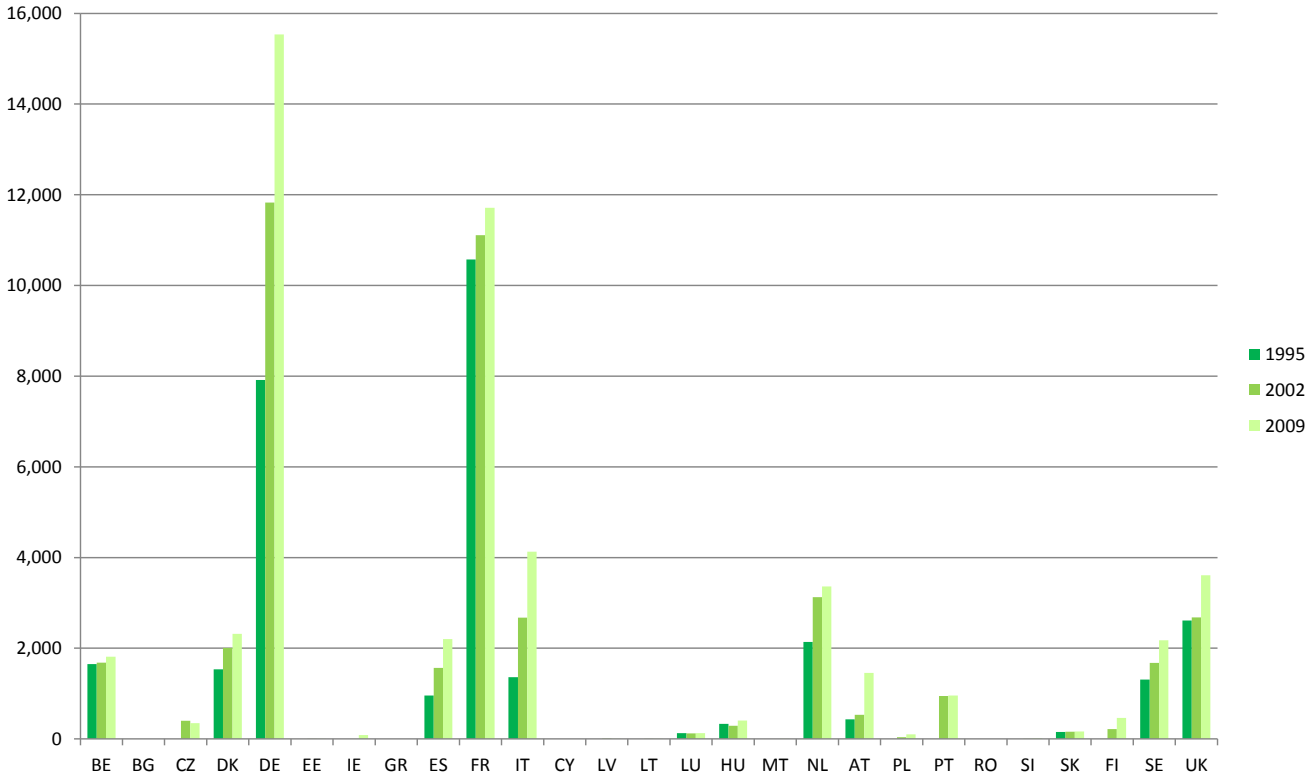


Figure 6 Municipal waste incinerated, EU 27, 1995-2002-2009 (kg/capita)

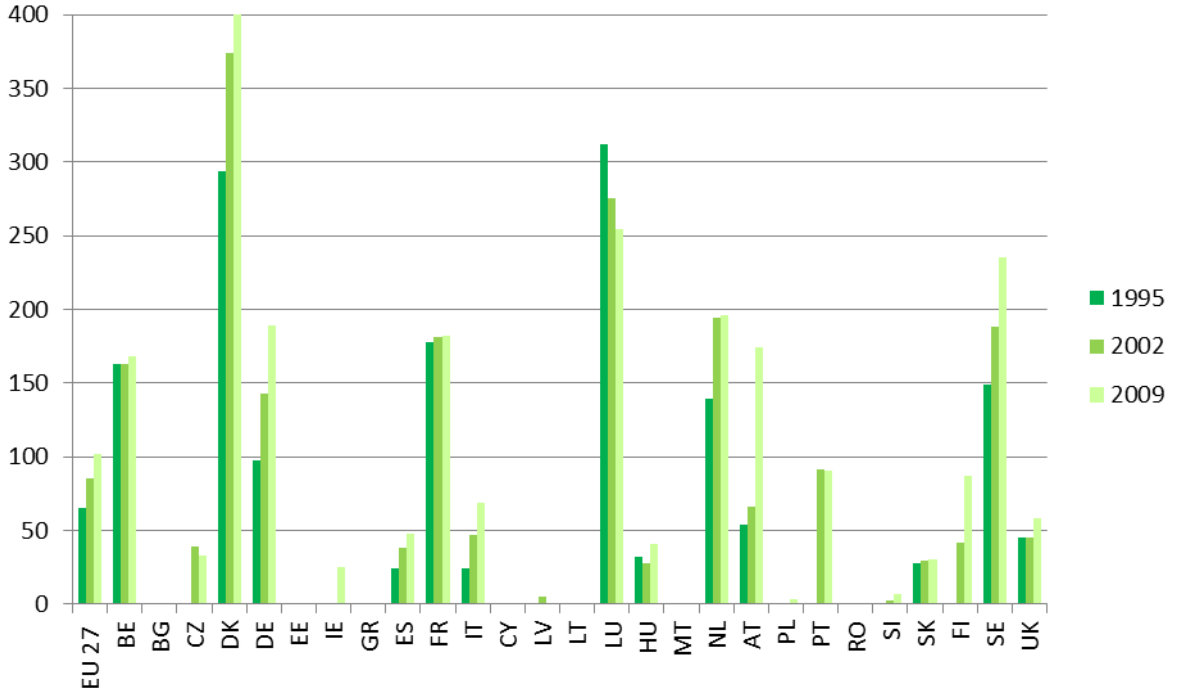


Figure 7 and Figure 8 show the progression in the amount of municipal waste recycled in the MS in 1995, 2002 and 2009. All MS (except Bulgaria, which still reports no recycling) increased the amount recycled from 1995 to 2009, both in absolute terms and in kg per capita. For those MS where such a figure can be calculated (i.e. MS that had higher than zero recycling in 1995), the amount recycled in kg per capita increased by between 42% and 99%. Again, this significant increase in recycling rates suggests that waste is being moved up the waste hierarchy.

Figure 7 Municipal waste recycled, EU 27, 1995-2002-2009 (1,000 tonnes)

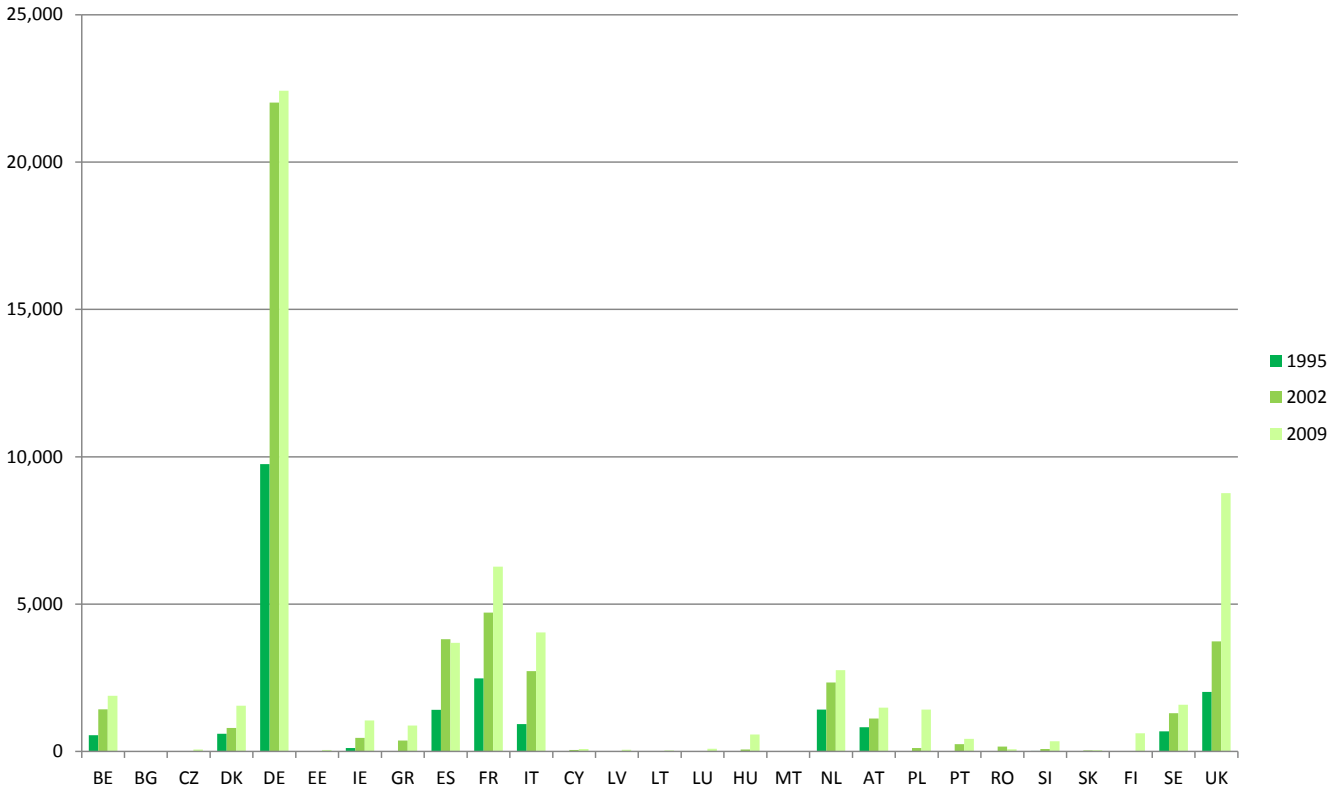


Figure 8 Municipal waste recycled, EU 27, 1995-2002-2009 (kg/capita)

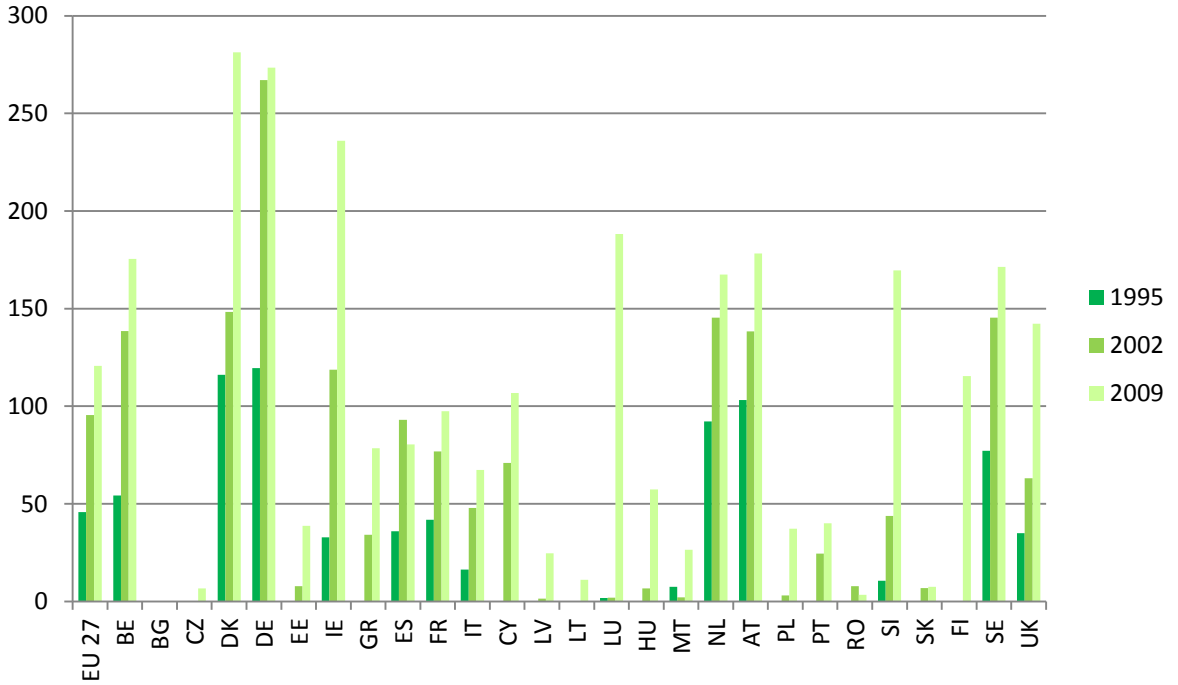


Figure 9 and Figure 10 show the amount of municipal waste composted in the MS in 1995, 2002 and 2009. All MS (except Bulgaria and Cyprus, which still reports no composting) increased the amount composted from 1995 to 2009, both in absolute terms and in kg per capita. The smallest kg per capita increase in the amount composted was observed in the Netherlands (from 131 to 144kg), and the largest kg per capita increase was observed in Italy (from 6 to 192kg). Again, this is a significant increase in the amount of waste being treated higher up the waste hierarchy. In addition, it has undoubtedly had an impact on the rate of landfilling of biodegradable waste, which is likely at least in part to be a response to the targets of the Landfill Directive.

Figure 9 Municipal waste composted, EU 27, 1995-2002-2009 (1,000 tonnes)

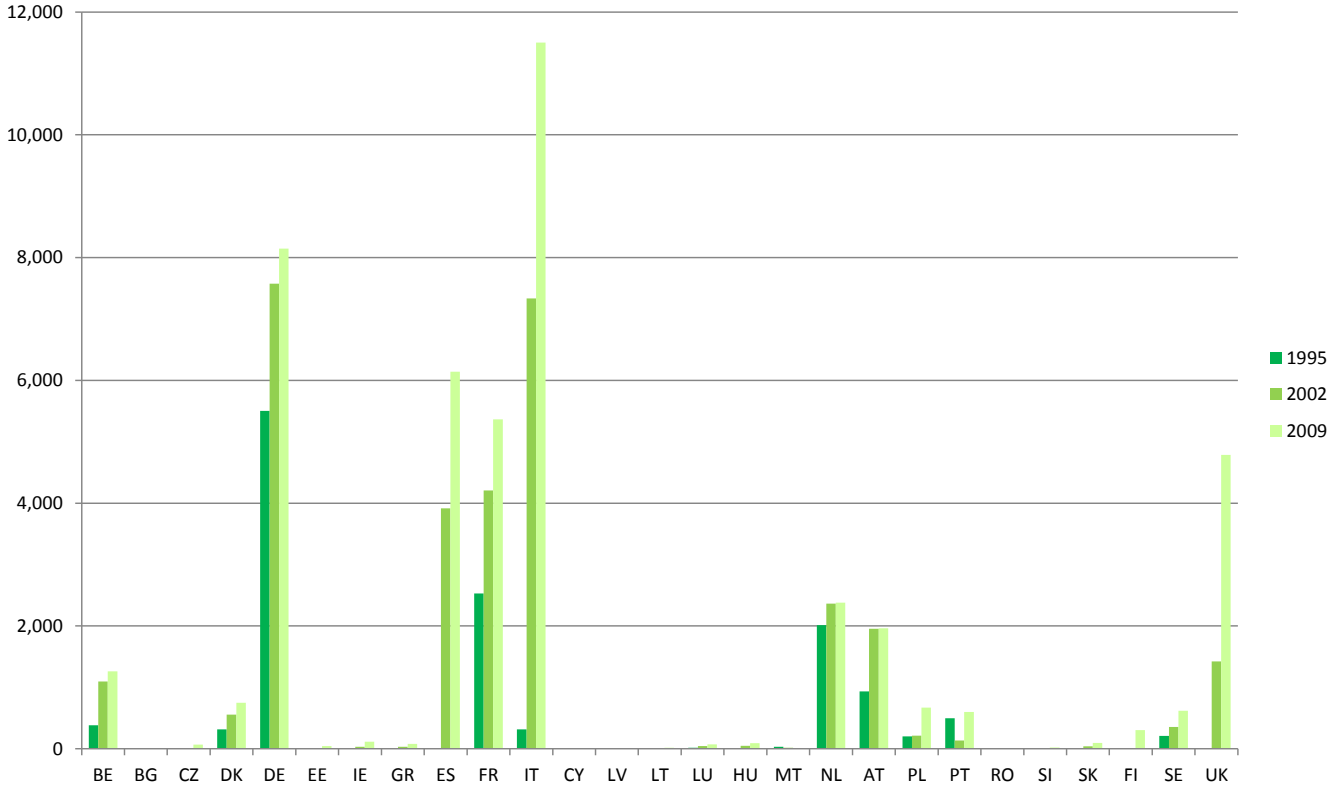


Figure 10 Municipal waste composted, EU 27, 1995-2002-2009 (kg/capita)

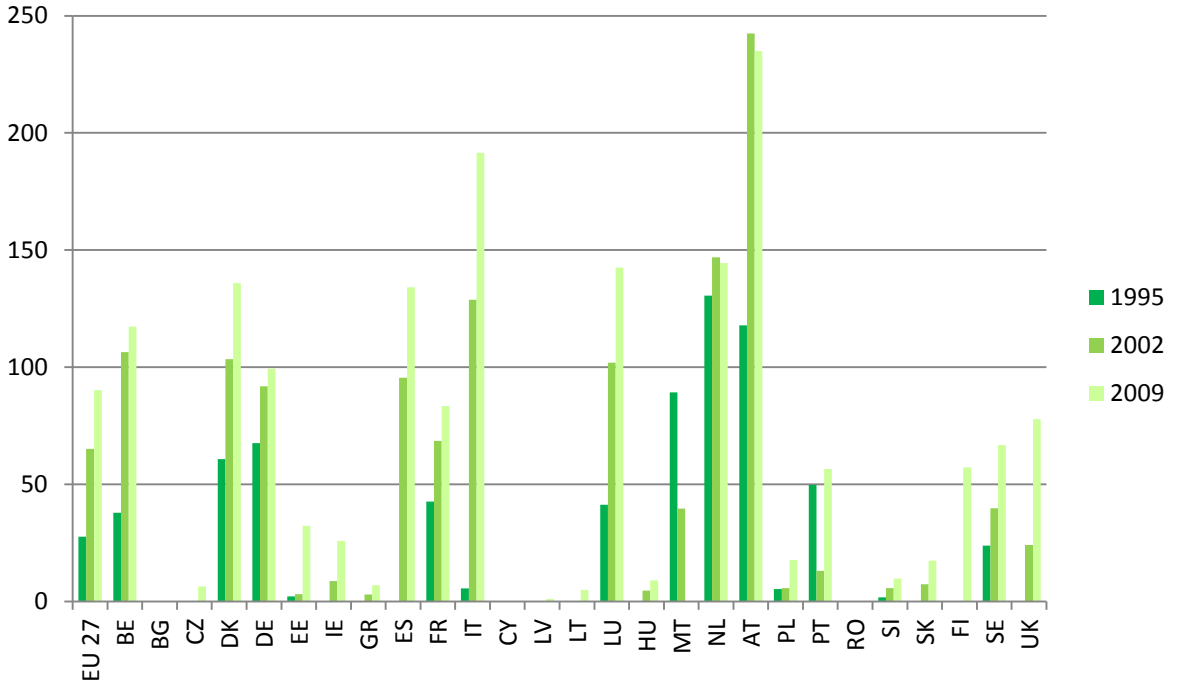
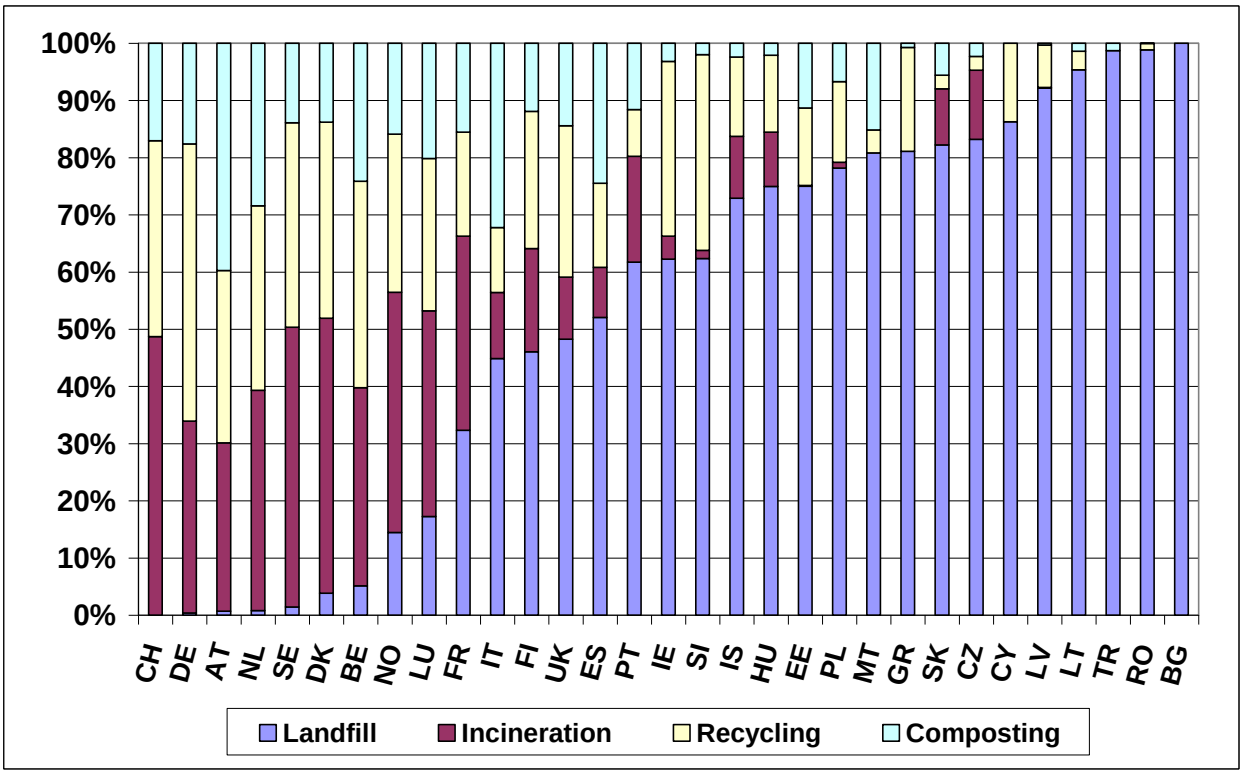


Figure 11 below provides an overview of whole picture of municipal waste management (the percentage of municipal waste landfilled, incinerated, recycled and composted) in the EU-27 (plus Switzerland, Norway, Iceland and Turkey) in 2009. The MS towards the right of the figure have the furthest to move in terms of reducing the percentage of waste sent to landfill, whilst those towards the right tend to be undertaking the highest percentages of recycling and/or composting, demonstrating movement of waste management up the waste hierarchy.

Figure 11 EU-27 municipal waste management (Eurostat, waste data centre 2010)



2.3.2 OTHER WASTE STREAMS SUBJECT TO EU TARGETS

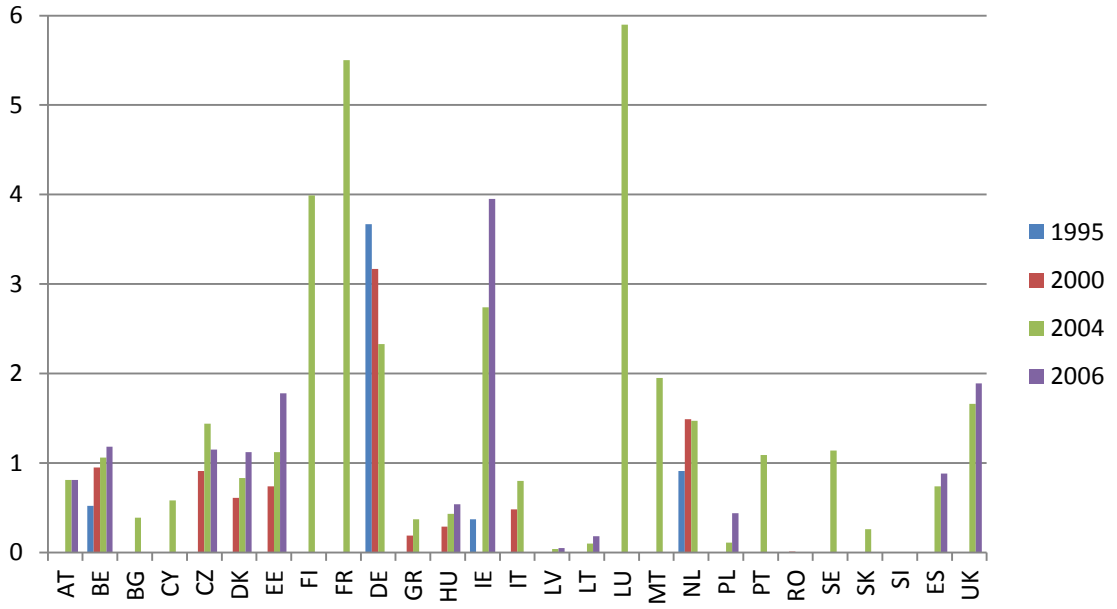
The following text and figures provide an overview of the waste management performances of the MS with regards to specific waste streams, including a summary of the targets set by relevant EU Directives.

With regards to the generation and recycling of **construction and demolition (C&D) waste**, it should be noted that data availability is generally poor. The figures below are based on data from the European Topic Centre on Resource and Waste Management.[3]

Around 850 million tonnes of C&D waste is generated in the EU per year. This represents 31% of the total waste generation in the EU. Figure 12 below shows the generation of C&D waste in the MS in tonnes per capita. The generation per capita varies considerably, from 0.04 tonnes in Latvia to 5.9 tonnes in Luxembourg. Apart from Germany, all countries where data for more than one year is available saw an increase in generation per capita in the period 1995 to 2006. The differences between countries in generation of C&D waste per capita are much higher than the differences in generation of municipal waste.

[3] ETC/SCP working paper 2/2009: EU as a Recycling Society - Present recycling levels of Municipal Waste and Construction & Demolition Waste in the EU, http://scp.eionet.europa.eu/publications/wp2009_2/wp/wp2009_2, 2009

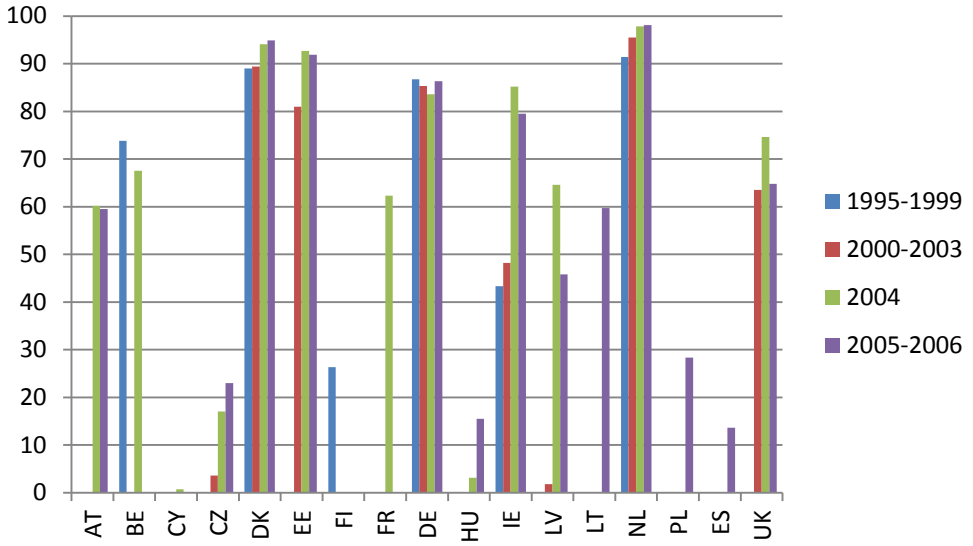
Figure 12 Generation of C&D waste (tonnes per capita), EU-27



The Waste Framework Directive (2008/98/EC) sets a target for 70% by weight preparation for re-use, recycling and other material recovery (including backfilling operations using waste to substitute other materials) of non-hazardous C&D waste by 2020.

Figure 13 shows the recycling rate for C&D waste in those MS for which data are available. The recycling rate in tonnes per capita differs among countries, but the differences are not so large as those seen for municipal waste. In addition, there is not a great difference in recycling rates between the old and the new MS. In 2004 (the year for which the most data is available), the recycling rate was 60% or over in 10 MS.

Figure 13 Recycling of C&D waste (% of amount generated)

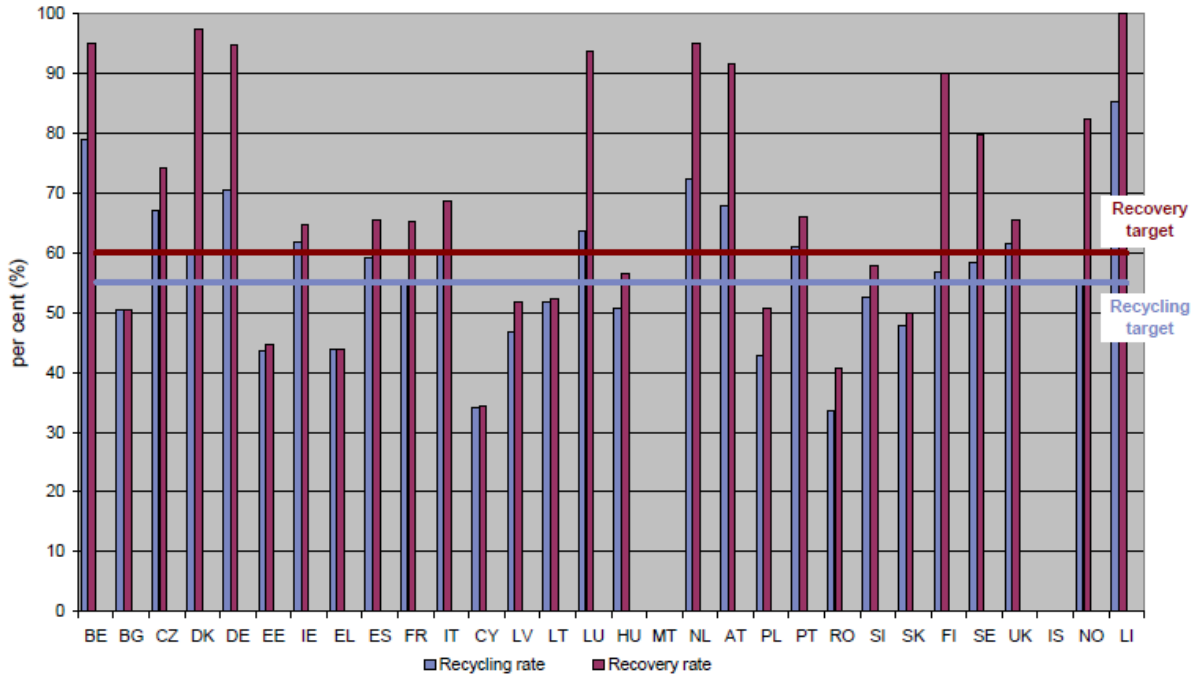


The **Packaging and Packaging Waste Directive** set the following targets for 2008:

- 60% recovery; and
- 55% recycling (50% for metal, 60% for glass, paper/cardboard, 22.5% for plastics and 15% for wood).

Figure 14 below shows the recycling and recovery rates for packaging waste in the 27 MS for 2008. It shows that, by 2008, 15 MS had met or exceeded the 55% overall recycling target and the 60% recovery target (Bulgaria, Cyprus, Estonia, Greece, Hungary, Latvia, Lithuania, Malta, Poland, Romania, Slovakia and Slovenia had not).

Figure 14 Recycling and recovery rates for packaging waste, 2008



The **WEEE Directive** set the following targets for 2006:

- Minimum collection rate of 4kg per inhabitant per year;
- 70-80% recovery (depending on category of WEEE); and
- 50-80% recycling including reuse (depending on category of WEEE).

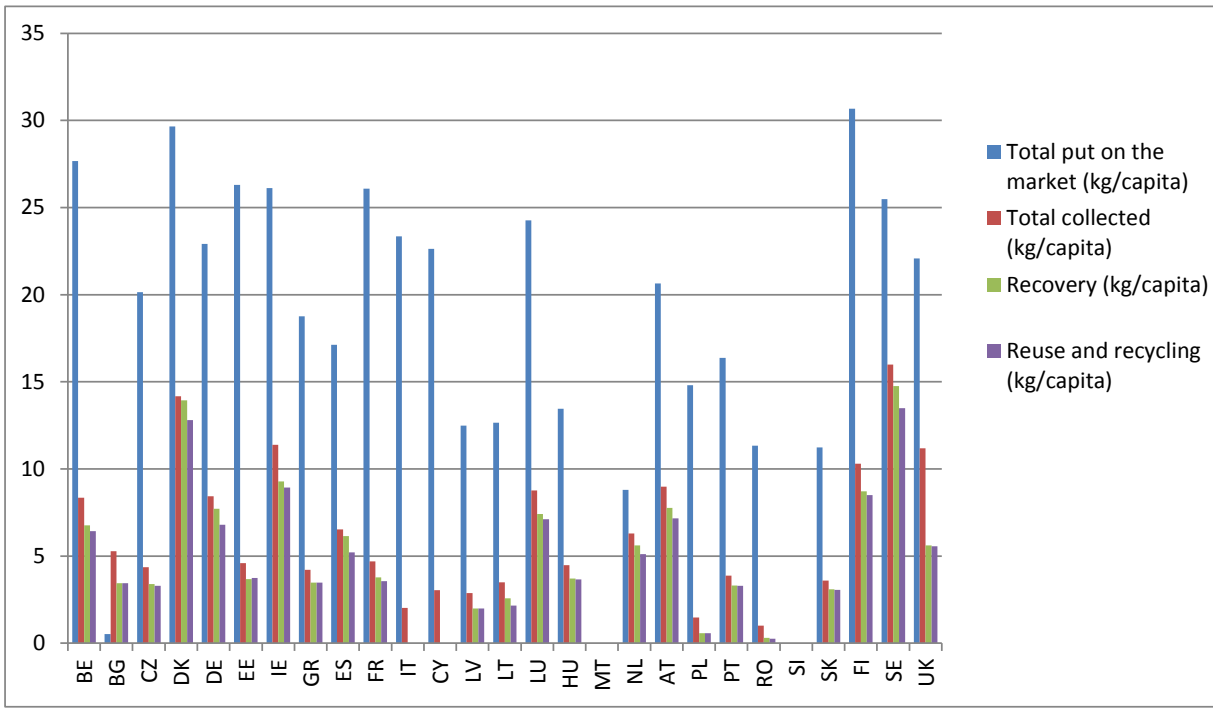
The **recast WEEE Directive** (which was in the final stages of approval at the conclusion of this study) will change these targets to the following by 2020:

- A collection target for 2020 of 65% of WEEE placed on the market (in the previous three years), (with an interim collection target of 45% by 2016); or
- A collection target of 85% of WEEE generated each year by 2020; plus
- 70-80% recovery by 2020 within 3 years of entry into force of the Directive (increasing to 75-85% after 6 years) (depending on category of WEEE); and
- 50-75% recycling including reuse within 3 years of entry into force of the Directive (increasing to 55-80% after 6 years) (depending on category of WEEE).

Figure 15 below shows that by 2008, 18 MS had reported meeting the 4kg per capita collection target (Cyprus, Italy, Latvia, Lithuania, Poland, Romania and Slovakia had not; data is not available for Malta and Slovenia). The average collection rate (for the 25 countries for which data were available, minus Bulgaria which appears to import large quantities of WEEE and would therefore skew the figures) was 31.5% by weight of amounts put on the market; an increase from 23% in 2006. It is likely that considerably more than this is collected but not reported, and that a

substantial part of this undergoes sub-standard treatment in the EU or is illegally exported. Where WEEE is collected separately, however, it is widely recycled: for 23 countries where recycling rates can be calculated, the average recycling rate was 75.8%.

Figure 15 WEEE put on the market, collected and recycled/recovered/reused (kg/capita), 2008



The **ELV Directive** set the following targets:

- Vehicles to be recoverable to a minimum of 95%, and reusable and/or recyclable to a minimum of 85% by 2005;
- 100% collection, 85% recovery and 80% recycling including reuse by 2006; and
- 100% collection, 95% recovery and 85% recycling including reuse by 2015.

Figure 16 below shows the reuse and recycling, and reuse and recovery rates for ELV in the 27 MS for 2008. By 2008, 22 MS had met or exceeded the 2006 reuse and recycling target (only Cyprus, France, Ireland and Poland had not; data was not available for Malta). Eight MS (Belgium, Estonia, Germany, Greece, Latvia, Lithuania, Slovakia and Slovenia) had already met or exceeded the 2015 reuse and recycling target. Also by 2008, 18 MS had met or exceeded the 2006 reuse and recovery target (Cyprus, Denmark, Finland, France, Hungary, Ireland, Poland and the UK had not; data was not available for Malta). Austria was alone in already having met or exceeded the 2015 reuse and recovery target.

Figure 16 ELV Reuse and recycling, and reuse and recovery rates, 2008

The **Batteries Directive** set the following targets:

- 100% recycling of collected batteries by 2009;
- 65% recycling for lead-acid batteries, 75% recycling for nickel-cadmium batteries and 50% recycling for other batteries by 2011;
- 25% collection rate by 2012; and
- 45% collection rate by 2016.

As the dates for the main targets have not yet arrived, data has not yet been gathered by Eurostat on the collection and recycling rates for batteries.

In addition to these specific targets, in September 2011 the European Commission published its Roadmap to a Resource Efficient Europe, which calls for a transition to an economy that recognises waste as a resource. According to the Roadmap, achieving resource efficiency will require that residual waste is close to zero, meaning there needs to be a much greater focus on re-use and recycling. The 2020 'vision' for waste is as follows:

> 'By 2020, waste is managed as a resource. Waste generated per capita is in absolute decline. Recycling and re-use of waste are economically attractive options for public and private actors due to widespread separate collection and the development of functional markets for secondary raw materials. More materials, including materials having a significant impact on the environment and critical raw materials, are recycled. Waste legislation is fully implemented. Illegal shipments of waste have been eradicated. Energy

recovery is limited to non recyclable materials, landfilling is virtually eliminated and high quality recycling is ensured.'[4]

The increased use of EIs in the waste management sector is anticipated by the Commission as a means to helping to attain this vision.

2.3.3 SUMMARY AND CONCLUSIONS

Data indicates that for 1995, 2002 and 2009, only six MS (Bulgaria, Estonia, Germany, Hungary, Lithuania and Slovenia) reported generating less municipal waste (both in absolute terms and in kg per capita) in 2009 than in 1995. The reasons for this are likely to be mixed. In the case of Germany in particular, it may include the implementation of very successful EIs (including landfill bans, PAYT schemes and producer responsibility schemes). For some of the newer MS, however, the decline in waste generation may instead be attributable to other factors, perhaps most notably the ongoing implementation of the EU waste *acquis* following their accession to the EU, and changes in data reporting and the definition of municipal waste.

In all MS that have municipal waste incineration, the amount sent to incineration in absolute terms was higher in 2009 than in 1995. This seems to suggest that waste treatment is at least moving away from landfill to incineration (with energy recovery), as a result of the progressive implementation of policies (and perhaps also EIs) to this end.

In the six MS with higher recycling rates and very low landfill rates, there is a combination of measures in place that help to achieve such waste management performance, including strong combinations of EIs and often restrictions/bans on landfilling of municipal waste.

The remainder of this report looks at the presence and impacts of EIs in the MS and attempts to relate these to the waste management performances outlined in this section.

[4] COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL COMMITTEE AND THE COMMITTEE OF THE REGIONS: Roadmap to a Resource Efficient Europe, COM(2011) 571 final, 20 September 2011

3 ECONOMIC INSTRUMENTS IN THE EU-27 MEMBER STATES

3.1 METHOD

In order to provide a solid foundation for the study, information was gathered from the beginning of the study on the use of EIs in the waste sector in the EU-27 MS. This information was verified and added to throughout the duration of the study.

A detailed template was developed to guide the information gathering process, and to try to ensure that the information gathered was as comparable as possible between MS. (The 27 MS templates are included in Annex 1 to this report.) This research was mainly desk-based and internet-based, consulting reports, journals and websites of relevant organisations (EEA, national ministries, OECD, industry associations etc). However, European-level industry associations were also sent tailored versions of the templates and several provided useful information on the use of EIs in their sectors. The associations contacted were: FEAD and CEWEP (waste management); ACR+ (local authorities/pay-as-you-throw); PRO-Europe (packaging); WEEE Forum (WEEE); EBRA and EPBA (batteries); ACEA and CLEPA (ELV); and CEPI (paper). The information gathered was also sent to the MS experts on the Technical Adaptation Committee (TAC) on waste management on two occasions, for verification and to help identify additional information sources to address current gaps in the information. By the time this draft final report was submitted, TAC members from the following MS had provided useful information: Austria, Belgium (Flanders), Cyprus, Ireland, Latvia, Luxembourg, the Netherlands, Portugal, Spain and the UK.

The latest headline data on waste management performance in the EU was gathered, most notably on waste generation, recycling (and composting) rates and waste disposal. The main focus was on municipal waste data, where the most complete data sets are available, and on wastes for which the EU has minimum recycling targets, namely packaging waste, WEEE, ELVs, batteries and construction and demolition (C&D) waste. Attempts are made in this report to identify relationships between this data on waste management performance and the use of EIs in the MS (bearing in mind that it can be difficult to draw causal links due to the influence of other factors, including the broader waste policy context in the MS).

Building on the data gathered in the 27 MS templates, more detailed information was gathered for nine MS (Austria, Belgium, Denmark, Estonia, France, Germany, Netherlands, Slovenia, and the UK), including information on: waste policy in the MS, waste management performance in the MS, more detailed information on the design and functioning of the EIs used (including their objectives and any problems/solutions/lessons learned from implementation), and the perceived costs (financial and administrative) of the EIs in use. For the nine selected MS, this more detailed version of the template was sent to the MS experts on the TAC on waste management, in particular to offer some assistance with the questions on problems/solutions/lessons learned and the perceived costs of the EIs.

It must be noted that throughout the study all efforts have been made to collect the maximum possible amount of data, but for some MS and some EIs data has been difficult to find. The study should not therefore be taken as presenting a fully comprehensive picture of the use of all EIs for waste management across the EU-27, but rather as presenting the most thorough picture possible given the time and budgetary constraints of the study.

3.2 LANDFILL TAXES

3.2.1 INTRODUCTION

Table 2 below provides an overview of landfill taxes, gate fees, restrictions (bans) and compliance with the Landfill Directive in the MS. It shows that 18 MS currently have **landfill** taxes in place for the disposal of non-hazardous municipal waste sent to legal landfills (a tax is planned for Lithuania for 2012). For this study, a distinction is made between:

- Taxes: a levy charged by public authorities (in most cases at national level, although is some cases (e.g. Italy, Spain) regional) for the disposal of waste in a landfill site, usually with an environmental purpose in mind, and where the revenue is accruing to the body responsible for the levy; and
- Gate fees: charges set by the operators of the landfills for the provision of the service (i.e. waste disposal) and which are designed to cover their costs and profit. NB This type of fee is subject to variation according to the landfill site used, and to other factors such as available landfill capacity and market variations. Gate fees do not always cover an operators' cost due to the market situation at a given time. In this report, the term 'gate fees' refers to the costs before the application of landfill taxes.

Throughout the report, the term 'charge' is used to refer to the sum of the prevailing level of tax and the gate fee, therefore representing the total cost of landfilling.

The level of taxation ranges very widely, and some countries levy no tax. Among the countries with taxes, the taxes vary from €3 per tonne in Bulgaria to up to €107.49 per tonne in the Netherlands. The total cost of disposal to landfill, however, depends also on the gate fee charged by the landfill for accepting waste. When this is factored in, the total cost to landfill one tonne of municipal waste in the EU appears to range from €17.50 in Lithuania to up to €155.50 in Sweden.

