The Incentive effect, Hypothecation and Polluter Responses to Environmental charges

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Abstract

Economists have long regarded environmental charges as essential for the effective management of industrial pollution. They argue that the source of many environmental problems lies in the failure of market systems to value environmental services fully, and that applying the polluter pays principle restores the financial incentive for businesses to engage in socially ‘optimal’ pollution abatement. Furthermore, the revenues accruing from environmental taxes or charges can be re-directed, or hypothecated, towards environmentally beneficial expenditures. This paper explores the empirical link between economic instruments and business environmental performance by examining British and German companies affected by the European Union Packaging Waste Directive. The study’s findings suggests that the incentive effect of environmental charges was less pronounced than anticipated, but that ‘closed-loop’ hypothecation was used to immediate, though arguably sub-optimal, effect. Factors affecting corporate environmental behaviour are assessed and the practical utility of environmental charges is discussed.

Introduction

Academics and policy-makers concerned with environmental affairs have long been concerned with the development of policy mechanisms for regulating industrial pollution (Segerson, 1996). Whilst legislation remains the predominant form of environmental regulation (Huppes et al., 1992), this approach has been criticised by economists and neo-liberal thinkers as inflexible, ill-informed, and, especially in the European Union (EU), for its poor record in arresting the general trend of environmental decline in Europe (Commission of the European Communities (CEC), 1992). This dissatisfaction, along with awareness of the need to integrate environmental concerns into other policy areas, has led to heightened interest in price-based environmental regulation (Pearce et al., 1989).

The basic principle behind price-based regulation is that by applying the polluter pays principle (PPP) alongside legislation, polluters will be encouraged to reduce the ecological footprint of their activities to ‘socially-optimal’ levels (Baumol and Oates, 1988; van den Bergh, 1996), the point where the marginal cost of additional abatement exceeds the marginal benefit to society. Whilst price-based regulation is becoming increasingly commonplace – Pearce and Barbier (2000) note that between 1987 and 1992 the number of price-based instruments in OECD countries increased from 100 to 169 – this paper critically examines their ability to generate significant pollution-reducing incentives and hypothecation benefits. Its findings are based on recent studies of British and German businesses affected by the EU

The question inevitably arises as to how generalisable conclusions based on a single case study can be and how representative the chosen study is of wider trends in price-based environmental regulation. No attempt is made here to claim that analysis of the Packaging Waste Directive captures all forms of price-based environmental regulation or their full effects on polluter behaviour. However, it is suggested that the example raises a number of important questions about the broader relationship between environmental charges and policy outcomes; issues that require further exploration if the effects of price-based regulation are to be better understood. While some of the study’s findings clearly relate solely to the issue of packaging waste, the conclusions suggest the need for additional case studies examining polluter responses to environmental charges and the policy outcomes engendered by their application.

The paper proceeds as follows. The first section reviews the theoretical underpinnings of environmental charges and, in particular, the incentives produced for changes in polluter behaviour and the role of revenue hypothecation. The primary aim of this section is to establish the key policy objectives that may be promoted using environmental charges, as well as to identify the debates within the literature concerning the practical efficacy of environmental charges. The second section outlines the policy context of the case study by examining the negotiation of the Packaging Waste Directive at the EU and its implementation in Britain and Germany. The third and fourth sections review the research methods employed and the results obtained, whilst the final sections address the broader practical benefits and limitations of price-based environmental regulation.

Theoretical Background

The price mechanism is generally regarded as the most efficient method for communicating the relative valuations consumers and producers ascribe to particular commodities (Pearce et al., 1989). Its inherent difficulty in relation to environmental resources, however, is that property rights over many ‘common goods’ such as air or water are either extremely ill-defined or entirely absent. In situations where there are no identifiable title holders for particular natural resources, environmental costs will remain largely external to market systems and economic agents even where externalities and losses in environmental utility can be readily identified (van den Bergh, 1996). Under such conditions the natural tendency is for economic systems to over-exploit the environment in terms of its resources and pollution sinks (Devlin and Grafton, 1998). The increasing sophistication and globalisation of industrial processes has exacerbated these trends by shifting externalities increasingly from the local to regional or global scales. Considering these problems, a key role of governments in the environmental sphere is the development of policy mechanisms that re-internalise environmental costs within economic systems (Turner, 1993).

The principal environmental objective of taxing polluting activities, therefore, is to encourage industries and households to curtail or discontinue their more environmentally damaging practices (Pearce et al., 1989, Gouldson and Murphy, 1996). The aim is rarely to eliminate particular pollutants altogether (in most cases, this would be unfeasible or economically damaging) but rather to achieve ‘optimum’ pollution where the marginal cost of any further
abatement exceeds the marginal benefit gained by society (van den Bergh, 1996). A significant by-product of the economic approach to environmental problems is the potential hypothecation of environmental-charge revenues, either for environmental expenditures or to shift the tax balance away from economic ‘goods’, such as employment (Gee and von Weizsäcker, 1994, Bohm, 1997). However, this benefit is often seen as outweighed by the economic distortions and rigidities hypothecation introduces into taxation systems (O’Riordan, 1997).

In terms of economic impact, there may be little to distinguish price-based policies from legislative techniques, as both impose abatement costs on polluters (Jacobs, 1991). Moreover, most economists acknowledge that economic instruments should complement rather than replace legislative standards (Pearce and Barbier, 2000). The critical distinction, however, is that legislative standards only impose threshold constraints on pollution, while economic instruments can monetise each unit of pollution and, thereby, create a constant pressure for improvement. Environmental charges consequently exact compensation for environmental damage and a rent on the use of natural capital (Jacobs, 1991). Furthermore, if market forces are utilised to govern environmental charges, as occurs with tradable permits, pollution prices should respond dynamically to changing economic, environmental and regulatory circumstances (Goddard, 1995). This notion of an incentive effect has become a central orthodoxy of the economic approach to environmental policy. Pearce et al. (1993: 96), for example, note that:

*Since price is instrumental in changing behaviour* it follows that taxation policy will also be an important influence on behaviour which affects the environment (emphasis added).

Similarly, Baumol and Oates (1979: 231):

In [the introduction of environmental charges or subsidies] the basic notion is the same: by offering virtue its just (financial) reward, we change the rules of the game to induce industry (and individual consumers) to alter their behaviour to promote an environmental objective.

The behaviour-changing potential of economic instruments is not undisputed, however. Jacobs (1991) argues that whereas legislation compels firms or households to observe pre-defined environmental standards, incentives merely encourage them to do so. Ultimately individuals may choose to pay charges without checking pollution. The point at which environmental charges effectively control particular pollutants therefore depends on the latter’s price elasticity; consequently, several charge iterations may be necessary before the desired abatement is achieved. Pearce et al. (1989) consider that sophisticated environmental-valuation techniques can help to combat some uncertainties in price behaviour, but More et al. (1996) question whether economics is adequately equipped to deal with such eclectic matters as future resource needs or the intrinsic value of the natural world.

Jones (1999) questions the incentive effect of environmental taxes and charges from a different angle, suggesting that polluting businesses base investment and technology decisions on total factor costs rather than individual components or marginal cost penalties. He therefore contends that the relationship between environmental taxes and business behaviour should not be over-exaggerated. Hahn (1989: 95), whilst supporting environmental valuation, confesses that the theoretical structure of environmental economics ‘often emphasises elegance at the expense of realism,’ while Jacobs (1991: 152) adds that
such models often: ‘fail to represent the complexities of the real world, in which “institutional” factors crucially affect corporate and consumer decision-making.’

Empirical evidence of the incentive effect of environmental charges and taxes is also somewhat uncertain (Opschoor and Vos, 1989; Hahn, 1989; Beder, 1996). The European Environment Agency’s (EEA) latest survey of environmental taxes in the EU reports significant successes in terms of switches to less-polluting technologies (EEA, 2000). However, Barde (1997) claims that most existing environmental taxes are set too low to produce pollution-controlling incentives, adding that carbon taxes must be raised significantly above $50 per tonne if CO\textsubscript{2} emissions are to be stabilised at their 1990 levels by 2050 (Sweden, a leader in such levies, currently taxes carbon at $41 per tonne). Tietenberg (1990: 32) nonetheless argues that even where environmental charges do not produce major pollution-reducing incentives, they have the ‘compelling virtue’ of achieving targets in a cost-effective manner. In further defence of the economic approach, Hahn (1989), Huppes et al. (1992) and Goddard (1995) note that charges and marketable permits are rarely introduced in their textbook form but are instead grafted (sometimes inappropriately) onto regulatory systems where legislation plays a dominant role. Thus, separating the impacts of regulation from those produced by environmental charges can be difficult (EEA, 2000).