Information gathered during the study suggests that 14 MS have landfill restrictions in place for materials other than those specified in the EU Landfill Directive[5]. Outline information on the materials subject to landfill restrictions is also included in Table 2. Some of these are not as

[5] Article 5 of the Landfill Directive (1999/31/EC) effectively bans the landfilling of: liquid wastes; explosive, corrosive, oxidising, highly flammable or flammable wastes; hospital and other clinical wastes; whole used tyres; any other type of waste which does not fulfil the acceptance criteria determined in accordance with Annex II of the Directive.

restrictive as others (for example, the Estonia restriction), and some, such as those in Finland and France, do not appear to have been strictly enforced.[6]

Table 2 also provides an overview of the reported rate of compliance of non-hazardous and inert landfills in the MS with the Landfill Directive (1999/31/EC). This is based on information reported by MS to the Commission for the period 2004-2006[7] and represents the level of compliance of landfills in the MS with the requirements of the Landfill Directive as a whole. Existing landfills had until 16 July 2009 to comply with the requirements of the Directive, and the Commission is currently checking whether these targets have been met. The reported rate of compliance for non-hazardous landfills ranges from 5% or below (in Bulgaria, Cyprus and Latvia) to 100% (in Austria, Belgium, France, Ireland, Italy and Luxembourg). The reported rate of compliance for inert landfills ranges from 25% in Sweden to 100% in Austria, Belgium, Estonia, France, Ireland, Italy and Luxembourg. It is worth bearing in mind these compliance rates when reading the following analysis as it will affect the overall environmental impact of the quantities of waste being sent to landfill. However it does not necessarily follow that the compliance rate will have an impact on the prices charged for landfilling; gate fees are strongly influenced by the prevailing market conditions.

[6] For a review of bans, see Green Alliance (2009) Landfill Bans and Restrictions in the EU and US: Summary Evidence Review, Report for Defra, August 2009 Eunomia (2010) Landfill Bans: Feasibility Research, Final Report to WRAP, February 2010

[7] Information taken from MS responses to Commission Decision (2000/738/EC): Questionnaire relating to Directive (1999/31/EC) on the landfill of waste (Section II, Question 5)

Table 2 Overview of landfill taxes, fees and restrictions (non-hazardous municipal waste, legal landfills)

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
AT (1989)	€26	approx. €58 – 219	€70	approx.€96	Waste with a total organic carbon (TOC) content over 5% unless it has been biologically treated so that its fermentability is below 7mg O_2/g d.m. Hazardous waste (except asbestos) may not be surface landfilled. These rules are effectively implemented.	**Tax:** Inert/C&D: €8 Other stable and non-reactive waste: €18-26 **Gate fees:** Inert/C&D: €14 (sorted, clean); €84 (unsorted/ contaminated)	**Non-hazardous:** 146 (146) **Inert:** 303 (303)

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
BE (Fl. 1989; Wall. 2007)	**Fl.** €82.03; **Wall.** €65	approx. €50	€50	Fl. approx. €132.03 Wall. approx. €105	**Flanders:** unsorted household/industrial waste; sorted and non-sorted wastes collected specifically for recovery; combustible residues from sorting of household/ comparable industrial waste; wastes suitable for incineration; waste pharmaceuticals. **Wallonia:** separated household waste; non-hazardous industrial waste; waste from pre-treatment/sorting facilities; non-pre-treated fine residual household waste; bulky household waste (shredded and non-shredded); various packaging; various metals; batteries; fly ash/bottom ash[+]	**Tax:** Inert: Fl. €11.67; Wall. €5[+]	**Non-hazardous:** 5 (5) **Non-hazardous (mono-landfill):** 17 (17) **Inert:** 6 (6)
BG (2004)	€3	No specific range found, but indication that gate fees are very low	?	?	n		**Non-hazardous:** 548 (24) **Inert:** 39

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
CY	Excise tax for C&D waste disposal	€56 (Larnaca-Ammochostos District plant)	€56	?	?		**Non-hazardous:** 116 (1)
CZ (1992)	€20	approx. €12 - €20	approx. €16	approx. €36	?		**Non-hazardous:** 159 **Inert:** 65
DK (1987)	€63	€10 - €95	€44	approx. €107	Waste suitable for incineration (i.e. combustible waste)	**Tax:** Inert: €63	
EE (1990)	€12	approx. €40	€40	approx. €52	Unsorted municipal waste (municipalities must organise source separation of paper/cardboard, green garden waste, hazardous waste, packaging waste).	**Tax:** Inert: €12 C&D: €0.70+ **Gate fee:** Inert: €6.50 (sorted, clean); €42.5 (unsorted, contaminated)	**Non-hazardous:** 17 (5) **Inert:** 3 (3)
FI (1996)	€30	€44.40 - €74.40	€59.40†	approx. €89.40†	Waste that has not been pre-treated; household waste where the major part of the biodegradable fraction has not been separately collected; biodegradable waste; compostable waste. This ban has not been effectively enforced. (A ban on landfilling biowaste is likely to be approved in 2011.)		**Non-hazardous:** 137 (64) **Inert:** 7 (2)

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
FR (1999)	€11 - €20 (adjusted based on site's env. performance)	€45 – 76 (50% of fees between €57 - €73)	€60.50	€80.50	Non-residual waste (waste that cannot be further treated according to current technical and economic conditions). Different definitions applied to residual waste and the ban is not fully respected.	**Gate fee:** Inert: €2.90-€7.10	**Non-hazardous:** 40 (40) **Inert:** 40 (40)
DE	-	approx. €60 - €220	€140†	approx. €140†	Municipal waste that has not been pre-treated (e.g. by MBT or incineration), waste wood. A 'fermentability threshold' applies of 5mg O_2/g DM		
GR	-	approx. €10 - €37	€23.50†	approx. €23.50†	?		**Non-hazardous:** 41 (23)
HU	-	€35	€35	€35	n		**Non-hazardous:** 192 (61) **Inert:** 9
IE (2002)	€50+	approx. €56 - €81	€70†	approx. €120†	Untreated municipal waste (except inert waste) (minimum pre-treatment requirements apply – this is a complex system of requirements which change over time)		**Non-hazardous:** 27 (27) **Inert:** 3 (3)

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
IT (1996)	€10 - €50 (varies by region)[+]	approx. €60 - €120	€90†	approx. €120†	Combustible waste (> 13 MJ/kg, as of 1 January 2011)	**Tax:** Inert: €1 - €10	**Non-hazardous:** 234 (234) **Inert:** 195 (195)
LV (1991)	€8	-	€30	€38	Sludge with over 80% water content; organic, food, industrial waste, wood treatment waste not for composting/landfill gas collection; unidentified chemical waste	**Tax:** C&D: €14.10[+]	**Non-hazardous:** 100 (5)
LT	(€22 tax to be introduced in 2012)	€15 - €17.50	€16.25	approx. €16.25 (2010), to rise to approx. €38.25 (2012)	Biodegradable waste from gardens and parks; batteries; waste that has not been pre-treated (from 2013)		**Non-hazardous:** 131 **Inert:** 3
LU	-	€123.95 - €175	€149.48†	€149.48†	Waste that is not treated, e.g. by MBT		**Non-hazardous:** 2 (2) **Inert:** 10 (10)
MT	-	-	€20	€20	No info found		
NL (1995)	€107.49	approx. €20 - €30	€25†	approx. €132.49†	35 categories of waste (combustible waste)	**Tax:** Inert: €16.79[+]	22 (most landfills receive both non-hazardous and hazardous waste) (19)

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
PL (2001)	€25	approx. €58 - €81	€69.50	€94.50	?		**Non-hazardous:** 1082 (428) **Inert:** 11 (9)
PT	€3.50	€6 - €15 (est. 2005)	€10.50	€14	n		**Non-hazardous:** 55 (53) **Inert:** 8 (6)
RO	-	€2.80-€4.60	€3.70	?	n		**Non-hazardous:** 370 (33) **Inert:** 2 (1)
SE (2000)	€49 (only charged to landfills over 50 tonnes capacity)[+]	approx. €78 - €135	€106.50†	approx. €155.50†	Combustible waste; organic waste		**Non-hazardous:** 217 (47) **Inert:** 36 (9)
SK (2004)	-	€0.33 - €13.28	€6.80	€6.80	Waste that is not inert or stabilised		**Non-hazardous:** 128 **Inert:** 18
SI (2001)	€11	approx. €36 - €175	approx. €105.50	approx. €116.50	Untreated biodegradable waste (e.g. by anaerobic digestion of at least 20 days of which at least 24 hours > 55°C). Enforcement is questionable.	**Tax:** Inert: €2.20	**Non-hazardous:** 69 (40) **Inert:** 12 (4)
ES (2004 in Catalonia)	**Catalonia:** €12 (€21 if biowaste not collected at source)	approx. €13-€52.50	€32.75	€44.75 (€53.75 if biowaste not collected at source)	n	**Tax:** **Madrid:** C&D: €3/m^3 **Murcia:** Inert: €3	

Member State (year tax introduced)	Current landfill tax rate (per tonne)	Current landfill gate fees (range per tonne)	Current landfill gate fee (typical per tonne)	Total typical charge for landfill (per tonne)	Landfill restrictions	Current maximum charges per tonne for non-MSW	Number of landfills (number complying with Dir. 1999/31/EC in brackets)
UK (1996)	approx. €64.40	approx. €12.69 - €50.76	€26.80	approx. €91.20	(Planned for Scotland for mixed unsorted waste, source-segregated dry recyclates (2015), source-segregated food waste (2015) and biodegradable waste (2017); under consideration in England and Wales.)	**Tax:** Inert: approx. €2.85	**Non-hazardous:** 339 (264) **Inert:** 416 (123)

NB The term 'typical' is intended to denote levels most likely to be observed in the MS concerned. Gate fees will vary for a number of reasons (as indicated by the ranges given in the table), but ranges often cover fairly extreme examples. The intention here is to indicate something approximating the most likely level of gate fee in the MS concerned, though this is not based upon a rigorous scientific analysis in all MS (in some, it does actually represent the median, or most likely value, of the gate fee).

The tax figures have been cross-checked with a recent EEA report[8], though the figures for Slovenia in the table above reflect a recent change in the landfill tax.

† *These figures are mean not median, therefore may not reflect actual fees in place.*

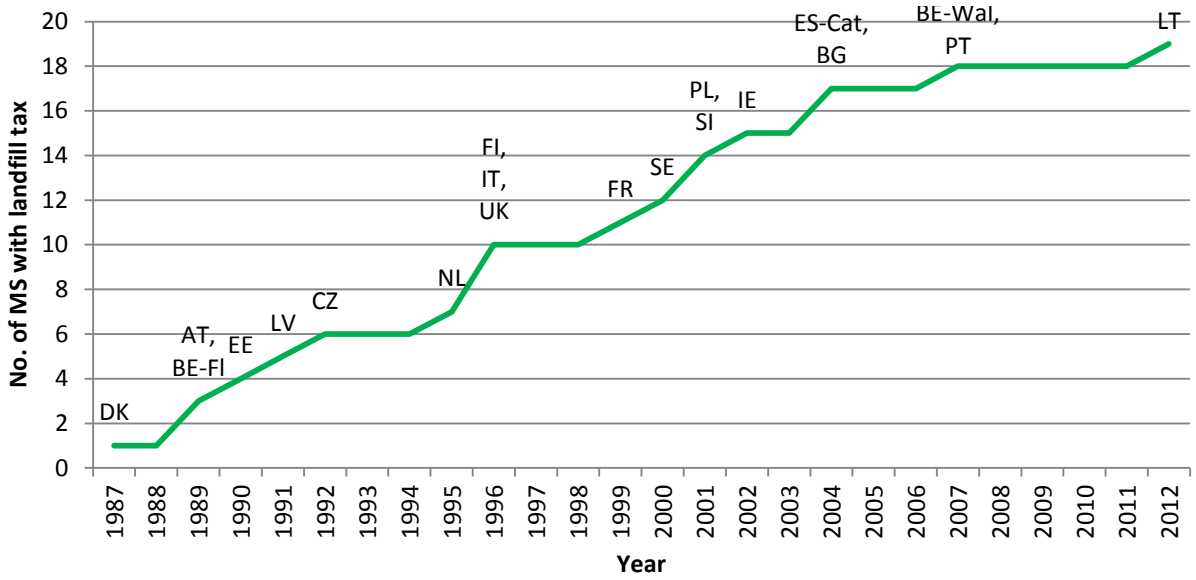
[+] *Data from: ETC/SCP Working Paper: Overview of the use of landfill taxes in Europe, Draft June 2011*

[8] ETC/SCP working paper (September 2011 draft), Overview of the use of landfill taxes in Europe, Christian Fischer and Mathias Lehner

3.2.2 MUNICIPAL WASTE

Figure 17 below shows the evolution in the number of MS putting in place landfill taxes for the disposal of non-hazardous municipal waste in legal landfills. Denmark was the first to introduce such a tax in 1987, 12 MS had taxes by 2000, and in 2012 the number will rise to 19 MS when Lithuania introduced its planned landfill tax, leaving only 7 MS with no tax.

Figure 17 Number of MS that have introduced a landfill tax for the disposal of non-hazardous municipal waste in legal landfills, 1987-2012

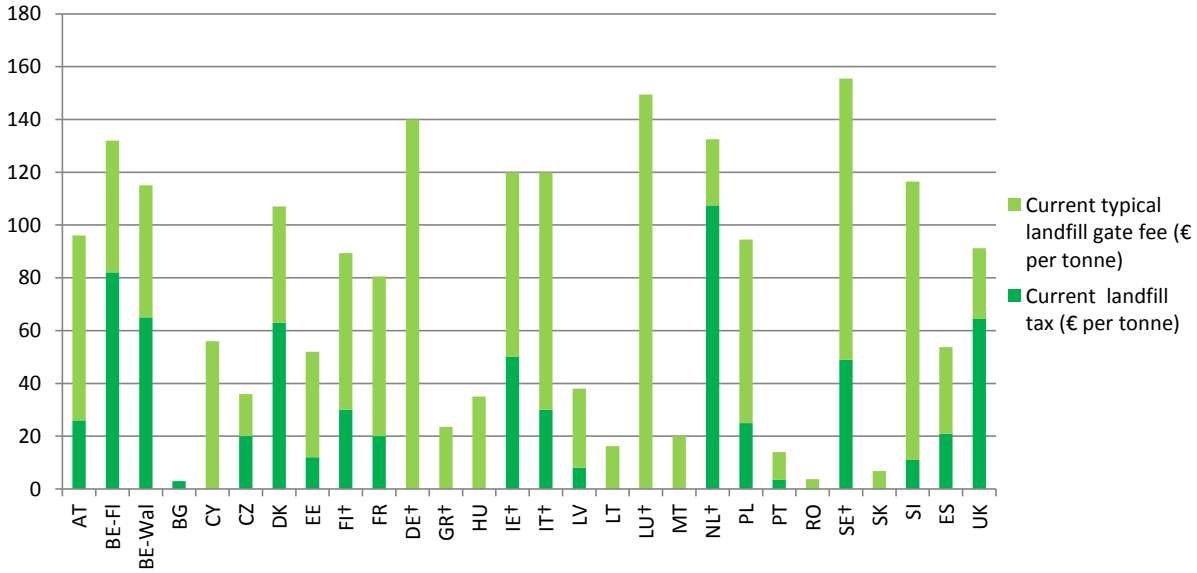


(BE-Fl = Flanders, Belgium; ES-Cat = Catalonia, Spain; BE-Wal = Wallonia, Belgium)

Figure 18 below provides a graphical overview of the data collected on the total typical cost of landfill for non-hazardous municipal waste deposited in legal landfills. It should be noted that these data refer to the typical level of gate fee prevailing, and the typical level of tax. It is well known that landfill gate fees may vary regionally for a variety of reasons, whilst some countries, notably Italy and Spain, are known to apply landfill taxes that vary between regions. The figures for taxes found during this study are broadly similar to those within the previously-mentioned EEA study (which contains no information on gate fees), although the exchange rates used are different in some cases (e.g. Sweden) and some more recent information has been used in this study (e.g. Slovenia).

Figure 19 below shows the total typical cost of landfill for non-hazardous municipal waste deposited in legal landfills, showing the range of gate fees. The total cost including the minimum gate fee is shown by the sum of the blue and red bars; the total cost including the maximum gate fee is shown by the sum of all three bars.

Figure 18 Overview of total typical cost of landfill (non-hazardous municipal waste, legal landfills)



NB Gate fee data are the latest available; tax data refers to 2011 as far as possible.

† *Figures based on the mean not median, therefore may not reflect actual fees in place.*

Figure 19 Overview of total cost of landfill (non-hazardous municipal waste, legal landfills) showing minimum and maximum gate fees

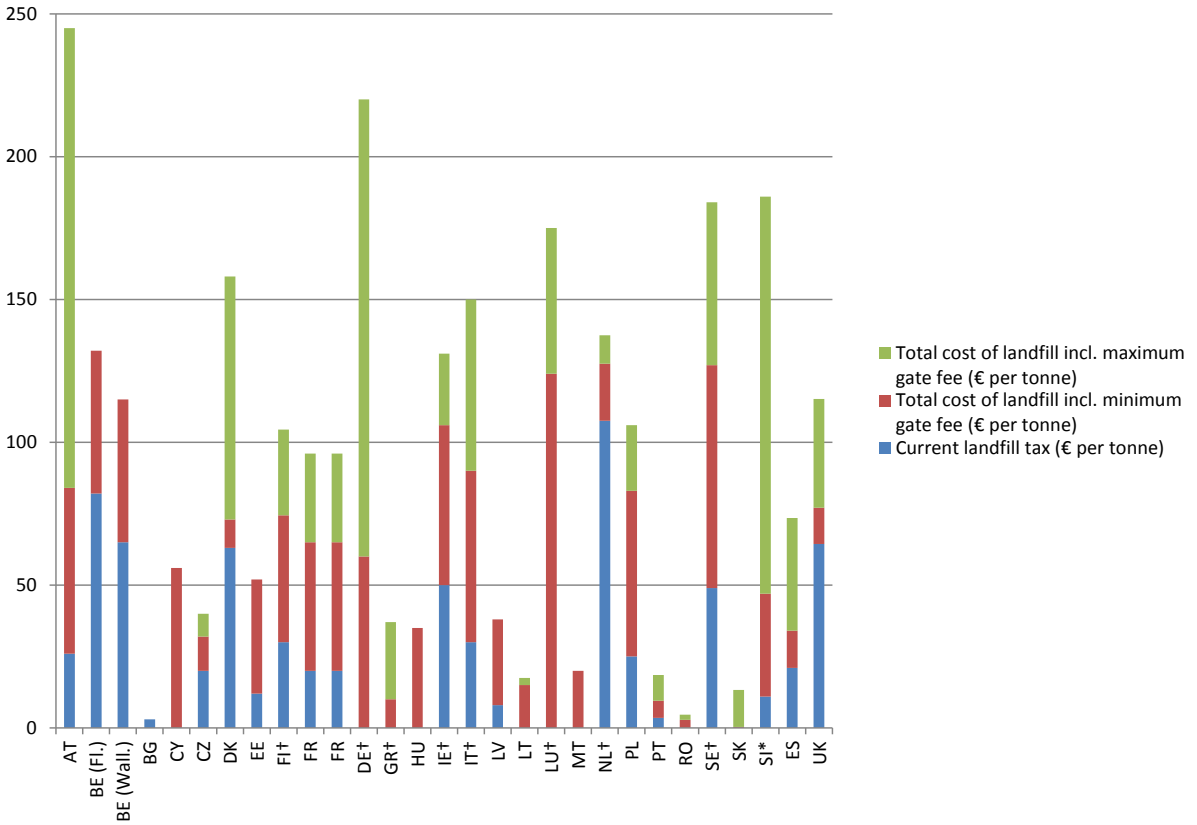


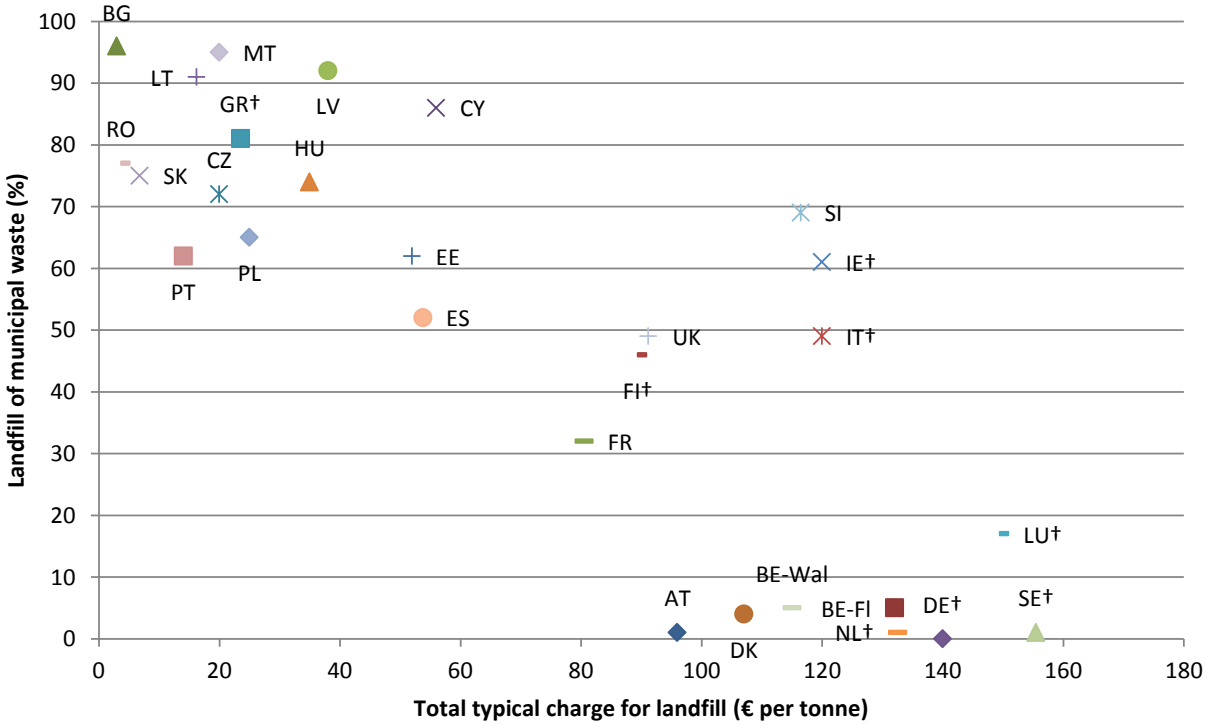
Figure 20 below compares the total typical landfill charge (based on available data; gate fees are missing for some MS) with the percentage of municipal waste sent to landfill in the MS. It shows three reasonably distinct groups of MS:

- MS with high total charges for landfill and low percentages of municipal waste landfilled (bottom right). It should be noted that these countries all have a landfill restriction in place which affects most residual municipal waste, and is well enforced. It is reasonable to believe that the bans also have an influence on forcing landfill rates down to low levels for perhaps obvious reasons (if a ban on landfilling is well-enforced, the landfill rate will, by definition, be at or close to zero);
- MS with mid- to high-range total charges and mid-range percentages landfilled (centre). These MS do not have bans in place that affect all residual municipal waste, so landfilling of residual waste continues. The effect of the tax is seen mainly in the level of recycling achieved; and
- MS with low total charges and high percentages landfilled (top left): all MS with total charges of less than €40 are landfilling more than 60% of their municipal waste.

This analysis suggests that there is a strong relationship between higher total landfill charges and lower percentages of municipal waste being sent to landfill. However, the picture is complicated by the existence of landfill restrictions. The interaction between the two has been examined for countries with bans in place in an earlier report. The presence of a well-enforced landfill restriction (particularly one for unsorted or untreated municipal waste) was shown to be in large part responsible for the movement of waste away from landfill and into alternative forms of residual waste treatment. The suggestion, therefore, is that the considerable gap in performance between the first (bottom right) and second (centre) groups of MS is explained not only by the landfill tax, but also by the implementation of well-enforced bans or restrictions that effectively demand that residual municipal waste cannot be landfilled without some form of pre-treatment (at least six MS have bans on the landfilling of untreated MSW, at least six have bans on organic/compostable/biodegradable waste, and at least five have bans on combustible waste).

Whilst this is not to say that the tax or total charge has a negligible impact, it offers an indication that in the majority of MS the rate of landfill tax tends to have an impact on the percentage of waste sent to landfill up to a certain point, and that further improvements in performance after that point can also be attributed to the restriction on landfilling. Indeed, close inspection of the sequencing of events shows that in several MS with landfill restrictions in place, high taxes were implemented only after the restriction was implemented, partly as a way of discouraging resort to exemptions from the restriction. (Examples from near-zero-landfilling MS include: Denmark, where the tax rose from €23 to €44 one year after the restriction; the Netherlands, where the tax was €12 when the restriction was announced and is now close to ten times that level; and Sweden, where the tax was 250 SEK before the restriction and has increased to more or less double that since the restriction was introduced. In Austria, a rise in tax was implemented in anticipation of a restriction entering into force.) Evidently, if taxes were much higher, they could achieve much the same affect as that of a restriction.

Figure 20 Total typical landfill charge and percentage of MSW landfilled, 2009



As a means to try to understand the effects of the tax (as opposed to the combined effect of restrictions and bans alongside taxes), Figure 21 below compares the total typical landfill charge (based on available data; gate fees are missing for some MS) with the percentage of waste recycled and composted in the MS (rather than the percentage of waste landfilled, which is by definition more directly affected by a restriction or ban). Landfill bans will, where effectively implemented, clearly shift waste from landfill to other residual waste treatments (such as incineration and MBT). Therefore where countries are not treating a large proportion of residual waste through incineration and MBT, this indicates the strength of the price effect on the motivation for recycling and composting. Figure 21 suggests a fairly clear and linear correlation; as MS charge more for landfilling, the percentage of MSW recycled and composted increases. Evidently, other policies (including those to promote recycling, extended producer responsibility schemes and PAYT schemes) also influence recycling and composting rates, but this does give an indication of the effect of the tax independent of any landfill restrictions and bans mentioned in the description of Figure 20. It does appear reasonable to state that in addition to simply reducing the amount of waste sent to landfill, higher landfill charges tend to push waste towards recycling and composting, therefore moving waste treatment up the waste hierarchy. Evidence in respect of waste prevention is much weaker where municipal waste is concerned, though clearly, higher costs of disposal provides a more fruitful environment for the application of pay-as-you-throw

schemes (discussed below).[9] In addition, the UK case study below suggests that the tax may have played a more important role in encouraging waste prevention in the commercial and industrial waste sectors.

It should be noted that some of these figures may include the effects of recycling from incinerator bottom ash and MBT so that they do not necessarily eliminate the influence of landfill restrictions and bans on the picture being presented (this may be significant for example for Denmark and the Netherlands; bottom ash may contribute almost half of the recycling rate in Denmark). It also indicates that – significantly given the targets under the Waste Framework Directive – Member States are much more likely to meet a 50% recycling target once landfill charges (or the cost of the cheapest disposal option) approach €100 per tonne. Such charges will tend to drive the economics of recycling and composting (which are dominated by the avoided costs of residual waste management).

Figure 21 Total typical landfill charge and percentage of MSW recycled and composted, 2009

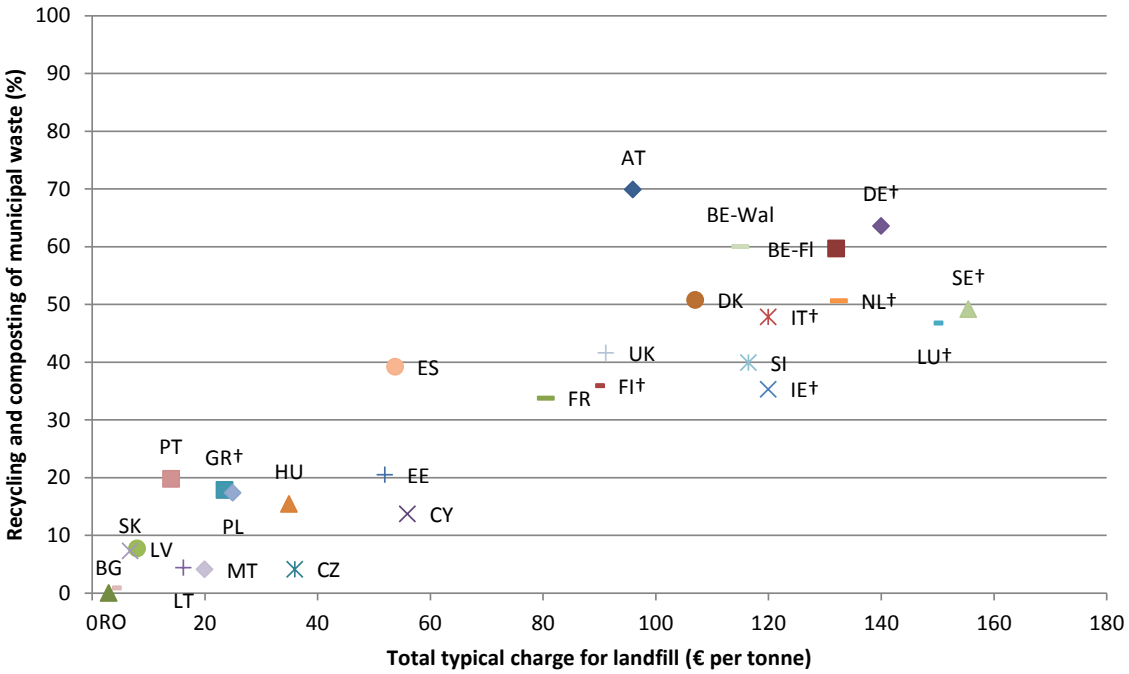


Figure 22 below compares the current highest landfill tax rates for municipal waste with GDP in the 18 MS with landfill taxes. This figure suggests that the majority of MS with GDP around

[9] See, for example, H. Bartelings, P. van Beukering, O. Kuik, V. Linderhof, F. Oosterhuis, L. Brander and A. Wagtendonk (2005) *Effectiveness of Landfill Taxation*, R-05/05, Report Commissioned by Ministerie von VROM, November 24, 2005; Eunomia Research & Consulting *et al.* (2009): International Review of Waste Management Policy: Annexes to Main Report. Bristol, UK. See URL http://www.environ.ie/en/Publications/Environment/Waste/WasteManagement/FileDownLoad,21598,en.pdf, accessed 09/09/2011.

€20,000 per capita or lower (Bulgaria, Czech Republic, Estonia, Latvia, Poland, Portugal and Slovenia) tend to have correspondingly lower landfill tax rates. There is a broader mixture of approaches in the MS with comparatively higher GDP (€25,000 per capita or above). Four MS (Austria, France, Italy and Spain) apply relatively low tax rates (€30 or below); six MS (Belgium (Wallonia), Denmark, Finland, Ireland, Sweden and the UK) apply mid-range tax rates (in the €40-€65 range); and Belgium (Flanders) and the Netherlands apply high landfill tax rates (over €80) (the Netherlands also has the highest GDP per capita).

Figure 22 Landfill tax rates compared with GDP for the 18 MS with landfill taxes, 2009

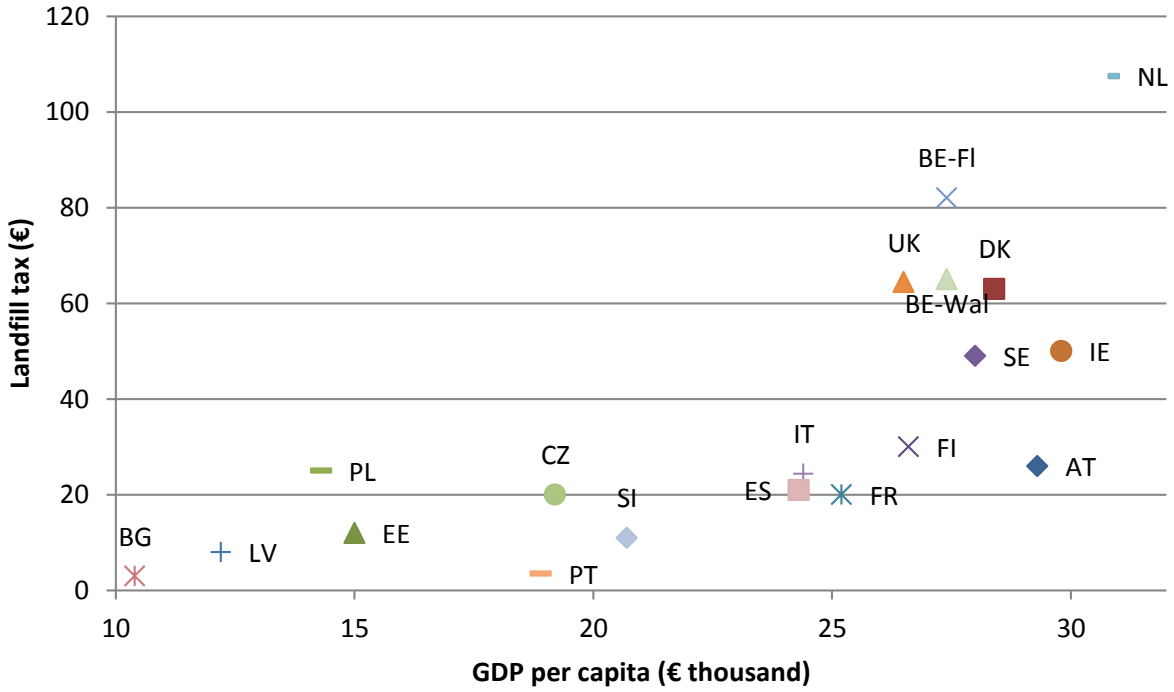


Figure 23 below shows the evolution of landfill tax rates over time for the 11 MS where adequate time series data have been found. The data clearly demonstrates that the tax rates tend to increase over time, though in some cases they remain constant for an extended period. A constant nominal tax rate implies that the incentive effect of the tax will be eroded by inflation. For this reason alone, it would seem important to increase tax rates over time. MS seeking to make greater progress in waste management are likely to increase tax rates over time above the rate of inflation (unless the approach is more to 'shock' the system, which might have unforeseen consequences).

Figure 23 Evolution of landfill tax rates over time, 11 MS

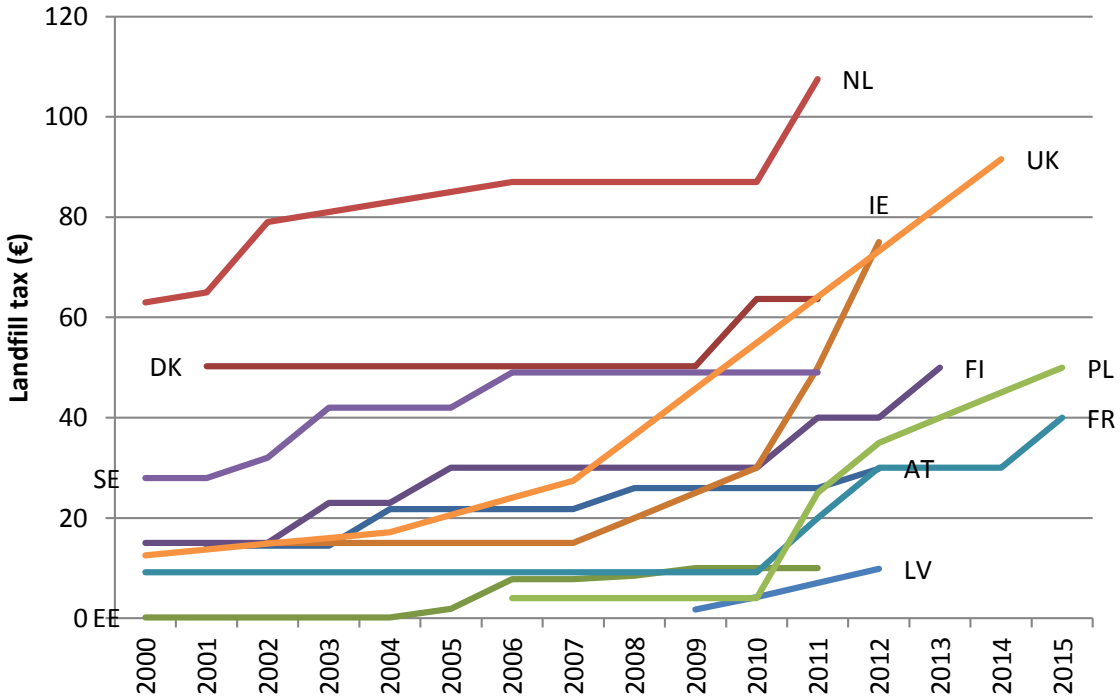


Figure 24 below compares, for the 11 MS where adequate time series data have been found, trends in the rate of landfill tax (for non-hazardous municipal waste) with trends in the percentage of municipal waste (MSW) generated that is sent to landfill. The figure takes into account landfill tax only, not the total charge (including gate fees). The figure shows strong *apparent* correlations between increasing landfill tax rates and decreasing rates of landfill for MSW in three MS in particular (Austria, Sweden and the UK). In Austria, the ban on landfilling of waste with total organic carbon (TOC) content over 5%, and waste exceeding the 7mg O_2 / g d.m., came into effect in 2004 (with exemptions until 2008); this is particularly relevant to MSW, hence the strong decline in the rate of MSW landfilled from 2004 onwards. In Sweden, a ban on landfilling combustible wastes was introduced in 2002 in order to reduce environmental burdens and promote resource efficiency, and a landfill ban on compostable waste was introduced a few years later (with exemptions for waste with a Total Organic Content (TOC) of less than 10%); both of these are likely to have contributed to corresponding drops in the rate of MSW landfilled, in conjunction with the increasing landfill tax rates. In the UK, the landfill tax was supported by two key measures. The first was the introduction of statutory recycling targets for local authorities in England, and the second was the Waste and Emissions Trading Act 2003,[10] which provided the statutory basis to set up the Landfill Allowance Schemes (LASs); in England, the LAS takes the form of a cap and trade scheme (the Landfill Allowances Trading Scheme) whereby local authorities are given annual allowances for the amount of biodegradable municipal waste they may send to landfill, and may trade those allowances based on a market price. The first full year of implementation of LATS was 2005-6,

[10] http://www.legislation.gov.uk/ukpga/2003/33/contents

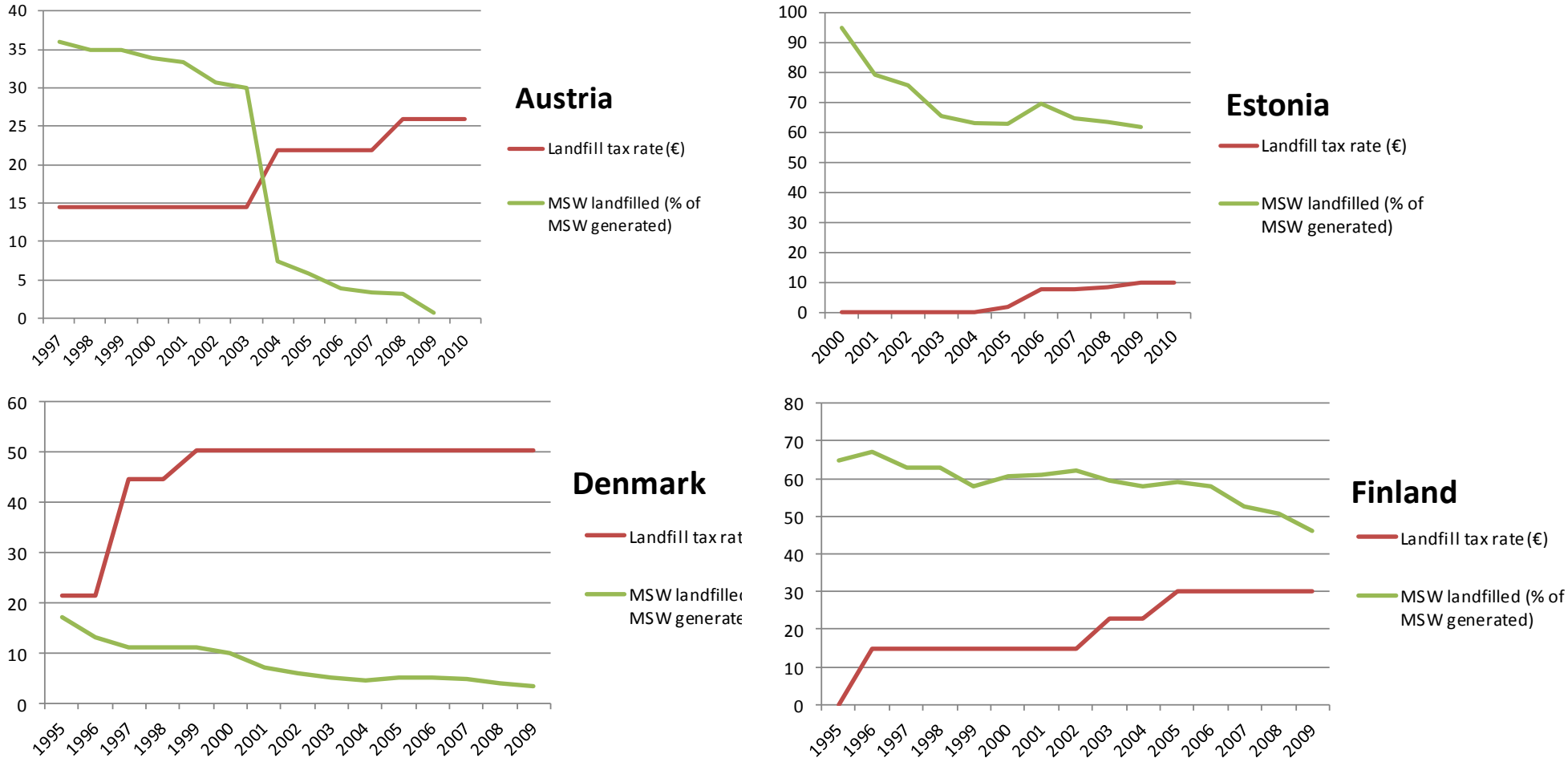
which may be a contributing factor in the maintained downward trend in rates of landfilling. The system will be disbanded in England at the end of 2012/13. The UK does provide some indication of a case where the landfill tax has exerted a strong influence, given that it has not implemented any landfill ban or restriction (other than those required under the Landfill Directive). The evidence for non-household waste is much stronger, and is examined below.

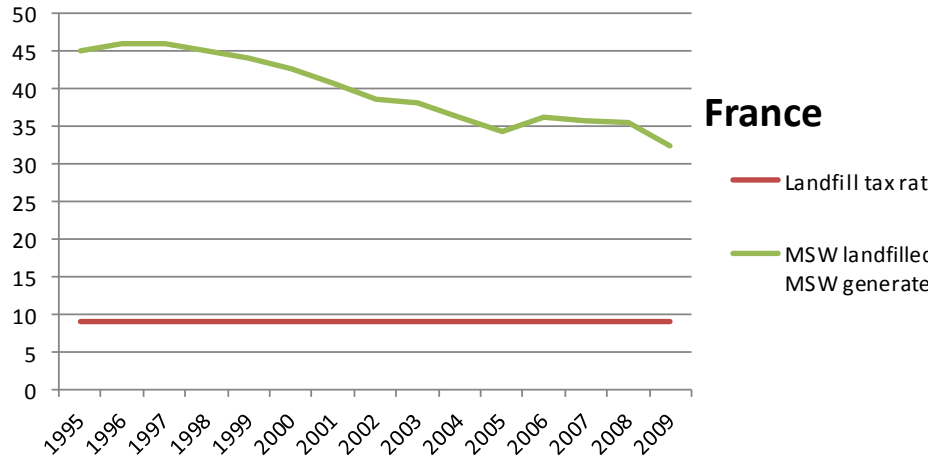
In three other MS (Estonia, Finland and the Netherlands), some slight *apparent* correlation can be observed. In a further five MS (Denmark, France, Ireland, Latvia and Poland), however, there is no distinguishable correlation between changes in the rate of landfill tax and the amount of MSW sent to landfill (which does not mean that this might not happen in future). However, it must be stressed that there will have been other factors at play, including broader waste policy in the MS (including the financing provided for different waste treatment infrastructures), the use of other EIs in the waste sector, the economic situation over time, changes in the gate fees charged by landfill sites (which may vary for reasons such as market conditions or changing technical or environmental standards), the available capacity of landfills and so on. Therefore we must be very wary, even where there are strong *apparent* correlations between the level of landfill tax and the amount of waste sent to landfill, of inferring *direct* correlations between the two factors. Equally, we should not assume that the apparent absence of an effect does not mean there has been no influence exerted by the tax.