The second component of environmental taxes, the hypothecation of revenues for environmental expenditures or to reduce tax penalties on activities that increase economic value, is also by no means universally welcomed. Excessive hypothecation may introduce rigidities into government expenditure by creating long-term commitments to programmes that may not be justified in terms of public or environmental need (O’Riordan, 1997). Barde (1997) adds that hypothecation detracts environmental taxes away from their main purpose of changing polluter behaviour, and that reductions in pollution may induce over-capacity in pollution-control facilities and, hence, economic inefficiencies. There is also the question of matching environmental taxes to expenditure requirements. Smith (1997) considers that only where this occurs naturally (arguably an improbable eventuality), can both be set at the correct level. Against this, where earmarking is tightly packaged to ameliorate specific problems targeted by the charge, effective loops between environmental charges and expenditure can be created (Spackman, 1997). Such systems can also assist public understanding of the reasons for the introduction of new charges or taxes (Tickell, 1995).

The incentive effects and hypothecation of environmental taxes and charges are therefore important, if controversial, elements of the economic approach to environmental policy. The literature has both defended the need to monetise environmental resources and recognised the practical weaknesses in this methodology. The principal issues are therefore; first, the extent and levels at which environmental taxes lead to reduced industrial pollution; and, second, whether other environmental effects accruing from this approach justify the imposition of new costs on industry.

**Policy Context - the Packaging Waste Directive**

The Packaging and Packaging Waste Directive (hereafter the Directive) was formally adopted by the EU Council of Ministers on 20 December 1994 (OJEC, 1994). Its primary purpose was to approximate Member State packaging laws following the introduction of the German *Verpackungsverordnung* (Packaging Ordinance) in 1991. Fearing that the recycling targets
imposed in Germany would become technical obstacles to free trade in the Single Market, other states and the European Commission pressed for harmonising legislation under Article 100a of the EU Treaty (Golub, 1996). The main objective of the Directive was that the Member States should introduce measures aimed:

as a first priority, at preventing the production of packaging waste and, as additional fundamental principles, at reusing packaging, at recycling and other forms of recovering packaging waste and, hence, at reducing the final disposal of such waste. (Article 1) (OJEC, 1994: 12).

However, reflecting the many disagreements between the Member States during policy negotiations, only three targets were actually quantified. The first two established national recovery and recycling targets for packaging waste of 50-65% and 25-45% respectively by June 2001; the other stipulated reduced concentrations of prescribed heavy metals in packaging (Article 11). The objectives of preventing, re-using, and developing end-use markets for packaging waste were instead contained as general conditions in the Directive’s ‘Essential Requirements’ (Annex II) and Article 6 correspondingly. Two points are immediately obvious. In common with much EU legislation, the Directive sought legislative approximation rather than harmonisation and employed banded targets and derogations to cater for the specific exigencies and capabilities of individual Member States (for an extended discussion, see Bailey, 1999a). In addition, the Directive itself only introduced legislative standards, though Article 15 states that: ‘acting on the basis of the relevant provisions of the Treaty, the Council adopts economic instruments to promote the implementation of the objectives set by this Directive.’ It is therefore apparent that although the Directive was primarily a legislative instrument, many Member States anticipated implementing its requirements using some combination of regulation and price-based measures (Brisson, 1993).

As with other directives, the Member States are bound in terms of the overall objectives set by the Packaging Directive but are entitled to determine the detailed arrangements for putting them into practice (Jordan, 1999). However, two features are common to most state measures, the introduction of economic instruments and the legal onus on industries engaged in the manufacture, distribution or sale of packaging or packaged products (the packaging chain) (Hagengut, 1997). The Directive’s flexibility has nonetheless led to strikingly different forms of legislation and implementation in the Member States, a diversity which prevents an explication of each in this paper. The two main models examined for this research, those of Germany and Britain, make a useful comparison for two reasons. First, by virtue of the fact that Britain and Germany are two of the largest EU economies, their policies have a major impact on the production of packaging waste in Europe (Hagengut, 1997). Second, the two countries have developed generally divergent styles of environmental policy implementation. British policy has traditionally promoted flexible systems of environmental governance in order to achieve objectives in a cost-effective manner (Lowe and Ward, 1998), whereas Germany has favoured precautionary environmental policies based on stringent standards and environmentally ambitious objectives (Knill and Lenschow, 1998). Although these distinctions have been partly eroded by the mutual adoption of price-based policy

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1 Although Articles 100 and 100a were amended to Articles 94 and 95 in the Amsterdam Treaty, this paper refers to the articles in force at the time the Packaging Waste Directive was negotiated.

2 The term recovery denotes the collection of packaging waste for the recouping of value. It includes composting, combustion with energy recovery, and recycling (OJEC, 1994: Article 3).
instruments in respect of packaging waste, both waste management systems remain heavily influenced by these ideological preferences (Haverland, 1999).

The British Model

According to the industry consultation paper circulated prior to the Producer Responsibility Obligations (Packaging Waste) Regulations 1997 (hereafter the Regulations), the British government was committed to three aims; a more sustainable approach to dealing with packaging waste, the implementation of the Directive, and the achievement of these goals in the most efficient and least burdensome manner (Department of the Environment, 1996). Reflecting these aims, the recovery and recycling targets established in the Regulations are the lowest permitted by the Directive and are being phased in over an interim period between 1998 and 2001. Responsibility for their attainment is shared across all sectors of the packaging chain (raw materials producers, packaging manufacturers, product manufacturers and retailers – companies collectively termed *packaging producers*). However, the Regulations exclude businesses with an annual turnover below £5 million or handling under 50 tonnes of packaging from direct recovery obligations (from 2000, the turnover threshold became £2 million). The Regulations also sanction the creation of *compliance schemes* to help affected businesses manage their recycling responsibilities (Bailey, 1999b).

In order to prove their compliance with the Regulations, producers or compliance schemes must submit evidence annually to the Environment Agency or Scottish Environmental Protection Agency (SEPA). The standard, though not only, form of evidence are certificates termed Packaging waste Recovery Notes (PRNs) produced by accredited reprocessors as they recycle or incinerate waste collected from industrial or household sources. Under this scheme, reprocessors are entitled to charge producers for PRNs according to the prevailing conditions in the recycling market, whilst prices are calculated according to packaging material (with those that are more expensive to recycle attracting a higher market price) and weight. The PRN scheme therefore operates both as a *de facto* tradable permit system and, indirectly, as an environmental charge. Its main purpose is to generate funds for investment in new recycling infrastructure without the need for direct taxes, while the behaviour of the recycling market is theoretically guided by the targets contained in the Regulations such that environmental objectives are achieved in a cost-effective manner (Department of the Environment, Transport and the Regions (DETR), 1999). The expectation is that market forces and competition between reprocessors (currently 210 are licenced by the Environment Agency) will lead to ‘optimum’ pricing of PRNs at each stage of the transition period, re-emphasising the commitment to cost-effective compliance with EU requirements.

The German Model

The Ordinance on the Avoidance of Packaging Waste of 1991 (amended in 1998) makes manufacturers and distributors responsible for the re-use and recycling of their packaging waste outside the public waste disposal system and requires retailers to set up in-store facilities for collecting used packaging (*Umweltsbundessamt*, 1991, 1998). The recycling targets established in the Ordinances are above the maxima specified in the Directive, whilst the amended legislation also provides for the imposition of mandatory deposit-refund systems for beverage containers should the national market share of re-fillable containers fall below 72% for two consecutive years (Michaelis, 1995). However, these obligations are waived for manufacturers and distributors taking part in an industry-led ‘Dual System’ for collecting, sorting and recycling used packaging, the *Duales System Deutschland* (DSD). In contrast with the British model, the DSD is a non-profit organisation which co-ordinates the collection
of all sales packaging in Germany and emphasises industry co-operation, rather than
competition, as the preferred strategy for achieving national targets (Haverland, 1999).