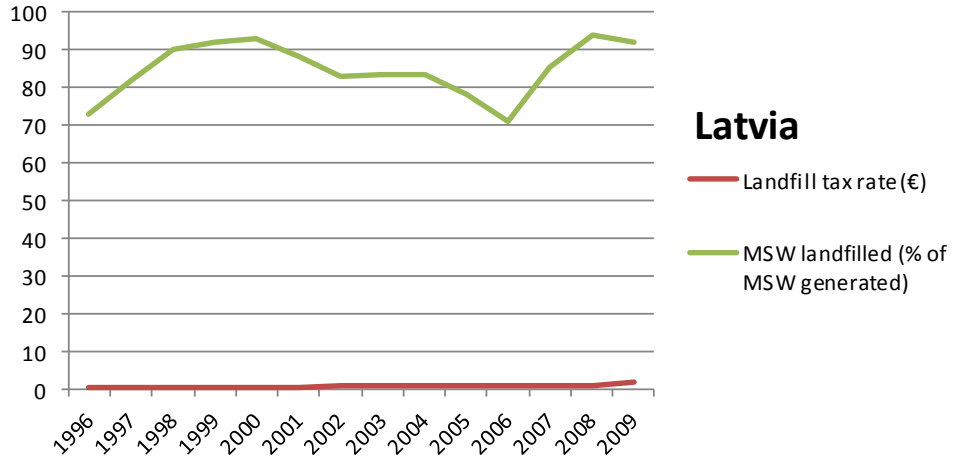
A 2009 European Environment Agency (EEA)[11] report suggests that to be effective a landfill tax level should be relatively high, but that public perceptions of the tax burden are arguably as important as the tax rate itself. The example of Estonia is cited, where although the tax rate is low compared with many other MS, and fairly low in the context of Estonian prices, it did increase considerably within a few years, and the rapid increase had the effect of Estonian waste companies and municipalities considering it to be a 'high' tax, which had some effect on reducing waste sent to landfill.

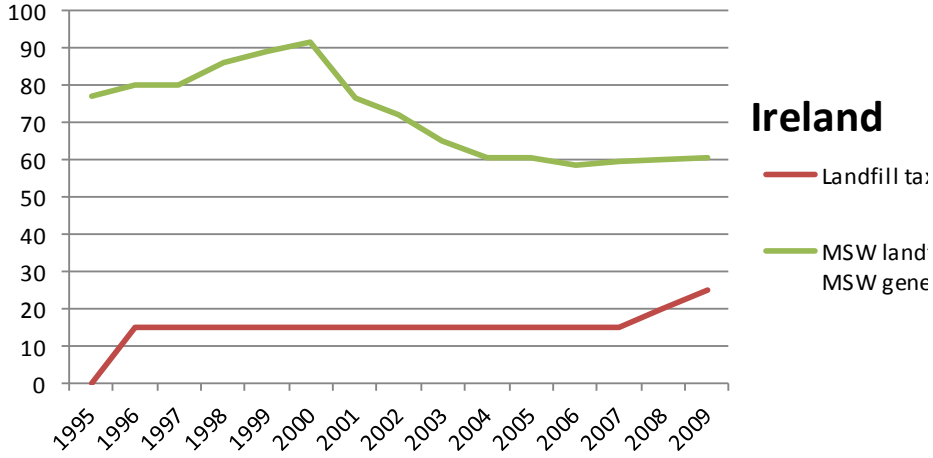
[11] EEA, Diverting waste from landfill: Effectiveness of waste management policies in the European Union, Report 7/2009, Cophenhagen, 2009, http://www.eea.europa.eu/publications/diverting-waste-from-landfill-effectiveness-of-waste-management-policies-in-the-european-union

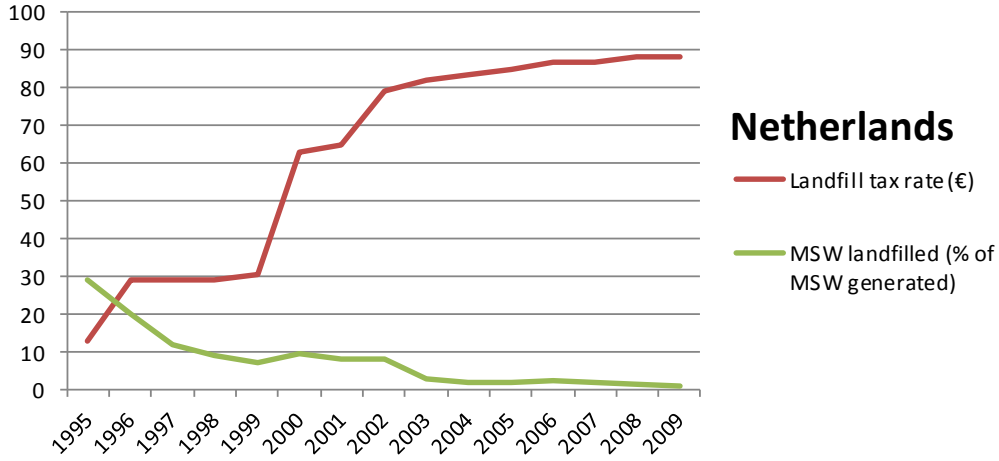
Figure 24 Landfill tax rates compared with percentage of municipal waste sent to landfill (AT, DK, EE, FI, FR, IE, NL, PL, SE, SK, UK)

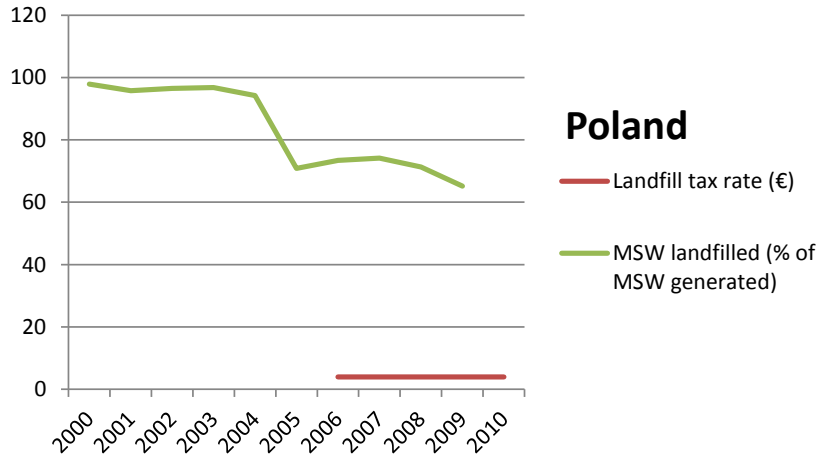


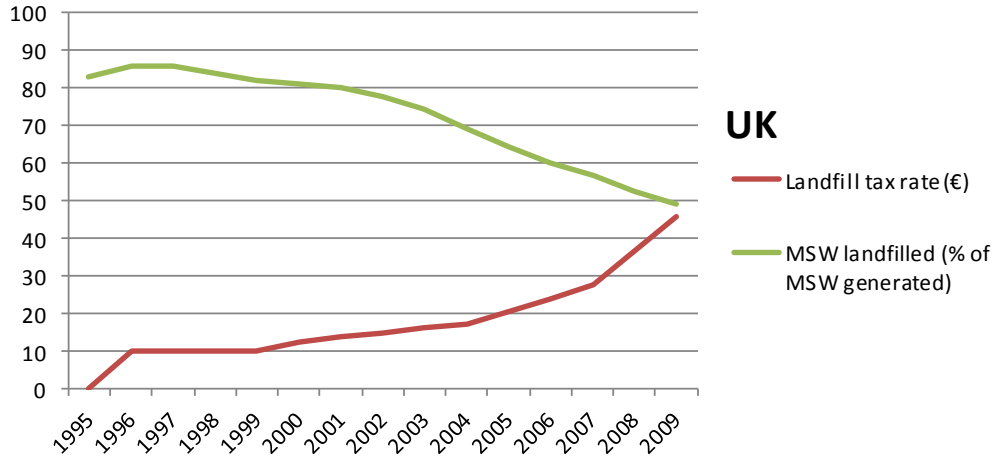

France
Landfill tax rat
MSW landfille
MSW generate

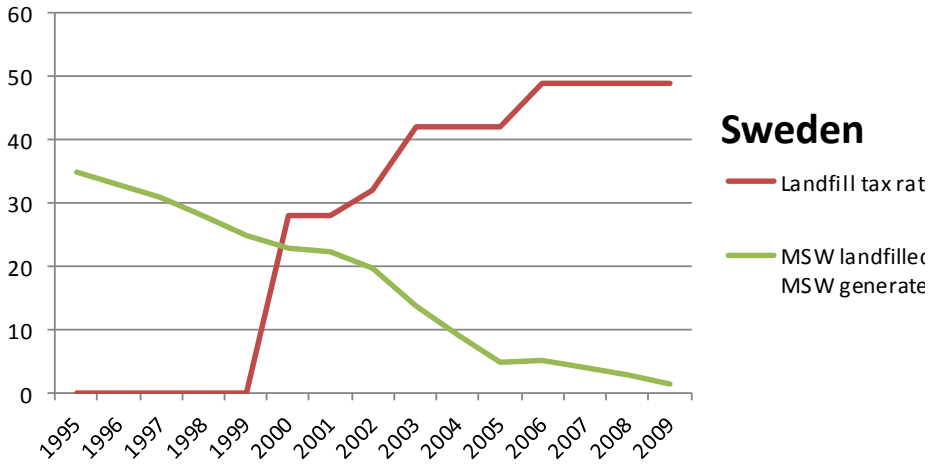

Latvia
Landfill tax rate (€)
MSW landfilled (% of MSW generated)


Ireland
Landfill ta
MSW land
MSW gene


Netherlands
Landfill tax rate (€)
MSW landfilled (% of MSW generated)





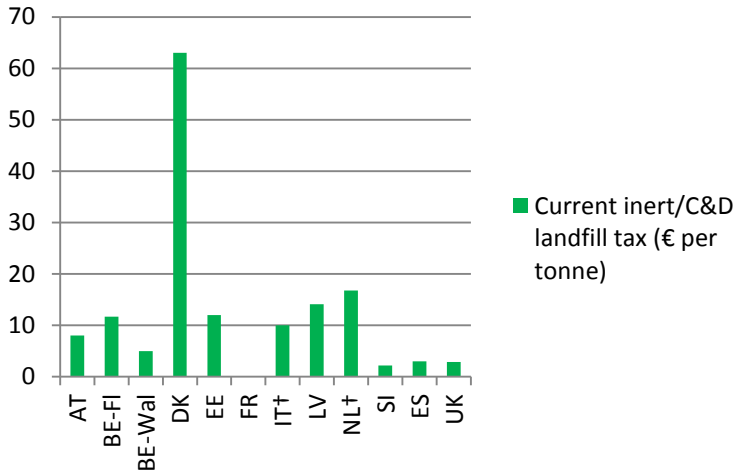


All MSW landfilled data for the above figure from Eurostat; data on landfill tax rates from various sources

3.2.3 INERT AND CONSTRUCTION AND DEMOLITION (C&D) WASTE

Figure 25 below provides an overview of the landfill tax for inert waste (including construction and demolition (C&D) waste) deposited in legal landfills, for ten MS. Gate fees are not included as for inert and C&D waste they are simply not as well known, and also tend to vary over time. It should be noted that such waste is sometimes received free of charge at landfills in order to help manage the sites (in some MS, some landfill sites may even pay for receiving such waste if the material is mainly e.g. soil or clay, especially if a landfill tax is in place). In the majority of MS where data was collected on the cost of both municipal and inert/C&D waste, the cost of landfilling inert/C&D waste appears lower than that of municipal waste. Indeed, for MS, it is likely to be close to zero. No regulatory distinction is made in Denmark between the two types of waste for tax/gate fee purposes. In Estonia and Latvia, the tax applied to inert/C&D waste is higher than that applied to municipal waste.

Figure 25 Overview of landfill tax for inert waste (including C&D waste) (legal landfills), selected MS

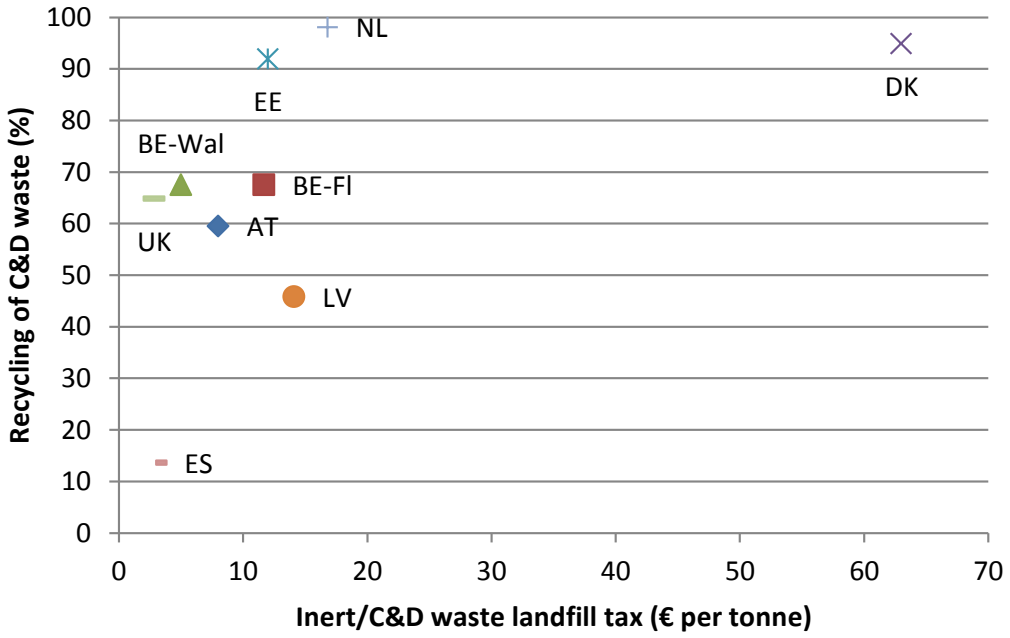


3.2.3.1 CONSTRUCTION AND DEMOLITION WASTE MANAGEMENT PERFORMANCES

It should be noted that the availability of data on the generation and recycling of C&D waste is generally poor, and it is therefore difficult to draw meaningful comparisons or conclusions of the MS waste management performance for this type of waste.

Figure 26 below, however, provides a graphical overview of the inert/C&D tax rate and the percentage of C&D waste recycled (it should be noted that the tax rates are the latest available, whereas data on recycling have only been found from 2004 or 2005). Based on the figures obtained, whilst the MS with the highest tax rate (Denmark) is amongst the best performing in terms of recycling and the MS with the lowest tax rate (Spain) is amongst the worst performing, there does not appear to be a particularly significant relationship between the tax rate and the amount of waste recycled.

Figure 26 Inert/C&D waste landfill tax and percentage of C&D waste recycled



NB Tax rates are the latest available; recycling rates are from 2005 in all cases except Belgium, where figures are from 2004

3.2.4 COUNTRY CASE STUDIES

In the following sections, we consider the situation in three countries:

- The UK, where the major instrument affecting the landfill of waste has been the landfill tax;
- Austria, where use has been made of both a tax and a landfill restriction; and
- Germany, where no landfill tax is in place, and where the major effect on landfill has come through a ban.

3.2.4.1 LANDFILL TAX IN THE UK

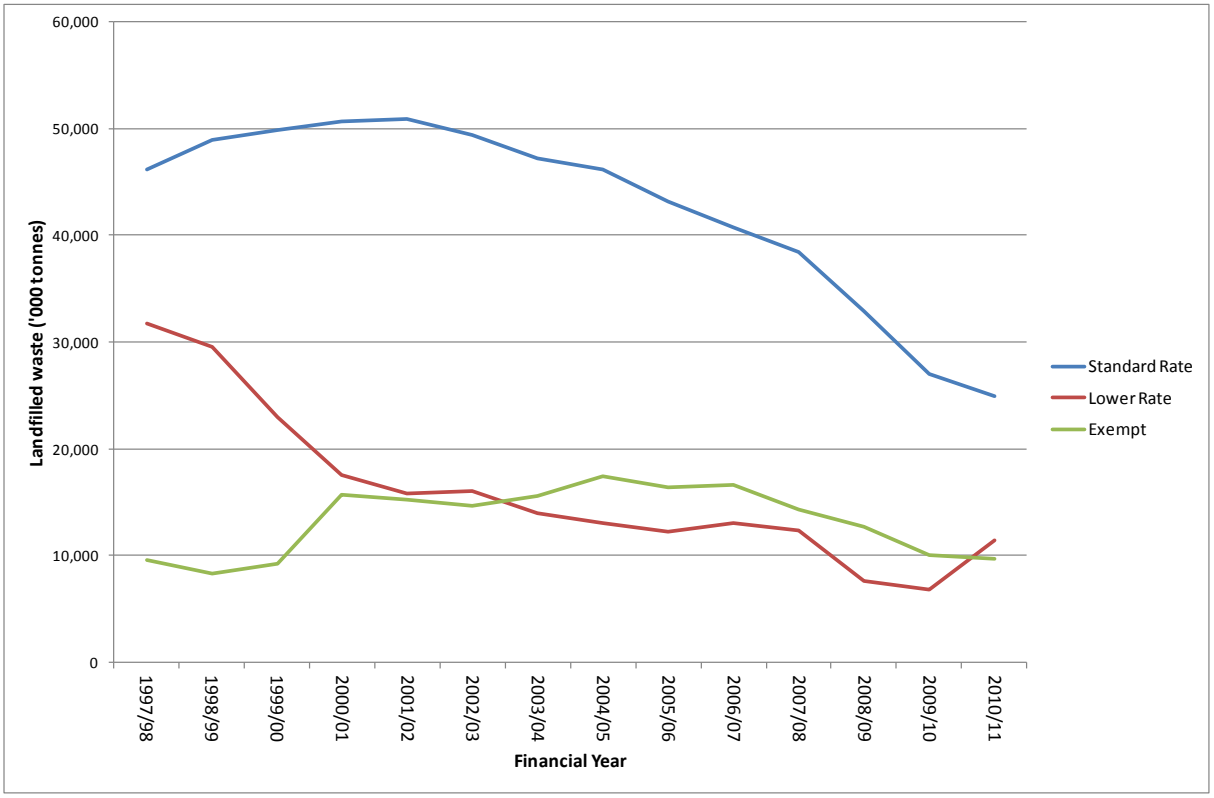
The UK Landfill Tax on landfilled waste was introduced in October 1996 and affects all sectors of the economy. At the time of inception, its purpose was to reflect the environmental impact of landfilling, to prevent waste generation and 'to recover value from more of the waste that is produced.' The tax is applied at two rates: a standard rate, applied to a range of materials, including household waste; and a lower rate, applying to specific 'qualifying materials', typically, those deemed to be 'inert', including materials such as rubble. Some materials are exempt from tax. Many of these exemptions relate to the use of material in the operation of the landfill itself, or in some necessary access roads to the site.

As of April 2011, the standard tax rate was approximately €67 (£56) per tonne. This is expected to rise to approximately €96 (£80) per tonne, maximum, by April 2014. The tax rate for inert wastes has remained relatively steady with only a slight increase to €3 (£2.50) per tonne in 2007, since which time the level has remained the same. (The rates in € vary somewhat, of course, with the exchange rate; the conversions here are based on the rate at the end of January 2012.) The revenues generated from the tax are directed to the Landfill Communities Fund, WRAP, LASU, WIP and various other Defra schemes.

Impacts

Figure 27 demonstrates how rates of waste diverted to landfills at the standard and lower tax rate have changed with tax escalation.

Figure 27 Quantities of Waste Landfilled in UK ('000 tonnes) at Different Tax Rates

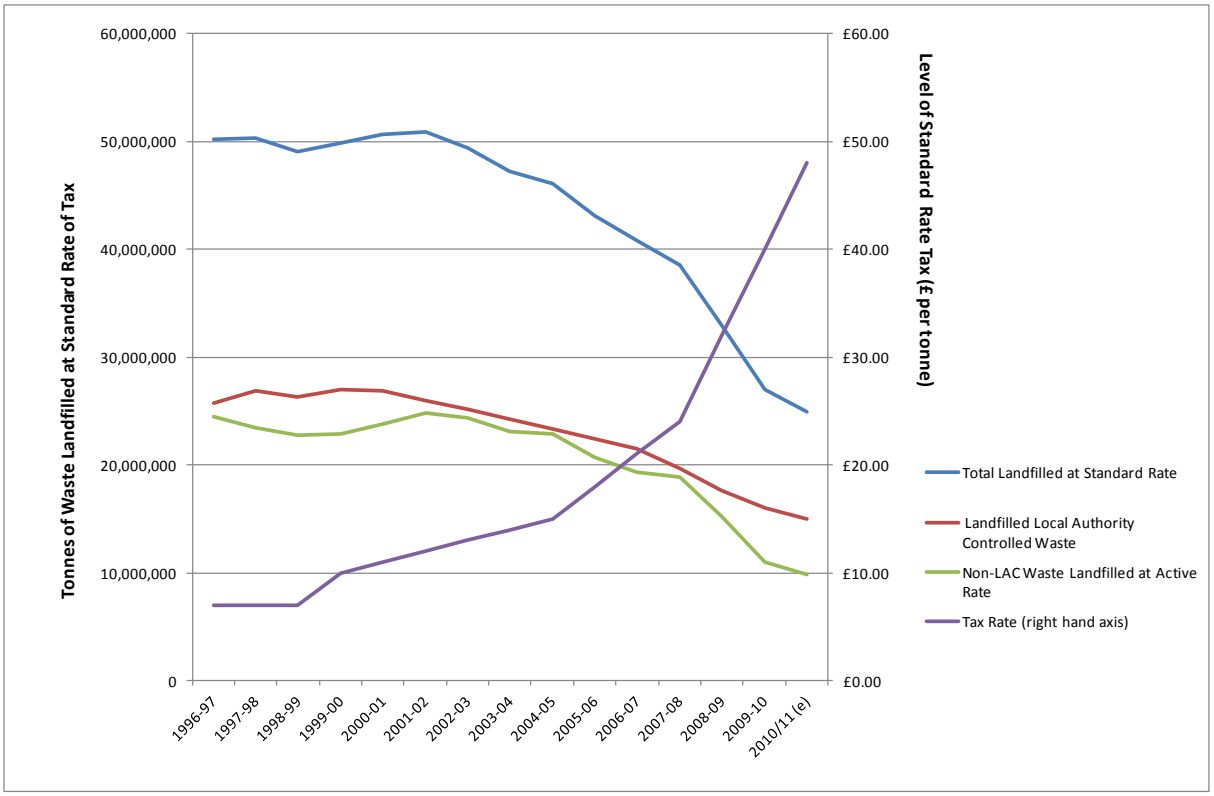


Source: HMRC data

As the standard tax rate has increased, the effect on waste subject to this tax rate has become more dramatic, with a turning point being reached around 2001/2002, and a decline in landfilled waste continuing to the present day. In the UK, the introduction of recycling targets for local authorities and the Landfill Allowance Schemes (LASs) has also had a strong effect on the management of waste collected by local authorities (local authority collected, or LAC, waste). It therefore becomes important to examine the extent to which the drop in landfilling of waste collected was due to wastes which were not affected by recycling targets and the LASs. Figure 28 demonstrates the impact of the tax upon the landfilling of LAC waste and non-LAC waste. The non-

LAC waste shows a decline in waste landfilled, with the rate of this decline quickening as the landfill tax escalator has increased.

Figure 28 Waste Landfilled in UK at Standard Rate Split by Local-Authority Collected, and Other Waste



Source: HMRC and estimate for 2010/11

It is important to note that reduction in the quantities of waste landfilled (broadly speaking, a halving from 50 to 25 million tonnes) has largely been achieved through waste prevention, reuse, preparation for reuse, and separate collection for recycling, composting and anaerobic digestion. The effect of the landfill tax in each of these respects is not entirely certain, but it should be recalled that revenue generated from it has been directed towards waste prevention activities across industry in the past.

3.2.4.2 LANDFILL TAX IN AUSTRIA[12]

Main features

The Act on the Remediation of Contaminated Sites (ALSAG) was first established in 1989 and has since been frequently amended. It introduced a tax which served a dual purpose: one, to finance the remediation of contaminated sites; and two to provide an incentive for improved waste management. The rate of the tax is dependent upon the environmental impact associated with different waste treatment options, and is differentiated by waste type, resulting in a lower tax for C&D waste or for incineration, and a higher tax for reactive waste and for landfilling.

Until 2004 it was possible to deposit waste in old non-compliant landfills. Use of such landfills incurred an additional charge which was as high as €73 per tonne in 2000. For landfilling of pre-treated (by MBT) MSW in compliant landfills, the tax gradually grew from €4.40 per tonne in 1995 to the current rate of €26 per tonne; it is due to increase to €29.80 per tonne in 2012. Waste may only be landfilled if its total organic carbon contents (TOC) is lower than 5%, which effectively bans the landfilling of untreated municipal waste (a higher rate of €87 per tonne applies to non pre-treated waste, and although this rate is now more or less obsolete since the ban on non pre-treated waste entered into force in 2004, it was important in incentivising the development of infrastructure in the period before the ban entered into force). Surcharges of €29 per tonne apply in cases where the destination landfill has no basement seal system or no vertical enclosure, and where municipal waste landfills are operated without a landfill gas collection and treatment system. Biological treatment, recycling and reuse does not incur a tax, so the landfill tax provides an incentive to move waste management up the waste hierarchy.

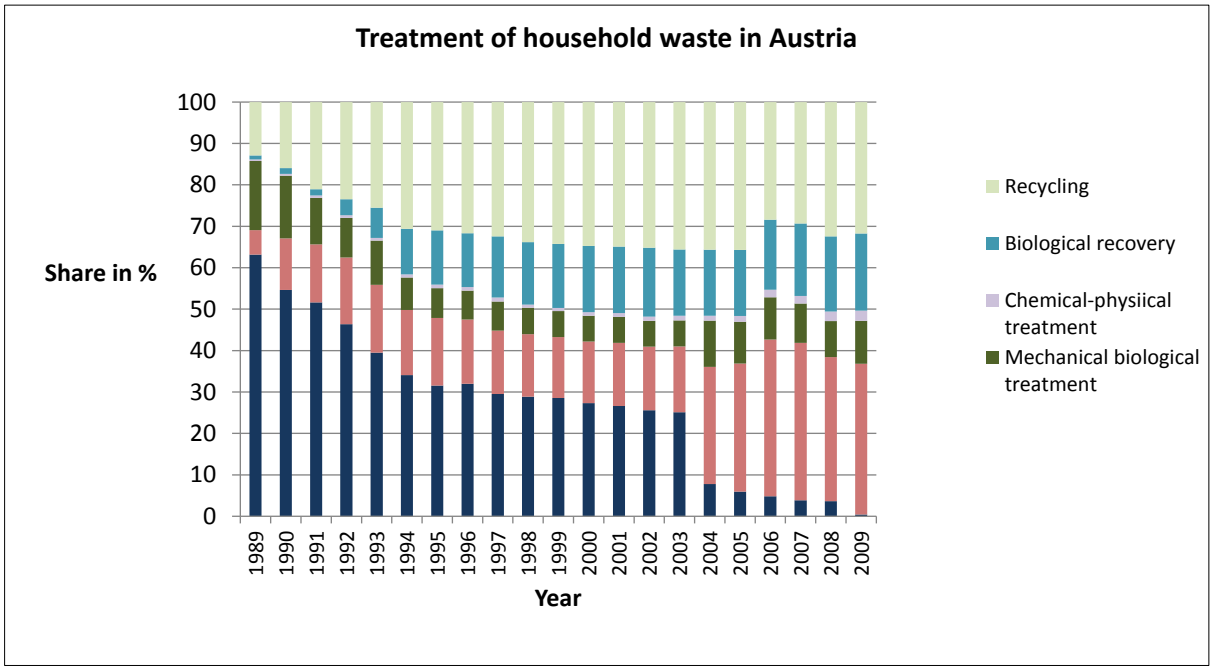
Impacts

Figure 29 shows the progress in MSW treatment in Austria from 1989 to 2009. In 1995 less than 50% of MSW was landfilled in Austria; this was relatively low in comparison to other EU countries. Between 1995 and 2003 MSW generation increased by 42% but the amount of MSW landfilled decreased by 9%. This decrease can be attributed to several factors, including the establishment of separate collection and recycling systems for packaging and paper, an increase in incineration and MBT capacity, the phasing out of non-compliant landfills, the unpopularity of introducing new landfills (limiting the available landfill capacity), in addition to the increasing landfill tax. In 2004 the landfilling of MSW dropped by a further 75% and had virtually disappeared by 2009; this can

[12] Lebensministerium (2011): Bundes-Abfallwirtschaftsplan 2011. Vienna. www.bundesabfallwirtschaftsplan.at and VOEB (2007): Entwurf der Novelle des Altlastensanierungsgesetzes 2007 – Stellungnahme des Verband Österreichischer Entsorgungsbetriebe 27. April 2007. http://voeb.at/media/2007/voeb_stellungnahme_alsag2007.pdf

clearly be related to the introduction and application of the landfill restriction as opposed to the landfill tax.

Figure 29 Municipal solid waste treatment in Austria, 1989 – 2009 (Eurostat 2011, Wirtschaftskammer 2011, Umweltbundesamt 2000)



Strengths/weaknesses

The strengths of the landfill tax include:

- It can be adapted to the varying environmental impacts of different waste types and treatment options. It is therefore a good instrument for driving waste management further up the waste hierarchy;
- Costing on average less than €10 per citizen per year the landfill tax is affordable whilst applying the polluter pays principle and therefore provides a (small) incentive for reducing waste generation; and
- The tax raises around €50 to 80 million per year to be spent on the actual remediation of contaminated sites.

The weaknesses of the landfill tax include:

- It is difficult to define the level of the tax where the effect of pushing waste up the waste hierarchy is achieved whilst market barriers are avoided (VOEB 2007); and

The tax may provide an incentive for illegal landfilling and waste exports. In order to prevent illegal dumping Austria undertook intensive awareness and information campaigns. Improper backfilling of C&D waste and illegal waste exports in particular increased after the rise in the tax. This

unintended side effect is still being addressed through intensified cooperation with the waste management industry, increased control and enforcement activities, and improved electronic recording of waste streams and waste management.

3.2.4.3 POLICY INSTRUMENTS RELATED TO LANDFILL IN GERMANY

Three national ordinances, unified in 2009 under the Landfill Ordinance, transposed the EU Landfill Directive into German national law, though imposing stricter requirements than those of the Directive. The range for landfill fees varies widely across Germany, as municipal authorities are responsible for determining fees through by-laws. One source identified the landfill fees range from €60 to €220 per tonne.[13]

Impacts of policies related to landfilling in Germany

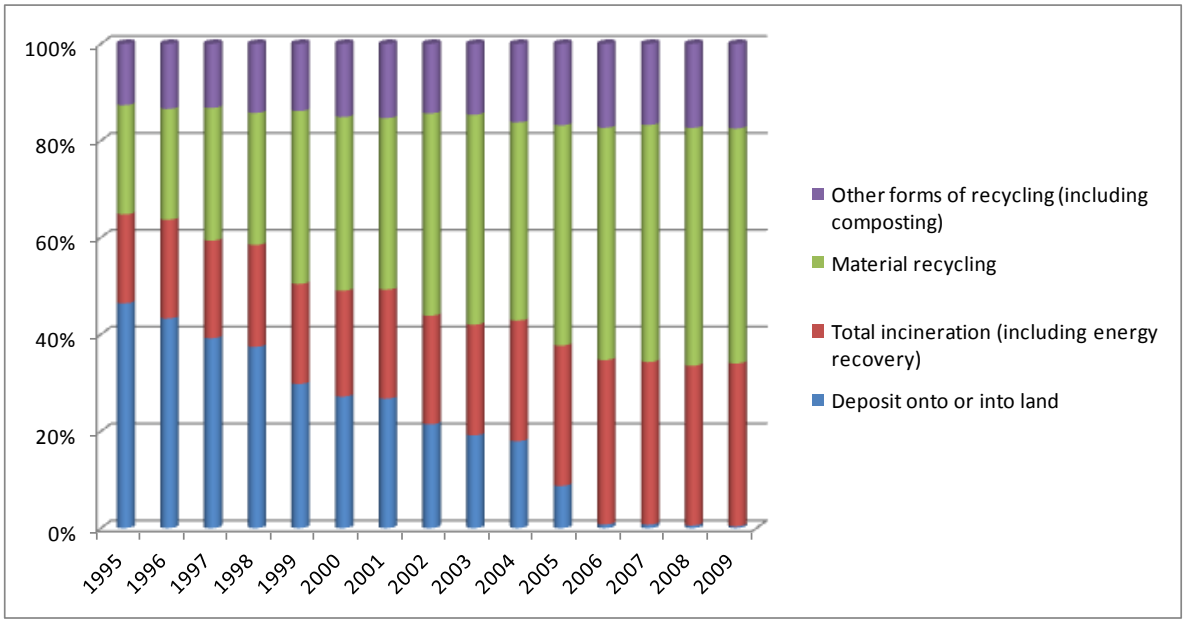
The amount of waste sent to landfill has drastically decreased in the last five years to 35 million tonnes (all waste fractions), and to virtually nothing for untreated MSW (2009).[14] According to available data, 48% of municipal waste in 2009 was recycled, 34% thermally treated and 18% composted.[15] Figure 30 below shows the treatment for municipal waste in Germany from 1995 to 2009. In particular it shows two contrary trends: the amount landfilled significantly decreased (to almost zero from 2006) whilst the amount incinerated increased. This reflects the effect of the landfill ban on untreated MSW, which has brought about an increase in pre-treatment by incineration.

[13] http://www.voithpaper.de/media/vp_de_twogether17_05_meri_d.pdf, in German, accessed 07/09/2011.

[14] http://www.destatis.de/jetspeed/portal/cms/Sites/destatis/Internet/DE/Presse/pm/2011/01/PD11__032__321,templateId=renderPrint.psml, in German, accessed 07/09/2011.

[15] http://www.eds-destatis.de/de/press/download/2011/03/037-2011-03-08.pdf, in German, accessed 07/09/2011.

Figure 30 Municipal waste treatment in Germany, 1995 – to 2009 (kg per capita)



Source: Eurostat

Thus, it can be assumed that the landfill ban on untreated MSW is well enforced and has greatly reduced the amount of waste going to landfill.

Strengths / factors for success / weaknesses of the policies

The landfill ban, which originates from the 2001 Waste Disposal Ordinance and the 1993 administrative regulation on municipal waste (Technische Anleitung Siedlungsabfall, TASi), legally requires that waste undergo pre-treatment procedures (waste incineration or MBT) before final disposal, and hence imposing higher costs for landfilling. There has been much debate on this issue, with specific stakeholders lobbying for far-reaching exemptions from the landfill ban. Furthermore, between 2001 and 2005 many municipalities and landfill operators encouraged landfilling and substantially lowered gate fees in order to fill as much of the landfill capacities as possible whilst this was still permitted, thus generating additional revenue.[16,17]

In some cases the landfill ban triggered the illegal dumping of untreated MSW in landfills and quarries. Available data suggest that illegal dumping of waste amounted in total to several

[16] WRAP and Eunomia Research & Consulting (2010): Landfill Bans: Feasibility Research. Appendices. Banbury, UK. See URL http://www.wrap.org.uk/downloads/Landfill_Bans_Feasibility_Research_Appendices_Final.cc333c39.8866.pdf, accessed 08/09/2011.

[17] Eunomia Research & Consulting *et al.* (2009): International Review of Waste Management Policy: Annexes to Main Report. Bristol, UK. See URL http://www.environ.ie/en/Publications/Environment/Waste/WasteManagement/FileDownLoad,21598,en.pdf, accessed 09/09/2011.

hundred thousand tonnes.[6] However, the landfill ban did yield environmental benefits. In particular, the higher costs for pre-treating waste going to landfill triggered intelligent waste management and also prevention activities at the municipal level, which contributed to an increase in recycling rates and also a moderate decrease in waste generation.

3.2.5 SUMMARY AND CONCLUSIONS

Landfill taxes appear to have a strong impact on the amount of municipal solid waste (MSW) sent to landfill, up to a certain point. Further improvements in performance after that point would appear to be attributable to a combination of landfill restrictions with adapted tariff policies.

There appears to be a clear correlation between the total cost of landfilling and the percentage of MSW recycled and composted in the MS; the higher the cost of landfilling, the more MSW is pushed up the waste hierarchy towards treatment via recycling and composting.

Member States appear much more likely to meet a 50% recycling target once landfill charges (or the cost of the cheapest disposal option) approach €100 per tonne. Such charges will tend to drive the economics of recycling and composting (which are dominated by the avoided costs of residual waste management).

In the context of this study, it has been possible to establish a strong apparent correlation between increasing landfill tax rates and decreasing rates of landfill for MSW in at least three MS (Austria, Sweden and the UK) (see Figure 24), even if these are not due to the taxes alone. For MS where there is an apparent weak correlation or no distinguishable correlation between landfill tax rates and the percentage of MSW landfilled, it must be stressed that other factors may be at play, including broader waste policy, the use of other EIs, the economic situation over time, changes in gate fees charged by landfill sites, and the available capacity of landfills. Whilst this study notes some apparent correlations between levels of landfill tax and total cost of landfilling and the amount of MSW sent to landfill, care must be taken not to infer direct correlations at the MS level, even where strong apparent correlations exist (and equally, we should not assume that the apparent absence of an effect does not mean there has been no influence exerted by a landfill tax).

With regards to landfilling of C&D/inert waste, the MS with the highest tax rate (Denmark) is amongst the best performing in terms of recycling and the MS with the lowest tax rate (Spain) is amongst the worst performing. Nevertheless, due to the lack of reliable historical statistics, it has not been possible to establish a significant relationship between the tax rate and the amount of waste recycled.

The country case studies included in this report indicate that there are different routes to reducing the landfilling of waste. It appears difficult to 'eliminate landfilling' through a tax alone. The effect of taxes has tended to be a shift away from landfilling towards material recovery. It appears that taxes have not been set at rates which give certainty to enable a complete switch away from landfilling based on a tax alone. On the other hand, the effect of bans appears to have been to reduce the landfilling of waste (close to zero in the case of bans which are effectively implemented

and enforced) and increase the amount of waste incinerated or sent to MBT. In the cases of the UK and Austria, landfill taxes have demonstrably contributed to a reduction in the amount of municipal waste being sent to landfill.

There are some concerns regarding the effect of bans on the way in which capacity for non-landfill residual waste treatment is developed. In Germany in particular, the ban has led to increases in capacity beyond what is required for Germany itself, and similar effects would seem to be observable in other countries with bans in place (such as the Netherlands). The issue with bans is that they remove flexibility from the system. Whatever the drawbacks of landfilling, it is more flexible to changes in throughput than treatments such as incineration and MBT.

Feedback gathered from stakeholders during the study suggests that landfill (and incineration) taxes are seen as appropriate in MS that are lagging behind in their waste management performance. They can help to tilt the balance of costs of waste management in favour of other waste treatment methods, particularly if tax levels progressively increase. However, taxes should be applied only when waste management legislation is well enforced to prevent an increase in illegal dumping. Some stakeholders raised concerns over what happens to the revenues from landfill (and incineration) taxes in some MS, and stressed that these revenues should be reinvested in waste infrastructures.

Overall, it would seem that the use of landfill taxes (adapted to the conditions in each MS) can help to shift waste into recycling options, with progressively higher taxes generating higher performance. Additional reductions in landfilling can be achieved using landfill restrictions or bans, but these have to deployed carefully to ensure that the waste management system does not become over-reliant on residual waste treatment at the expense of further progress in recycling. Where they are judiciously deployed, however, these instruments seem to be indispensable (in combination with other policies) to meet the existing minimum EU targets and furthermore to ensure MS progress towards the goal of the Resource Efficiency Roadmap to achieve zero residual waste.

3.3 INCINERATION TAXES

Incineration taxes can be imposed to attempt to stop waste simply being diverted from landfill to incineration.

Table 3 below suggests that only 6 MS currently have incineration taxes in place for the disposal of municipal waste (the Netherlands has an incineration tax but the rate is currently set at €0; the Czech Republic is considering introduction of an incineration tax; Sweden introduced a tax in 2006 that was subsequently abolished in 2010).

We distinguish in this study between:

- Taxes: a levy charged by public authorities (usually national but occasionally regional) for the disposal of waste in a landfill site, usually with an environmental purpose in mind, and where the revenue is accruing to the body responsible for the levy; and
- Gate fees: charges set by the operators of the landfills for the provision of the service (i.e. waste disposal) and which are designed to cover their costs and profit. NB This type of fee is subject to variation according to the incinerator used (including the age of the facility), and to other factors such as available capacity and market variations. Gate fees do not always cover an operators' cost due to the market situation at a given time.

The level of taxation varies considerably, from as little as €2.40 per tonne in France to €44 per tonne in Denmark. The total cost of incineration, however, depends also on the gate fee charged by the incinerator for accepting waste. When this is factored in, the total median cost to incinerate one tonne of municipal waste in the EU appears to range from €46 in the Czech Republic to €174 in Germany.

The research undertaken within this study suggests that only 2 MS (Latvia and Luxembourg) have restrictions in place for the incineration of certain materials.

Table 3 Overview of incineration taxes, fees and restrictions (municipal waste), and incentives for energy from waste

Member State	Current incineration tax (per tonne)	Current incineration gate fees (range per tonne)	Current incineration gate fee (typical per tonne)	Total typical charge for incineration (per tonne)	Incineration restrictions (y/n)	Number of incinerators (waste-to-energy plants treating MSW unless otherwise specified) and quantity treated in 2009[18]	Incentives for energy from waste (y/n)
AT	€7 (+ €18 residue)	€100 - €150	€125	approx. €150	N	10 (2.3 Mt/year) + 49 other incineration plants (2.2 Mt/year) operated according to Directive 2010/75/EC + approx. 130 'smaller' co-incineration plants (2010)	y
BE	€7.66	approx. €70 - €130	€110	approx. €117.66	N	16 (2.8 Mt/year) (2009)	y
CZ	Under consideration	approx. €36 - €56	€46	approx. €46	?	3 (0.4 Mt/year) (2009)	?
DK	€44 (waste to energy levy)	€36 (for thermal treatment)	€36	c €80	?	31 (3.5 Mt/year) (2009)	?
FI	-	?	?	?	?	3 (0.3 Mt/year) (2009)	y
FR	€2.40 - €11.20 (adjusted based on site's env. performance)	€77 - €121 (50% of municipalities between €85 - €104)	€99	approx. €110.20	?	130 (13.7 Mt/year) (2009)	?
DE	-	€80 - €250	€174	approx. €174	N	70 (19.1 Mt/year) (2009)	?
HU	?	?	?	?	?	1 mixed municipal waste incinerator (0.4 Mt/year) (2009)	y
IE	-	-	?	?	?	0	?

[18] Data from CEWEP, http://www.cewep.eu/information/data/studies/138.Map_European_Waste-to-Energy_Plants_in_.html

Member State	Current incineration tax (per tonne)	Current incineration gate fees (range per tonne)	Current incineration gate fee (typical per tonne)	Total typical charge for incineration (per tonne)	Incineration restrictions (y/n)	Number of incinerators (waste-to-energy plants treating MSW unless otherwise specified) and quantity treated in 2009[18]	Incentives for energy from waste (y/n)
IT	-	approx. €100 - €150	€125†	approx. €125†	N	49 (4.5 Mt/year) (2009)	?
LV	-	?	?	?	Industrial and vehicle batteries	0	?
LU	-	€96.68	€96.68	€96.68	Y	1 (0.1 Mt/year) (2009)	y
NL	€0[+]	€70 - €120	€95†	approx. €95†	N	12 (6.3 Mt/year) (2009)	y
PL	?	€130	€130	?	?	1 (0.04 Mt/year) (2009)	?
PT	?	?	?	?	?	3 (1.1 Mt/year) (2009)	?
SE	-	?	?	?	?	31 (4.7 Mt/year) (2009)	?
SK	?	?	?	?	?	2 (0.2 Mt/year) (2009)	?
SI	-	?	€113	?	?	1 (0.02 Mt/year) (2009)	?
ES	**Catalonia:** €5.50; €16 if biowaste not collected at source	€35 - €79	€57	€62.50 (€73 if biowaste not collected at source)	N	10 (2.2 Mt/year) (2009)	y
UK	-	€36.90 - €138.20	€87.55	approx. €87.55	N	23 (3.4 Mt/year) (2009)	y

NB No municipal waste incineration reported in 2009 in BG, CY, EE, GR, LT, LV, MT, RO

† *These figures are mean not median, therefore may not reflect actual fees in place.*

[+] *Data from: ETC/SCP Working Paper: Overview of the use of landfill taxes in Europe, Draft June 2011*

Figure 31 below provides a graphical overview of the data collected on the total typical cost of incineration for non-hazardous municipal waste.