The DSD’s operations are financed by means of licence agreements with companies affected
by the Ordinance. The licence fees are paid by firms on each unit of packaged product and
are calculated on the basis of the packaging materials used, as well as its weight, area and
volume (see Figure 1). Payment of fees entitles manufacturers to mark their packaging with
the DSD’s Green Dot logo to identify it to consumers as participating in the Dual System
(DSD, 1998). These provisions only apply to sales packaging and separate arrangements
must be made for transport and other packaging. Although agreements to establish the DSD
were concluded between industry and the federal government prior to the Ordinance’s
introduction, commentaries note that this was the lesser of two evils for industries faced with
unstinting regulatory pressure (Haverland, 1999). In place of competition between
reprocessing companies, co-ordinating organisations were founded for each major packaging
material. Their main roles are to produce global reprocessing certificates – the Mass Flow
Verification – to raise finance for reprocessing facilities, and to act as guarantors of the Dual
System (Eichstädt et al., 1999).

![Figure 1 Green Dot Licence Fees](image)

<table>
<thead>
<tr>
<th>Item Fee in Pfennigs (including statutory VAT)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Volume item fee</strong></td>
</tr>
<tr>
<td>&lt; 50-200 ml and &gt; 3 g</td>
</tr>
<tr>
<td>&gt; 200 ml - 3 litres</td>
</tr>
</tbody>
</table>
Research Methods

Aims and Evaluation Criteria

As noted earlier, the main aim of the research was to examine the effect of environmental charges introduced to implement the Packaging Waste Directive on industry behaviour. The first element, therefore, was to determine the extent to which national packaging policies (in both cases a combination of legislative and economic instruments) have precipitated changes in corporate waste-management practices. The second was to assess the extent to which any changes are attributable to the imposition of environmental charges. The final element was to identify other benefits arising from the introduction of environmental charges, paying particular attention to the role of hypothecation.

In developing evaluation criteria, it was immediately acknowledged that environmental charges are only one of several instruments used by the Member States to implement the Packaging Directive. In both Britain and Germany the principal means of intervention have been legislative standards covering recovery and recycling. Furthermore, neither the PRN nor the Green Dot schemes are direct government taxes, nor were they designed primarily as incentive charges; in each case their chief aim is to facilitate investment in recycling infrastructure. Whilst the introduction of multiple and multiple-objective policy mechanisms makes it difficult to isolate the effects of individual measures (EEA, 2000), both economic instruments should arguably produce some indirect incentive for pollution prevention by virtue of the fact that each increases the cost of packaging production and consumption. This is particularly the case with the German model, as Green Dot fees are significantly higher than PRN certificates. In some cases, this is by orders of magnitude (see Figure 2).\(^3\) Whilst the incentive effect in both cases may be slight because this is not their main function, the link between Green Dot charges and changes in corporate waste management practice should logically be clearer than in the PRN scheme.

\(^3\) Personal correspondences with national enforcement agencies.
These suppositions were tested by studying British and German businesses affected by national measures introduced to implement the Packaging Waste Directive. The first task was to develop an analytical framework that adequately disentangled the effects of environmental charges from those of legislative standards. Because the recovery and recycling objectives of the Directive are subject to legislative standards, environmental charges play only a peripheral role in promoting these targets. However, for those policy objectives not supported by defined standards – prevention, re-use and the development of end markets for recycled packaging – the main methods of intervention have been voluntary codes of conduct (Industry Council for Packaging and the Environment (INCPEN), 1998) and environmental charges. Again recognising that the effect on polluter behaviour may be concealed by other factors, the relationship between environmental charges and these latter, qualitative objectives formed the main focus of inquiry.

**Sampling**

The selection of research techniques was primarily determined by the need for data that accurately represented the attitudes and behaviours of the 4,500 British and 17,000 German companies affected by national packaging measures. Postal surveys were therefore preferred to interviews as they enabled access to a larger and more representative sample of the overall population being investigated. The UK sampling frame was compiled from a public database of companies obligated by the Regulations (Environment Agency, 1998); however, the lack of corresponding data for Germany necessitated the compilation of an independent register from electronic business directories. Two search criteria were used to match the German sampling frame with that for Britain; first, the main sectors targeted by the Ordinance – manufacturers, distributors and retailers – were included (Hagengut, 1997; Perchards, 1998); second, companies were selected on the basis of company turnover. As the UK Regulations exempt businesses falling below certain turnover and packaging-consumption thresholds, similar filters were used for Germany. A database of 4,500 German companies was created using this method. The exclusion of smaller businesses and, inevitably, the limitations of the electronic directories used explain the discrepancy between this figure and the total number of
affected businesses. However, the imperfections in the sampling process made further analysis of respondent characteristics an essential prerequisite of the comparison.

To achieve this, respondents were requested to provide basic profile data on company turnover, number of employees, and business sector. These were then assessed by means of Chi-Square tests (Table 1). In terms of turnover, the variance between the British and German groups is largely explained by the greater proportion of German businesses with an annual turnover exceeding £1,000 million, though this difference was not statistically significant. Less variation was found in the number of employees, though firms with 100-999 employees were more heavily represented in the German sample. Finally, the most noticeable difference in terms of business sector concerned the low representation of packaging manufacturers in the German set, a factor explained by the Packaging Ordinance’s stronger emphasis on product manufacturers. Though no statistically significant differences were found between the respondent-group profiles, the variances observed were sufficient to necessitate analysis of their impact on the main survey indicators.

**Survey Methods**

A total of 1800 potential respondents (900 from each country) were selected for the survey using simple random sampling. The questionnaire targeted environmental or other managers within respondent companies with responsibility for compliance with national packaging-waste measures. All questionnaires were accompanied by cover letters stating the aims and potential uses of the study. Replies were received from 469 British firms (52.1%) and 309 Germany companies (34.3%). A filter question was used to eliminate businesses not affected by either the Regulations or the Ordinance; 96.4% of UK respondents and 76.4% of German businesses confirmed being obligated by national regulations. This reduced the number of useful responses to 450 (50.0%) and 236 (26.2%) respectively.

**Table 1. Analysis of respondent characteristics**

<table>
<thead>
<tr>
<th></th>
<th>Germany</th>
<th>UK</th>
<th>Chi-Square ($\chi^2$) and sig.</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Turnover (£ million per annum)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-49</td>
<td>151</td>
<td>275</td>
<td></td>
</tr>
<tr>
<td>50-99</td>
<td>27</td>
<td>65</td>
<td></td>
</tr>
<tr>
<td>100-999</td>
<td>36</td>
<td>79</td>
<td>$\chi^2 = 7.036$ sig. = 0.070</td>
</tr>
<tr>
<td>1,000+</td>
<td>15</td>
<td>12</td>
<td></td>
</tr>
<tr>
<td>Total</td>
<td>229</td>
<td>431</td>
<td></td>
</tr>
<tr>
<td><strong>Number of employees</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-99</td>
<td>57</td>
<td>127</td>
<td></td>
</tr>
<tr>
<td>100-999</td>
<td>119</td>
<td>252</td>
<td></td>
</tr>
<tr>
<td>1000-4,999</td>
<td>13</td>
<td>45</td>
<td></td>
</tr>
<tr>
<td>5,000+</td>
<td>8</td>
<td>24</td>
<td>$\chi^2 = 2.697$ sig. = 0.210</td>
</tr>
<tr>
<td>Total</td>
<td>197</td>
<td>448</td>
<td></td>
</tr>
<tr>
<td><strong>Packaging chain sector</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Raw materials producer</td>
<td>2</td>
<td>8</td>
<td></td>
</tr>
</tbody>
</table>
Results

The Response of British and German firms to Packaging Regulation

The proportion of UK and German respondent businesses engaged in each waste-management action is shown in Table 2. These results apparently verify the link between environmental charges and corporate behaviour, in that a significantly higher proportion of German firms confirmed they had plans to promote the reduction and re-use of packaging, as well as the purchase of packaging made from recycled materials. This difference was particularly pronounced for waste minimisation, where 57.1% of German firms reported introducing packaging reduction plans compared with only 12.7% of British companies. A similar distinction was apparent in respect of waste management targets, with German respondents reporting more ambitious objectives than their UK counterparts (Table 2). At the same time, there are indications that these differentials may diminish in the future; for example, mean UK re-use rates were 43.2% those in Germany in 1997 but should rise to 49.2% by 2001. The data nonetheless suggest it will be several years before this gap is no longer significant. Despite the concerns about the comparability of the data, few company characteristics were found to influence corporate actions significantly. The only noteworthy relationships were that German businesses with more employees and those in the product-manufacturer sector were more inclined to buy recycled packaging ($\chi^2 = 16.44$, $n = 213$, significance = 0.006).