Figure 31 Overview of total typical cost of incineration (municipal waste)

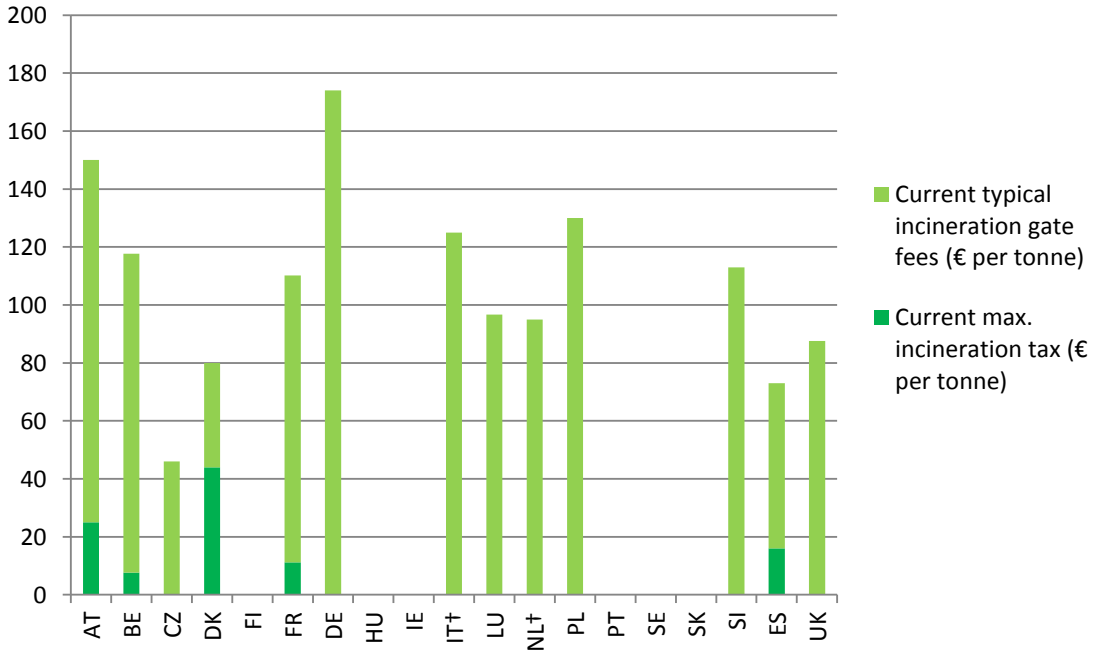
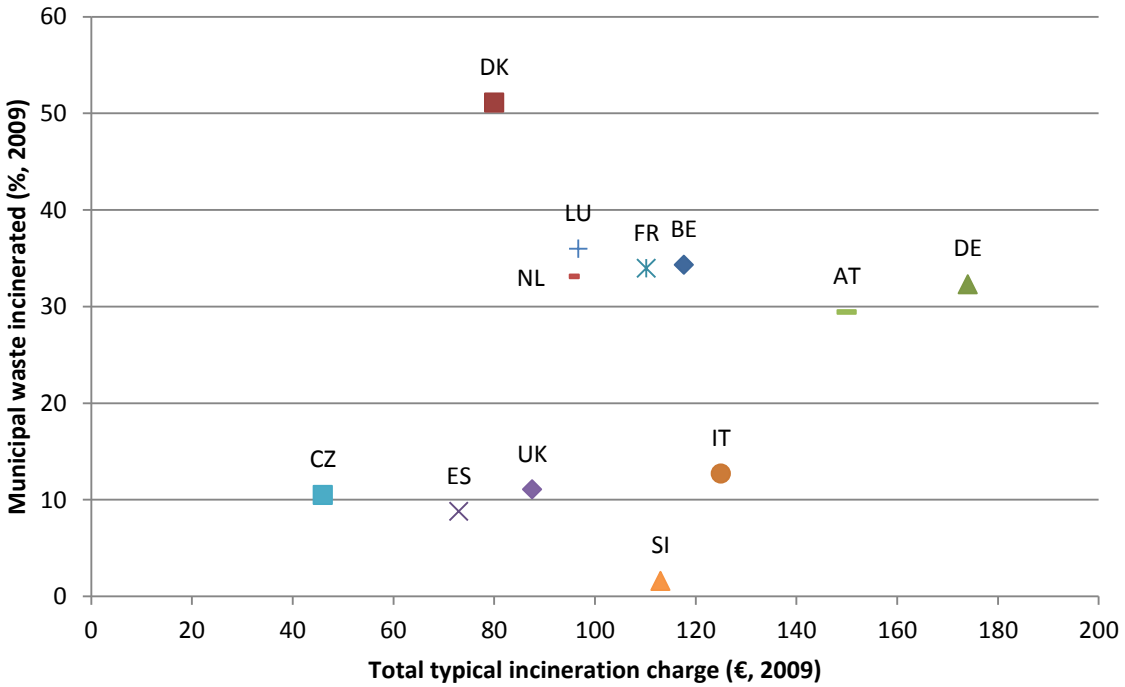


Figure 32 below compares the total typical charge for incineration with the percentage of MSW incinerated, for the MS where data has been found. There is not the same correlation as in the case of landfill, where higher costs tend to result in less MSW being sent to landfill; indeed, one would not necessarily expect any correlation. Incineration is, in most countries, a relatively costly treatment option, meaning that the amount of residual waste sent to incineration is generally affected by the extent to which landfill taxes make incineration cost competitive with landfill, and more importantly, whether there is a ban on landfilling of waste. In cases where the latter applies, residual waste has to be dealt with through means other than landfilling, thereby acting as a significant driver for increased incineration.

Figure 32 Total typical incineration charge and percentage of MSW incinerated, 2009



Due to a lack of time series data on the rates of incineration taxes, it has not been possible within the study to further analyse the impact of the rate of incineration tax on the amount of MSW treated by incineration.

Figure 33 below makes a comparison between the overall typical cost of landfilling and the overall typical cost of incinerating municipal waste. All MS that have incineration taxes also have landfill taxes, and in every case the landfill tax is higher than the incineration tax. However, in MS where information has been found for the total typical cost of both landfill and incineration, the total typical cost of incineration appears higher in 8 MS (Austria, Belgium (Wallonia), the Czech Republic, France, Germany, Italy, Poland and Spain). For Austria and Germany, where there are effectively bans on landfilling MSW-type waste, this is of little concern; however, it suggests that landfilling may preferable in purely economic terms in the other MS. The total charges applied to landfill are higher in 6 MS (Belgium (Flanders), Denmark, Luxembourg, the Netherlands, Slovenia and the UK). It should be noted that the fact that charges can be higher for landfill than incineration does not automatically lead to a switch from landfill to incineration. Where facilities are financed privately, financial backers may seek a form of guarantee that ensures that a significant proportion of an incinerator's capacity will be filled through contracted flows of waste for a significant period of time. Such contractual guarantees are not always easily secured. Furthermore, the 'typical' charges illustrated in the figure mask significant local variation, and developers of new incinerators have to consider the competing local facilities, including landfills, which may be cheaper than a new incinerator.

Figure 33 Comparison between the cost of landfilling and the cost of incineration (municipal waste)

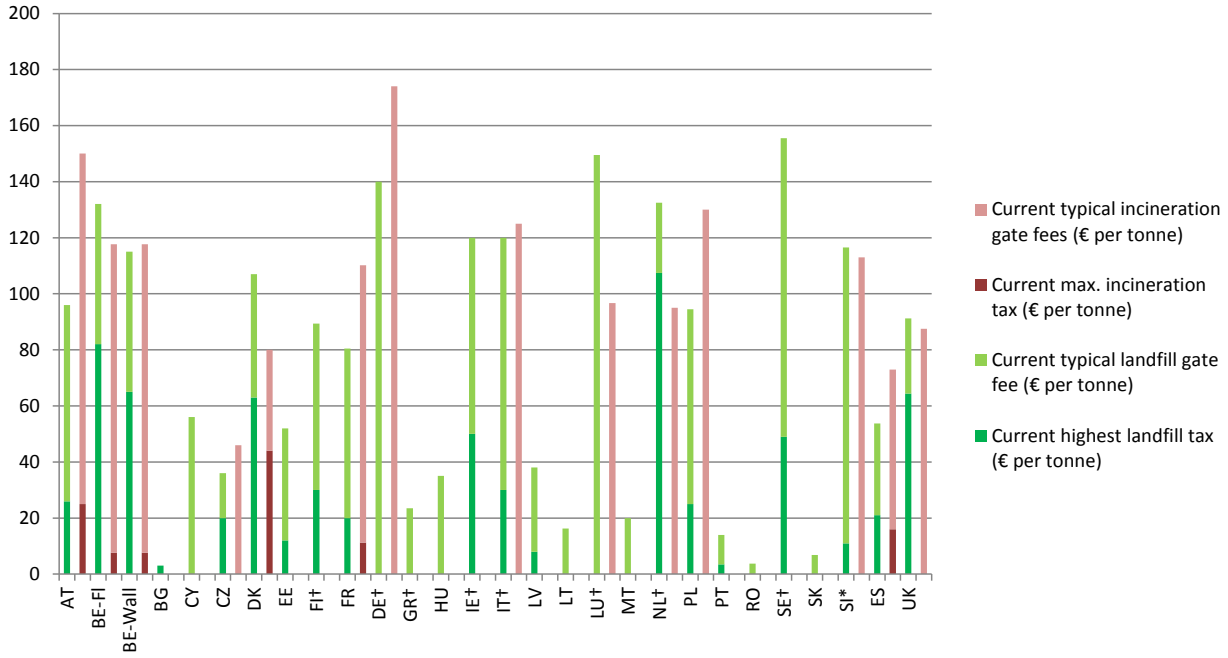
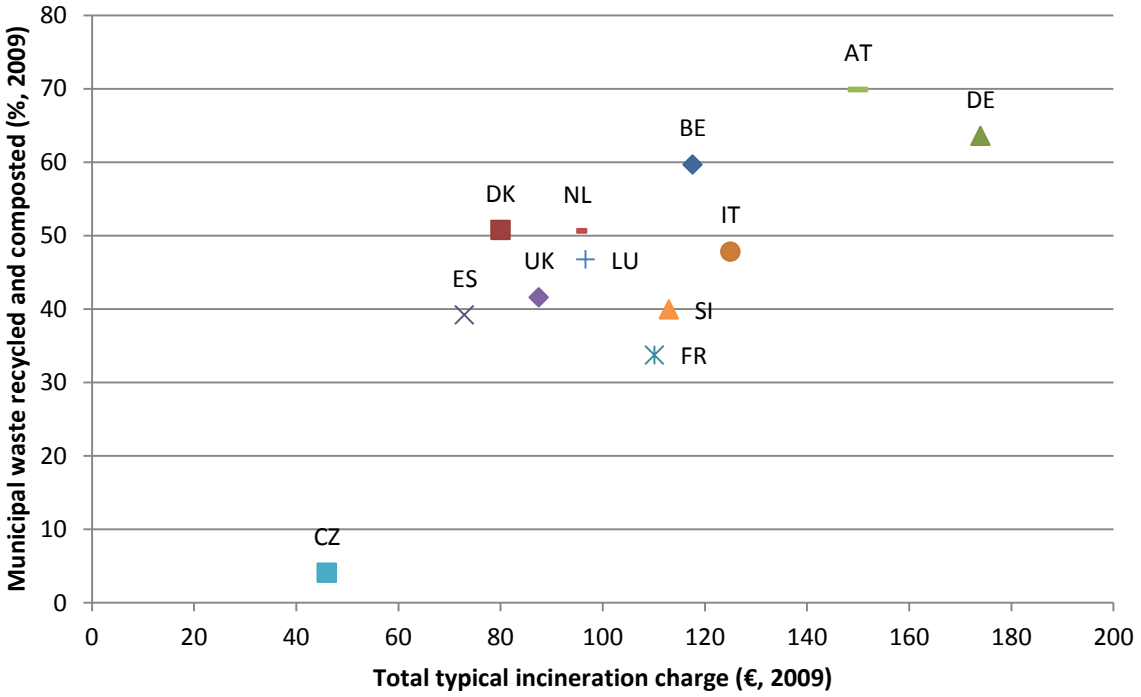


Figure 34 below compares the total typical incineration charge (for MS where data on both tax and gate fee has been found) with the percentage of waste recycled and composted in the MS. It shows a broad overall trend that higher incineration charges are generally associated with higher percentages of municipal waste being recycled and composted, indicating that higher incineration charges may help to push waste treatment up the waste hierarchy. The chain of causality may be indirect – incineration facilities tend to be more prevalent where landfill is either restricted or banned, or where the costs of landfill are made high through the deployment of taxes. Where incineration is more common, this often reflects higher costs of all residual waste management options. The higher the avoided costs of residual waste disposal or treatment become, the stronger the incentive for waste prevention and recycling.

Figure 34 Total typical incineration charge and percentage of MSW recycled and composted, 2009



3.3.1 INCENTIVES FOR ENERGY FROM WASTE

Directive 2009/28/EC specifies an overall goal of 20% energy generation from renewable energy sources (RES) by the year 2020. MS have determined their own evolving targets and have in place variable market-based (economic) instruments and incentive schemes to achieve these.

Specific incentives for energy from waste have been identified in six MS. Austria offers financial support (in general up to 25% of investment costs) for investments to incinerate biogenic raw materials and residual materials and for measures to substitute fossil fuels with biogenic materials and residues. In Belgium, Flanders issues green energy certificates and WKK (co-generation/combined heat and power (CHP)) certificates. These are both applicable to waste incinerating facilities for the renewable energy component part. Wallonia is considering energy from waste subsidies. Brussels does not grant such subsidies. In Finland, there is State support for electricity generated through the incineration of renewable fuels. Hungary recognises energy produced in waste-to-energy and RDF plants as renewable and provides a special price for such energy. In the Netherlands, a feed-in tariff has been provided for energy produced from waste incineration under certain circumstances. In the UK, the Renewables Obligation provides support for energy from waste with CHP.

RES-Electricity (Electricity from renewable energy sources)

In the MS surveyed (all except Bulgaria), the principal market-based instruments (MBIs) in place to achieve respective renewables targets are quota/purchase obligations coupled with green certificates (GC) trading systems, price premiums/minimum prices, feed-in tariffs and tax

exemptions (specifically exemption from electricity and consumption taxes). However, it should be noted that very few MS afford priority grid access and distribution to RES-E producers. Methods of using waste as an energy source include the use of biomass, biogas production from biodegradable waste and sewage treatment, other forms of biogas, and the burning of waste.

By far the most prominent MBI used throughout the EU to promote RES-Electricity generation is the feed-in tariff. Of the 26 MS surveyed for this study, 19 had feed-in-tariff schemes in place as their main RES support policy. While one waste type does not appear to be favoured over another, a review of the relevant literature observed that waste RES was commonly specified as biogas and biomass. The length of the guaranteed tariff ranged from one year in Slovakia to 25 years in the UK, with the average tariff being guaranteed for 15 years. Alongside the feed-in tariff scheme there are sometimes feed-in premiums, which afford plant operators the option of foregoing the tariff and instead receiving a premium payment on top of the market rate for a specific form of RES-generated electricity. Table 4 below provides an overview of feed-in tariffs and premiums for various types of waste. Some 22 MS have such tariffs, with 18 MS using differentiated tariffs. Premiums are used in Denmark (for 10 years) and the Netherlands. The Czech Republic, Estonia, Slovenia and Spain offer a choice between feed-in tariffs and premiums. Six MS use green certificates, and others are considering their introduction.

Table 4 Tariffs in €/MWh for various types of waste in the MS

Member State	Type of waste	Tariff in €/MWh
AT	Wood into 500 kW plant	150
	Wood into >10 MW plant	100
	Wood into power plant MW plant	61.2
	Bark, saw dust into 500 kW plant	112
	Waste with high biomass share	50
	Landfill gas	50
BE	Co-firing of biomass in coal plants	60
	Biogas production from bio degradable waste and sewage treatment	60
	Burning of waste	60
	Solid of liquid biomass	90
	Biomass waste	90
	Unspecified biogas	90
BG	Wood waste	111
CY	Landfill gas	63
CZ	Landfill gas and sewage gas	91.1 (feed-in) 55.3 (premium)
DE	**Feed-in tariff per kWH**	

Member State	Type of waste	Tariff in €/MWh
	Biomass	0.07 to 0.12
	Sewage and landfill gas	0.05 to 0.08
	Energy from organic waste via fermentation	0.14 to 0.16
	Incineration of mixed municipal waste is not included.	
DK	Electricity produced from biologically or thermally gasified biomass	100.1
	Mixed biogas	54.4
	Biomass	20.2
EE	Peat, waste	52
ES	**Average selling price** (Eurocent/kWh) –Sold via tariff through distribution company	
	Waste	5.853
FI	**Electricity tax aid (per kWh)**	
	Forestry waste	0.69
	Biomass systems	0.42
FR	Biogas/ Methanisation	75 to 90 Energy efficiency bonus of 0 to 30 €/MWh and a methanisation bonus of 20 €/MWh
	For all other biomass there is no feed-in tariff available upon request.	
GR	RES type not specified	80.14 -91.74
HU	**Feed-in tariff per kWH**	
	Gas from waste yard	3.9 and 10.7
IE	Biomass (landfill gas)	81.486
	Other biomass	83.814
	Anaerobic digestion	120
IT	**Feed-in tariff per kWh**	
	Biogas and biomass	0.28
	Landfill gas and gas resulting from purification processes	0.18
LV	Biomass, biogas Installed capacity of biomass PP < 4 MW	

Member State	Type of waste	Tariff in €/MWh
	Biomass	(support level 1) 91.05-176.99 (support level 2) 68.79-133.73
	Biogas	(support level 1) 75.48-141.60 (support level 2) 57.03-106.98
	Biomass Installed capacity > 4 MW	(support level 1) 60.38-110.19 (support level 2) 50.32-91.83
	Biogas Installed capacity < 2 MW	(support level 1) 133.18-163.84 (support level 2) 106.55-131.07
LT	**Feed-in tariff per kWH**	
	Biogas	0.30 LTL
	Biomass	0.30 LTL
LU	Waste wood ≤ 1 MW =	129.35
	Waste wood > 1 MW =	109.45
MT	-	-
NL	Electricity production in waste incineration plants (if efficiency of the installation is above 22%)	25 - 48
	Electricity production from landfills and sewage treatment (for power stations)	15
	Biomass electricity from Combustion (10-50 MW)	71-112
	Biomass electricity from fermentation of biodegradable waste	85-105
	Biomass electricity from Co-fermentation and small-scale combustion (up to 10 MW)	108-133
	Other fermentation (liquid biomass)	114
PL	All technologies used in the generation of RES-E are promoted through price regulation.	
PT	Unsorted urban waste	53-54
	Sorted/prepared urban waste	74-76
RO	-	-
SK	Biomass combustion	107-130

Member State	Type of waste	Tariff in €/MWh
SI	Biodegradable waste up to 1 MW	77.4
	Biodegradable waste up to 10 MW	74.3
	Landfill gas 50 kW-1 MW	67.5
SE	All technologies, (wind, solar, geothermal, biogas, biomass, hydro, wave energy) used in generation of RES-E are covered by the quota system.	
UK	**Feed-in tariff per kW (£)**	
	Anaerobic digestion	
	≤ 250 kW	14.0p
	>250 kW – ≤ 500 kW	13.0p

RES-C&H (Cooling and heating from renewable energy sources)

For cooling and heating (C&H) generation, incentives for investment in RES include project permitting and tax exemptions, obligations/targets for RES-C&H, and the wide availability of feed-in tariffs, investment subsidies, grants and low-interest loans, some of which apply to domestic installations of RES technology (e.g. biomass stove).

Advancement of RES-C&H throughout the EU comes mainly in the form of investment subsidies and grants, and tax relief/exemption. Investment subsidies and grants are widely available for biomass using plants and anaerobic digestion CHP, as well as RES-H installations sited on industrial, commercial, public and community premises. Low-interest loans and some government and EU grants are typically available for the same projects. Several MS have also made grant schemes available to homeowners intending to install RES-H systems (i.e. Ireland's Greener Homes Scheme). Indirect tax relief on RES-H is also widely observed throughout the MS surveyed. This is usually brought about by higher taxes levied on fossil fuel generated sources of heating, as well as personal income tax exemptions for those who invest in energy saving measures.

3.3.2 SUMMARY AND CONCLUSIONS

There is no apparent correlation between the total cost of incineration and the amount of MSW sent to incineration; this is in contrast to the situation for landfill, where higher costs tend to result in less MSW being sent to landfill. Due to a lack of time series data on the rates of incineration taxes, the study has not been able to analyse in depth the impact of the rate of incineration tax on the amount of MSW treated by incineration. However, there is a broad overall trend that higher incineration charges are generally associated with higher percentages of municipal waste being recycled and composted. This indicates that higher incineration charges may help to push waste treatment up the waste hierarchy, although the causality may be indirect. This is because the higher charges for incineration are typically found in countries with higher landfill charges and/or landfill bans, increasing the avoided cost of the main options for residual waste management.

All MS that have incineration taxes also have landfill taxes. In every case the landfill tax is higher than the incineration tax. This can be taken as an indication that MS are attempting to push waste

up the waste hierarchy by increasing the cost of all forms of residual waste management, and tilting the economic balance in favour of incineration. In MS where information has been found for the total typical cost of both landfill and incineration, the total charges applied to incineration appear higher in 8 MS (AT, BE (Wallonia), CZ, DE, ES, FR, IT, PL), suggesting that landfilling may be preferable in purely economic terms. (The total charges applied to landfill are higher in 6 MS – BE (Flanders), DK, LU, NL, SI, UK.)

The picture on incineration is complicated by the presence of support mechanisms (e.g. feed-in tariffs, premiums and green certificates) for the production of energy from waste. This raises issues for policy coherence. Support is provided through energy policy for the generation of energy from waste, which is considered a renewable resource, whereas waste policy is attempting to both reduce the amount of waste generated and move remaining waste further up the waste hierarchy towards prevention, preparation for reuse, recycling and composting. It may be appropriate, if support is provided for energy generation, to also reward savings of embodied energy by rewarding energy saved through recycling. Further application of incineration taxes would more accurately reflect the negative externalities of incineration and also reduce the likelihood of a simple switch from (untreated) landfill to incineration as a waste treatment method.

Feedback gathered from stakeholders during the study suggests that incineration (and landfill) taxes are seen as appropriate in MS that are lagging behind in their waste management performance. They can help to tilt the balance of costs of waste management in favour of other waste treatment methods, particularly if tax levels progressively increase. However, taxes should be applied on the condition that waste management legislation is well enforced and also that there are sufficient waste management/treatment infrastructures to accept waste that is diverted from incineration (and landfill). Given such conditions, incineration (and landfill) taxes can act as a trigger to the separate collection of recyclables.

Some stakeholders pointed out that incineration should not be considered as representing the bottom of the waste hierarchy, as it is a preferred option when compared to landfill. One stakeholder also suggested that taxing waste-to-energy incineration could have some detrimental impacts, such as making recycling more expensive (as recycling residues often need to be treated thermally, though this clearly reflects the quality of the material collected for recycling) and making the production of local energy from waste more expensive (which would reduce the benefits of waste-to-energy as an alternative to fossil fuels). The latter point needs to be contextualised by the fact that fossil fuel generation technologies are themselves subject to a range of economic instruments, both at the EU level (the EU-Emissions Trading Scheme) and the MS level.

3.4 PAY-AS-YOU-THROW SYSTEMS

3.3.3 OVERVIEW OF PAYT SYSTEMS IN THE EU-27

Table 5 below suggests, according to the data found within this study, that 17 MS have established one or more **pay-as-you-throw (PAYT) systems** for municipal waste, with one trial also in operation in Greece. It makes relatively little sense, however, to speak in terms of MS policy where PAYT schemes are concerned since these are systems which are designed and implemented by municipalities. Hence, in the majority of MS with some PAYT schemes in place, implementation is far from being 'universal'. The use of PAYT by local authorities is required by relatively few MS, and even where this is the case, coverage has not always been universal. Some MS, such as Italy, offer guidance on the structuring of charges. Some regional authorities (German Länder, for example) have also sought to mandate the use of PAYT schemes.

In order to fund waste management services, most MS without PAYT schemes require households to pay for the collection/disposal of their waste, but through flat charges or municipal taxes rather than variable charging schemes. These are not considered PAYT schemes in this study since they imply that the amount paid is independent of any change in waste management behaviour.

In the vast majority of PAYT schemes, the overall cost of the service is funded through a combination of flat rate fees or taxes (deemed important to give some certainty over the level of revenue generated, which is a requirement to ensure costs are fully recovered), and a variable element which may be linked to one or more of:

- the choice of container size (volume-based schemes);
- the number of sacks set out for collection (sack-based schemes);
- the frequency with which a container is set out for collection (frequency-based schemes); and
- the weight of material collected in a given container (weight-based schemes).

A review of the literature indicates that volume-based schemes generally impart the weakest incentive for waste prevention and recycling, partly related to the fact that once a bin of a specific size has been purchased (or subscribed to), the marginal cost of reducing the quantity of waste sent to the residual waste container is effectively zero.[19] In all other schemes, this is not the case, and each has its merits in specific circumstances (as does each combination).

As PAYT schemes are largely devised and implemented at the local level, it is not possible within the scope of this study to present a complete overview of PAYT schemes in the EU; indeed, a variety of experiences can be found within a number of MS. Table 5 therefore provides overview information on:

- whether PAYT schemes are in place in each MS;

[19] For a review, see D. Hogg, D. Wilson, A. Gibbs, M. Astley and J. Papineschi (2006) *Modelling the Impact of Household Charging for Waste in England*, Final Report to Defra. www.defra.gov.uk/environment/waste/strategy/incentives/pdf/wasteincentives-research-0507.pdf;

- the estimated coverage, where such information exists (in terms of population, number of households or number of municipalities covered);
- the types of scheme which are known to be used (few MS have detailed information on charging structures in their territory, therefore this may not be completely accurate); and
- details of the level of charges used in a number of example schemes.

The example schemes listed in the table indicate a broad range of variable rates used in the various PAYT schemes. Although the schemes are not necessarily directly comparable, and also cover MS with varying levels of household income and waste management costs (including disposal costs – see sections 3.2 and 3.3 above), a summary of findings is presented below:

- **Fixed annual fees per household (as an element of a PAYT scheme)** range from €40 (Miravet and Rasquera Municipalities, Tarragona region, Spain) to up to €2,415 (for a large 1,100l bin in Stuttgart, Germany);
- **Fees for the purchase of mandatory refuse bags for residual waste** range from €0.65 for a 17 litre bag (Argentona Municipality, Barcelona region, Spain) to €5.50 for a 70l bag (for bags over and above standard volume collected, Stuttgart, Germany);
- **Fees per emptying of a bin** (120l or 140l only, for comparison purposes) range from €0.50 in France (in the context of a scheme combining volume and frequency elements) to €4.20 (north Helsinki, Finland); and
- **Fees per kg** range from €0.17 (Slovakia) to €0.36 (Sweden).

It should be noted that schemes do not necessarily charge only for residual waste. Those schemes which collect garden waste frequently also charge for the collection of biowaste. In the UK, where no examples of 'pure' PAYT schemes exist, it is common for households to be charged for garden waste collections (since this is seen as a special service provided by the local authorities).

Table 5 Overview of use of PAYT schemes (municipal waste)

Member State	PAYT schemes in place (y/n)?	Estimated coverage (population/ households)	Types of scheme observed (e.g. weight/volume/ sack/frequency based)	Details of charges under example schemes
AT	Y	Nationwide	Volume and frequency,	
BE	Y	Flanders; around 100% coverage	Volume, sack, frequency and weight	**Flanders:** Rubbish bag = €1.50; collection of mixed waste = €0.75-2.5 per 60l bag; for a 120l bin = €2.5-3.76 per volume, €0.15-0.2 per kg, €0.25-1 per collection
BE	Y	Wallonia; around 50% coverage	Volume, sack, frequency and weight	**Namur:** Rubbish bag = €1.00 per 60l bag; recycling bag €0.13 per 60l bag; biowaste = €0.25 per bag
BG	N	-		
CY	N	-		
CZ	Y	1 'pure' PAYT scheme and 24 'combined schemes' out of 162 municipalities (in 2002)	Volume and frequency	
DK	Y	20 out of 275 municipalities (in 2006)	Weight	**Bogense Municipality (2000):** fixed fee per household per year (for 5kg/fortnight): €142.53; variable fee per extra kg = €0.50; fixed fee per summerhouse per year = €109.01
EE	Y	Unknown	Volume and/or frequency	
FI	Y	All municipalities	Typically volume and frequency	**Kerava, Järvenpää, Tuusula Mäntsälä (north of Helsinki):** per collection: 140l container = €4.20; 240l = €5.90; 360l = €7.50; 600l = €10.70 **Kuopio (East Finland):** per collection: 140l container = €3.17; 240l = €3.78; 360l = €4.53; 660l = €5.92

Member State	PAYT schemes in place (y/n)?	Estimated coverage (population/ households)	Types of scheme observed (e.g. weight/volume/ sack/frequency based)	Details of charges under example schemes
FR	Y	30 municipalities (to be extended to 17 more in 2011)	Volume, frequency and weight	**Ribeauvillé:** annual fixed fee = €82.80; container maintenance/ rent: 120l = €5, 240l = €8, 770l = €25; variable fee = €0.50 per regular collection, €1.10 per additional collection, €0.27 per kg
DE	Y	Unknown	Sack, volume, frequency and weight	**Stuttgart (2011):** fixed annual fee (fortnightly collections): 60l = €114, 120l = €211.80, 240l = €376.20; fixed annual fee (weekly collections): 120l = €445.20, 240l = €790.20, 1,100l = €2,415; fee of €5.50 per additional 70l bag.
GR	N	-		
HU	Y	?	Volume and frequency	
IE	Y	In theory nationwide, but in reality this is not achieved; mandatory charging since 2005	Volume, sack, frequency and weight	
IT	Y	Estimated around 2 million inhabitants	Volume and frequency	
LV	N			
LT	?	?		

Member State	PAYT schemes in place (y/n)?	Estimated coverage (population/ households)	Types of scheme observed (e.g. weight/volume/ sack/frequency based)	Details of charges under example schemes
LU	Y	21 out of 116 municipalities (around 40% of the population)	Weight + frequency, volume + frequency, frequency (with a set minimum)	**SICA Region:** annual fee = €120; emptying fee = €1.30 (120l), €2 (240l); weight fee = €0.15 per kg; separately collected biowaste weight fee = €0.10 per kg **SYCOSAL Region:** A) annual fee = €91.20 - €144; container size fee = €46.80 - €66 (120l), €82.80 - €128 (240l); emptying fee = €2.25 - €3.50 (120l), €4.50 - €6.50 (240l) B) annual fee = €174 - €264; emptying fee = €3.75 - €4 (120l), €7.50 - €8 (240l). Minimum number of emptyings: 19-28 (120l), 10-28 (240l)
MT	?			
NL	Y	Over 20% of 538 municipalities (in 2000)	Frequency (54 municipalities), volume (29), sack (20) weight (13) unspecified (10)	
PL	?			
PT	N			
RO	N			
SE	Y	26 out of 290 municipalities (around 9% of the population) use weight-based PAYT; 215 municipalities volume-based	Volume, frequency, weight	In all cases a fixed cost of basic fee + bin fee + cost based on weight; in some cases an additional fee per kg per collection of €0.12-0.36.
SK	Y	Unknown	Municipal waste fees, based on volume or weight	Municipal waste fees: €0.0033-€0.0531 per l, €0.0066-0.1659 per kg, or €2.40-40 per person per year (framework set by Act No. 582/2004)

Member State	PAYT schemes in place (y/n)?	Estimated coverage (population/ households)	Types of scheme observed (e.g. weight/volume/ sack/frequency based)	Details of charges under example schemes
SI	Y	Unknown	Frequency and volume based	
ES	Y	Very small proportion; 15 more local authorities in Catalonia are studying the implementation of PAYT schemes	Sack	**Esporles Municipality (Mallorca region):** fixed annual fee = €90 per household; variable fee = €1 per bag **Miravet and Rasquera Municipalities (Catalonia region):** fixed annual fee = €40 per urban households, €30 per rural household; variable fee = €0.30 per packaging bag (35l) and €0.70 per residual waste bag (17l) **Argentona Municipality (Catalonia region):** fixed annual fee = €95 per household; variable fee = €0.35 per packaging bag (35l) and €0.65 per residual waste bag (17l)
UK	Y	Extremely limited	Volume	**Blaby:** Each household is issued with one 140l wheeled bin; larger bins are available at an annual cost (additional 140l bin = €42 (£34.80); 240l bin = €42 (£34.80); 360l bin = €62 (£51)); discounts apply to households of five or more residents; 'side waste bags' are available for occasional extra waste = £6 for 3.

As stated above, it has not been possible within the scope of this study to undertake a comprehensive analysis of PAYT schemes across the EU-27. The decision was therefore taken to focus on example schemes from a small number of MS (Austria, Finland, Germany and Ireland) which have the most widespread use of PAYT schemes. The following sections present an overview of PAYT schemes in those MS.

3.3.4 COUNTRY CASE STUDIES

3.3.4.1 PAYT IN AUSTRIA

Main features of PAYT in Austria

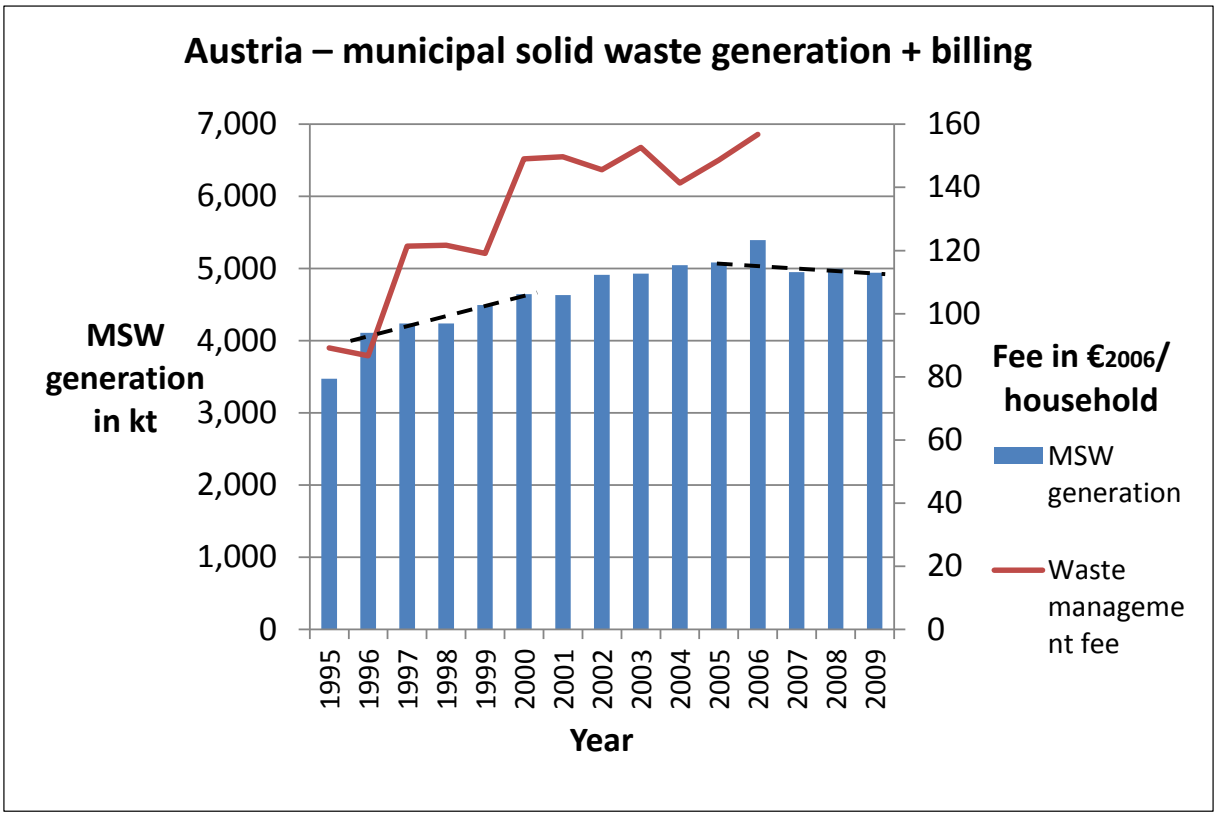
Waste management fees in Austria must be based on the size of a household's residual waste bin and the frequency of emptying. Fees for the collection and treatment of household waste must also correspond to the costs of the services rendered, as opposed to being profit-oriented. The fee levied per household is typically comprised of a system charge (which is used to finance waste management activities, such as the removal of litter) and a treatment charge covering the costs of waste collection and treatment. Possessing the legal authority to do so, most municipalities have formed waste management associations through which the collection and treatment of all waste is facilitated.

Impacts of PAYT schemes in Austria

Waste collection and treatment fees collected by Austrian municipalities increased from €72 per household in 1995 to €155 per household in 2006, due to a 23% increase in waste generation per household and the conversion of a landfill based waste disposal system to a recycling/incineration/biological based waste management system, which provides for the collection of different categories of household waste.

Figure 35 below compares the progression of annual municipal solid waste generation in Austria with the annual average waste management fee levied per household. From 1996 to 2000 annual MSW generation grew at a rate of 3.1% per year. From 2005 to 2009 annual MSW generation decreased. While the increased waste management fee likely influenced this downturn, it is not thought to be wholly responsible, as data uncertainty and the economic decline from 2008-2009 probably also had an effect. The doubling of waste management fees from 1995 to 2006 may have had a limited impact on household behaviour. This could be for several reasons, including: the fee being absorbed in an annual operation costs bill (which at €16 per household at its peak was not perceived as costly), increased household incomes offsetting the cost, or greater quantities of waste being produced.

Figure 35 Municipal solid waste generation and development of average household waste management fee (without tax) in year 2006 € in Austria (Eurostat 2011, Denkstatt 2009, Statistik Austria 2011)



Local case study – City of Vienna

At the local level, the picture is similar to the Austrian average, i.e. that an increased tariff possibly had a dampening effect on waste generation. However, the data do not provide conclusive proof of this. In the city of Vienna, for example, MSW generation grew at an average rate of 2.1% per year from 1996 to 2000 and levelled off after 2004. The waste management fee for the weekly emptying of a 110 litre bin decreased in real terms between 1995 and 2001 (as during this period the fee rate was not adjusted for inflation), but increased thereafter. In real terms, the waste management fee was 15% higher in 2009 than in 1995.

Strengths/Factors for Success/Weaknesses of the PAYT Schemes

A limiting factor of Austrian PAYT schemes concerning their ability to encourage waste prevention is that waste management fees are not felt to be significant enough to prompt wide-scale behavioural change from the public. The accounting method for determining fees contributes to the inefficacy of schemes due in large part to their broad variability. The following three methods for calculating waste fees provide some or no financial incentive to reduce waste generation:

- There is only one waste bin size (240 l) and every emptying costs €9.72;
- The annual fixed price is €22, and every emptying of a 240 l container costs €1.80; and

- The annual fixed price is €0.29 per litre of waste bin (€69.60 for a 240 l bin) and every emptying costs €7.90.

In Austria true PAYT schemes are only provided for single-family houses, with waste collection and treatment fees for apartment buildings being determined by apartment size (m²) as opposed to the size of the waste container.

3.3.4.2 PAYT IN FINLAND

Main Features of PAYT in Finland

Under the Finnish Waste Act of 1996, municipalities must provide for the collection and disposal of household waste. As such, they may levy a waste management charge covering not only the collection of waste, but investments in, and operation of, waste treatment facilities. Whilst PAYT schemes in Finland are not legally mandated, charges for municipal household waste are primarily determined by the quantity and type of waste being disposed of, as well as the frequency of collection. However, the charge must also be of enough significance to serve as a deterrent to waste production and encourage recycling.[20] In addition to a weight-based waste charge, an annual fixed fee is applied based on house type (e.g. single family home, apartment block). Residents also have the option of disposing of waste at collection points, though access to these collection points also incurs the levying of a fixed fee.[21]

Impacts of PAYT Schemes in Finland

On average, the annual fixed waste management fee for a single family house is €159, with the collection and treatment of a 240l bin averaging €6 per emptying, and a 600l bin averaging €9.50. The charges also vary depending on the type of waste disposed of. For example, residents who compost waste at home realise large savings over those who separate their compost from household waste for separate collection, and those who do not separate compost from household waste.[22]

Strengths/Factors for Success/Weaknesses of the PAYT Schemes

Regional cooperation in waste management activities has been emphasised as a means of improving the collection and processing of municipal waste throughout Finland. Currently, there are 30 regional municipal waste treatment organisations, with only some municipalities choosing to individually manage their waste activities. Further regional cooperation may enhance the effectiveness of collection route planning, ease the implementation of separate waste collection services (e.g. biowaste, packaging, recyclables, hazardous waste), and reduce the unit costs of waste management. Consumer awareness and advice through increased media coverage, the harmonising of waste rules and regulations throughout a region, the broader application of schemes (so that

[20] http://scp.eionet.europa.eu/facts/factsheets_waste/2009_edition/factsheet?country=FI

[21] http://www.finlex.fi/fi/laki/ajantasa/1993/19931072

[22] REPORT: Tietoa jätehuollosta - Vuoden 2009 jätemaksutiedot, p19

larger volumes are collected and unit prices reduced) have also been advanced as ways to optimise waste management.

Reliance on the private sector in the provision of services where they have achieved efficiency (i.e. collection and pre-treatment), as well as open competitive tendering processes, have also been recommended for the optimisation of MSW waste management.[23]

3.3.4.3 PAYT IN GERMANY

Main features of PAYT schemes in Germany

In Germany, there is a legal obligation for all citizens to utilize the local waste management system in order to dispose of household waste. While there is no federal legislation requiring the use of PAYT schemes, some federal states' waste legislation mandates that waste fees should promote waste prevention and recovery (e.g. § 8 of the Berlin Waste Law or § 6 of the Waste Law in Saxony-Anhalt). PAYT schemes in Germany encompass bin volume-based systems, sack volume-based systems, frequency-based systems and weight-based systems, with the majority of local disposal systems being volume- and frequency-based. The household subscription cost is comprised of an annual fee for waste collection dependent upon bin volume, as well as the frequency of collection. The fee levels differ regionally depending on the waste disposal capacities and the population serviced by the local disposal system.[24]

Impacts of PAYT schemes in Germany

Generally, PAYT schemes establish incentives for households to reduce the amount of waste they generate and thus lower collection costs. In many cases there is a basic fee and a variable fee, the latter applying in particular to residual waste and biowaste. The rate of the variable fee can be manipulated through household waste prevention and recovery activities. However, in order to effectively do so – and thus to stimulate cost savings for households and local authorities - appropriate infrastructure and services must be established.

Local Case Study – County of Aschaffenburg[25]

In 1997 the county of Aschaffenburg introduced a weight-based waste collection system, which was expanded in 1999 to include bulky waste. Under this system, costs are determined based on the

[23] Slide presentation

[24] Eunomia Research & Consulting *et al.* (2009): International Review of Waste Management Policy: Annexes to Main Report. Bristol, UK. See URL http://www.environ.ie/en/Publications/Environment/Waste/WasteManagement/FileDownLoad,21598,en.pdf, accessed 09/09/2011.

[25] Landratsamt Aschaffenburg (2008): Landkreis Aschaffenburg Umweltschutzbericht 2003 – 2007. Aschaffenburg. See URL http://shop.landkreis-aschaffenburg.de/artikel_28.ahtml, accessed 09/09/2011.

weight of waste collected in wheeled bins (which are weighed by the waste collection vehicle), with the separation of waste and recyclables, as well as general waste prevention activities, resulting in lower waste management fees. In the first year of the scheme, household waste generation fell to, and appears to have levelled off at, 8,000 tonnes. Accordingly, waste fees per 4-person household have fallen, from 157.99 € prior to 2003 to 149.45 € in 2007.

Despite significant investment and transformation costs, the weight-based system has helped to save between €1.53 and €1.79 million per year since its introduction. While it was feared that the weight-based system would increase illegal waste diversion, data from 1995 to 2007 reveals that quantities of illegally diverted waste did not increase as a result of the new scheme.[26]

Strengths / factors for success / weaknesses of PAYT schemes

In the course of establishing PAYT schemes, courts have fostered the legal position for households to be charged collection fees according to the actual amount of waste they generate. The courts have upheld several cases of residents requesting that municipalities provide smaller waste containers (40 l as opposed to the standard 80 l) and adjust charging accordingly.

In some places in Germany, illegal dumping of household waste has been reported. However, these cases are isolated and rare, and are not believed to reduce the overall effectiveness of PAYT in Germany.

3.3.4.4 PAYT IN IRELAND

Main Features of PAYT in Ireland

National legislation passed in 2003 mandated the transition from the previous fixed rate waste disposal charge to a weight/volume-based charge by 1 January 2005. Local authorities and waste collectors that have not yet transitioned to PAYT schemes continue to charge a flat-rate for waste disposal services.[27] Three systems of PAYT/PBU schemes are observable throughout Ireland, namely volume-based systems, tag-based systems and weight-based systems. The waste collection sector in Ireland is almost completely privatised; companies establish contracts with households and determine pricing, and only a limited direct role is played by the local authority.[28] Local authority waste services do compete in this market in a range of counties.

Impacts of PAYT/PBU Schemes in Ireland

Weight-based PAYT/PBU systems have been observed to bring about greater reductions in household waste production than tag-based systems, with a weight-based PAYT/PBU generating a 49% waste reduction from the year prior to its introduction to the year subsequent to its

[26] Landratsamt Aschaffenburg (2008): Landkreis Aschaffenburg Umweltschutzbericht 2003 – 2007. Aschaffenburg, p. 61.

[27] EPA doc, 1.3

introduction, while a tag-based PAYT/PBU generated a 23% decrease. Recycling rates have also appeared to benefit from PAYT/PBU schemes, with some local authorities reporting up to 46% recycling of waste (e.g. Waterford City Council, 2005) – significantly higher than the 32% average national rate in Ireland. Causality, however, is difficult to determine.[29] However, the introduction of PAYT has also been observed to encourage the environmentally damaging practice of waste burning among households, presumably to lower waste collection costs as revealed by a 2006 Environmental Protection Agency (EPA) study (2006).

Local Case Study

A notable example of the successful implementation and operation of a PAYT/PBU is the case of county Monaghan, which in 2003 transitioned from a fixed rate bin charge to a weight-based charge and kerbside recycling. In the first year of the scheme, waste sent to landfill fell by 25%, and by 40% in 2005. This reduction in waste production has been attributed primarily to an increase in recycling rates, and to home composting of organic waste and greater vigilance about bin contents.[30]

Strengths/Factors for Success/Weaknesses of the PAYT Schemes

A potentially confounding barrier to achieving significant reductions in household waste generation is that households in Ireland are not required to avail themselves to any private or municipal waste collection service. In general, dissatisfaction with PAYT/PBU schemes has been expressed over the difficulty of their implementation, primarily public resistance to charges, difficulties enforcing the adoption of the scheme, and additional pressure being placed on the resources of local authorities.[31]

Factors contributing to the success of PAYT/PBU schemes in Ireland include the creation of improved recycling and composting infrastructure, the justification of schemes on environmental grounds with careful selection of terminology, the simplification of administrative processes, plenty of notice before a scheme is implemented, public information campaigns on waste issues, and a thorough review of contractor/local authority responsibilities. Good monitoring and measuring techniques are also imperative if waste management policy is to be effective.[32]

3.3.4.5 PAYT IN ITALY

Main Features of PAYT in Italy

Since 1994, the twenty administrative regions of Italy have delegated the responsibility of waste management to the office Ambito Territoriale Ottimale (Optimal Territorial Scope, ATO), which sets targets for the landfilling of biodegradable municipal solid waste and the separate collection of sorted waste. Districts/provinces are responsible for meeting the targets established by their ATO,

[29] Dunne et al., pp.2831

[30] Dunne et al., pp.2828

[31] EPA doc, section 1.10

[32] Dunne et al., pp.2832

but are free to implement a waste management system of their choosing. Frequently, this is realised through the creation of waste consortia, which determine waste management policy.

Households and other producers of urban waste are compelled to finance their municipal waste management system by paying either a tax or a tariff/PAYT which covers the costs of waste management and related administrative activities. Taxes are determined according to the surface area (in square meters) of the house or business establishment under consideration. The tariff is composed of a) a fixed part (which funds essential components of the waste management infrastructure), and b) a variable element which is determined by (i) the quantity of waste generated, (ii) the standard of the service provided by the municipality, (iii) the costs of waste management. The variable component is determined by using either a presumptive calculative method or a precise method. The presumptive method is based on estimated quantities of waste set out for collection, while the precise rate is based on actual quantities of waste generated by a business or household.

Impacts of PAYT/PBU Schemes in Italy

While 1,000 out of 8,100 Italian municipalities have implemented PAYT waste management systems, the extant data on PAYT presents a somewhat contradictory account of the true effects of this economic instrument, in particular its ability to prevent/reduce waste generation. For example, a survey of four municipalities examining the effects of PAYT reported significant decreases in waste generation as a result of the implementation of this waste management system. However, several other studies have emphasized an increase in sorted waste as the main effect of PAYT.

Local Case Study

A survey of ninety-five municipalities of the Province of Trevisio evaluated the effects of PAYT and the usefulness of this economic instrument in improving waste behaviour. While not having developed their own unique waste management programmes, the municipalities voluntarily subscribed to regional waste consortia to which they delegated responsibility for these activities. As of 2008 forty-one of the municipalities had adopted door-to-door collection and flat fee pricing, while fifty-three had implemented door-to-door collection and PAYT pricing. The results of the study revealed that the fifty-three municipalities participating in PAYT/door-to-door waste management systems witnessed a 12.2% increase in their recycling ratio through PAYT pricing upon the 18.1% increase already achieved through the implementation of door-to-door collections. PAYT is observed to increase the quantity of sorted waste by 9.6%, while the generation of total waste is unaltered. A learning effect is also observed within these municipalities, as each additional year the PAYT system is in operation, the recycling ratio increases by 1.8%.

Strengths/Factors for Success/Weaknesses of the PAYT Schemes

PAYT has been observed to perform best when pricing is based upon the quantification of actual waste produced, as opposed to the volume of the waste container, or using the presumptive method. It also works best when coupled with door-to-door collection and domestic composting of organic waste. Annual public information campaigns to increase awareness of PAYT systems and

their environmental purpose, and the monetary savings achievable through sorting and reducing waste, are advised to optimize the performance of these systems.

3.3.5 SUMMARY AND CONCLUSIONS

PAYT schemes appear to be most effective when the fees payable by households are at levels high enough to encourage reflection by householders on their waste generation behaviour, and engagement with separate collection systems. At the same time, there are arguments for not making charges so high that they give a strong incentive for illegal dumping. PAYT schemes have been shown to both decrease the generation of waste (i.e. waste prevention) and increase the proportion of generated waste that is separately collected. The observed effects depend not only on the charging scheme, but the design of the waste management system, and the way in which waste was being managed before the implementation of PAYT.

PAYT systems will work best where the infrastructure for recycling is both comprehensive and convenient for the user; in addition, higher costs for disposal will make it more likely that a PAYT scheme will be financially viable and attractive to a local authority, whilst increasing the likelihood that residents will make savings when they reduce waste generation and/or increase recycling.

With regards to waste prevention, weight-based PAYT schemes (i.e. determining waste collection and treatment fees based on the weight of the waste collected) tend to be the most successful, followed by volume and frequency-based/sack-based systems, and then volume-based systems. Weight-based systems may require more expensive infrastructure, but this can be justified by their better results.

PAYT schemes are decided and implemented at the local (municipal) level, and as such a legal framework may be useful to promote the more widespread use of PAYT. Regional cooperation in waste management may enhance the effectiveness of PAYT, e.g. by optimising the planning of collection routes, easing the implementation of separate waste collection services and economising overall waste processing.

Greater importance should be placed on increasing consumer awareness through public campaigns, harmonising waste rules and regulations to avoid confusion, and also taking steps to combat the illegal dumping, littering or burning of waste.

Feedback gathered from stakeholders during the study suggests that PAYT schemes are welcomed as a means of sharing waste management (notably collection and recycling) responsibilities among public authorities, industry and consumers. They can help to achieve a higher level of selective household collection. They are also better aligned with the polluter pays principle.

Stakeholders suggested that in order to achieve optimal performance, PAYT schemes should be based on the precise quantification of actual generated household waste, they should establish door to door collection services, the public should be educated through regular awareness campaigns, and schemes should encourage the domestic composting of organic waste.

With regards to assessing the impacts of PAYT schemes, stakeholders pointed out that it is important to compare levels of waste generated prior to and subsequent to their introduction, rather than looking simply at the total amount of waste recovered by these schemes. Stakeholders also stated the importance of trying to isolate the effects of PAYT and extended producer responsibility (EPR) schemes independent of similar policies, in order to observe their true effect and to avoid the 'double counting' of improvements in waste management performance.

3.4 PRODUCER RESPONSIBILITY SCHEMES

In terms of producer responsibility schemes, this study primarily addresses schemes that have been developed to implement the requirements of EU legislation (the Packaging, WEEE, ELV and Batteries Directives).

The concept of producer responsibility includes a wide range of instruments, from ecodesign measures to financial contributions towards separate collection and recycling and so on. The focus of this study is on those schemes in which companies obligated under national legislation are required to pay fees to support the costs of organising the collection and recycling of specific waste streams to meet national targets (referred to as 'producer fee schemes'). Other measures used by MS, including deposit refund schemes and taxes, are not addressed in detail in this report.

This section summarises the information gathered on producer responsibility schemes. The targets of the various national schemes are summarised in an annex to this report.

3.4.1 OVERVIEW OF PRODUCER RESPONSIBILITY SCHEMES

Table 6 below provides a synthetic overview of the presence of producer fee schemes for packaging, WEEE, ELV, batteries and other waste streams in the 27 MS. Whilst attempts have been made to be as complete as possible, this is not presented as a fully comprehensive summary of all producer schemes in the MS.

Table 6 Summary of presence of producer responsibility schemes in the 27 MS

Member State	Packaging	WEEE	ELV	Batteries	Other
AT	•	•	•	•	Tyres; waste mineral and edible oils; paper; bulky metals, glass, plastics and wood; plastic foils; medical waste; compound packaging (Tetra-Pak); expanded polystyrene
BE	•	•	•	•	Paper/card; plastic bags; disposable plastic kitchenware; car batteries; unused/old medicines; oils; tyres; photo-chemicals
BG	•	•	•	?	Tyres; plastic bags; tax on products generating widespread waste
CY	•	•	•	•	Paper/card; motor oils; tyres
CZ	•	•	•	•	?
DK	-	•	•	•	Paper/card; tyres
EE	•	•	•	•	Tyres; old medicines; paper
FI	•	•	•	•	Paper; tyres
FR	•	•	•	•	Paper/card; tyres; unused medicines; textiles
DE	•	•	•	•	Commercial waste; construction waste; waste oil
GR	•	•	•	•	?
HU	•	•	?	?	Tyres
IE	•	•	•	•	Tyres; farm plastics; plastic bags
IT	•	•	•	•	Tyres

Member State	Packaging	WEEE	ELV	Batteries	Other
LV	•	•	•	•	Paper/card; tyres; tax on tyres, lubricating oils and oil filters
LT	•	•	•	•	Paper/card; tyres
LU	•	•	Participates in BE scheme	•	n
MT	•	?	?	•	Eco-contribution Act
NL	-	•	•	•	Paper/card; tyres
PL	•	•	•	•	?
PT	•	•	•	•	?
RO	•	•	?	?	?
SE	•	•	•	•	?
SK	•	•	•	•	Paper/card; tyres
SI	•	•	•	•	Waste from hazardous pesticides; graveside candles; medical waste; tyres; lubricating oils
ES	•	•	•	•	Tyres; mineral oils
UK	•	•	•	•	?

3.4.2 PACKAGING

Producer fee schemes for **packaging waste** have been identified in 24 MS. These schemes essentially oblige producers to financially support (to varying degrees) the implementation of recycling schemes for packaging waste. The majority of MS have implemented a form of producer fee scheme in which the fees support a proportion of the costs of packaging waste collection and recycling; the UK has opted to put in place a system of tradable credits for packaging. Three MS (Denmark, Hungary and the Netherlands) have opted to apply taxation systems and/or deposit-refund systems. The main types of schemes used for packaging waste in the MS are summarised in Table 7 below.

Table 7 Overview of main type of producer responsibility schemes for packaging in the EU-27

Member State	Main type of scheme	Member State	Main type of scheme
AT	Producer fee scheme (some deposit-refund schemes)	**LV**	Producer fee scheme
BE	Producer fee scheme (some deposit-refund schemes	**LT**	Producer fee scheme
BG	Producer fee scheme	**LU**	Producer fee scheme
CY	Producer fee scheme	**MT**	Producer fee scheme
CZ	Producer fee scheme	**NL**	Tax (some deposit-refund schemes)
DK	Tax and deposit-refund schemes	**PL**	Producer fee scheme
EE	Producer fee scheme / deposit-refund schemes	**PT**	Producer fee scheme
FI	Producer fee scheme / deposit-refund schemes	**RO**	Producer fee scheme
FR	Producer fee scheme	**SE**	Producer fee scheme / deposit-refund schemes
DE	Producer fee scheme / deposit-refund schemes	**SK**	Producer fee scheme
GR	Producer fee scheme	**SI**	Producer fee scheme
HU	Tax	**ES**	Producer fee scheme
IE	Producer fee scheme	**UK**	Producer fee scheme (tradable credits)
IT	Producer fee scheme		

This study focused primarily on producer fee schemes for packaging waste. Details of packaging taxes and deposit-refund schemes for packaging that have been implemented in the MS are included in an annex to this report.

3.4.2.1 PRODUCER FEE SCHEMES FOR PACKAGING

Table 8 below focuses on the level of fees paid by producers to the different producer fee schemes. Producers generally pay a fee to join the scheme, together with ongoing fixed and variable fees. The broad aim of such schemes is to cover the costs of collection, sorting and recycling of sufficient packaging to meet relevant national targets once the packaging becomes waste.

Table 8 Overview of fees paid to producer fee schemes for packaging

Member State	Annual fixed participation fee	Maximum average fee (latest available data) per tonne of:					
		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	?
BG	min. €75 for producers placing ≤10t on market	€80	€40	€100	€30	€130	€50
CY[3]	?	€47.14	€29.06	€21.38	€95.39	€105.89	?
CZ[2]	€65 registration fee	€106.44	€58.67	€81.76	€61.39	€215.99	€42.14
DK	No producer fee scheme	-	-	-	-	-	-
EE	?	€110	€100	€260	€260	€410	€40
FI	min. €203 for producers with turnover over €1m	€23.50	€10	€21	€3	€21	€0.40
FR[1]	?	€163.30	€4.80	€60.60	€30.20	€237.80	?
DE[3]	?	€175	€74	?	?	€1,296	?
GR	€150	€52.50	€10.90	€8.80	€21	€66	€9.50
IE	?	€22.73	€9.18	€83.62	€78.51	€89.16	€10.60
IT	?	€22	€17.82	€52	€31	€140	€8
LV	€84 min. licence fee	€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

Member State	Annual fixed participation fee	Maximum average fee (latest available data) per tonne of:					
		Paper	Glass	Aluminium	Steel	Plastic	Wood
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]	max. €500 one-off joining fee	€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
UK	-	Varies depending upon value of tradable certificates					

[1] *Scheme(s) known to cover only household packaging waste*

[2] *Scheme(s) known to cover household and commercial packaging waste*

[3] *Scheme(s) known to cover household, commercial and industrial packaging waste*

The data obtained shows huge variation in the level of fees paid per tonne of packaging material placed on the market in the MS, as shown in Figure 36 and Figure 37 below. It should be noted that there are some limitations to these comparisons. Fees may vary for several reasons, including:

- The schemes cover a varying percentage of the costs of collecting and recycling packaging;
- Because MS targets vary and because fees tend to relate to packaging placed on the market, where targets are lower the fees per unit of packaging placed on the market will also be lower;
- The use of different collection systems within the schemes (partly reflecting different targets), and the differing cost profiles of these collection systems; and
- Schemes in some MS cover household, commercial and industrial packaging waste, whilst some cover only household and commercial, and others only cover household packaging waste.

Notwithstanding these factors, the variation across otherwise similar schemes is considerable, suggesting varying levels of efficiency in the schemes' operation. This highlights the potential for efficiency savings within the operation of the schemes.

Figure 36 Maximum fee charged under producer fee schemes (€ per tonne) for paper, glass and wood packaging (latest available data)

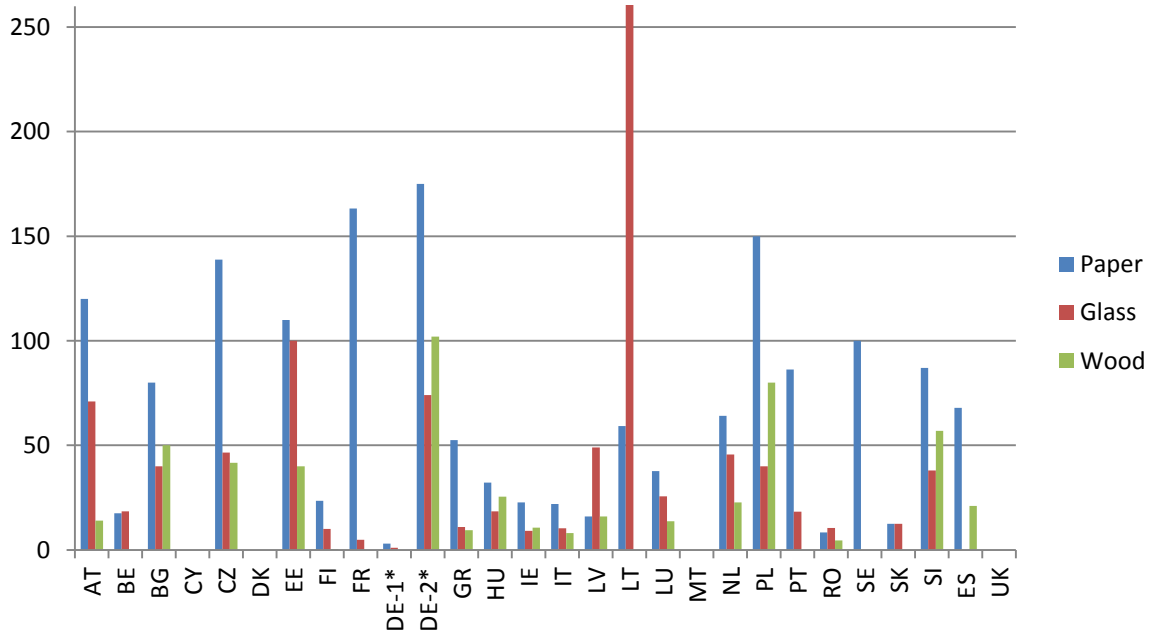
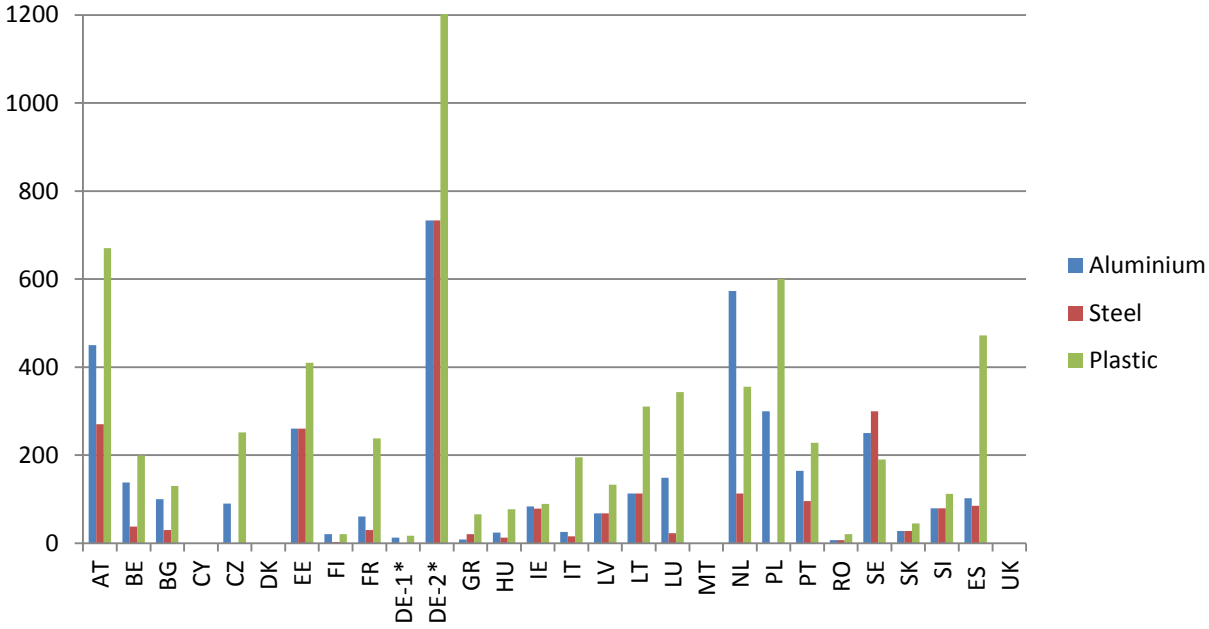


Figure 37 Maximum fee charged under producer fee schemes (€ per tonne) for aluminium, steel and plastic packaging (latest available data)



**Two figures are included for Germany as different sets of figures were found from the same source and could not be verified*

The efficiency and effectiveness of the schemes also depends on the proportion of costs of collection, sorting and recycling of waste packaging that are covered by producers' contributions.

Research under this study suggests that the schemes in only 3 MS (Austria, Belgium and Germany) currently appear to cover the full costs to local authorities/waste collection authorities of these activities. In Finland, after implementation of the new waste law (likely to be in 2013), producers will cover 100% of the costs of packaging waste management. In other countries it is unclear whether there is full cost coverage by producers.

Table 9 Percentage of costs covered by producers and public/private nature of packaging producer responsibility schemes

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
AT	100% (recycling, recovery of packaging waste).	Producer-led. PRO Altstoff Recycling Austria – ARA
BE	Packaging/Producer fee scheme 100% (collection, sorting and recycling of the packaging waste, as well as costs of collecting and disposing of/treating packaging as residual waste)	Producer-led. PRO FOST Plus VAL-I-PAC
BG	Producer fee scheme Information not found. Participating producers/importers finance the collection, recovery and recycling of packaging waste, and the implementation of information and communication campaigns to promote separate collection and recycling.	Information not found. PRO ECOPACK BULGARIA JSC
CY	Producer fee scheme Believed to be around 80% of collection, recycling and recovery costs	Information not found. PRO Green Dot Cyprus
CZ	Producer fee scheme Information not found. Participating producers/importers contribute financially to the collection, sorting and reprocessing of packaging waste.	Producer led and funded. PRO EKO-KOM

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
DK	N/A Packaging waste management costs are included in the budgets of local authorities and are financed via household, consumer, retailer and industry taxation and management fees.	Government-led
	Deposit refund All scheme costs covered by producers and unclaimed deposits. Registration and collection/logistics fees charged to importers and producers (domestic and external) of beverages.	PRO Dansk Retursystem (DRS) Appointed by government to lead the recovery/recycling effort. Private operators (recycling) and local authorities (treatment) are responsible for management of household and commercial packaging waste.
EE	Deposit Refund All scheme costs covered by unclaimed deposits. A Packaging Excise Duty is levied on importers and producers of beverage packaging that fail to establish a packaging waste collection/recovery system.	Information not found. PRO Eesti Pandipakend
	Producer fee scheme Information not found.	Producer-led. PRO Estonian Recovery Organisation
FI	Deposit Refund All scheme costs covered by unclaimed deposits.	Government led, producer funded.
FR	Producer Fee Scheme Information not found. Fillers of packaging pay for the collection, recycling and recovery of packaging. The scheme compensates local authorities for their role in collecting household packaging	Producer-led. PRO Eco-emballages Adelphe Cyclamed
DE	Deposit Refund Mostly covered by producer fees	Producer-led
	Producer fee scheme 100% of costs covered by producers	Producer-led PRO Nine schemes

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
GR	Producer fee scheme Information not found. Participating producers/importers contribute financially to the functioning of the scheme.	Government led, producer funded.
HU	Producer fee scheme Information not found.	Producer-led. PRO ÖKO-Pannon
IE	Producer fee scheme Information not found. Fillers of packaging pay for the collection, recycling and recovery of packaging. Municipalities are reimbursed for 40% of the costs of the operation of the recycling scheme.	Voluntary agreement between industry and the Government of Ireland, in response to the EU Packaging Directive. PRO Repak
IT	Producer fee scheme Information not found. Participating producer/importer contribute financially to the activities of CONAI.	Producer-led. PRO CONAI Collaborates with ANCI (national association of Italian municipalities).
LV	Deposit Refunds 100%	Voluntary deposit system for reusable packaging Producer-led.
LT	Producer fee scheme Information not found. Participating producers/importers contribute financially to the management of waste packaging.	Information not found. PRO Žaliasis tašas
LU	Producer fee scheme Information not found.	Producer-led. PRO VAROLUX

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
MT	Producer fee scheme Information not found.	Government-led, producer funded. PRO GreenPak
NL	Deposit Refunds Information not found.	The Netherlands has a voluntary deposit refund scheme on single use beverage packaging.
	Producer fee scheme Information not found. Importer/producer is responsible for the management and separate collection of packaging waste.	Producer-led, public authorities responsible for collection of household glass and paper/cardboard packaging waste.
PL	Producer fee scheme Information not found.	Producer-led. PRO Rekopol Recovery Organisation S.A.
PT	Producer fee scheme Information not found.	Producer-led. PRO Ponto Verde
RO	Producer fee scheme Information not found.	Producer-led. PRO ECO - ROM AMBALAJE (ERA) INTERSEROH ECOLOGIC 3 R SC SOTA GRUP21 SRL SC ECO-X SRL
SE	Deposit Refund 100% funded by producers and unclaimed deposits	Producer-led PRO Ab Svenska Returpack-Pet Svenska Returglas 50-Cl Ab Ab Svenska Returpack.

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
SK	Information not found.	Producer-led. PRO ENVI-PAK
SI	Producer fee scheme Information not found. Producers/importers cover the costs of collecting, and reprocessing or disposal of packaging waste.	Information not found. PROs SLOPAK INTERSEROH d.o.o. EKODIN d.o.o. SUROVINA d.d.
ES	Producer fee scheme Information not found.	Information not found. PROs ECOEMBES, ECOVIDRIO
UK	Producer fee scheme Varies, but typically of the order of 5-10%. The UK system was not intended to cover the full cost of collection, recovery and recycling of packaging waste.	More than 20 compliance schemes

NB The lists of PROs mentioned in the table for each MS may not exhaustive; other PROs may also be present.

3.4.2.2 PACKAGING PRODUCER RESPONSIBILITY IN BELGIUM

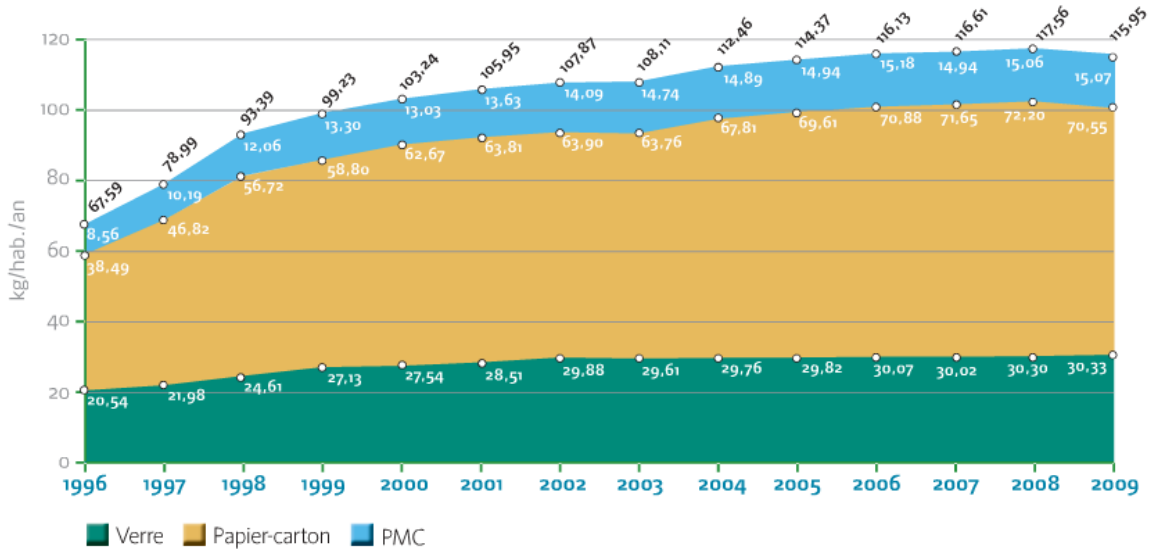
Main features

Directive 94/62/EC was transposed into national Belgian law as a Cooperation Agreement between the three Belgian regions on 5 March 1997. It obliges parties responsible for placing packaging on the Belgian market to comply with three legal obligations: 1) a collection obligation (80% recycling and 90% recovery for household packaging waste by 2009; 2) an information obligation (to inform the governing entity (IPC) of the nature of packaging produced and recycling percentages achieved); and 3) a prevention plan describing steps to reduce the quantity and the harmfulness of packaging, as well as detailing the objectives the company wishes to achieved.

Packaging compliance in Belgium has been largely privatised, with two commercial 'Green dot' organisations responsible for coordinating and financing the collection and end-of-life treatment of packaging waste across the country: Fost Plus and VAL-I-PAC. Subscription to a 'Green dot' organisation is not obligatory, however. Companies may choose to discharge their packaging waste obligations themselves.

Figure 38 provides an overview of the evolution/progression of waste packaging collection rates. Glass is represented in green, paper and cardboard in orange, and PMC in blue.

Figure 38 Packaging waste collection rates in Belgium from 1996 to 2009



A **deposit refund scheme** for drinks containers has operated in Belgium at Federal level since 1993. In 2007 a revised ecotax was introduced for both disposable and reusable drinks containers (€1.41/hectolitre for reusable drinks containers, and €9.86/hl for disposable drinks containers). In April 2004, the Federal government introduced a series of **packaging taxes** to include taxes on: single use drinks packaging (April 2004), reusable drinks packaging (March 2007), and plastic bags, disposable cutlery, plastic clingfilm and aluminium foil (April 2007).

Impacts

In 2010, out of the 755,000 tonnes of household packaging put on the Belgian market by Fost Plus members, 91.5% was recycled and 94.9% recovered, exceeding the minimum overall recycling target of 80% and the 90% recovery target.

Strengths / factors for success / weaknesses

The success of the collection, sorting and recycling of packaging waste in Belgium is attributable to close cooperation between the many actors involved throughout the process, as well as a coherent strategy. However, Fost Plus identifies additional areas for improvement, including: reinforcement of responsibility-sharing between operators involved in waste management; the simplification of the legal framework directing the Green dot scheme; and ensuring the transparent traceability of funds (€5.5 million) allocated every year to regions to improve waste packaging management.

3.4.2.3 PACKAGING PRODUCER RESPONSIBILITY IN GERMANY

Main features

The EU Packaging Waste Directive was transposed into German national law via the waste packaging ordinance of 1991[33], which specifies producer responsibilities concerning various waste categories, though primarily beverage packaging (deposit on beverage packaging scheme) and all sales packaging aimed at private consumers (Green dot scheme).

The deposit scheme for beverage packaging, which came into force in January 2003, levies a €0.25 deposit on non-environmentally friendly one-way beverage packaging for mineral water, beer, alcoholic mix-drinks and soft drinks (0.1-3 litres), paid by the consumer at the time of purchase and refunded when the packaging is returned. The Green dot scheme, introduced in 1990 by the Duales System Deutschland GmbH, covers all packaging that is subject to German waste law and packaging aimed at private consumers. Under this scheme, manufacturers and fillers of such packaging are obliged to take part in one of the nine existing compliance systems for sales packaging management. The collection, sorting and recycling of used sales packaging is financed via participation payments paid into the dual systems by trade and industry manufacturers and distributors. These costs are usually integrated into product prices, and thus passed on to end consumers. Though enacted by government legislation, the scheme is operated by industry.

The waste packaging ordinance set a deadline of 31 December 2008 for the following targets to be met for the packaging of consumer goods:

- 65% (in weight) must be recovered; and
- 55% (in weight) must be recycled.

Impacts

For several years Germany has achieved the highest rates for recycled plastic packaging in the EU (47% in 2008), while the share of recovered waste packaging steadily increased between 1991 and 1997. Since then, the share has remained relatively stable (between 73% and 78%).

Strengths / factors for success / weaknesses

Although the 'Green Dot' is displayed on relevant packaging the system also has an impact on other fractions of household waste. Contamination of the recycling container with non-packaging materials has been observed. For example, products such as plastic cups or spoons have been disposed of in Green Dot containers, as opposed to residual waste bins.[34]

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[33] http://www.bmu.de/english/waste_management/downloads/doc/37115.php, accessed 07/09/2011

[34] Eunomia Research & Consulting *et al.* (2009): International Review of Waste Management Policy: Annexes to Main Report. Bristol, UK. See URL http://www.environ.ie/en/Publications/Environment/Waste/WasteManagement/FileDownLoad,21598,en.pdf, accessed 09/09/2011.

Opinions on the cost/benefit ratio of the system differ. For example, one German research institute opined that the system is a burden for the German economy when compared to a system based on incineration and landfills.[35] Another report stated that alternative systems of waste treatment cannot achieve the significant external benefits of the 'Green Dot' Scheme.[36] It should be noted that in Germany (and several other MS) there is competition between a number of schemes, which may have the tendency to increase their cost.

A strength of the system is that it motivates industry to reduce packaging by putting greater thought into product design. Industry has a clear incentive to reduce the amount of packaging it generates, as it is faced with high tariffs determined by the material composition of its packaging waste, and bears full responsibility for operating the waste packaging management system. Furthermore, as payments to contractors responsible for collection are based on material specific prices there is a greater incentive to collect larger amounts of packaging waste.[37]

3.4.2.4 PACKAGING PRODUCER RESPONSIBILITY IN THE NETHERLANDS

Main features

The Dutch Packaging Decree of 2005, which is based on European Directive 94/62/EC and the revised Packaging Directive 2004/12/EC, stipulates that Dutch producers and importers of packaged products are responsible for the separate collection and recycling of packaging waste and also for waste prevention. The packaging regulation covers plastic, paper and cardboard, metal, wood, textile and glass packaging.[38]

The producer or importer is required to ensure that 70% by weight of the total quantity of packaging is put to good use (75% from 2010), whilst 65% by weight must be recycled (70% from 2010). 'Good use' is understood as the re-use or recycling of material, primary use as a fuel or primary use for energy generation. Individual recycling percentages per material have also been defined. According to the Decree, producers and importers of packaged products may either achieve the targets on an individual basis or join a collective organisation (i.e. Nedvang).

In 2008 a packaging tax was introduced for companies placing more than 15,000kg of packaging onto the Dutch market, with the exception of logistics packaging (e.g. pallets, trolleys, large

[35] Prognos AG (2002): Assessment of Sustainability and the Perspectives of the DSD, Duesseldorf, June 2002.
36 Öko-Institut (2002): Advantage of the Green Dot for the Environment, Duesseldorf, March 2002.

[37] EU Commission (2000): European Packaging Waste Management Systems. Final Report. Brussels.

[38] Oosterhuis, F. H. (2009): Economic instruments and waste policies in the Netherlands. Inventory and options for extended use.
http://www.recyclingnetwerk.org/andere_themas/2009/09/statiegeld.php, assessed 09/09/2011,
See URL: http://www.verpakkingsmanagement.nl/thema's/duurzaamheid/42-statiegeld+voor+pet+niet+eco-effici%C3%ABnt.html, assessed 09/09/2011

crates). Last-minute packaging (e.g. bags and fast food containers) are also exempt - in this case the tax is paid by the manufacturer or importer of the (empty) packaging. The overall objective of the packaging tax is to meet the national recycling targets of 32% of plastic packaging by 2009, 38% by 2010 and 42% by 2012. These targets are aligned with Dutch obligations under the Packaging Directive. The tax finances a Waste Fund, which is to be used to assist in the provision of the separate collection of plastic packaging material from households at the municipality level.

Impacts

Figure 39 shows that the amount of different types of waste packaging recycled increased between 1997 and 2008. Figure 40 shows that plastic packaging had the largest increase between 1997 and 2008, with three times more packaging being recycled. Recycling of wooden packaging nearly doubled between 2000 and 2008.

Figure 39 Recycled packaging waste in Netherlands (Source: Eurostat)

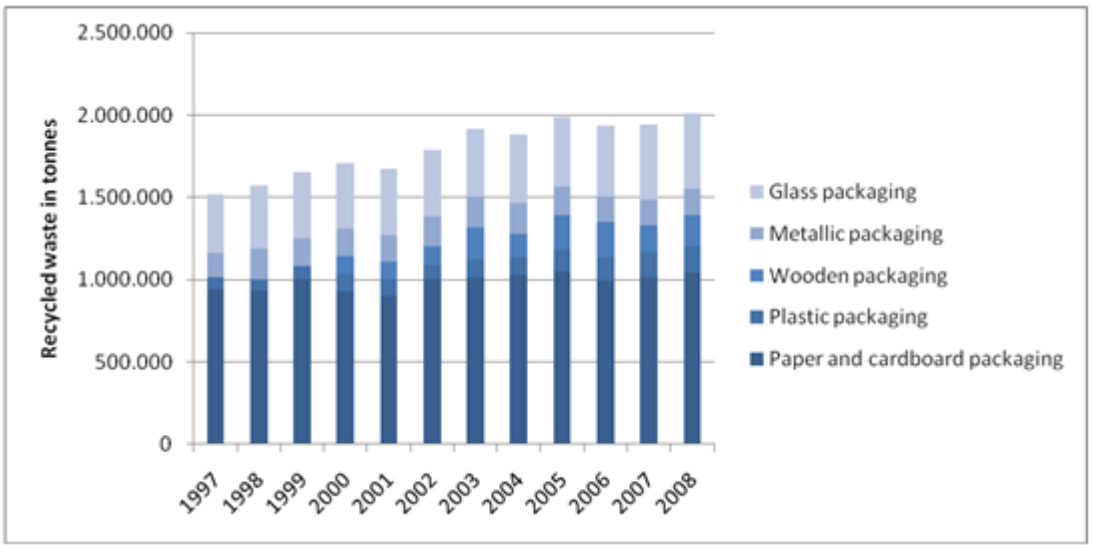
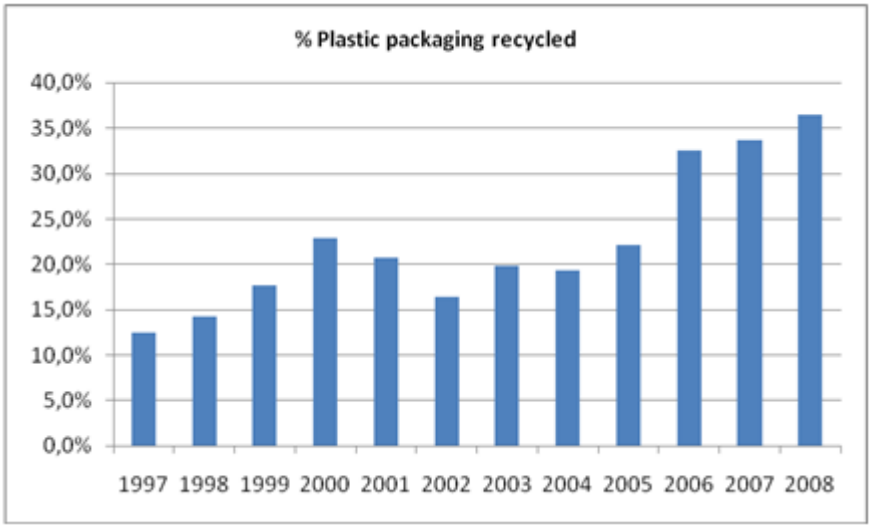


Figure 40 Recycled plastic packaging as percentage of total plastic packaging waste (Netherlands) (Source: Eurostat)



The statistical data shows that the Dutch regulation has already achieved increased recycling rates for packaging waste.

Strengths / factors for success / weaknesses

The simplification of the packaging tax in early 2008 along with the convening of tailor-made agreements for sectors and individual companies, and the narrowing of the definition of packaging are believed to have improved producer performance. These changes demonstrate that it was necessary to adjust the packaging tax to the situation in practice, to reduce the administrative burden and simplify the practical implementation of the regulation.

3.4.2.5 PACKAGING WASTE MANAGEMENT PERFORMANCES

Table 10 below attempts to compare the overall cost of contributions to the producer fee schemes with the packaging waste recovery and recycling performance of the MS for which data has been found. (NB The overall cost ranking was calculated by ranking the MS in terms of the cost per tonne charged to producers for the packaging materials for which costs were found, then averaging these rankings to provide an overall cost ranking). A figure is also included for the cost per capita in 2008 for paper plus plastic plus glass (calculated by multiplying the waste generation per capita of plastics, glass and paper and cost per tonne charged to producers/users for the respective waste streams). Similar figures are also included for the cost per capita in 2008 aluminium and steel, and for wood, where sufficient data has been found. The highest recovery and recycling rates and the lowest costs are indicated in dark green; the lowest recovery and recycling rates and the highest costs are indicated in dark red.

Table 10 Cost of contributions made to producer fee schemes compared with recovery and recycling performance

Member State	%recycled	%recovered	Overall cost ranking	EUR per capita per year, 2008 (paper, plastic and glass)	EUR per capita per year, 2008 (aluminum and steel)	EUR per capita per year, 2008 (wood)
BE	78	95	15	7.37	n.a	n.a
DE	71	95	1	17.41	8.14	3.19
NL	73	95	5	15.24	n.a	0.74
LU	63	93	12	19.54	n.a	0.25
AT	68	91	3	29.70	n.a	0.11
IT	60	78	17	7.46	0.34	0.36
CZ	67	74	7	9.37	0.30	0.43
PT	61	67	9	14.91	n.a	n.a
ES	59	66	8	21.72	n.a	0.36
FR	50	66	13	18.71	0.36	n.a
SI	52	58	10	6.70	n.a	1.01
FI	52	58	20	1.70	n.a	0.02
HU	51	57	16	3.03	n.a	0.57
LT	52	53	6	13.88	n.a	n.a
LV	47	52	14	4.29	n.a	0.44
BG	51	51	11	2.81	n.a	0.05
PL	43	51	2	16.49	0.63	2.11
SK	48	50	19	1.15	n.a	n.a
EE	43	44	3	31.89	n.a	0.22
GR	44	44	17	3.63	0.24	0.06
RO	34	41	21	0.55	n.a	0.05

The overall cost ranking was calculated by ranking the MS in terms of the cost per tonne charged to producers for the packaging materials for which costs were found, then averaging these rankings to provide an overall cost ranking (1 = highest cost, 21 = lowest cost).

The cost per capita per year (representing the cost of the packaging schemes per citizen) was calculated by multiplying the waste generation per capita of plastics, glass and paper and cost per tonne charged to producers/users for the respective waste streams.

Table 10 must be viewed with certain caveats in mind, notably that the data is not complete for all MS, that the schemes in different MS cover different types of packaging – only household, household and commercial, or household, commercial and industrial – and that the schemes have varying degrees of cost coverage by producers' contributions. The analysis in Table 10 does not therefore show any clear relationship between the costs charged under producer responsibility schemes and the packaging recovery and recycling performance of the MS, with some relatively 'cheap' schemes demonstrating high levels of recovery/recycling (most notably Belgium and Luxembourg) and some 'expensive' schemes demonstrating low levels of recovery/recycling (most notably Poland and Estonia). Given that the Belgian scheme covers 100% of the costs of collection

and recycling of packaging, this serves to re-emphasise the point made earlier – that there appears to be considerable inefficiency in a number of existing schemes.

The Belgian Fost Plus scheme for municipal packaging waste (as described in section 3.4.2.2) could perhaps be judged to be the best in terms of 'value for money', if value for money is judged in terms of the amount of packaging waste recovered and recycled compared with the cost of contributions to the scheme. According to figures from Fost Plus, the cost of the system per inhabitant in 2010 was €6.50 (indicating a lower cost even than that indicated in Table 10 for only paper, plastic and glass in 2008, which suggests a further increase in efficiency over those two years). The scheme has the best recovery and recycling rates of the schemes reviewed during the current study, and was the 7th cheapest out of 21 MS schemes for which producer contribution data was found. In 2010, the scheme recovered 94.9% of the household packaging put on the Belgian market by its members, and recycled 91.5% of that packaging (13.5% higher than the overall reported national recycling rate). Its members represent around 93% of the packaging market in Belgium.