Respondents were then asked to provide details of their response to packaging-waste charges, with particular emphasis on the following categories:

- The existence of corporate plans for waste management activities promoted by the Directive but not subject to statutory targets
- Corporate participation in each activity during 1998, measured as a percentage of total company packaging production/use
- Anticipated participation rates for 2001, the first commitment deadline for the Directive

Finally, respondents were requested to estimate the costs incurred as Green Dot or PRN charges by their companies, the aim being to correlate waste-management targets against compliance costs in order to assess the strength of any relationship between them.

### Table 2. Waste management actions and targets of UK and German respondents

<table>
<thead>
<tr>
<th>Actions</th>
<th>No. of businesses engaged in waste management actions</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>UK</td>
</tr>
<tr>
<td>Packaging manufacturer</td>
<td>4</td>
</tr>
<tr>
<td>Product manufacturer</td>
<td>141</td>
</tr>
<tr>
<td>Wholesaler</td>
<td>21</td>
</tr>
<tr>
<td>Retailer</td>
<td>22</td>
</tr>
<tr>
<td>More than one category</td>
<td>42</td>
</tr>
<tr>
<td>Total</td>
<td>232</td>
</tr>
</tbody>
</table>
According to this survey, therefore, the Ordinance has been substantially more successful than the Regulations in achieving the broader objectives set by the Directive. Not only are more German businesses engaged in active and preventative waste management, their average level of participation is also far higher. Moreover, German firms appear to be moving away from ‘end-of-pipe’ management towards waste prevention and are, by virtue of their willingness to buy recycled packaging, more involved in the development of closed-loop recycling. Whilst the re-use and recycling figures indicate that the Regulations are beginning to affect British businesses, the data suggest the transition may be lengthy.

### Relationships between Environmental Charges and Corporate Actions

As noted earlier, the assessment of relationships between environmental charges and business behaviour was achieved by correlating corporate waste-management targets against Green Dot or PRN fees (Table 3). In order to produce a standardised picture of the proportionate environmental-charge burden for larger and smaller businesses, packaging-waste charges were expressed as a percentage of company turnover. The results of this analysis revealed no significant relationships between any corporate waste-management action and the proportionate economic burden imposed by packaging charges. In order to test this finding further, corporate actions were also correlated against environmental charges without factoring to take account of company size\(^4\). This revealed some significant correlations between charges and the purchase of packaging made from recycled materials, though the maximum correlation observed was only 0.243 for the purchase of recycled packaging by German firms in 1998 (significance = 0.10, n = 91).

#### Table 3. Correlation of packaging charges and respondent waste management targets

\(^4\) The correlation of corporate responses to policy pressures in terms of proportionate economic burden is considered to be a crude method of measurement by some economists, though respondent firms did confirm this as a significant factor in determining corporate policy (also Watkins, 2000).
The second element of survey therefore failed to uncover a clear link between packaging charges and business actions, despite evidence from previous studies that show overall packaging consumption in Germany to have fallen by approximately 11% between 1991 and 1998 (Gesellschaft für Verpackungsmarktforschung, 1996). This suggests that these reductions are by-products of other factors – the stringency of the German legislation, the additional time period the Ordinance has been in operation, and a more proactive attitude amongst German firms towards environmental issues – rather than direct responses to the environmental costs imposed by the Dual System. Considering that neither scheme was designed primarily as an incentive charge, this result may not be particularly surprising. What is noteworthy, however, is the absence of a discernible difference between the responses of German and British respondents, especially considering the sizeable disparities in environmental-charge rates. The next section examines the reasons for this.