According to Fost Plus, its success can be attributed to having one single accredited producer responsibility organisation with solidarity amongst producers, close cooperation between all the actors involved in the packaging/packaging waste chain (industry, retailers, public authorities and consumers), good communication with – and active participation of – citizens, and a gradual roll-out of the scheme across the country to ensure its successful application.

3.4.3 WEEE

Reliable information on the nature of producer responsibility schemes for **WEEE** has been found in 25 MS. However, comparable data on costs paid into the schemes proved rather more difficult to find; the way in which the costs are determined varies between MS, and in several cases information on costs is only readily available to members of the schemes. Figure 41 and Figure 42 below therefore present the limited amount of data of this type that was found during the study.

Figure 41 Maximum payment to WEEE scheme, € per item sold

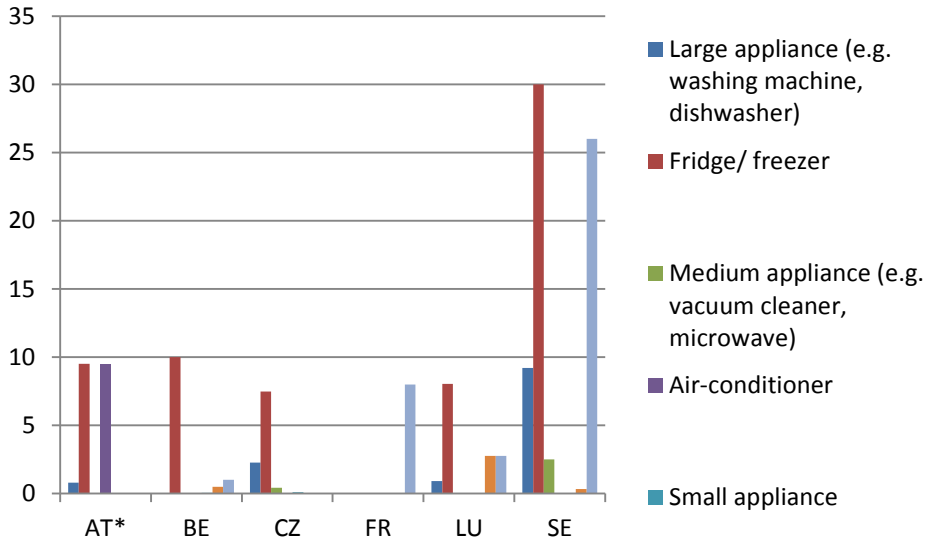
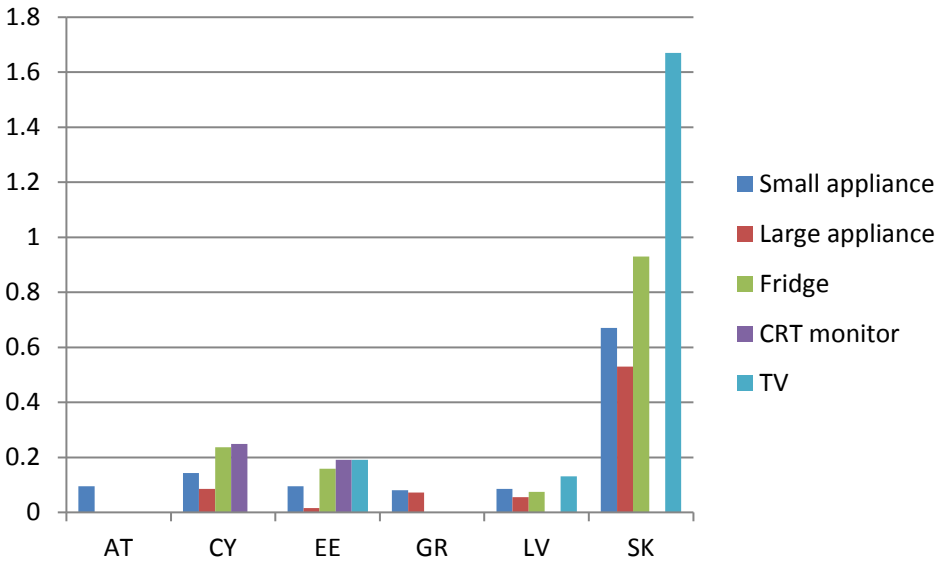


Figure 42 Maximum payment to WEEE scheme, € per kg of type of appliance sold



Despite the lack of data on the actual amounts charged to producers, information found on the vast majority of WEEE schemes suggests that the fees charged are based on the amount placed on the market by producers (either per unit, or per kg/per tonne). This helps to ensure that the cost to producers is in proportion to their market share of EEE sold in the country in question. In Ireland, fees are based more specifically on the EEE-related turnover of the company (used as an equivalent to market share) and the length of time for which the company has been placing EEE on the Irish market. Fees paid in Germany vary according to the contracts negotiated with waste management firms. Research indicates that the fees are translated to 'visible fees' for consumers (i.e. the fee paid to the scheme is displayed to the consumer at the point of sale) in Belgium,

Romania and Slovakia (although the fee will no longer be displayed to the consumer in Romania after February 2013).

Again, the extent to which the schemes provide incentives to producers to change behaviour can be considered (other things being equal) to depend on the proportion of costs of collection, recycling and recovery of WEEE that are actually covered by producers' contributions. The research suggests that producers' contributions to the identified schemes in 8 MS (Austria, Belgium, Cyprus, the Czech Republic, Denmark, Ireland, Latvia and Poland) cover either the full costs of these activities, or the full costs once the WEEE has been delivered to specific collection points. In Finland, producers pay for 100% of the costs for management of business-to-consumer WEEE; for business-to-business WEEE, producers and end users may agree to share costs. In other MS it is unclear whether there is full cost coverage by producers. Where producers pay only when material has been delivered to collection points, it is likely that the costs of organising and implementing separate collection schemes may – depending upon existing schemes – be the most significant cost in the overall system.

Table 11 Percentage of costs covered by producers and public/private nature of WEEE producer responsibility schemes

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
AT	100% of costs from the WEEE collection sites onwards	Producer-led. PRO UFH (Umweltforum Haushalt) Elektro Recycling Austria GmbH (ERA) Erfassen und Verwerten von Altstoffen GmbH (EVA) European Recycling Platform (ERP)
BE	Information not found.	Producer-led PRO RECUPEL
BG	Information not found.	Information not found. PRO Ecobultech ELTECH Resource
CY	100% (covers collection, recycling and recovery).	Information not found. PRO Green Dot Cyprus

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
CZ	Information not found.	Government- led, producer funded.
DK	Producers responsible for providing collection equipment and treating WEEE from private households; local authorities responsible for collection and sorting.	Government-led. PRO DPA-System (Danish Manufacturer Liability System)
EE	Information not found.	Information not found. PRO MTÜ EES-Ringlus MTÜ Eesti Elektroonikaromu Ekogaisma SIA Eesti filiaal
FI	100%	Producer-led, government regulated
FR	Information not found.	Producer-led.
DE	Producers are responsible for providing collection equipment and ensuring their emptying; municipal collection services are responsible for the collection of B2C WEEE.	Producer-led, government-regulated. Public authorities responsible for collection of household WEEE. PRO Stiftung Elektro-Altgeräte Register
GR	Information not found.	Government-led, producer funded.
HU	Information not found.	Information not found.
IE	100%	Voluntary agreement between Irish Industry and the Government of Ireland, in response to the EU WEEE Directive. PRO WEEE Ireland ERP Ireland
IT	Information not found.	Producer-led. PRO ReMedia Consortium Ecodom

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
LV	100% (covers collection and recycling).	Producer-led.
LT	Information not found.	Producer-led.
LU	Information not found.	Producer-led. PRO Ecotrel
MT	Information not found.	Information not found.
NL	Information not found.	Producer-led
PL	100%	Government led, producer funded.
PT	Information not found.	Producer-led. PRO AMB3E ERP Portugal
RO	Information not found.	Producer-led. PRO Ecotic RoRec
SE	100% (covers collection of WEEE from municipal collection points and their treatment/processing).	Producer-led, public authorities responsible for collection of household WEEE. PRO El-Kretsen
SK		Producer-led, public authorities responsible for collection of WEEE. PRO SEWA (Slovak Electronic Waste Agency) ENVIDOM
SI	Information not found.	Producer-led. PRO ZEOS Slopa Interseroh
ES	Information not found.	Information not found.

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
UK	In principle producers cover costs from the point of collection for B2C; some contribution is made to local authorities for collection costs (but does not cover their costs in full).	PRO Valpak Appointed by government to lead the recovery/recycling effort. Other PRO REPIC

In order to identify some of the features of successful WEEE producer responsibility schemes, and due to the lack of comparable data for the EU-27, a small number of case studies have been developed for those MS demonstrating some of the best performance levels with regards to WEEE recycling and recovery (Germany, Denmark and Sweden). These are presented below.

3.4.3.1 WEEE PRODUCER RESPONSIBILITY IN GERMANY

Main features

The principal German policy concerning waste electrical and electronic equipment is the ElektroG, Electrical and Electronic Equipment (EEE) Act of 2005,[39] which transposed the WEEE Directive into German national law.[40] This act obliges producers of EEE to establish a take-back system for waste equipment either for reuse, recycling or disposal according to environmental standards. Thus, it aims to reduce WEEE generation and also the release of hazardous substances into the environment.[41] The ElektroG distinguishes legal obligations depending on whether WEEE originates from the commercial sector[42] (B2B) or from private households (B2C).

B2B producers of EEE placed on the market after 13 August 2005 are obliged to dispose of these products, while commercial end-users are responsible for the disposal of equipment placed on the market before this date. B2B WEEE must not be disposed of at public collection sites made available for household WEEE and producers who are exempt from collecting B2B WEEE.[2] While the municipal collection services are responsible for the collection of B2C WEEE, producers must provide adequate containers for collection at designated collection points and ensure timely emptying of these containers.[43] As WEEE disposal with residual waste has been banned citizens are obliged to set aside WEEE for separate, free of charge collection.

The producers are responsible for ensuring that the WEEE they sell is collected from the municipal collection sites and properly treated. The contracts are negotiated with waste managers e.g. by the different sectors of the industry. WEEE producers must pay a fee to register with the 'Stiftung Elektro-Altgeräte Register' (stiftung EAR), an organisation that monitors the collection of WEEE and

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[39] http://www.bmu.de/files/pdfs/allgemein/application/pdf/elektrog_uk.pdf, accessed 07/09/2011

[40] See URL, http://www.koord.hs-mannheim.de/horizonte/h32_Schmidt.pdf, accessed 07/09/2011.

[41] Eunomia Research & Consulting *et al.* (2009): International Review of Waste Management Policy: Annexes to Main Report. Bristol, UK. See URL http://www.environ.ie/en/Publications/Environment/Waste/WasteManagement/FileDownLoad,21598,en.pdf, accessed 09/09/2011.

[42] To be commercial WEEE, the equipment requires to be exclusively used by businesses and be characteristic for business use, e.g. as regards the equipment's place of use or the qualification of the personnel using it. L.c.

[43] Eunomia Research & Consulting *et al.* (2009): International Review of Waste Management Policy: Annexes to Main Report. Bristol, UK. See URL http://www.environ.ie/en/Publications/Environment/Waste/WasteManagement/FileDownLoad,21598,en.pdf, accessed 09/09/2011.

coordinates the siting of WEEE collection points. EAR also determines the financial obligations of individual producers according to market share.

Impacts

Monitoring data indicate that since the introduction of the ElektroG the amount of WEEE landfilled has decreased. While ElektroG mandates the collection targets to the rate of 4kg of WEEE/per capita/per year, data from the Federal Environment Agency (2006) reveals that the amount of WEEE actually collected amounted to 8kgper capita.[44] Increased rates of WEEE being returned to producers has prompted more consideration of the environmental impact of products during their design phase, as producers are now required to finance the costs of recycling. This has prompted many producers to reduce or substitute the use of hazardous substances contained within their products.

Strengths / factors for success / weaknesses

Despite the increase in WEEE collected, and in the number of producers registered with EAR, doubt has been cast on the achievements and the overall aims of the ElektroG to prevent WEEE and to promote reuse, recycling and recovery. Most importantly, a large fraction of WEEE returned to collection points is damaged during the process of collection, handling or transport thus severely impacting its reusability.[45]

Furthermore, since WEEE reuse is prioritised over its recycling and disposal, exports of WEEE to other countries, Africa in particular, has ensued. From an environmental perspective this is a serious issue, as, on the one hand, the equipment available for recycling (and thus increasing resource efficiency) is reduced, and, on the other hand, though increasing the availability of affordable equipment benefits social well-being in the receiving countries (e.g. employment creation in equipment trade and repair as well as IT-based education),the standards for further handling of WEEE are often poor, resulting in environmental and health impacts.[46,]

3.4.3.2 WEEE Producer Responsibility in Sweden[47]

[44] See URL http://umweltbundesamt.de/uba-info-presse/2008/pd08-019.htm, accessed 09/09/2011.

[45] SRU (2008): Umweltgutachten 2008. Berlin. See URL http://www.umweltrat.de/SharedDocs/Downloads/DE/01_Umweltgutachten/2008_Umweltgutachten_BTD.pdf?__blob=publicationFile, accessed 09/09/2011.

[46] SRU (2008): Umweltgutachten 2008. Berlin. See URL http://www.umweltrat.de/SharedDocs/Downloads/DE/01_Umweltgutachten/2008_Umweltgutachten_BTD.pdf?__blob=publicationFile, accessed 09/09/2011.

[47] Global perspectives on e-waste (Rolf Widmeran, Heidi Oswald-Krapf, Deepali Sinha-Khetriwalb, Max Schnellmann, Heinz Böni); Producer responsibility for WEEE as a driver of

Even prior to the entry into force of the WEEE Directive 2002/96/EC, Sweden had long since been considered a leader in the management of WEEE. National producer responsibility legislation was passed in 2000, and a national compliance scheme has been in place for the past seven years. The structure of WEEE producer responsibility in Sweden today retains the emphasis on producer responsibility originally contained in national law, while the adoption of the 'Extended Producer Responsibility' (EPR) paradigm into Swedish e-waste management schemes formalises producer responsibility through the post-consumer stage of a product's life. EPR, which is closely aligned with the Polluter-pays Principle, in essence finances WEEE treatment and final disposal by incorporating the costs of these services into a product's retail price.

In Sweden a single national producer responsibility organisation (PRO), El-Kretsen, exists to fulfil the WEEE collection and treatment obligation of producers. El-Kresten is owned jointly by 20 business associations and has agreements with 1,300 affiliated companies for electrical/electronic products recycling, and with 700 for battery recycling. Membership dues for affiliated companies are comprised of a fixed entry fee, an annual membership fee, and a fee based on product type and the subscribing company's market share. Through these dues, El-Kretsen facilitates the collection (from El-Kretsen collection points) and treatment of household and commercial WEEE. EEE producers who do not wish to participate in the El-Kretsen compliance scheme have the option of facilitating the collection and recycling of WEEE themselves. Producers are also required to fulfil old-for-new equipment responsibilities, in that when supplying a new product to consumers, they must take back a similar product for disposal where they are asked to do so. Responsibility for historical WEEE, or WEEE placed on the market before 13 August 2005, is a collective responsibility, with management costs determined by a producer's market share of household products on the Swedish market.

Impacts

The El-Kretsen compliance scheme has been remarkably successful in its activities and has achieved the highest WEEE collection rates in the EU at 15.8kg per capita/per year. In Sweden municipalities bear responsibility for the household collection of WEEE and the delivery of it to El-Kretsen collection points. From here, physical and financial responsibility for transport to recycling/treatment centres and final processing of WEEE is transferred to El-Kretsen. Since operations began in 2001 El-Kresten has recycled 1 million tonnes of WEEE (as of 2010) and has witnessed a steady increase in WEEE collection from 5.7kg per person in 2001 to 15.8kg per person in 2006.

Strengths/factors for success/weaknesses

The existence of producer responsibility legislation and a national compliance scheme before the introduction of the WEEE Directive undoubtedly made its transposition and the meeting of objectives much easier than in MS with no prior experience with e-waste management. In terms of the success realised by El-Kresten, they have espoused high levels of cooperation between partners, municipalities and contractors, along with the creation of environmental awareness

ecodesign: case studies of business responses to producer responsibility charges (Annika Gottberg); and Electronic waste management (Ronald E. Hester, Roy M. Harrison)

among the public as paramount to achieving high WEEE collection and recycling rates. The effectiveness of public collection points – one place where everything can be deposited – has been highlighted by El-Kretsen as being particularly effective.

3.4.3.3 WEEE PRODUCER RESPONSIBILITY IN DENMARK[48]

Prior to the introduction of Directive 2002/96 EC on waste electrical and electronic equipment (WEEE), Denmark had already established national legislation mandating the collection and recycling of WEEE. This legislation placed responsibility for the recovery and treatment of WEEE with local authorities, the costs of which were covered through taxation and/or collection fees. Transposition of the WEEE Directive into Danish law has not resulted in diminished government involvement in meeting WEEE targets (75% of equipment falling under Annex 1B categories 1 and 10, and 65% of equipment falling under the categories 3 and 4). Within the WEEE administrative framework establish by the Danish Environmental Protection Agency, local authorities remain responsible for the collection and sorting of WEEE from private households, while responsibility for supplying collection equipment and treating WEEE has shifted to producers. The framework also stipulates that all producers of EEE and portable batteries must register with and submit annual reports to Danish Producer Responsibility (DPA), a non-profit organisation which monitors producer compliance, among other tasks, and is funded through annual fixed fees charged to producers.

Producer responsibility is determined annually on an individual basis via DPA-system calculations, which allocates responsibility according to producer market share of household WEEE and/or waste portable batteries, and delineates the geographical area from which WEEE and/or waste portable batteries much be collected. As opposed to fulfilling DPA obligations individually, producers may subscribe to a compliance scheme which assumes registration, reporting, and WEEE collection/treatment activities on behalf of its members. El-Retur is the largest compliance scheme operating in Denmark, and consequently collects the largest amount of WEEE. While it operates on a non-profit basis, a fixed annual membership fee of DKK 3,000 is levied upon members. Additional to this is a variable environmental fee determined by quantities of WEEE placed on the market by an individual member. This fee covers the costs of collection and treatment of WEEE, with rates highly dependent upon the raw materials market.

Impacts

DPA statistics reveal that rates of EEE being put on the Danish market appear to be trending downwards, while WEEE collection is on the increase. The electrical and electronic equipment included in the DPA survey are: large household appliances; small household appliances; IT and

[48] Electronic waste management (Ronald E. Hester, Roy M. Harrison); Stemming the Tide of Patchwork Policies – The Case of E-Waste; The Producer Responsibility Principle of the WEEE Directive; Distribution of responsibilities between DPA-System, Producers and Collective Schemes

telecommunications equipment; consumer equipment; lighting equipment-luminaries; lighting equipment – light sources, electrical and electronic tools; toys, leisure and sports equipment; medical devices, monitoring and control instruments, and automatic dispensers.

Strengths/factors for success/weaknesses

A core weakness of the Danish transposition of the WEEE Directive is the virtual removal of individual producer responsibility for EEE placed on the market after 13 August 2005 (i.e. 'new' WEEE). Instead of requiring individual producer responsibility for future products, the Danish WEEE system holds producers jointly responsible for these products. Allocation of financial responsibility for new WEEE is determined by a current market share when costs are incurred, as in the historical WEEE financing mechanism.

3.4.3.4 WEEE MANAGEMENT PERFORMANCES

Table 12 below attempts to compare the cost of contributions to WEEE producer responsibility schemes and the WEEE recovery and recycling performance of the MS for which data has been found. The highest recovery and recycling rates and the lowest cost are indicated in dark green; the lowest recovery and recycling rates and the highest cost are indicated in dark red. This analysis is based on a very limited data set and therefore cannot be taken as an accurate overall representation of the cost and effectiveness of WEEE schemes.

Based on the only 3 MS for which comparable data were available, it does appear that where contributions for large appliances are made per item, there may be a relationship with the percentage of those appliances recovered and recycled (i.e. a higher cost of contribution is matched with higher recovery and recycling rates). However, for schemes where contributions for large appliances are based on a cost per kg, there does not appear to be any relationship between the cost of contributions and the rates of recovery and recycling.

For schemes for small appliances, there does not appear to be any relationship between the cost of contributions and the rates of recovery and recycling, whether the contributions are per item or per kg.

Table 12 Cost of contributions to WEEE schemes compared with recovery and recycling performance

	Large appliances (€/item)		
Member State	% recycled	% recovered	Cost ranking
CZ	93	93	1
LU	87	89	2
AT	84	89	3
	Large appliances (€/kg)		
	% recycled	% recovered	Cost ranking
EE	91	91	4
SK	87	87	1
LV	83	83	3
GR	81	81	2
	Small appliances (€/item)		
	% recycled	% recovered	Cost ranking
BE	70	74	2
CZ	82	89	1
LU	72	77	3
	Small appliances (€/kg)		
	% recycled	% recovered	Cost ranking
EE	91	91	2
AT	73	85	2
GR	81	81	4
LV	81	81	4
SK	74	78	1

In 2008, Sweden had the second highest rate of collection as a percentage of total WEEE placed on the market at around 63% (only the Netherlands is higher at around 72%), and also by far the highest rate of collection in kg per capital at 16kg (the next highest being the UK at 11kg and Finland at 10kg). Sweden also has the third highest recovery rate (in percentage of WEEE collected per capita) at 92% (behind Germany at 98% and Spain at 94%), and the fourth highest reuse/recycling rate (in percentage of collected per capita) at 84% (behind Denmark at 90% and both Portugal and Slovakia at 85%).

These performances already put Sweden in advance of the majority of the 2020 targets set in the recast WEEE Directive, and suggest that the Swedish WEEE producer responsibility scheme El-Kretsen may be able to provide some insight into the successful design of a WEEE producer responsibility scheme.

El-Kretsen claims that it has low running costs (although no data on the costs paid by producers has been found during the current study). According to El-Kretsen, the key factors contributing to its success include: a willingness of producers to accept responsibility, good collaboration and a clear division of responsibility with local authorities (who collect household WEEE and deliver it to El-Kretsen collection points, where responsibility passes to El-Kretsen), awareness-raising amongst the public, and the high convenience of available public collection points.

3.4.4 ELV

Reliable information on the nature of producer responsibility schemes for **ELV** has been found in 24 MS (including Luxembourg which participates in the Belgian scheme). Where producer responsibility schemes exist, they all specify that ELV must be taken back at no cost to the final owner of the vehicle (in several cases unless the vehicle has been modified or is no longer intact). Several MS (Belgium, Bulgaria, Denmark, Greece, Latvia, Lithuania, Netherlands, Portugal and the UK) have organisations or 'eco-organisms' that coordinate the take back and recovery of ELV on behalf of producers. Only limited information has been found on financial contributions to producer responsibility schemes for ELV. These are not directly comparable, therefore they are presented in Table 13 below rather than in graphical form.

Table 13 Identified financial contributions to ELV producer responsibility schemes

Member State	Detail of financial contribution	Paid by
CZ	EURO III or higher standard: no fee EURO II standard: €122/vehicle EURO I standard: €205/vehicle	The person who registers the used M1 or N1 vehicle
DK	From January 2012: €21.54 per tonne for landfilled shredder residue; from January 2015: €63.70 per tonne €12.10 annual environmental fee	Landfiller of residue Car owner
FI	One-off joining fee per producer (typically less than €1,000) and less than €5 per car sold	Producer/importer
NL	€15 per new car registration	Producer/importer, but ultimately passed on to the consumer
PT	<500 vehicles: €250 500 to 9,999: €500 From 10,000 to 20,000: €1,000 > 20,000: €1,500	Producer/importer
SK	€66.67/kg contribution to recycling fund	Producer/importer
SI	€40 per tonne of new vehicle	

Table 14 below summarises the information found on the percentage of costs covered by producers, and whether the schemes are public- or private-led.

Table 14 Percentage of costs covered by producers and public/private nature of ELV producer responsibility schemes

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
AT	Information not found.	Information not found.
BE	Information not found. Federations finance the monitoring body. Cost of collection, recycling and recovery are covered by the value of parts and materials.	Producer-led. PRO Febelauto
BG	Information not found	Information not found PRO Avtoecobul Ecobulcar
CY	Information not found	Information not found
CZ	Information not found	Government-led, producer funded
DK	Information not found.	Government-led. PRO DPA-System (Danish Manufacturer Liability System)
EE	100% (covers collection and treatment of ELV).	Information not found.
FI	100 % (covers collection, transport and treatment of ELV).	Producer-led, regulated by government.
FR	Information not found.	Producer-led.
DE	100% (covers collection and treatment).	Producer-led.
GR	Information not found.	Government-led, producer funded.
HU	Information not found.	Information not found.
IE	Information not found.	Information not found.
IT	Information not found.	Producer-led.
LV	Information not found.	Producer-led.
LT	Information not found.	Producer-led.
LU	Information not found.	Producer-led.

Member State	% of costs covered by producers	Public or private (i.e. producer) led / relationship between public and private organisations
MT	Information not found.	Information not found.
NL	Information not found. Cost spilt between producers, importers and consumers.	Producer-led PRO Auto Recycling Nederland (ARN)
PL	Information not found.	Information not found.
PT	Information not found.	Information not found. PRO Valorcar
RO	Information not found.	Information not found.
SE	Information not found.	Information not found.
SK	Information not found.	Information not found.
SI	Information not found.	Information not found.
ES	Information not found.	Information not found.
UK	Information not found.	Producer-led. PRO Autogreen Cartakeback

In order to identify some of the features of successful ELV producer responsibility schemes, and due to the lack of comparable data for the EU-27, a small number of case studies have been developed for the MS demonstrating some of the best performances with regards to ELV recycling and recovery (Austria, Belgium and Denmark). These are presented below.

3.4.4.1 ELV Producer Responsibility in Austria

Main features

The Austrian Ordinance on End-of-life Vehicles (ELV) stipulates that producers and importers of cars must take back ELVs from the brand they market, with the average distance to a take back point not being greater than the average distance to a selling point. As such, most selling points are also take-back points.

Returning ELVs must be free of charge for the last holder or owner. However, the holder or owner may be charged when waste has been added to the vehicle. In addition, the last holder or owner

may be charged when parts of the vehicle which determine the value of the ELV (as a source of recycling material) are missing.

Producers and importers of cars are responsible for the treatment of ELVs. They are obliged to ensure that reusable parts are re-used and that recyclable materials are recycled, so that 85% (or 95% as of 2015) of the ELV is re-used, recycled or recovered in another way. For the most part, the take back and treatment of ELVs is organised separately by each individual brand. However one organisation, ÖCAR Automobilrecycling GmbH, is registered as the responsible collection and treatment system for 15 different brands of ELVs.

Impacts

The free-of-charge take back obligation together with the commercial value of the car material has resulted in the collection and proper treatment of ELVs when they have no further commercial value. As can be seen in Table 15, around 96% of ELVs brought back to car producers and importers for waste treatment were recovered or reused in 2008 (the majority were recycled). Almost 84% of ELVs collected for treatment were recycled or reused.

Table 15 Treatment of ELV in Austria (Eurostat 2011b)

	Unit	Year		
		2006	2007	2008
ELV for treatment	Number of cars	87,277	62,042	63,975
	Tonnes	69,329	50,805	52,202
ELV recovered and reused	Tonnes	59,471	43,769	50,182
Share of recovered + reused	%	85.8	86.2	96.1
ELV recycled	Tonnes	52,628	37,932	41,255
ELV reused	Tonnes	2,722	2,430	2,413
Share of recycled + reused	%	79.8	79.4	83.7

However, only 25% of deregistered old cars are treated as ELVs; 15% are exported officially as second hand cars while the destiny of the remaining 60% of cars is unknown. It can be assumed that almost all of the missing 60% is exported without documentation as second-hand-cars, mainly to Eastern Europe.

Strengths / factors for success / weaknesses

In terms of legal compliance, the take-back system works very well. However, the incentive to sell ELVs for use abroad seems to be greater than for collection and treatment within the country, so that a large proportion of ELVs are handled outside the formal system.

In consideration of the waste hierarchy it is beneficial for the environment that cars stay in use as long as they are safe, meet the EU emission limits and are treated according to EU standards after their use. There are considerations to introduce control mechanisms for assuring that used-cars may be only exported if the environmental standards in the receiving country comply with EU regulations.

3.4.4.2 ELV PRODUCER RESPONSIBILITY IN BELGIUM

Main features

As waste management in Belgium falls under regional jurisdiction, it was by regional environmental convention in 1999 that the governments of the three Belgian regions, together with the concerned industries (e.g. automotive federations, dismantlers, shredders, etc.), first adopted regulations on end-of-life vehicles (ELV), which have been subsequently updated to transpose requirements contained within the ELV Directive into national law. Specific goals concerning ELVs are an 85% reuse, recycling or recovery rate of the weight of life vehicles by 2006, and 95% by 2015.

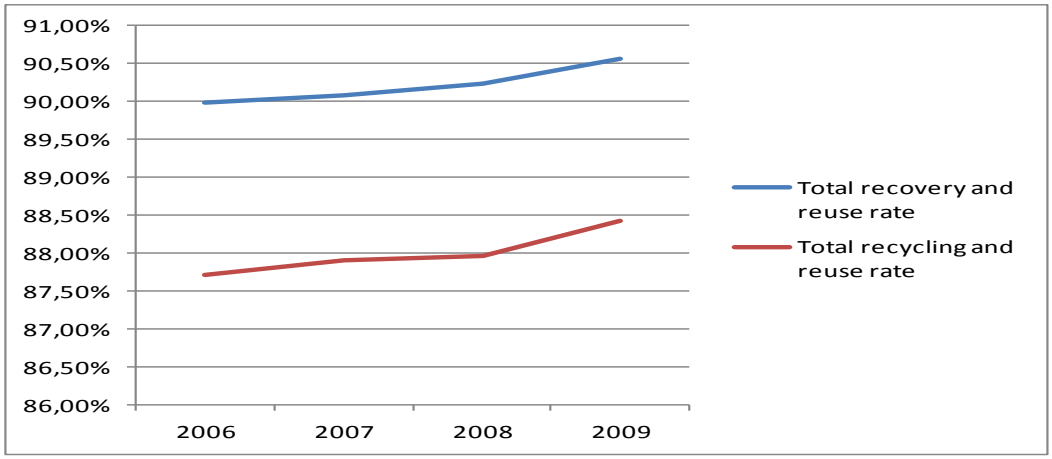
In Belgium, automobile retailers are obliged to take back one ELV for each new vehicle sold, and to provide a certificate documenting this transaction to the final owner of the ELV. ELVs are typically taken back free-of-charge to the owner, provided that specific preconditions are met (e.g. the ELV is still operative). Manufacturers and importers of automobiles are obliged to establish an adequate number of collection points throughout the regional territory, and for 90% of the Belgian population, there is at least one collection point within a maximum distance of 30km.

Febelauto, an ELV producer responsibility organisation, was founded in 1999 by professional federations representing the automotive industry, dismantlers and shredders and subcontractors for the purposes of facilitating and monitoring the collection and treatment of ELVs, with the overall aim of meeting the requirements of EC 2000/53. The monitoring of the registration and the dismantling of end-of-life-vehicles is conducted via Febulauto's EMS-system (end-of-life vehicle monitoring system). ELV-responsibility tasks undertaken by Febelauto on behalf of industry also include activities in the areas of: coordination (creation of annual management plans; monitoring (creation of a monitoring system to verify that industrial sectors are meeting objectives contained in management plans); certification (creation of reporting procedures for participating bodies); awareness-raising (communication with members on treatment problems relating to ELVs, information campaigns on take-back obligations); prevention (reduction of the use of hazardous substances and products in order to increase the recyclability of vehicles); social (the overall management of ELVs).

Impacts

According to a report[49] conducted in 2008 by the Public Waste Agency of Flanders (Openbare Afvalstoffenmaatschappij voor het Vlaams Gewest, OVAM), levels of recycling and energetic valorisation in Belgium ranged between 78% and 93% in 2006. Furthermore, overall ELV recycling rates reached 87% in 2006, with total recovery nearing 90%. Figure 43 below shows the total recovery and recycling/reuse rates for ELV in Belgium for 2006-2008.

Figure 43 Total recovery and recycling/reuse rates on ELV in Belgium between 2006 and 2008 (Eurostat)



Strengths / factors for success / weaknesses

- Strengths

In Belgium, data collection and monitoring on ELVs is performed on a daily basis through a dynamic system. This allows almost real time estimates on what percentages can be reached. In addition, it enables a proactive approach to the choice of the different processing centres according to the type of ELVs, which can eventually lead to higher recycling rates. In addition a dynamic monitoring system may lead to commercial advantages, since real-time estimates of the recycling rates allows better management and planning.

Febelauto[50] believes that the fact the scheme is driven by the market with strong consensus between the operator, the members and the shredder operators ensures a harmonised operation of the scheme. The good relations between the different actors together with the good level of understanding of the market have made possible a minimum level of government intervention. When problems are encountered, new regulation can adjust the scheme and impose improvements. For example, the low performance on the collection of tyres made it necessary to integrate the collection of tyres in the ELV scheme in 2010. Another important factor that contributed towards the success of the scheme was Belgian R&D investment in post-shredder technologies, which have resulted in good levels of material reuse and recovery. The high cost of

49 OVAM (2008), Validation of the recycling percentages for end-of-life vehicles at shredder companies and flotation units, available at http://www.ovam.be/jahia/Jahia/cache/offonce/pid/176?actionReq=actionPubDetail&fileItem=1740 (assessed on 24/08/11)

50 Phone Interview performed on 30/08/2011

waste disposal has also acted as a motivating factor for industry to reduce the amount of waste by achieving better levels of reuse and recovery.

- Weaknesses

In general, OVAM reports weaknesses concerning the reliability of ELV recycling rates due to inefficiencies contained within the monitoring system, while Febelauto argues that consensus needs to be reached on the taxation of ELV shredder residues in Belgium and Europe. High levels of such taxes would be likely to act as a driver to the improvement of the level of ELV recycling and reuse, since the development of more efficient technologies would become more cost-effective. However, the idea that some countries would set up an administrative system to establish a tax on shredder residues where no other taxes are in place may be unrealistic.

Finally, there still seems to be a considerable rate of ELV shredded in non-certified facilities. This practice imposes unfair competition for the certified companies and might also lead to destructive environmental impacts as the process of vehicles is not monitored.

3.4.4.3 ELV PRODUCER RESPONSIBILITY IN GERMANY

Main features

The German End-of-Life Vehicle Act of 1998 (Altfahrzeuggesetz) and amendments in 2002 transposed the ELV Directive into national law[51], with the 2002 addenda requiring that producers establish a free take-back system obligatory for cars registered in the EU. The free take-back of an ELV can be denied if the car documents (German document or EU Document) have not been handed in. Car producers must take back all the vehicles of their brand in an authorised permitted collection facility or an authorised dismantling facility designated by the car producer. At the collection or dismantling facility, the owner is given a certificate of destruction. The German ordinance also requires the installation of a 'sufficiently comprehensive network of authorised collection facilities or authorised dismantling facilities'.

Impacts

For 2006 and 2015 the following recycling, recovery and reuse targets have been specified:

- 2006: at least 85% recovery and reuse, and 80% recycling and reuse; and
- 2015: at least 95% recovery and reuse, and 85% recycling and reuse.

[51] Information on the ELV ordinance have been taken from http://www.europarl.europa.eu/comparl/envi/pdf/externalexpertise/end_of_life_vehicles.pdf, accessed 07/09/2011 as well as from http://www.umweltbundesamt-daten-zur-umwelt.de/umweltdaten/public/theme.do?nodeIdent=2304, accessed 07/09/2011.

On the basis of data from the statistical federal office and calculations from the Umweltbundesamt the following recycling and reuse rates have been achieved for ELVs (in 2008)[52], meeting 2006 targets.

- 89.2% for recycling and reuse
- 92.9% for recovery and reuse

Strengths / factors for success / weaknesses

No practical problems with the cost-free take-back system have been cited. Dismantling facilities, however, have reported a reduction in the number of ELVs being returned for dismantling, making it difficult for them to operate at capacity. The assumption here is that ELVs are being illegally shipped abroad. The continued use of vehicles can be seen positively from a resource perspective, however, responsibility for recovery and recycling ends at the EU border. If a vehicle is exported to a non-EU country the responsibility no longer exists, enhancing the incentive for exportation over recovery and recycling. To deter illegal overseas shipments of ELVs, requirements for exported vehicles are under discussion regarding their roadworthiness and/or CO_2-emissions.[53]

Occasional enforcement problems have been reported by local authorities. For example, the submission of a certificate of destruction in line with the ELV ordinance as a precondition of car deregistration is not always respected.

A further objective of the regulation is the optimisation of product design and environmentally friendly recovery. However, it is not known whether these targets have been reached, as statistics on ELV do not include information concerning product design.

3.4.4.4 ELV WASTE MANAGEMENT PERFORMANCES

Austria (84% reuse and recycling, 96% reuse and recovery), Belgium (88% reuse and recycling, 90% reuse and recovery) and Germany (89% reuse and recycling, 93% reuse and recovery) are amongst the best-performing MS with regards to ELV waste management performance. These performances suggest that these MS may provide some insights into the successful design of an ELV producer responsibility scheme.

The Belgian Febelauto scheme is based on strong cooperation between the operator, the members and the shredder operators, ensuring the smooth functioning of the scheme with a minimum level of government intervention. The scheme also collects and monitors data on ELVs on a daily basis, allowing almost real time estimates on the reuse, recycling and recovery percentages that can be reached. This allows better management and planning, and for a

[52] UBA (2008): Altfahrzeugaufkommen und –verwertung. http://www.umweltbundesamt-daten-zur-umwelt.de/umweltdaten/public/theme.do?nodeIdent=2304#f1

[53] SRU (2008): Umweltgutachten 2008. Berlin. See URL http://www.umweltrat.de/SharedDocs/Downloads/DE/01_Umweltgutachten/2008_Umweltgutachten_BTD.pdf?__blob=publicationFile, accessed 09/09/2011.

proactive approach to be taken to the choice of processing centre, which can eventually lead to higher recycling rates. There has also been significant R&D investment in post-shredder technologies in Belgium, which has resulted in good levels of material reuse and recovery.

In both Austria and Germany, there appears to still be an incentive of some kind for vehicles to be exported for use abroad rather than domestic treatment (Austria is considering introducing control mechanisms to allow exports only if certain environmental standards are met in the destination country; Germany is considering measures to place roadworthiness and/or CO_2-emissions criteria on exported vehicles). In Belgium, there still seems to be a considerable proportion of ELV shredded in non-certified facilities. In both of these circumstances, it can make it difficult for ELV dismantling facilities to operate at capacity. In general, the problem with even some of the best performing schemes in terms of recycling and recovery is not the achievement of high rates, but ensuring the responsible management of all ELV (not just those being recycled and recovered through the systems established for the purpose).

However, based on the lack of comparable data collected within the study on producer responsibility schemes for ELV, it has not been possible to draw meaningful conclusions on the performance of MS in relation to collection, recovery and recycling/reuse of ELV and the impacts of producer responsibility schemes for this waste stream, or the cost-effectiveness of the existing schemes.

3.4.5 BATTERIES

With the entry into force of the amended Directive on Waste Batteries and Accumulators in 2006, the EU established maximum content thresholds for mercury and cadmium in batteries, and required Member States to promote and maximise the separate collection of spent batteries and accumulators, as well as to apply the best available techniques to the treatment and recycling of these products. The Directive legally obliges producers (manufacturers and importers) that place more than 1 tonne of product on the market to finance the costs of collection, treatment and recycling of their market share of spent batteries and accumulators, as well as the costs associated with informing citizens of these arrangements. The Directive also entitles final consumers to dispose of spent batteries at no cost at battery retailers or other collection points, including those operated by local authorities.

As with the collection treatment and recycling obligations for packaging waste and WEEE, privately owned, not-for-profit collection schemes have emerged to fulfil the compliance obligations of their members, who generally pay annual fees determined according to their market share of batteries and accumulators. Retailers that place more than 32kg of portable batteries on the market are required to assist in the collection and recovery of spent batteries, commonly through the hosting of battery take-back containers on their premises which are emptied, free-of-charge, by the collection scheme to which the battery producer has subscribed.

Reliable information on the nature of producer responsibility schemes for waste **batteries** was found in 24 MS. Comparable data on the amount charged to producers, however, proved extremely difficult to find within the constraints of the present study.

Essentially, the schemes charge fees to producers based on the amount of batteries placed on the market, either per kg, per battery or according to market share (in Lithuania the Law on environmental pollution taxes batteries at a rate of €0.70 per kg). However, the MS determine the cost based on the type of battery, and the classification varies from country to country, e.g. consumer/vehicle/industrial battery in Austria and Lithuania; the chemical content of the battery (e.g. lead-acid/nickel-cadmium/alkaline/zinc carbon/lithium/button/lithium ion) in Latvia and Portugal; and size or weight of battery in Cyprus and Slovakia.

Table 16 below summarises the information found on the percentage of costs covered by producers, and whether the schemes are public- or private-led.

Table 16 Percentage of costs covered by producers and public/private nature of batteries producer responsibility schemes

Member State	% of costs covered by producers	Public or private (i.e. producer led? Relationship between public and private organisations?
AT	100% (covers collection and treatment)	Producer-led. PRO UFH Elektro Recycling Austria GmbH (ERA) Erfassen und Verwerten von Altstoffen GmbH (EVA) European Recycling Platform (ERP)
BE	Information not found.	Government-led, funded by government, consumers and producers. PRO BEBAT
BG	Information not found.	Public authorities responsible for collection. Licenses for recycling and reuse of lead-acid batteries and accumulators have been issued to the private companies Monbat Montana, KCM Plovdiv, and LZC Kardzhali.
DE	Information not found.	Producer-led
DK	Information not found.	Government-led. PRO DPA-System (Danish Manufacturer Liability System)

Member State	% of costs covered by producers	Public or private (i.e. producer led? Relationship between public and private organisations?
EE	100% (covers collection and treatment of waste batteries and accumulators).	Information not found.
FR	Information not found.	Producer-led, authorized by public authorities.
CY	100% (covers collection, recycling and recovery).	Information not found. PRO Green Dot Cyprus
CZ	Information not found.	Government-led, producer funded.
FI	100%	Producer-led, regulated by government.
GR	100% (covers collection, transportation, recycling, awareness raising activities and shipment of batteries abroad for recycling).	Government-led, producer funded.
HU	Information not found.	Information not found.
IE	Producers are required to finance any net costs arising from – the collection, storage, treatment and recovery and/or disposal of waste batteries and/or accumulators, and any public information campaign on the collection, storage, treatment and recovery and/or disposal of portable batteries and/or portable.	Voluntary agreement between Irish Industry and the Government of Ireland, in response to the EU Batteries Directive 2006/66/EC. PRO WEEE Ireland ERP Ireland
IT	Information not found.	Producer-led.
LT	Information not found.	Producer-led.
LU	Information not found.	Established by government, led and funded by producers, importers, distributors and retailers. PRO Ecobatterien
LV	Information not found.	Producer-led.
MT	Information not found.	Information not found.

Member State	% of costs covered by producers	Public or private (i.e. producer led? Relationship between public and private organisations?
NL	Information not found.	Producer-led.
PL	Information not found.	Information not found.
PT	Information not found.	Producer-led. PRO Valorcar Amb3E ERP Portugal Ecopilhas GVB.
RO	Information not found.	Information not found.
SE	Information not found.	Information not found.
SK	Information not found.	Information not found.
SI	Information not found. Costs split between producers, importers, retailers and final consumers.	Information not found. PRO ZEOS Slopak Interseroh
ES	Information not found.	Information not found.
UK	Producers are obliged to fund the net costs arising from the collection, treatment and recycling of batteries.	Producer-led. PRO BatteryBack

It is not possible within the present study to draw meaningful comparisons between the fees paid to the schemes in the different MS based on the data collected. Batteries producer responsibility schemes would therefore be an area additional future research, should the Commission so wish.

3.4.5.1 TWO EXAMPLES: BATTERIES PRODUCER RESPONSIBILITY IN GERMANY AND THE UK

As Europe's largest battery collection scheme and the leading recycler of batteries in Germany, Stiftung GRS Batterien partners with nearly 2,400 producers in fulfilling their compliance

obligations under the Directive, collecting and treating both portable and industrial batteries. Stiftung GRS Batterien services 170,000 collection points, of which 140,000 are sited at retail locations, and in 2010 collected 14,507 tonnes of batteries, or approximately 177 grams per inhabitant. This figure represents an increase of 103 tonnes over the previous year. Having achieved a collection rate of 44% in 2010, the scheme is one percentage point away from meeting the 2016 national collection target.

BatteryBack, a collection scheme representing battery producers active within the UK market, operates a nearly identical compliance system for its members. With the goal of increasing the number of its collection points to 60,000 by the end of 2012, BatteryBack provides retailers and distributors of batteries with a variety of spent battery containers ('BatteryCans'), ranging from 5 litres to 120 litres. These containers are also provided free of charge to schools, colleges, libraries and other public buildings to facilitate easier battery collection from members of the public. Compliance costs for paying members are stated to be no more than £0.01 (€0.0119) per battery placed on the market. As of 2011, BatteryBack had achieved a recycling rate of 55% of the gross weight of batteries treated, falling well within EU targets.

3.4.5.2 BATTERIES WASTE MANAGEMENT PERFORMANCES

The dates for the main targets in the Batteries Directive have not yet arrived, and data therefore has not yet been gathered by Eurostat on collection and recycling rates. In addition to this, the study has not collected adequate data on producer responsibility schemes for batteries to enable any conclusions to be drawn on the performance of MS in relation to collection and recycling of batteries and the impacts of producer responsibility schemes for this waste stream.

3.4.6 PRODUCER RESPONSIBILITY SCHEMES DEALING WITH OTHER WASTE STREAMS

Although not the focus of the study, attempts were made to identify producer responsibility schemes dealing with other materials, to provide a somewhat more complete picture of the application of producer responsibility in the EU-27 (although it is worth noting that the information included here is almost certainly incomplete) and to provide a source of inspiration for the broader application of producer responsibility schemes in the MS. The additional materials subject to producer responsibility schemes range from paper/card (frequent) to photochemicals (rare). Of these 'other' materials covered, producer responsibility schemes were most commonly found for the recovery of tyres, paper/card, oils (including mineral, motor, edible and lubricating oils), and the collection and disposal of old and unused medicines.

Sixteen of the 27 MS currently have in place producer responsibility schemes covering one or more of the additional waste streams identified, as summarised in Table 17 below. Fifteen MS have recovery programmes in place for tyres, nine for paper/card, seven for oils, and three for medicines.

Table 17 Producer responsibility schemes for other materials in the EU-27

Member State	Tyres	Paper/Card	Oils	Medicines
AT	•	•	•	
BE	•	•	•	•
BG	•			
CY	•	•	•	
CZ				
DK	•	•		
EE	•			•
FI	•	•		
FR	•	•		•
DE			•	
GR				
HU	•			
IE	•			
IT	•			
LV	•	•	•	
LT				
LU				
MT				
NL	•	•		
PL				
PT				
RO				
SE				
SK	•	•		
SI	•		•	
ES			•	
UK				

3.4.6.1 EXAMPLES OF OTHER PRODUCER RESPONSIBILITY SCHEMES IN FRANCE AND BELGIUM

Several countries have gone further in terms of producer responsibility than the requirements of the various EU Directives. Examples of a limited number of schemes from France and Belgium are presented here.

France has a number of producer responsibility schemes in place for other waste streams based on existing and forthcoming national legislation. These include obligations for the collection and treatment of unused medicines, tyres, textiles, graphic paper, furniture, self-treatment medical waste with a piercing risk, and products of agricultural supply (a voluntary initiative). Two producer responsibility organisations (PROs), FRP and Aliapur, service the tyre industry, Adivalor the agricultural supply industry, while two PROs are currently being established to service the domestic furniture market and the professional market.

Tyres

As of July 2002, it has been illegal in France to dispose of tyres in landfills. Since 2004 producers placing tyres on the French market have been obliged to recover and treat their products at the end of their life cycle. Aliapur, a non-profit PRO founded by the tyre manufacturers Bridgestone, Continental, Dunlop Goodyear, Kléber, Michelin and Pirelli represents 70% of the French tyre market, servicing 240 producers, to include its founders, in the collection and processing and recycling of tyres. Aliapur fulfils the take-back obligations of its subscribing members by collecting tyres from over 44,000 owners and distributors throughout the country, sorting and preparing them for processing (shredding) and recovering material and energy through recycling. For the 2011-2013 period, Aliapur has contracted 35 collectors and 13 transformation sites to aid in fulfilling member compliance obligations. In 2010, Aliapur's members declared 268,274 tonnes of product. In 2010, 278,756 tonnes were collected. Recovered, or shredded, tyre is typically exported to other countries where it is used as material or a fuel source by various industries. Figures from 2010 reveal that of the 66,135 tonnes of shredded tyre – equivalent to 8.8 million passenger car tyres – exported by Aliapur, 34,884 tonnes went to Switzerland, Luxembourg and Morocco, where it served as a material input into the cement works industry, 20,363 tonnes to Germany, the Netherlands and Belgium for use in granulation, and 10,888 tonnes to Sweden, where it served as a source of urban heating.

Packaging Waste for Products of Agricultural Supply

The recovery of packaging waste for products of agricultural supply in France by producers is wholly voluntary, with no national legislation requiring producer responsibility in this field. ADIVALOR, a non-profit PRO, was founded in 2001 through an initiative of the Union of Industrial and Plant Protection (UIPP) and with the cooperation and input of distributors, trade unions, agricultural wholesalers, the National Federation of Farmers' Unions (FNSEA), and French Chambers of Agriculture, in order to fulfil the voluntary commitments made by these industries to ensure safe and responsible disposal of waste generated by products of agricultural supply (i.e. crop protection products). Currently, ADIVALOR represents 240 producers and partners with 1,000 distributors and retailers in the recovery of waste packaging and waste product. Waste packaging and products subject to voluntary collection by ADIVALOR are: used packaging (phytopharmaceutical products, fertilizers and soil enhancers, seeds and plants, dairy farm hygiene

products), unusable phytopharmaceutical products and used plastic films. Participating farmers must store products once they have reached the end of their life and return waste packaging to depositories on dates determined by distributors, while participating distributors and retailers are responsible for organising the collection and sorting of waste. Producers (manufacturers and importers) finance the collection and treatment of waste via an eco-tax, which also funds ADIVALOR's administrative and operational activities. Chambers of Agriculture are relied upon to coordinate collections throughout the country as well as relay information to farmers.

As there are no national targets to guide ADIVALOR's performance, the organisation has established the following objectives for itself (2010):

- Collect 70% of empty phytopharmaceutical products;
- Collect 50% of fertilizer, plant and seed packaging;
- Participate in the collection and optimisation of 30,000 tonnes of used agricultural film;
- Collect and eliminate new stocks of obsolete phytopharmaceutical products;
- Put in place a progressive self-financing system to collect and optimise used agricultural film;
- Examine recovery and recycling solutions for other agri-supply waste;
- Increase awareness of recycling issues and prevention amongst stakeholders; and
- Offer a nationwide collection and recovery solution for dairy farm hygiene packaging by 2013, with a view to collecting 75% of that waste by 2015.

To date, ADIVALOR has established 4,300 collection points, recovered 41,000 tonnes of plastic, 66% of empty phytopharmaceutical packaging, 15,000 tonnes of used plastic films, 80% of old or obsolete stocks, and recycled 35% of large volume bags collected.
Like France, Belgium has a number of producer responsibility schemes in place for other waste streams, based on regional environmental conventions. These include obligations for the collection and treatment of plastic bags, disposable plastic kitchenware, unused and expired medicines, waste oil, tyres and photochemicals.

Tyres

The Belgian PRO Recyctyre, was founded in 1998 by the manufacturers Michelin, Continental, Goodyear, Pirelli, Vredestine, Bridgestone, and a number of organisations represented by Federauto, to provide environmentally sound solutions to tyre disposal. Since 2005, regional environmental conventions have required tyre producers to take back all tyres placed on the Belgian market at the end of their life, with the exception of bicycle tyres. A similar scheme to France's Aliapur, regarding participants and how the system is financed, Recyctyre represents 350 producers and services 6,000 collection points.

Waste Oil

Legislation enacted in Wallonia (2002), Brussels (2003) and Flanders (2004) established producer take-back obligations for waste oil, with a collection target of 100% of all usable waste oil currently in place. Regeneration or recycling of waste oils has been prioritised through the imposition of minimum recycling targets for producers. In Flanders, at least 85% of recovered waste oil must be recycled and at most 15% processed for energy generation, while in Wallonia and Brussels at least 60% of recovered waste oil must be recycled and at most 40% processed for energy generation. Valorlub, a PRO founded by the Belgian Petrol Federation (BPF), Lubricants Association Belgium (LAB), the Belgian federation of distributors (Comeos), and Federauto, which represents the Belgian automotive sector, was established to aid subscribing members (importers and manufacturers (producers), retailers and distributors) in meeting their take back requirements through the collection and processing of waste oils. By law, producers must finance the collection and processing of waste oils, meet the 100% collection target for usable waste oil, achieve the stated recycling targets while keeping within the maximum allowances for energy generation from waste oil, and inform and sensitise consumers and collectors/processors of these arrangements and individual obligations. Producers, distributors and retailers must accept waste oil being returned by customers, with no purchase obligation in Flanders and Wallonia, and with a purchase obligation in Brussels.

Many oil products are subject to producer take-back obligations under current legislation on waste oil, including: disperse oil; machining oils, emulsions and solutions; hydraulic oils; engine, gear and lubricating oils; insulating and heat transmission oils; and other oils and fats.

3.4.7 SUMMARY AND CONCLUSIONS

Although performance targets are set for various waste streams as a result of EU legislation (notably the Packaging, WEEE, ELV and Batteries Directives), there is considerable variation among the MS with regards to how they execute the associated producer responsibility obligations. The design, success and cost-effectiveness of schemes appears to vary considerably.

Although efforts have been made during the current study to obtain data to make judgements on the success and cost-effectiveness of producer responsibility schemes, it has not always been possible to obtain adequate data to make meaningful comparisons possible. This implies a certain lack of transparency of some schemes, which could perhaps be an issue to be addressed in the future. Better availability of data, in particular on the costs of the schemes, would enable an improved analysis to be undertaken which could feed into more substantial conclusions on the factors for the success of producer responsibility schemes.

It is also important to take into account competition policy. The creation, design and functioning of producer responsibility schemes should aim to strike a balance between fair competition and the consistency of the schemes created, particularly with regards to the collection systems put in place by the schemes. Too many competing schemes may result in fragmented collection, which may in turn result in lower efficiency and cost-effectiveness of the schemes.

The following sections highlight some of the key conclusions for the various waste streams addressed by producer responsibility schemes.

3.4.7.1 GENERAL CONCLUSIONS ON PRODUCER RESPONSIBILITY SCHEMES

Feedback gathered from stakeholders during the study suggests that for producer responsibility schemes to function successfully, waste management legislation must first be enforced. In addition, producer responsibility schemes require quite sophisticated interaction between government and industry. High quality industry bodies/producer organisations are needed, and they must have sufficient resources to provide a service to, and influence, their members. Sectoral coordination amongst producers enables the costs of producer responsibility to be shared by producers, rather than redirecting fees to customers.

Stakeholders raised concerns over any future shift from product-based (e.g. packaging, WEEE) to material-based (e.g. plastics, metals) schemes (e.g. plastics and not packaging), as this would require separate systems to deal with separate materials, whereas in reality materials often end up mixed together in products.

Stakeholders felt that producer responsibility schemes do actively contribute to meeting EU recycling targets, and that they could therefore be an important tool for improving the recycling and recovery rates of under-performing waste streams.

Suggestions from stakeholders to optimise producer responsibility schemes include involving all relevant players and clarifying their respective roles, increasing the transparency of schemes and publicising them (e.g. through awareness-raising actions), exchanging information and best practices between schemes, support for the eco-design of products, improving EU level monitoring, and undertaking further research to observe the impacts that schemes have on markets and access to waste.

3.4.7.2 PACKAGING

The analysis in this study has focussed predominantly on producer fee schemes. Other approaches to increasing recycling and recovery of packaging have been used in the MS, including packaging taxes, deposit-refund schemes and tradable compliance credits.

The current study has not found any clear relationship between the costs charged under producer responsibility schemes and the packaging recovery and recycling performance of the MS. Some relatively 'cheap' schemes demonstrate high levels of recovery/recycling (e.g. Belgium and Luxembourg) and some 'expensive' schemes demonstrate low levels of recovery/recycling (e.g. Poland and Estonia). The conclusion can therefore be drawn that the cost effectiveness of the schemes varies greatly, and that there is significant room for improvement in terms of improving the cost effectiveness of many schemes.

The Belgian Fost Plus scheme for municipal packaging waste (as described in section 3.4.2.2) could perhaps be judged to be the most cost-effective (the best performing in terms of recovery and recycling rates, yet the 7th cheapest of 21 schemes investigated in terms of producer contributions,

and costing only around €6.50 per citizen in 2010), and it is therefore worth investigating further the lessons that can be learned from this scheme.

Based on the Fost Plus experience and other research undertaken during the study, several factors for the success of packaging waste producer responsibility schemes (both in terms of recovery and recycling rates and cost-effectiveness) have been identified.

The creation of one single accredited producer responsibility organisation with solidarity amongst producers appears to be beneficial. This is the case of the Fost Plus scheme, whose members account for around 95% of household packaging placed on the market in Belgium. Stakeholder feedback also suggested that the establishment of a single, producer-led, scheme has proven to be the most effective way to collect used household packaging. MS with more than one compliance scheme may face challenges such as free-riders and 'cherry-picking', where schemes concentrate only on the waste fractions which are the least expensive to collect. In such cases, legislation setting minimum requirements is essential to ensure a level playing field, the collection and actual recycling of all household packaging, and transparency on the recycling levels achieved. Such minimum requirements could specify the types of waste and the geographical area to be covered by the schemes. An independent clearing-house could provide the mechanism needed to monitor the performance of competing schemes and compliance by all obligated parties. This may help to address potential competition-related concerns.

Close cooperation between all the actors involved in the packaging/packaging waste chain (industry, retailers, public authorities and consumers) is also important to success. Clearly defined roles for those actors, including good communication with – and active participation of – citizens, can aid the smooth functioning of a scheme. Stakeholder feedback supported this view.

Although limited data was gathered on the proportion of costs of collection, sorting and recycling of waste packaging that are actually covered by producers' contributions, it may be the case that this may affect the efficiency and effectiveness of producer fee schemes. The three MS where there appears to be full cost coverage by producers (Austria, Belgium and Germany) are amongst the top 5 performing MS in terms of packaging recovery and recycling rates. Stakeholder feedback suggested that sharing the cost of waste management between public authorities, industry and consumers has proven to be the most appropriate approach to ensuring collection and recycling. In MS with well developed household waste collection and waste management infrastructure, producers can more easily take full responsibility for the cost of collection and recycling; in other MS, industry may only be able to take financial responsibility for the activities it actually controls, in order to keep the overall environment and economic costs to society as low as possible. This may be an area that is worth further investigation.

Simplifying the legal framework governing producer fee schemes, maintaining the transparency/traceability of allocated funds, and adopting an interregional approach could also enhance the effectiveness of schemes.

Schemes could usefully increase their focus on waste prevention activities; this would ensure that the schemes are contributing to the practical implementation of the waste hierarchy and prioritising the prevention of waste at source.

Narrowing of the 'packaging' definition has been cited as leading to better performance by producer responsibility organisations.

Establishing sectoral agreements can simplify the practical implementation of producer responsibility.

3.4.7.3 WEEE

Based on the very limited data set found during the current study, no clear relationship has been identified between the cost of producer contributions to WEEE producer responsibility schemes and the WEEE recovery and recycling performance of the MS. Based on the only 3 MS for which comparable data were available, it does appear that where contributions for large appliances are made per item, a higher cost of contribution corresponds to higher recovery and recycling rates. However, for schemes where contributions are based on a cost per kg, there does not appear to be any relationship between the cost of contributions and the rates of recovery and recycling.

The Swedish El-Kretsen scheme (described in section 3.4.3.2) has contributed greatly to Sweden's outstanding performance in terms of collection (16kg per capita and 63% of total WEEE placed on the market), recovery (92% of WEEE collected per capita) and reuse/recycling (84% of WEEE collected per capita) of WEEE. These performances already put Sweden in advance of the majority of the 2020 targets set in the recast WEEE Directive.

Based on the El-Kretsen experience and other research undertaken during the study, several factors for the success of WEEE waste producer responsibility schemes have been identified.

The ready availability of public collection points for WEEE is crucial to the success of producer responsibility schemes; it enables consumer WEEE to easily pass into the producer responsibility scheme, which in turn contributes to high collection and recovery rates.

As with packaging, close cooperation between all the actors involved in the WEEE chain (industry, public authorities and consumers) is also important to success. Clearly defined roles for those actors (in the case of El-Kretsen, local authorities are responsible for delivering WEEE to El-Kretsen collection points, then responsibility passes to El-Kretsen), including good communication with – and active participation of – citizens, can aid the smooth functioning of a scheme. Stakeholder feedback supported this view.

One limitation to the success (in terms of the amount of WEEE recycled and reused) of WEEE PR schemes is that a large fraction of the WEEE returned to collection points is damaged during the process of collection, handling or transport; this severely impacts its reusability.

Since WEEE reuse tends to be prioritised over recycling and disposal, the (legal and illegal) export of WEEE to other countries (in particular African countries) has become increasingly widespread; this phenomenon poses its own environmental issues.

Stakeholder feedback suggested that WEEE targets should not be weight-based (this will indeed be addressed by the forthcoming revision of the WEEE Directive). Also, given the specific nature of WEEE (e.g. hazardous or environmentally harmful content), particular vigilance is required to avoid

the leakage (e.g. through illegal dumping or exports) of WEEE from producer responsibility schemes.

3.4.7.4 ELV

Based on the lack of comparable data collected within the study on producer responsibility schemes for ELV, it has not been possible to draw meaningful conclusions on the performance of MS in relation to collection, recovery and recycling/reuse of ELV and the impacts of producer responsibility schemes for this waste stream, or the cost-effectiveness of the existing schemes.

Some factors for success have been identified based on case studies conducted on the Austrian, Belgian and German ELV producer responsibility schemes.

Strong cooperation between the operator, the members and the shredder operators can ensure the smooth functioning of an ELV producer responsibility scheme.

Regular monitoring of data can allow accurate estimates to be made on the reuse, recycling and recovery percentages that can be attained, allowing better management and planning which can eventually lead to higher recycling rates.

In both Austria and Germany, there is a suspicion that there is still an incentive of some kind for vehicles to be exported for use abroad rather than domestic treatment (Austria is considering introducing control mechanisms to allow exports only if certain environmental standards are met in the destination country; Germany is considering measures to place roadworthiness and/or CO2-emissions criteria on exported vehicles). In Belgium, there still seems to be a considerable rate of ELV shredded in non-certified facilities. In both of these circumstances, it can make it difficult for ELV dismantling facilities to operate at capacity.

According to operators, the high cost of waste disposal has acted as a motivating factor to industry to reduce the amount of waste by achieving better levels of reuse and recovery.

A general consensus on the level of taxation on ELV shredder residues across the EU may be beneficial; high levels of such taxes would be likely to act drive an increase in ELV recycling and reuse, since the development of more efficient technologies would become more cost-effective.

3.4.7.5 BATTERIES

It has not been possible based on the data collected within the present study to draw meaningful comparisons between the fees paid to the batteries producer responsibility schemes in the MS. In addition, the dates for the main targets in the Batteries Directive have not yet arrived, and data therefore has not yet been gathered by Eurostat on collection and recycling rates. In addition to this, the study has not collected adequate data on producer responsibility schemes for batteries to enable any conclusions to be drawn on the performance of MS in relation to collection and recycling of batteries and the impacts of producer responsibility schemes for this waste stream.

3.4.7.6 OTHER WASTE STREAMS

Based on the limited scope of this study, only limited information has been collected on the presence of producer responsibility schemes in the MS for other waste streams. It is anticipated that this does not represent a full picture of producer responsibility schemes in all the MS.

The study did however find that a very wide range of additional waste streams are addressed by specific producer responsibility schemes. These additional waste streams include: tyres, oils (edible, motor and lubricant), paper/card, bulky waste, plastic foils, medical/infectious waste, compound packaging (Tetra-Pak), expanded polystyrene, plastic bags, disposable plastic kitchenware, unused/old medicines, photo-chemicals, textiles, shoes, fluorinated refrigerants, gas cylinders, chemical products, furniture, unused pesticides, pesticides and fertilizer packaging, seed and plant packaging, agricultural films and graveside candles.

The study found that the additional waste streams where producer responsibility schemes are present are: tyres (15 MS), paper/card (9 MS), oils (7 MS), and medicines (3 MS).

These findings may be used to provide some inspiration for the application of producer responsibility to additional waste streams in more MS.

In terms of potentially applying producer responsibility to additional waste streams, stakeholders felt that one of the key limiting factors is the capacity of a MS to adequately set up, manage and regulate such schemes, each of which requires the engagement of an economic sector/group of producers.

3.5 SUMMARY OF USE OF ECONOMIC INSTRUMENTS IN THE EU-27 MEMBER STATES

Table 18 below provides a synthetic overview of the use of EIs in the waste sector in the 27 MS.

Table 18 Overall (current) presence of economic instruments in the 27 EU Member States

Member State	Disposal			Producer responsibility				
	Landfill tax	Incineration tax	PAYT schemes (NB local level)	Packaging	WEEE	ELV	Batteries	Other
AT	•	•	•	•	•	•	•	Tyres; waste mineral and edible oils; paper; bulky metals, glass, plastics and wood; plastic foils; medical waste; compound packaging (Tetra-Pak); expanded polystyrene
BE	•	•	•	•	•	•	•	Paper/card; plastic bags; disposable plastic kitchenware; car batteries; unused/old medicines; oils; tyres; photo-chemicals
BG	•	No MSW incineration	?	•	•	•	?	Tyres; plastic bags; tax on products generating widespread waste
CY	Excise tax for C&D waste disposal	No MSW incineration	-	•	•	•	•	Paper/card; motor oils; tyres
CZ	•	C	•	•	•	•	•	Tyres, motor oils
DK	•	•	•	-	•	•	•	Paper/card; tyres

Member State	Disposal			Producer responsibility				
	Landfill tax	Incineration tax	PAYT schemes (NB local level)	Packaging	WEEE	ELV	Batteries	Other
EE	•	No MSW incineration	•	•	?	•	•	Tyres; old medicines; paper
FI	•	-	•	•	•	•	•	Paper; tyres
FR	•	•	•	•	•	•	•	Paper/card; tyres; unused medicines; textiles, shoes; mineral/ synthetic oils; fluorinated refrigerants; infectious waste from healthcare activities; gas cylinders; chemical products; furniture; office and IT supplies; pesticide packaging; unused pesticides; fertilizer packaging; agricultural films; seed and plant packaging
DE	-	-	•	•	•	•	•	Commercial waste; construction waste; waste oil
GR	-	No MSW incineration	P	•	•	•	•	Tyres, motor oils
HU	-	?	•	-	•	?	?	Tyres
IE	•	-	•	•	•	•	•	Tyres; farm plastics; plastic bags
IT	•	-	•	•	•	•	•	Tyres
LV	•	-	-	•	•	•	•	Paper/card; tyres; tax on tyres, lubricating oils and oil filters
LT	(planned for 2012)	-	?	•	•	•	•	Paper/card; tyres

Member State	Disposal			Producer responsibility				
	Landfill tax	Incineration tax	PAYT schemes (NB local level)	Packaging	WEEE	ELV	Batteries	Other
LU	-	-	•	•	•	Participates in BE scheme	•	n
MT	-	-	?	•	?	?	•	Eco-contribution Act
NL	•	• (but €0)	•	-	•	•	•	Paper/card; tyres
PL	•	?	?	•	•	•	•	?
PT	•	?	-	•	•	•	•	Tyres; motor oils
RO	-	No MSW incineration	-	•	•	?	?	?
SE	•	-	•	•	•	•	•	?
SK	•	?	•	•	•	•	•	Paper/card; tyres
SI	•	-	•	•	•	?	•	Waste from hazardous pesticides; graveside candles; medical waste; tyres; lubricating oils
ES	• (Catalonia, Madrid & Murcia)	• (Catalonia & Madrid)	•	•	•	•	•	Tyres; mineral oils
UK	•	-	•	•	•	•	•	?

• = EI in use ? = information not found - = EI not in use P = pilot/trial scheme(s) c = under consideration

4 MODELLING

4.1 BASELINE SCENARIO

4.1.1 SCOPE AND METHODOLOGY

The scope of the modelling exercise within the study was to investigate what might be the effect, in quantitative and qualitative terms, of implementing new (sets of) EIs in the waste sector. These effects have to be expressed in terms of the change relative to a baseline situation in which these instruments are not present. For this reason a theoretical baseline scenario 'do-nothing' scenario was defined, i.e. what would happen in the EU-27 and its individual MS if no supplementary policy measures are taken, and if the existing waste treatment options persist in future. The modelling of the baseline scenario intentionally excludes the assumption of compliance with the EU recycling and landfill diversion targets. This is necessary because new EIs will be needed to achieve compliance with these targets.

The baseline scenario models MSW generation driven by:

- Varying degrees of empirically observed decoupling of average MSW generation from household consumption;
- Demographic evolutions in the Member States; and
- Evolutions in the composition of MSW.

The waste treatment is modelled assuming a status quo in the distribution of MSW in terms of percentage over the operational waste treatment options in the MS.

The methodology is described in detail in the full modelling report annexed to this main report (Annex 3).

4.1.2 OUTCOME

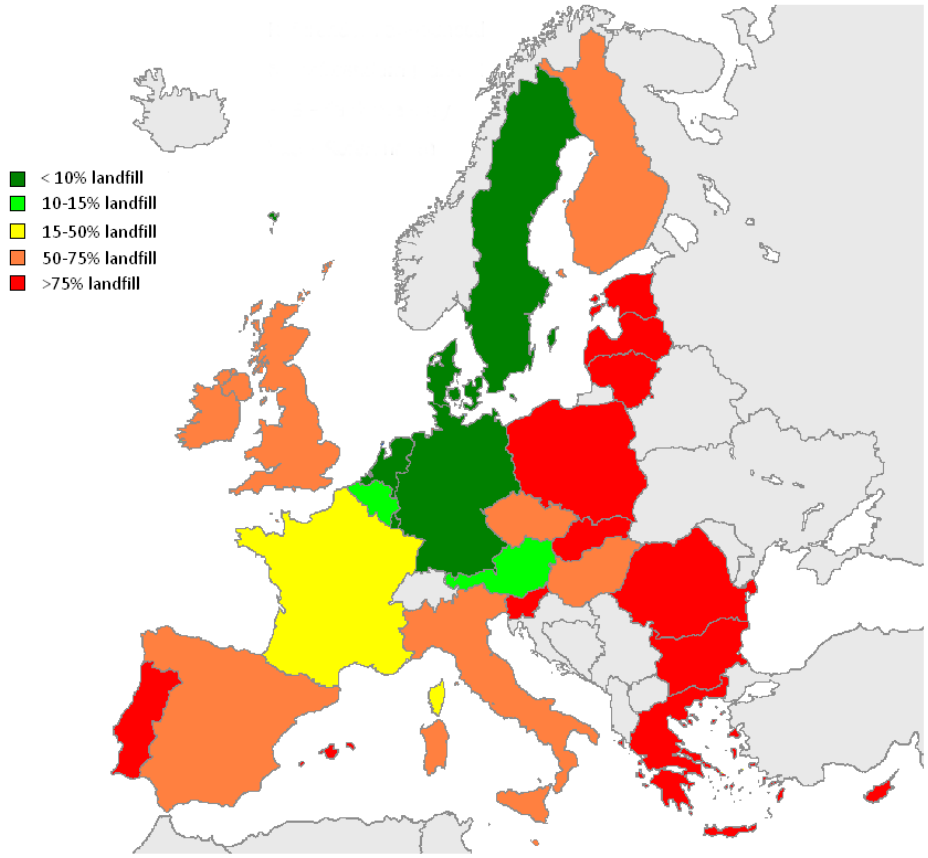
Average municipal waste generation per capita increases from 446 kg per capita to 532 kg per capita between 2008 and 2025. The driving force is the coupling or only relative decoupling of MSW generation from consumption in 20 MS. Only 7 MS show absolute decoupling, mainly based on their level and type of household consumption.

Population increases in the EU-27 from 495,809,146 inhabitants in 2008 to 521,883,935 in 2025. Combined with an increasing MSW generation per capita this gives an increase from 221 million tonnes in 2008 to 277.5 million tonnes in 2025. Biowaste remains the largest constituent element, and the impact of paper increases.

The distribution of MSW over the different waste treatment options for 2008 remains, in the scope of this baseline scenario, unaltered in 2025. It is acknowledged that this neglects the effect of

autonomous market evolutions (driven by prices, resource scarcity and energy scarcity) towards increasing material recovery or energy recovery. Of MSW generated and collected, 45.6% goes to landfill, 18.7% to incineration, 8.8% to paper cardboard recycling, 2.1% to plastic recycling, 3.8% to glass recycling, 1.1% to metals recycling, 13% to other recycling options, 6.1% to composting, 0.5% to home composting, and 0.2% to anaerobic digestion.

Figure 44 Geographic distribution of average percentages of MSW being landfilled in the baseline scenario

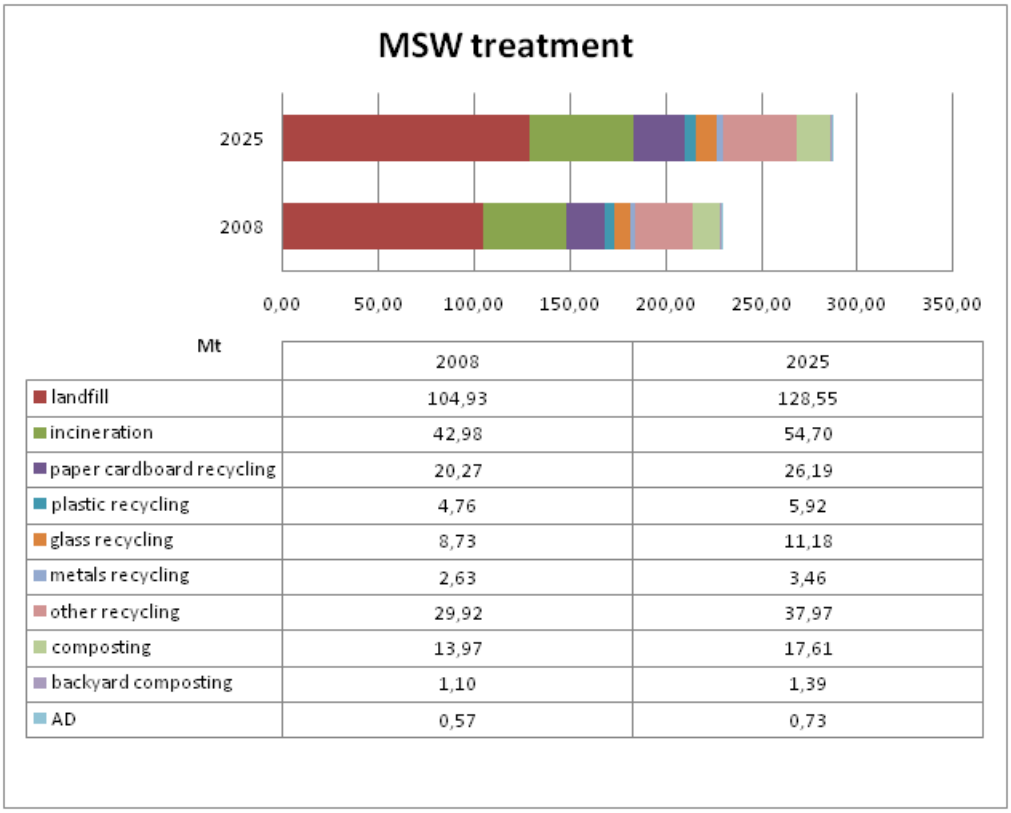


The figures include the total amount of MSW directly landfilled plus the landfilled MSW recycling residues.

In absolute figures the increase in necessary landfill capacity and incineration capacity is obvious. Recycling and composting also increases in this scenario in line with the increasing overall waste generation.

An overview of the increase of MSW production, and its split over the different treatment options is expressed in million tonnes (Mt) in Table 19. This table is composed as a sum of the baseline scenarios developed for each individual MS:

Table 19 Overview EU-27: MSW treatment in the Baseline Scenario, assuming increasing waste generation and no shifts in the MSW treatment options



Mt	2008	2025
landfill	104,93	128,55
incineration	42,98	54,70
paper cardboard recycling	20,27	26,19
plastic recycling	4,76	5,92
glass recycling	8,73	11,18
metals recycling	2,63	3,46
other recycling	29,92	37,97
composting	13,97	17,61
backyard composting	1,10	1,39
AD	0,57	0,73

Data for individual MS, both on generation and treatment, as well as EU-27 overviews, are included in Annex 3.

4.1.3 DEFINITION OF SCENARIOS IMPLEMENTING A MIX OF ECONOMIC INSTRUMENTS

Empirical research was undertaken on the relation between the rate of the landfill tax and the percentage of MSW being landfilled for time series in Austria, Denmark, Finland, France, Ireland, Latvia, Netherlands, Poland, Sweden, Slovakia and the UK. This approach is limited to directly available data. It would be useful to repeat the exercise using data from more MS, and including total landfill costs (i.e. tax plus gate fee), to check the consistency of the outcome of the preliminary exercise in this chapter.

Two scenarios are defined. The data show that:

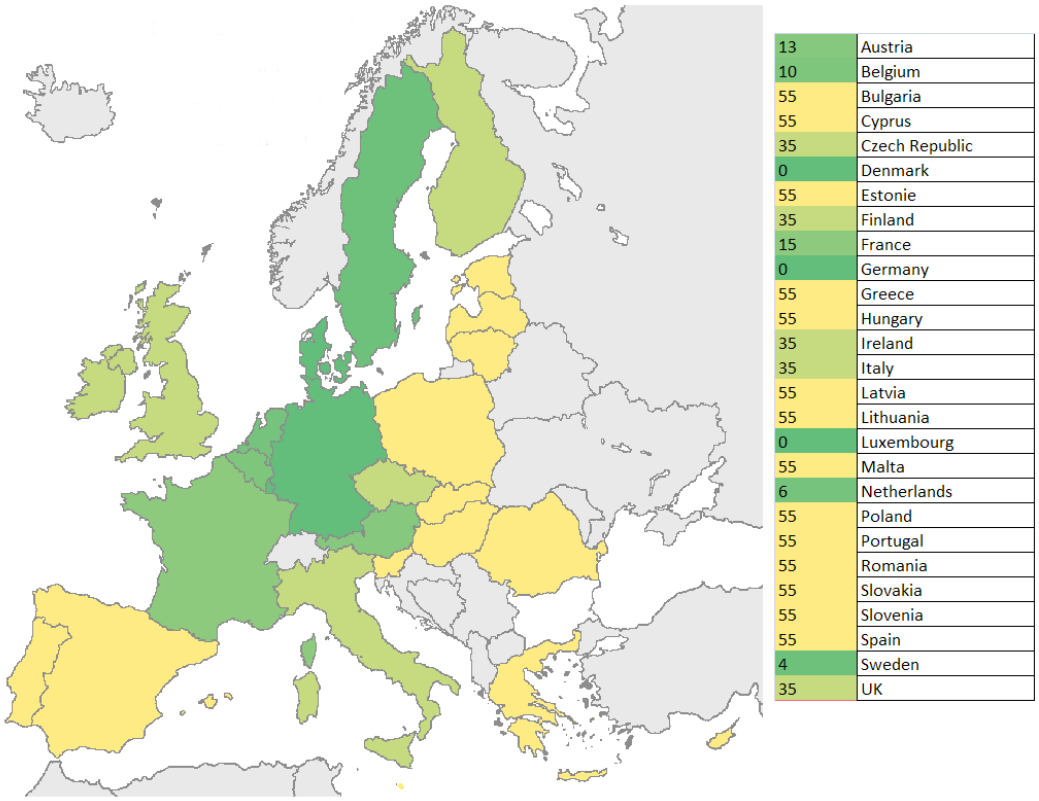
- MS depending largely on landfilling can reduce landfill to 55% of the generated MSW by imposing landfill taxes up to €40 per tonne; and
- MS depending largely on other waste treatment techniques can reduce landfill to 15% of the generated MSW by imposing landfill taxes up to €40 per tonne.

The scenarios were defined as a situation in which all MS reach a level of landfill tax of at least €40 per tonne, which leads to landfill diversion as follows:

High landfilling Member States: BG, CY, EE, GR, HU, LV, LT, MT, PL, PT, RO, SK, SI, ES	<55%
Low landfilling Member States: AT, BG, DK, FR, DE, LU, NL, SE	<15%
Intermediate Landfilling Member States: CZ, FI, IE, IT, UK	<35%

For MS with an intermediate position, a reduction to a landfill dependence of 35% is assumed.

Figure 45 Percentages of landfilling MSW, assuming that in 2025 a landfill tax of €40 per tonne or more would be imposed



13	Austria
10	Belgium
55	Bulgaria
55	Cyprus
35	Czech Republic
0	Denmark
55	Estonie
35	Finland
15	France
0	Germany
55	Greece
55	Hungary
35	Ireland
35	Italy
55	Latvia
55	Lithuania
0	Luxembourg
55	Malta
6	Netherlands
55	Poland
55	Portugal
55	Romania
55	Slovakia
55	Slovenia
55	Spain
4	Sweden
35	UK

Percentages of landfilling MSW, assuming that in 2025 a landfill tax of 40 €/t or more would be imposed

It is assumed that these percentages will be reached rather synchronous with reaching the level of €40 per tonne. In this analysis it is assumed that these targets will be reached in 2025, and that there will be a gradual increase in landfill taxes from the current level towards the level of €40 per tonne.

In **Scenario A** the distribution of waste derived from landfills will be distributed over the different alternatives (recycling, composting, incineration) in line with the current distribution.

Scenario B is a variation on scenario A. Next to the increased landfill tax, a mixture of EIs (of which PAYT and EPR may be the most important) is taken into account. These instruments favour source separated collection and recycling. Scenario B is developed in a way such that recycling and composting are the final destination of all wastes diverted from landfill. All bio-waste diverted would

be composted and all non-biowaste would be recycled. It is recognised that this is a maximalist scenario.

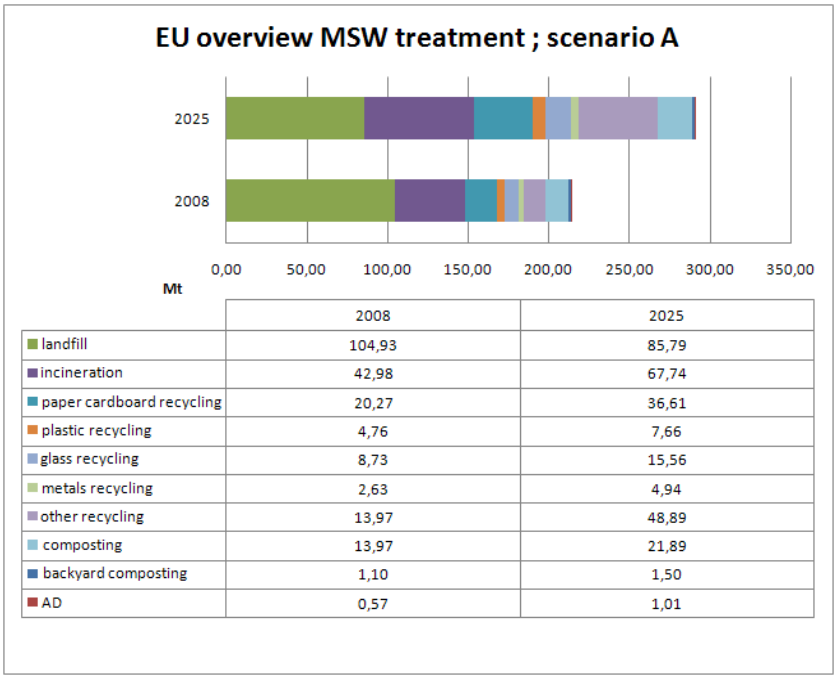
4.1.4 QUANTITATIVE OUTCOME OF SCENARIO A

The amount of MSW diverted from landfill in 2025 in EU-27 is 43 Mt compared to the baseline scenario or 19 Mt compared to the 2008 quantities.

Of the total of 291.5 Mt MSW generated and collected in 2025, 29.4% goes to landfill, 23.2% to incineration, 12.6% to paper cardboard recycling, 2.6% to plastic recycling, 5.3% to glass recycling, 1.7% to metals recycling, 16.8% to other (or non specified) recycling options, 7.5% to composting, 0.5% to home composting, and 0.3% to anaerobic digestion.

Data for individual MS, as well as more detailed EU-27 overviews, are included in Annex 3.

Table 20 Quantities of MSW treated in 2025, applying scenario A



	2008	2025
landfill	104,93	85,79
incineration	42,98	67,74
paper cardboard recycling	20,27	36,61
plastic recycling	4,76	7,66
glass recycling	8,73	15,56
metals recycling	2,63	4,94
other recycling	13,97	48,89
composting	13,97	21,89
backyard composting	1,10	1,50
AD	0,57	1,01

Table 21 Results of scenario A, compared to the start situation in 2008 and to the baseline situation in 2025

	EU : result of scenario A				
Mt	**2008**	**baseline 2025**	**scenario A 2025**	**Δ 2008**	**Δ baseline 2025**
landfill	104,93	128,55	85,79	-19,15	-42,76
incineration	42,98	54,70	67,74	24,76	13,04
paper cardboard recycling	20,27	26,19	36,61	16,34	10,42
plastic recycling	4,76	5,92	7,66	2,90	1,74
glass recycling	8,73	11,18	15,56	6,83	4,38
metals recycling	2,63	3,46	4,94	2,31	1,48
other recycling	13,97	37,97	48,89	34,92	10,93
composting	13,97	17,61	21,89	7,91	4,28
backyard composting	1,10	1,39	1,50	0,40	0,11
AD	0,57	0,73	1,01	0,44	0,27

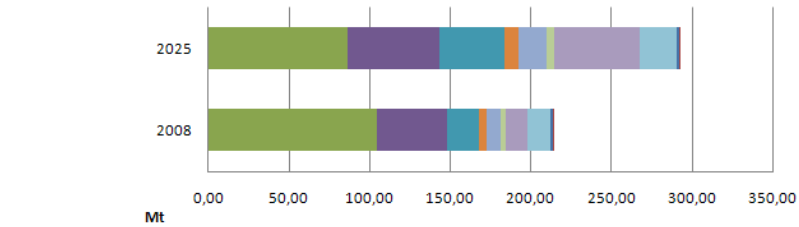
4.1.5 QUANTITATIVE OUTCOME OF SCENARIO B

The amount of MSW diverted from landfill in 2025 in EU-27 is 43 Mt compared to the baseline scenario or 19 Mt compared to the 2008 quantities, as in scenario A. The difference is to be found in the way in which the diverted waste will be treated, which is described below.

Of the total of 291.5 Mt MSW generated and collected in 2025, 29.5% goes to landfill, 19.5% to incineration, 13.7% to paper cardboard recycling, 2.8% to plastic recycling, 5.9% to glass recycling, 1.8% to metals recycling, 17.9% to other (or non specified) recycling options, 7.9% to composting, 0.5% to home composting, and 0.4% to anaerobic digestion.

Data for individual MS, as well as more detailed EU-27 overviews, are included in Annex 3.

Table 22 Quantities of MSW treated in 2025, applying scenario B



	2008	2025
landfill	104,93	86,47
incineration	42,98	57,21
paper cardboard recycling	20,27	40,25
plastic recycling	4,76	8,26
glass recycling	8,73	17,23
metals recycling	2,63	5,41
other recycling	13,97	52,35
composting	13,97	23,09
backyard composting	1,10	1,52
AD	0,57	1,15

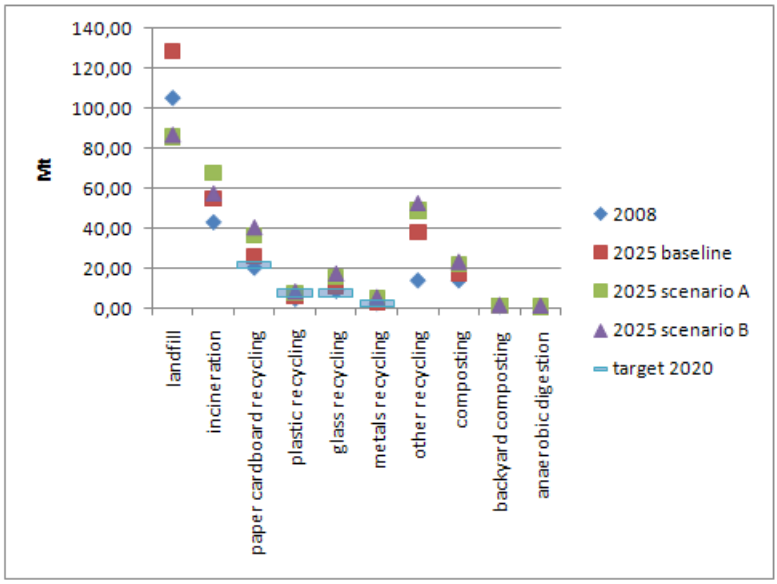
Table 23 Results of scenario B, compared to the start situation in 2008 and to the baseline situation in 2025

	EU : result of scenario B				
Mt	**2008**	**baseline 2025**	**scenario B 2025**	**Δ 2008**	**Δ baseline 2025**
landfill	104,93	128,55	86,47	-18,47	-42,08
incineration	42,98	54,70	57,21	14,24	2,51
paper cardboard recycling	20,27	26,19	40,25	19,98	14,06
plastic recycling	4,76	5,92	8,26	3,51	2,34
glass recycling	8,73	11,18	17,23	8,50	6,05
metals recycling	2,63	3,46	5,41	2,78	1,95
other recycling	13,97	37,97	52,35	38,38	14,38
composting	13,97	17,61	23,09	9,12	5,49
backyard composting	1,10	1,39	1,52	0,42	0,13
AD	0,57	0,73	1,15	0,58	0,42

4.1.6 OVERVIEW AND CHECK ON COMPLIANCE WITH EU RECYCLING TARGETS

For the purposes of the modelling exercise, the EU targets for recycling are assessed as follows: 50% of the glass, metals, plastics and paper/card waste generated as MSW in 2020 should be recycled in 2020. To assess this we have used the composition of MSW generation, in percentages, as predicted for 2025 (as a proxy for 2020). These percentages were applied to the assessed quantity of MSW generated in 2020 (258.83 Mt), and the target values were calculated as 50% of waste generation. These target values can be compared with the outcome of 2008, the baseline scenario and scenario A and B.

Figure 46 Projected MSW treatment (landfill, incineration and materials recycling) compared with EU targets



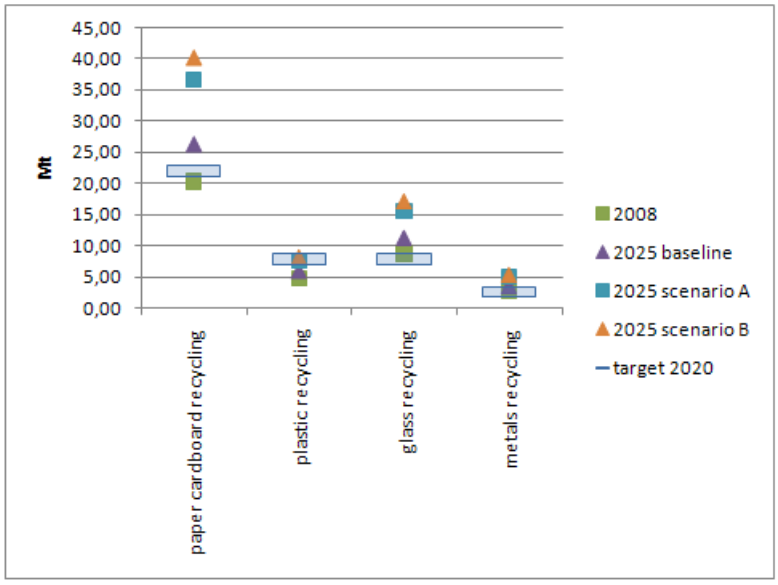


Table 24 Projected MSW treatment (landfill, incineration and materials recycling) compared with EU targets

Mt	2008	2025 baseline	2025 scenario A	2025 scenario B	target 2020
landfill	104,93	128,55	85,79	86,47	
incineration	42,98	54,70	67,74	57,21	
paper cardboard recycling	20,27	26,19	36,61	40,25	22,00
plastic recycling	4,76	5,92	7,66	8,26	7,76
glass recycling	8,73	11,18	15,56	17,23	7,76
metals recycling	2,63	3,46	4,94	5,41	2,59
other recycling	13,97	37,97	48,89	52,35	
composting	13,97	17,61	21,89	23,09	
backyard composting	1,10	1,39	1,50	1,52	
anaerobic digestion	0,57	0,73	1,01	1,15	

By 2025, Scenario B would reach values in excess of all 2020 recycling targets. Scenario A would only fail to meet the target for plastics recycling. However, as the recycling targets need to be reached by 2020 (not 2025), measures in addition to landfill taxes may be needed.

4.1.7 Environmental Impacts

For **scenario A** it was calculated that, as a consequence of introducing a landfill tax of €40 per tonne, and compared to the 'do nothing' baseline scenario:

- Landfill of MSW will decrease by 42.76 million tonnes;
- Incineration of MSW will increase by 13.04 million tonnes;
- Recycling will increase by: 10.42 million tonnes for paper, 1.74 million tonnes for plastics, 4.38 million tonnes for glass, and 1.48 million tonnes for metals;
- An additional 10.93 million tonnes of other waste fractions, or of the above mentioned materials in other fractions, will be recycled; and
- Composting will increase by 4.28 million tonnes, home composting by 0.11 million tonnes and anaerobic digestion by 0.27 million tonnes.

The environmental effect of these shifts can be assessed as follows:

- The reduction in landfilled MSW will result in a reduction of 46,213,200 tonnes of CO_2 equivalent (t CO_2e) (when considering landfills with 20% landfill recovery) or 33,376,200 t CO_2e (if taking into account the carbon sink effect);
- The increase in incineration corresponds to a reduction of 3,129,600 tonnes of CO_2 emissions;
- Increased recycling of paper, plastics, glass and metals will result in a total reduction of 10,532,204 t CO_2e. The associated avoided natural resource depletion is estimated to be 230,580 tn Sb eq. Fossil resource depletion is estimated at 10,488,180 toe. For other waste fractions the supplementary recycling will reduce CO_2 emissions by 821,882 tonnes, avoided natural resource depletion is estimated at 20,494 tn Sb eq, and fossil resource depletion at 913,202 toe;

- Composting and home composting will reduce CO_2 emissions by 35,120 t CO_2e or 263,400 t CO_2e (if carbon sink effects are taken into account); and
- Anaerobic digestion will reduce CO_2 emissions by 21,870 t CO_2e or 39,420 t CO_2e (if carbon sink effects are taken into account).

The aggregate environmental impacts can be summarised as follows:

- Total reduction of CO_2 emissions: 60,753,876 CO_2e, or 48,162,706 CO_2e if the carbon sink effects of landfilling and composting are taken into account;
- Total avoided natural resource depletion: 251,074 t Sb eq; and
- Total avoided fossil resource depletion: 11,401,382 toe.

As pointed out above, when compared to the 2020 EU targets, Scenario A achieves higher levels of recycling in all waste streams with the exception of plastics. Specifically in Scenario A, 14.61 million tonnes of paper, 7.8 million tonnes of glass and 2.35 million tonnes of metals will be recycled in addition to the 2020 target. For plastics, the achieved levels in Scenario A will be lower by 0.1 million tonnes. In terms of environmental impacts, this is translated to 11,980,200 t CO_2e for paper recycling, 1,404,000 t CO_2e for glass recycling and 4,158,910 t CO_2e for metal recycling. Concerning plastics, the 2020 target corresponds to higher avoided emissions and specifically 98,612 t CO_2e.

In comparison to the 2020 target, in Scenario A, the overall avoided environmental impacts are as follows:

- CO_2 emissions: 17,444,497 t CO_2e
- Total avoided natural resource depletion: 487,800 t Sb eq; and
- Total avoided fossil resource depletion: 22,120,780 toe.

For **scenario B** it was calculated that, as a consequence of introducing a landfill tax of €40 per tonne and introducing supplementary EIs which lead to the effect that all supplementary diverted MSW goes to recycling or composting, and compared to the 'do nothing' baseline scenario:

- Landfill of MSW will decrease by 42.08 million tonnes;
- Incineration of MSW will increase by 2.51 million tonnes (mainly recycling residues);
- Recycling will increase by: 14.06 million tonnes for paper, 2.34 million tonnes for plastics, 6.05 million tonnes for glass, and 1,95 million tonnes for metals;
- An additional 14.38 million tonnes of other waste fractions, or of the above mentioned materials in other fractions, will be recycled; and
- Composting will increase by 5.49 million tonnes, home composting by 0.13 million tonnes and anaerobic digestion by 0.42 million tonnes.

The environmental effect of these shifts can be assessed as follows:

- The reduction in landfilled MSW will result to a reduction of 45,446,400 t CO_2e (when considering landfilling with 20% landfill recovery) or 32,822,400 t CO_2e (if taking into account the carbon sink effect);
- The increase in incineration corresponds to a reduction of 602,400 tonnes of CO_2 emissions;

- Increased recycling of paper, plastics, glass and metals will result in a total reduction of 16,754,444 t CO_2e. The associated avoided natural resource depletion is estimated to be 310,870 tn Sb eq. Fossil resource depletion is estimated at 14,145,480 toe. For other waste fractions the supplementary recycling will reduce CO_2 emissions by 1,081,305, avoided natural resource depletion is estimated at 26,963 tn Sb eq, and fossil resource depletion at 1,201,449 toe;
- Composting and home composting will reduce CO_2 emissions by 44,960 t CO_2e or 337,200 t CO_2e (if carbon sink effects are taken into account); and
- Anaerobic Digestion will reduce CO_2 emissions by 34,020 t CO_2e or 61,320 t CO_2e (if carbon sink effects are taken into account).

The aggregate environmental impacts can be summarised as follows:

- Total reduction of CO_2 emissions: 63,963,529 CO_2e, or 51,659,069 CO_2e if the carbon sink effects of landfilling and composting are taken into account;
- Total avoided natural resource depletion: 337,833 t Sb eq; and
- Total avoided fossil resource depletion: 15,346,929 toe.

In scenario B, 18.25 million tonnes of paper, 0.5 million tonnes of plastics, 9.47 million tonnes of glass and 2.82 million tonnes of metals will be recycled in addition to the 2020 EU targets. In terms of environmental impacts, this is translated to 14,965,000 t CO_2e for paper, 663,084 t CO_2e for plastics, 1,704,600 t CO_2e for glass and 5,479,645 t CO_2e for metals.

In comparison to the 2020 targets, the overall avoided environmental impacts of scenario B are as follows:

- CO_2 emissions: 22,812,330 t CO_2e
- Total avoided natural resource depletion: 648,380 t Sb eq; and
- Total avoided fossil resource depletion: 29,435,380 toe.

In conclusion, scenarios A and B both demonstrate considerably reduced environmental impacts. This occurs both when comparing to the overall impacts of the baseline scenario (which takes into account all possible means of waste treatment) and when estimating the avoided impacts of recycling which are additional to the 2020 EU targets. Concerning the second case, the impacts of the 2020 EU targets are lower in plastics recycling in scenario A, but when taking into account the overall impacts (of all waste streams), scenario A demonstrates the better environmental performance.

5 POLICY OPTIONS

5.1 INTRODUCTION

Based on the work undertaken in the study, the project team was asked to suggest a selection of potential policy options for consideration by the Commission; these may then be further investigated and developed at the EU level to promote an 'optimal' use of economic instruments (EIs) by the Member States (MS). It will of course be important for the environmental, social and economic impacts of policy options to be considered during the design of future policies and EIs.

Some general principles could be applied to the implementation of the various policy options presented below, to ensure that they are as effective as possible. Most notably, these principles could include:

- Allowing some flexibility for MS to implement EIs in the most appropriate way for their own particular conditions (i.e. respect of the subsidiarity principle). This principle was particularly highlighted as important by stakeholders during the study;
- Ensuring an appropriate balance between regulatory instruments (e.g. targets, technical standards, bans) and EIs. Whilst EIs are intended to incentivise an improvement in waste management behaviour, they can equally generate additional stimulus for illegal activity, so regulatory approaches need to be developed in parallel;
- Considering carefully what should be done with revenues generated from EIs. Financial orthodoxy tends not to support ear-marking (or hypothecation) of revenue. However, innovative refunding mechanisms intended to strengthen the incentive of the instrument (such as that used for refunding landfill tax revenues in Catalunya) may be worth considering. There may also be arguments for some use of revenue, preferably for a fixed time period, to support development of infrastructure, environmental protection measures, and communication and awareness-raising activities;
- Providing a clear policy framework for the foreseeable future within which the waste management industry can operate. This is a pre-requisite for rational investment in infrastructure. MS should consider the need to announce rates for instruments such as landfill and incineration taxes for several years ahead. If the rates change from one year to the next, industry is likely to hold back from investing;
- Fully taking into account the economics of the waste management sector, which many MS are currently failing to do. The development of EIs must rest on rational analysis of the problem in hand, including the costs for collection of different waste streams, and the costs of recycling, treatment and disposal. In the absence of this, MS will struggle to achieve various waste management objectives;

- A full understanding by MS of the external costs and benefits of various waste management options. This is essential if the MS are to feel confident in developing EIs. It is more than a decade since the publication of a study[54] commissioned by the European Commission on the externalities of landfill and incineration, and much has changed since then; and
- Requiring better reporting by MS on waste generally and on the use of EIs specifically. Enhanced reporting requirements (e.g. through the MS waste management plans required by the WFD, through the reporting questionnaires attached to the Packaging, ELV, WEEE, Batteries and Landfill Directives, and through better integration of reporting requirements with the Waste Statistics Regulation (2150/2002/EC)) would generate more consistent and comparable data, could encourage MS to use certain EIs, and could be used to gather more information on the design, costs and impacts of EIs. This could in turn help to further develop or refine any future EU level guidance on EI use in the waste management sector. Improved data clarity has been highlighted by stakeholders as an important foundation for the development of future waste policy and the encouragement of the use of EIs.

5.2 POLICY OPTIONS BASED ON THE CURRENT STUDY

The policy options below draw on the research undertaken during the current study. These options have been developed with a view to encouraging the movement of waste management up the waste hierarchy as defined in Article 4 of the Waste Framework Directive (2008/98/EC). EIs, or combinations of EIs, should ultimately support waste prevention.

5.2.1 OPTION 1 – SETTING A MINIMUM LEVEL OF LANDFILL TAX TO BE APPLIED IN ALL MS

The study has found that MS have increasingly been introducing landfill taxes for non-hazardous MSW. (Although not the focus of this study, it is also worth noting here that landfill taxes also tend to be an effective way of dealing with commercial and industrial (C&I) waste.) Denmark was the first to introduce a landfill tax for MSW in 1987, 12 MS had taxes by 2000, and in 2012, the number will rise to 19 MS with Lithuania's planned tax (see Figure 17). The tax rates, materials exempted from the tax, and the purposes for which tax revenues are used, all vary from country to country.

Disposal of waste in landfill is the least preferred option according to the EU waste hierarchy as laid out in the EU Waste Framework Directive. Article 4 requires MS to give effect to the hierarchy in waste management policy and law. Economic theory suggests that environmental taxes should be set at a level where the marginal external costs from additional activities and the marginal benefits from undertaking the activity are equal. This theoretical ideal is, however, not so straightforward to achieve in practice with all wastes since the marginal damages from landfilling (related, to a large

[54] European Commission, DG Environment, A Study on the Economic Valuation of Environmental Externalities from Landfill Disposal and Incineration of Waste (Final Main Report), October 2000, http://ec.europa.eu/environment/waste/studies/econ_eva_landfill.htm

extent, to uncaptured methane emissions) are different for different waste streams, depending upon the extent to which they degrade in landfills.

As a result, MS have not tended to set landfill taxes on the basis of an attempt to precisely reflect the environmental externalities associated with landfilling (the UK tax, which was initially designed on this basis, has long since departed from this approach). It is more common for landfill taxes to be set to achieve higher rates of recovery, in particular recognising the external benefits from improved recycling rates. In part, it could be argued that landfill taxes are being set to capture the difference between the external damages caused by landfill, and some notional value of the external benefit from recycling.

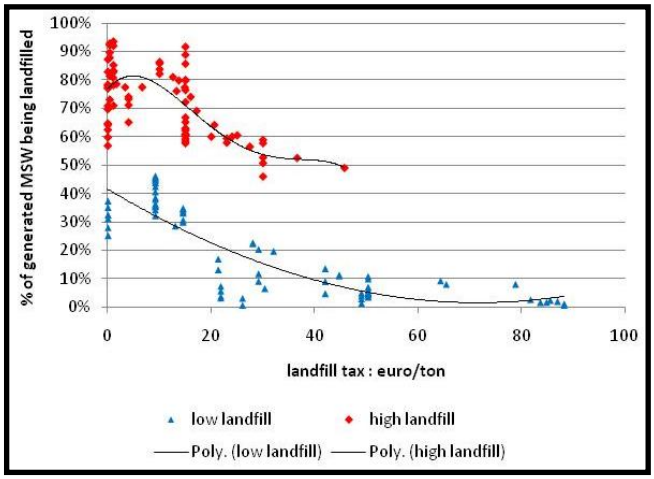
There appears to be a good case, therefore, for MS to implement landfill taxes with the aim of increasing the rate of recovery of materials in line with the waste hierarchy. Landfill taxes increase the cost of disposal so that the financial benefits of avoiding disposal increase. This acts to encourage waste prevention, reuse, recycling and recovery. At the same time, tax rates should take into account the fact that there are some materials which, due to their particular properties, do not generate the same level of environmental problems (in particular methane emissions) when sent to landfill.

The Commission could investigate the implied external costs of landfilling, to explore a potential basis for tax variation related to the composition of waste landfilled. Such an 'externality tax' could form the basis for a minimum tax rate for MS. It may not be appropriate to recommend the same minimum level for all MS, given their varying economic conditions; the minimum rate could be supported with some guidance for MS regarding the level of total disposal costs (i.e. gate fee plus tax) that appears necessary to achieve specific outcome(s), though again, this would likely vary across countries. This could, for example, draw upon estimates of the typical benefits from recycling a 'basket' of dry recyclables. In this way, the tax would 'top up' the prevailing gate fee to ensure that the total disposal cost would make most forms of source segregation for recycling financially viable, whilst allowing for the economic conditions in each MS to be taken into account.

There is a precedent for EU action on minimum rates of taxation, with the Energy Tax Directive. The results of the modelling exercise undertaken within this study suggest that a reasonable minimum level for a landfill tax for untreated MSW may be around €40 per tonne. A lower rate of tax (or a total exemption from the tax) could be applied for waste for which the environmental impacts of landfilling are known to be lower, or where wastes had been treated so as to significantly reduce their methane emissions.

The modelling exercise within this study conducted empirical research on the ratio between the landfill tax in € per tonne and the percentage of MSW which still is landfilled. The research suggested that low landfill taxes had little effect on the MSW percentages being landfilled, but that landfill taxes higher than €40 per tonne only affected a marginal further decrease in MSW quantities being landfilled. This is at least the case for MS that have already reached a rather low level of landfilling, though it must be borne in mind that most of these have implemented some form of ban or restriction on landfilling as well as a tax (with only Germany having a landfill restriction, but no tax).

Figure 47 Results of the modelling exercise



For MS that have bans that effectively prohibit the landfilling of untreated MSW (e.g. Germany), it may not be necessary to impose a tax (although in a few cases countries have increased taxes once a ban has been implemented to reduce the financial advantage to be gained from seeking exemptions from the ban).

In all MS, landfill taxes have tended to increase over time, as shown by Figure 23. Those MS with tax at lower rates should be encouraged to increase them to improve their effectiveness in pushing waste higher up the waste hierarchy. When imposing and increasing a landfill tax, a MS should decide on the trajectory to be taken (i.e. how often and by how much it will increase) and should ensure that the changes in tax are announced as far in advance as practicable. This will allow the operators of landfills and also the producers of waste to take into account the impact of the changes in tax. MS should also ensure that the tax does not fall in real terms, to avoid any increased incentive to return to landfilling.

It must be noted that imposing a fixed, minimum tax level through EU legislation is not desirable due to differences in the economic conditions in the MS, and it would in any case be difficult to bring about due to the unanimity required by MS on taxation legislation. Instead, a common method for calculating a minimum tax level could be developed, taking into account the varying conditions in the MS, and taxes could be more strongly encouraged in the worst performing MS (e.g. an option not to impose a tax where landfilling is below e.g. 10%, requiring a tax to be put in place when landfilling is above 75%, and encouraging the application of a tax for 'intermediate' MS).

Means of executing this policy option could include a revision of the Landfill Directive, a Decision addressed to the MS, or, if a non-legislative approach is preferred (although this would be more difficult to enforce), guidance issued to the MS.

5.2.2 OPTION 2 – SETTING CRITERIA/PRODUCING GUIDANCE FOR THE DESIGN OF PRODUCER RESPONSIBILITY SCHEMES

There is considerable variation in the design and performance of (extended) producer responsibility (EPR) schemes in the MS. The most successful schemes as identified in this study appear to share some common features, including:

1. A common, fully private body that is created, run, owned and supported by the obligated producers. The body's responsibility should include the marketing of collected and sorted materials. Not for profit organisations appear to be appropriate vehicles;
2. Requiring producers to fully fund the collection and recycling scheme which is implemented; and
3. High targets (or a requirement for a high level of service, such as generalised take-back obligations, which may have the same impact as high targets).

Feature 1 ensures that the responsibility is truly visited on the obliged producers. Their 'investment' (financial and otherwise) in the EPR scheme will ensure that it is in their interests for the scheme to function efficiently. Giving the scheme body the responsibility for marketing recyclable or reusable materials and products provides an incentive to maximise this revenue stream (to reduce costs to the industry it serves), improving the likelihood that the necessary collection infrastructure will be designed and operated so as to deliver materials of high quality and value. Assurances should be provided by a regulator/legislation that all obligated producers are contributing as they should, to address the issue of 'free riders'. Legislation could also usefully highlight the expectation that local authority collections will be supported.

The schemes that most effectively attribute the costs of waste management to producers tend to be the most effective. This is evidenced, for example, by the Fost Plus packaging EPR scheme in Belgium, which covers the full cost of collection and treatment and achieves around 92% recycling and 95% recovery. Efforts should therefore be made in the design of schemes to incorporate **Feature 2** to ensure that 100% (or as close to 100% as possible) of the costs are paid for by producers, making them financially responsible for the full costs of running the scheme. This is the approach which is most closely aligned with the polluter pays principle, and has the merit of ensuring that municipalities do not incur financial costs for the waste streams covered by the EPR schemes. To the extent that EPR schemes can incentivise waste prevention through product design, ensuring the producers are exposed to the full costs of collection and recycling is most likely to encourage such activity.

In many existing EPR schemes, the costs of meeting a specific target are relatively low when considered per unit of product sold. One reason for this is that if the targets themselves are low (or if the performance is low), then fees raised per unit of product placed on the market are required to cover only the costs related to the fraction which is recycled. Under such a system, the lower the recycling rate, the lower the cost will be, which has the effect that obligated producers have an incentive to maintain performance at low levels. The implied incentive to prevent waste, or to design

for recyclability, is also weakened. **Feature 3** – setting high targets and monitoring performance closely – can overcome some of these problems.

There are of course many other design features that may influence the success of EPR schemes. It is beyond the scope of the current study to address all these in detail, but some additional suggestions and discussion are included below to help direct future research in this area:

- A requirement to fully fund all collection, sorting and treatment of the target waste, including that which remains in the residual waste stream (i.e. that which does not pass through the EPR scheme). The consortium understands that this is the case for the Fost Plus packaging scheme in Belgium, as well as for the schemes in Austria and Germany; all three of these MS are amongst the top performers with regards to packaging recovery and recycling rates;
- Targeting both households and consumption of products 'away from home';
- Incentivising or otherwise encouraging the involvement of producers in eco-design;
- Encouraging the greater application of individual producer responsibility (IPR);
- Encouraging communication, dissemination of knowledge and best practice between producers, and between all actors in the production and consumption chain;
- Where possible, basing variable fees within schemes on defined criteria (general performance of the producer, degree of eco-design, recyclability of materials etc);
- Setting targets to better differentiate between materials; and
- Appropriate involvement of governments, municipalities and waste management operators.

It may be useful to consider requiring the fees paid by producers to cover the **costs of treatment and disposal of the fraction of waste that is not recycled**, but which is treated or disposed of through the residual waste system. The cost of this would (at least in principle) be reasonably simple to calculate as it could be based on the weight of the waste that is not recycled. This would offer a far greater incentive for producers to ensure that their products are both designed to be recyclable, and are actually recycled. Such a step could be particularly effective when used in conjunction with a minimum level of landfill tax (as described in Option 1). Linked to this should be an attempt to ensure that **both household consumption and the consumption of products 'away from home'** are targeted, to guarantee the maximum possible coverage of the scheme.

One of the main goals of EPR schemes should be to encourage greater integration of **eco-design** into the producers' product development, in order to prevent waste at source by 'designing in' longevity, recyclability, reusability and the use of less environmentally damaging materials. This is an idea that was supported by stakeholders during the study.

Collective schemes that include the participation of many producers can serve to dilute responsibility, as the overall performance of the scheme can mask the performance of individual producers (some producers may be performing extremely well whilst others are performing less well, but only the overall performance of the scheme is visible). It may be the case that the extension

of **individual producer responsibility (IPR)** would encourage individual producers to consider more carefully the environmental impacts of their products. Encouraging more widespread individual producer responsibility may be easier for some product types than for others due to the nature of the products. For example, items such as WEEE or ELVs are immediately recognisable as originating from a specific producer, whereas it can be much less clear which producer is responsible for packaging. Consideration would therefore need to be given to how to move closer to the ideal of IPR (e.g. more specific targets for products/materials) whilst ensuring that it is possible both in practical and financial terms to adequately monitor performance. It should be recognised that in order for IPR to be realised, some means of identifying and sorting branded items, or arranging for take back of brand-specific products, would probably be required.

Within collective schemes, it would be beneficial to **encourage communication, knowledge and best practice exchange** between the members of the scheme, allowing producers to identify strengths, opportunities and weaknesses in order to improve their performance. This could help to plug gaps related to specific types of products and related waste streams (e.g. shredder residues for ELVs, which can sometimes be difficult to recover effectively). National and EU federations/producer responsibility organisations could have a role to play in bringing together producers, and also bringing together all actors in the production and consumption chain (producers, consumers, waste management and treatment operators), to share best practices to improve overall waste management performance (and value for money of schemes) with regards to specific waste streams.

Producers could pay **variable fees** to EPR schemes, based on defined criteria. Such criteria could include the past performance of a producer within the scheme, the degree of eco-design attempted/achieved by a producer, or the recyclability of the materials used by a producer. If variable fees are used, they should be set such that more environmentally sustainable products are subject to lower fees. It should be noted, however, that all schemes need to cover their costs. Unless the design improvements lead to improved scheme efficiency, whether by generating genuine reductions in the cost of operation or genuine improvements in performance (i.e. increased proportion of material recycled), then the case for such differentiation will remain weak.

For packaging schemes specifically, **differentiated targets** could be considered for different types of packaging. This would help to distinguish between different types of materials based on the cost of treatment and the related environmental impacts. For example, materials that cannot currently be recycled are still in use even though alternatives are available; better differentiation of fees could help to encourage the use of such alternatives. At the same time, the target recycling rates for various types of materials could be ramped up. These two measures taken together have significant potential to ensure that certain materials are more widely recycled, and that there is greater recyclability of materials more generally. Careful consideration would need to be given to which materials to address, however (e.g. it would be much more difficult to implement for different types of plastics than for different brands of TV). The **appropriate involvement of governments, municipalities and waste management operators** within an EPR scheme can also go a long way to ensuring its success. Governments can usefully play a support and control role to ensure that a scheme has political backing and is carefully monitored. Local municipalities can assist through implementing suitably designed PAYT schemes, encouraging households to use infrastructure for

separate collection of the targeted materials or products. Waste management operators have an important role to play in the service provision associated with the scheme. During the study, stakeholders highlighted the importance of ensuring that the role of various stakeholders is clear in any scheme.

Means of executing this policy option could include integration of aspects of the option within relevant waste Directives (the Packaging, WEEE, ELV, Batteries and/or Waste Framework Directives), a Decision (or Decisions) addressed to the MS, or guidance on the design of EPR schemes if a non-legislative approach is preferred. Measures regarding, for example, the extent to which schemes should cover costs have been addressed in such Directives, but the Directives have generally fallen short of ensuring that producers are made fully financially responsible (including all costs associated with collecting and sorting materials) for delivering their obligation. With regards to best practice exchange, a forum could be created to promote discussion, or EU funds could potentially be used to promote the more concrete application of successful schemes in less well performing MS.

For the guidance option, further research would be needed; this could fit well with the current timetable to revise waste targets by 2014 as it would allow guidance to be published to accompany the revised targets. Within such guidance, it would be beneficial to identify the key success factors for EPR schemes, in order to either:

- Require EPR schemes in all MS to comply with a set (or subset) of 'minimum standards'; or
- Provide the MS with guidance on the design and implementation of EPR schemes.

Guidance could also indicate which materials in addition to those currently covered by EU Directives might be considered appropriate targets for the introduction of EPR schemes (e.g. furniture, junk mail).

5.2.3 OPTION 3 – ENCOURAGING THE USE OF CHARGING THAT ENSURES WASTE GENERATORS FACE INCENTIVES IN LINE WITH THE WASTE HIERARCHY

In principle, all producers of waste should face some incentive to reduce waste generation and to make use of recycling services which are cost-effective under prevailing market conditions. Regarding the latter, **landfill taxes** will increase the range of recycling services that can be offered cost-effectively (see Policy Option 1 above), and the costs of providing some services may be fully supported through producer responsibility schemes (see Policy Option 2 above). The same could be said for **incineration taxes**. Landfill and incineration taxes will help discourage disposal of, or energy recovery from, waste that could be dealt with higher up the waste hierarchy.

The response of waste producers to instruments such as landfill taxes and incineration taxes will be mediated by the way in which they are charged for the services they use. Not all waste producers are charged in such a way that they benefit, financially, from actions that either reduce the waste they generate or lead to a reduction in residual waste (through recycling and composting etc.), even though this may reduce the cost of the services they use.

As regards **households**, as discussed in this report, the extent to which municipalities in the MS make use of **PAYT schemes** varies widely. The nature of the schemes used also varies considerably. Fundamentally, however, the literature is clear that PAYT systems will work best, and will be least problematic, where the infrastructure for recycling is both comprehensive and convenient for the user (this also reiterates the point that the higher the cost of disposal, the more likely it is that such a service becomes financially viable). It may be the case that, if recycling targets are set at an appropriately ambitious level, there will be a natural evolution of the MS towards the implementation of PAYT schemes.

Generally PAYT schemes should seek to charge:

- The highest variable fee for residual waste;
- A lower (but non-zero) fee for biowaste if garden waste is targeted by biowaste collection (to encourage home composting);
- A zero fee where only kitchen waste is targeted by biowaste collection (anecdotal evidence suggests that providing separate collection of kitchen waste has, independently of charging schemes, a waste prevention effect); and
- A low or zero fee for collected dry recyclables.

Schemes should also be encouraged to cover costs by splitting the revenue generation between a fixed fee element and the revenue from the variable fees. This also ensures that the marginal rates (i.e. the cost per pick up of residual waste) are not so high as to encourage illegal dumping of residual waste.

Evidence suggests that with regards to waste prevention, there is a ranking of systems as follows (from most to least successful):

- Weight-based systems;
- Volume and frequency-based/sack-based systems; and
- Volume-based systems (i.e. schemes where households simply choose a specific size of container).

It should be noted, however, that the costs of implementing the different schemes vary. The level of waste prevention achieved appears to be strongly affected by whether garden waste was collected free of charge prior to the introduction of PAYT, and by the approach for charging for biowaste collections under the PAYT scheme. However, wastes other than the organic fractions also appear to be affected.

The effects on recycling of the different approaches to PAYT are strongly affected by the nature of the collection scheme used and by the charging scheme. The proportion of total waste recycled tends to increase under PAYT, but the extent of the increase will be determined by the scope and convenience of the recycling infrastructure.

On balance, therefore, it seems reasonable both to encourage use of PAYT, and to encourage the use of PAYT systems that do not charge solely on a 'subscription basis' by using a specific sized

container for a whole year. The aim should be to structure charging systems so that they introduce incentives to reduce, continuously, the quantity of residual waste being generated.

It should be noted that where PAYT systems are concerned, the need (or otherwise) to generate revenue to support the costs of recycling provision will be affected by the scope of application of producer responsibility schemes, and the extent to which those schemes are required to cover the costs of the services provided by, or on behalf of, the municipality. In principle, therefore, for those waste streams covered by producer responsibility, there ought to be no need to raise revenue through PAYT schemes. However, there are still sound reasons (i.e. waste prevention) to levy small charges on dry recyclables, albeit at substantially lower rates than for residual waste.

It is likely that in many MS it will become increasingly common for the collection of **commercial and industrial (C&I) waste** to be charged based on the number of collections (with the price related to the container size), the number of sacks collected or the weight of waste produced. These approaches all offer some incentive for producers to reduce their waste, although charging per collection may generate no incentive if containers are simply less full as a result of waste prevention (or recycling) efforts. In principle, it would be desirable for C&I waste producers to be charged in such a way that there are stronger incentives, at the margins, for waste prevention and for source separation.

MS are required, under the Waste Framework Directive, to both implement the hierarchy in policy and law (Article 4) and to implement Waste Prevention Programmes (Article 29). It might reasonably be expected that MS should ensure the removal of all obstacles to implanting measures which enable the right incentives to be conveyed to waste producers, including households. Indeed, such programmes should encourage the implementation of PAYT, for example, recognising that it has a fundamental role to play in reducing household waste generation and increasing household recycling.

Guidance could usefully help to define how to design PAYT schemes so that they enhance the prospects for prevention and recycling, whilst ensuring charging structures recover costs. There are existing examples of such guidance in some MS, and guidance also been developed by the US Environmental Protection Agency in the United States.

5.3 POSSIBLE ALTERNATIVE POLICY OPTIONS

Below are listed several additional options which are not necessarily suggested based on the research undertaken during the current study, but which, based on the experience of the study team, may nevertheless be worth considering. During the course of the study, stakeholders also suggested that instruments outside the scope of the current study – such as product or materials taxes, fee-rebate schemes and recycling credits – be considered by the Commission to inform options for future policy. These are all discussed to some extent in the alternative policy options outlined below.

5.3.1 OPTION 4 – PRIMARY MATERIALS TAXES

It is important to acknowledge the potential for application of taxes on primary materials as a means to a) reduce the consumption of materials generally, b) encourage re-use and preparation for re-use, and c) increase the proportion of material use which is accounted for by recycled materials. In theory, this could be an attractive option for improving the sustainability of materials management.

In reality, there are a number of obstacles which confront the use of such an instrument. One of the largest challenges is presented by the ongoing trade in materials and products which occurs both within EU countries, and between the EU and the rest of the world. In principle, if such a system is not to place those affected by such an instrument at a competitive disadvantage, it would be necessary to implement some form of border tax adjustment (BTA) which enabled imports to be taxed at an equivalent rate to domestic production, and exports to be exempted from the tax. For individual MS, the option of using BTA effectively does not exist since it would appear to contravene principles regarding the operation of the internal market. BTA is a possibility for EU wide taxes, but even at the EU level, the information requirements in order to apply BTA in a fair way would be significant. This is because for all imports, an estimated 'default' material content would be needed, but there must also be flexibility for the importer to demonstrate a level of use differing from that default. Furthermore, the scope of application of such taxes would need to be considered very carefully since, unless all materials were covered, there is a risk that switching between materials occurs, placing some at a disadvantage relative to others.

Perhaps for these reasons, the scope of application of such taxes has been limited in the EU, with taxation generally being focused on materials such as aggregates and gravel. These materials are not so widely traded and such taxes have been applied successfully in, amongst others, Denmark, Sweden and the UK.

In practice, therefore, this may be a difficult policy option to execute. In addition to the reasons outlined above, any EU level taxation would require unanimity amongst MS, which could be difficult to achieve. It is therefore suggested that the other options below be considered as more practical and achievable alternatives, other than for a narrow range of materials such as aggregates.

5.3.2 OPTION 5 – LEVY ON EXCESS RESIDUAL HOUSEHOLD WASTE

Recycling targets for household waste suffer from a drawback in that they can reward schemes which simply collect large quantities of garden waste. This makes comparison across different municipalities somewhat awkward in that differentials may be affected strongly by whether or not garden waste is collected separately, whether it is collected free of charge, and the type of housing in any given municipality. Similarly, the generation of household waste varies considerably across MS. Some MS with lower recycling rates also generate smaller quantities of waste per inhabitant than MS with much higher recycling rates.

A measure which effectively incorporates the effectiveness of measures aimed at sustainable consumption, waste prevention and waste recycling is the quantity of *residual* household waste per inhabitant. Such a measure would likely even out the level of performance between MS with high

consumption levels and recycling rates, and those with lower consumption levels but lower recycling rates. To the extent that the overall impact on resource use is likely to be an important determinant of environmental impact, a focus on 'what is not kept in the cycle of use' is likely to be of more interest than a measure focused only on the percentage of waste recycled.

One EI which has been discussed elsewhere is the use of a levy for the generation of residual waste above an average level, expressed in terms of kg per inhabitant. The instrument generates a dynamic incentive for reducing residual waste per inhabitant by charging a levy for residual waste which is generated above the average level, and refunding the revenue in full to those whose residual waste is below the average level. The strength of the incentive is determined by the magnitude of the levy. Such a scheme could be introduced at MS level, or on an EU-wide basis. This would introduce an incentive for municipalities to reduce residual household waste by encouraging and incentivising waste prevention, preparation for re-use and recycling.

5.3.3 Option 6 – Deposit refunds for hazardous materials/materials containing valuable materials

The literature in respect of deposit refunds highlights their potential usefulness where the waste material or product concerned can give rise to problems if they are poorly managed. In this respect, hazardous materials, materials which may be mismanaged, materials of potentially high value (including critical materials) and materials with a propensity to arise as litter are suitable candidates for such schemes.

One interesting application of this could be the collection of small WEEE items. Such materials may be capable of being prepared for re-use, and may contain valuable materials for recycling. They can also be problematic in environmental terms at the end of their life. As such, to retain the material value, and to avoid improper management of the waste, a deposit refund scheme may provide a useful way in which to encourage high rates of take-back of such products.

5.3.4 Option 7 – Refundable compliance bonds

One policy which may (in conjunction with landfill taxes and taxes on primary aggregates) have a useful role to play in increasing recycling rates for C&D wastes is an approach using refunded compliance bonds. Under this arrangement, contractors would be required to pay a financial sum to the relevant regulatory land-use planning authority, related to the size of the project at its commencement, plus a small administrative fee (to cover the administrative costs of the system). The financial sum would be retained as a bond to ensure that the project exceeded a specified recycling rate, which could be set higher for public sector projects (in line with green procurement principles).

The size of bond paid would vary by project size, and all of the bond, excluding the administrative fee, would be returned on demonstrating achievement of the desired recycling rate. A proportion of the bond only would be refunded where partial compliance with the desired target was achieved.

The mechanism could require materials from the project to be sent to authorised facilities, which had demonstrated that they could recycle the proportion of material suggested by the developer's targets. As such, a procedure for authorising the reprocessors who receive the material could also be required.

5.3.5 Option 8 – Product Taxes and Charges

Many schemes implemented under producer responsibility make use of what are effectively product charges, designed to fund the operations of a compliance scheme. These charges do not generally bear any strong relation to the environmental impact of the product itself.

A number of MS make use of product taxes. Several of these have been shown to affect waste generation. Product taxes are likely to reduce waste when they are applied to products for which there are clear substitutes which lead to lower levels of waste generation. Most obvious, in this respect, are taxes on disposable items. Although plastic bag taxes have not always been portrayed in this way, they provide a classic example of how a tax on a disposable item can reduce waste generation.

If society genuinely wishes to reduce waste generation, then it seems reasonable to tax disposable items to discourage their consumption, and encourage the use of non-disposable alternatives.

5.3.6 Option 9 – Subsidies for Waste Prevention Activities

Subsidies for activities that promote waste prevention may be justified where they lead to behaviours that generate financial savings to the public purse and/or environmental benefits which exceed the level of subsidy being granted. Examples of such schemes already in place address mainly home composting and the use of reusable nappies. Support for home composting sometimes comes in the form of reduced levels of payment under PAYT, in which case, the provision of an additional subsidy may not be relevant.

There may be other relevant target waste streams which may justify support in order to enable households to switch more easily to lower-waste options for a given product or service.

5.3.7 Option 10 - Variable VAT Rates

Within the scope of the existing VAT Directive (2006/112/EC), MS are given some discretion to support particular activities through reduced levels of VAT. In particular, organisations focused on social wellbeing would appear to have a freer rein than commercial enterprises to offer reduced VAT over a wider range of services. The example of repairing white goods (large household appliances) is a category of service which would seem to comply with the spirit of the Directive on a number of points.

Another potential use of the option to vary VAT rates might be to encourage manufacturers to offer an extended warranty over and above the typical minimum period. VAT rates could be lowered in cases where warranties were for a longer period of time.

Other possible applications could be for products and services that are intrinsically less waste-intensive than close substitutes. The same types of product that could otherwise be targeted by product taxes could be the subject of a change in VAT. Similarly, the provision of a service that can replace the purchase of products could be supported by a reduction in VAT, if it can be shown that the service model is environmentally preferable.

Even if reduced rates of VAT may be effective for certain applications, it should be recognised that other approaches such as product taxes may be more cost-effective, and potentially more 'visible'. Further research is required to better understand the situations where one instrument would be preferable to the other.

5.3.8 OPTION 11 – RECYCLING CREDITS

The use of support schemes for renewable energy has increasingly been justified through reference to the savings related to reduced greenhouse gas emissions. Other factors are also important, such as the contribution made to diversifying the energy supply and improving energy security. Several schemes for supporting the price for renewable energy cover the generation of energy from waste, occasionally including energy generated from landfill gas, sometimes including incineration, and often including anaerobic digestion.

In terms of the contribution made to reducing greenhouse gas emissions, recycling receives no equivalent support, even though for most dry materials (e.g. paper, card, plastics, metals and glass) the savings appear to exceed those which are associated with the generation of energy from the same waste. There are therefore questions as to whether such support mechanisms are justified, but also whether there might be justification for some incentive to support the recycling of various materials.

In principle, it could be possible to adapt the EU ETS (Emission Trading Scheme) to recognise emissions savings related to recycling. At present, this is the case only to a limited extent, and under fairly specific circumstances due to the sectoral and geographical scope of the EU ETS. Establishing the principle of recognising the benefit of recycling relative to use of virgin materials within the EU ETS could therefore be an important first step to wider recognition of recycling within a future design of the ETS.

More work is needed to develop the exact mechanism by which a credit would be awarded, along with consideration of how the full range of materials could gain appropriate recognition, alongside issues relating to how to account for imports and exports, given the sectoral scope of the 'core' EU ETS at any given point in time.

7 LIST OF ANNEXES TO THE REPORT (SUBMITTED SEPARATELY)

Annex 1: Data fact sheets

This annex contains the completed data fact sheets for the 27 MS, which gathered much of the raw data on the use of economic instruments in the waste sector in the MS.

Annex 2: Full case studies

This annex contains the longer versions of the case studies presented throughout the main body of the report (on overall MS waste management performance, landfill, PAYT schemes and producer responsibility schemes).

Annex 3: Producer responsibility organisations in the EU-27

This annex provides a list of the producer responsibility organisations in the EU 27 as identified during the study. (The list may not be fully comprehensive.)

Annex 4: Producer responsibility targets and deadlines

This annex summarises the main producer responsibility recycling/recovery targets and deadlines in the MS.

Annex 5: Full modelling report

This annex comprises a separate report, prepared by Arcadis, detailing the modelling exercise undertaken within the study.

Annex 6: Report from stakeholder workshop

This annex presents the main discussions during the stakeholder workshop held in Brussels on 25 October 2011.

Annex 7: References

This annex provides a list of the references and information sources used throughout the study.