Factors Inhibiting the Incentive Effect of Environmental Taxes

The Financial Impact of Environmental Charges

As part of the research process, telephone interviews were conducted with fifteen manufacturing firms with representation in both Britain and Germany. Though these data are limited, the comments received help to explain the marginal relationship between environmental charges and business actions found in this study. One electronics manufacturer, for instance, stressed that as its annual compliance costs amounted to £30,000 compared with a turnover of £870 million, any possible savings from projects re-evaluating the design and consumption of packaging did not justify the expenditures involved. Five other respondents acknowledged that the Regulations had increased the pressure for packaging re-design but felt they were overshadowed by other business considerations (such as the logistics and marketing benefits of packaging, and the standardisation of product branding throughout Europe). This corroborates Jones’ (1999) earlier point that businesses rarely undertake major operational commitments in response to relatively minor cost pressures. Two other manufacturers claimed that as the business routinely explored all opportunities for packaging ‘optimisation’, neither the Regulations nor economic instruments

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5 Though data from the Gesellschaft für Verpackungsmarktforschung show an increase in packaging consumption from 11.51 million to 11.63 million tonnes between 1997 and 1998.
had influenced their decisions. The British Retail Consortium’s (BRC) submission to a review of the UK Regulations in 1998 echoes these sentiments (BRC, 1998: 3-6):

Many retailers have long been doing all they can to encourage recycling and minimise packaging use. Examples include provision of banks on car parks; specification of recycled materials in packaging; packaging minimisation programmes; increased use of reusable packaging as demonstrated by closed-loop reusable schemes; and heavy investment in equipment for recovery of backdoor packaging waste. However, packaging is necessary in terms of product protection and health and safety considerations. Reuse will occur if there is an economic benefit to it.

The foremost explanation for the limited relationship between environmental charges and polluter behaviour is therefore that packaging-waste charges are simply too low in this instance to encourage major shifts in business behaviour. This suggests that the question is simply one of price elasticity and that, if policy-makers so desire, environmental charges can be raised until abatement incentives are created (Pearce and Turner, 1992). However, whilst the downward trend in packaging consumption in Germany partly corroborates this assertion, the recycling costs already incurred by German industry may make it politically unfeasible to raise charges further. Jacobs (1991) makes a similar point, noting that whilst it was possible practically to eliminate many ozone-depleting substances because affordable substitutes were available, the demand for petrol has risen despite substantial increases in real prices. Higher charges may also encourage more firms to disregard the law, making enforcement costs a paramount issue. A further consideration for the PRN system is the fact that market forces are the sole mechanism currently determining environmental-charge rates. As recycling markets respond to numerous influences – regulatory regimes, producer willingness-to-pay, competitive pressures, commodity prices and general macro-economic conditions – it is dangerous to over-stylise the relationship between environmental charges and polluter behaviour. Whilst these findings in no way negate the incentive potential of environmental taxes, the House of Commons Select Committee for Environment, Transport and Regional Affairs ((HCSCETRA), 1999 par. 11) arrived at similar conclusions about the incentive properties of the UK Landfill Tax:

There is no evidence that the tax has encouraged any change in the behaviour of waste producers, and has had no impact in diverting waste away from landfill ... the Landfill Tax has so far had very little effect on disposal options ... it is having a minimal impact at the current levels ... and that is unlikely to change when the tax goes up to £10 a tonne. (At the time, Landfill Taxes had recently been increased from £7 to £10 per tonne for active waste).

Cost Diffusion

In addition to the direct incentive effect, it is recognised that firms subjected to environmental charges may attempt to push, or diffuse, some of their additional costs through the economy in the form of price increases (van den Bergh, 1996). The policy intention here is that such actions will disseminate the polluter and user-pays principles to other links in the ‘pollution chain’, thereby aiding the re-internalisation of environmental-impact costs (Jacobs, 1991). The nature of cost diffusion resulting from packaging-waste regulation was assessed by means of a follow-up survey. This showed that whilst neither respondent group has sought to diffuse all environmental-charge costs, 58.9% of German respondents have increased the price of packaged products because of packaging-waste fees, compared with 32% of British

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6 This point was underlined by the recent blockades of oil refineries in Britain and France. Here, taxes on demand-inelastic products became unacceptably high before they prompted discernible reductions in demand.
companies (Mann Whitney 27220.5, significance 0.000). This suggests, logically enough, that affected industries will attempt to diffuse costs more actively as environmental-charge costs rise. However, the extent of any resulting reduction in demand for polluting products or processes from other sectors of the pollution chain again depends on the scale of the original charge and, consequently, its financial impact on each indirectly affected party. Considering that direct packaging charges are too low to induce changes in the behaviour of directly affected industries, any indirect incentives in this case will be slight. Although 41.5% of German firms indicated they would inform their customers of the reasons for price increases (an action which spreads the message of producer and user-pays as well as its impact), this view was shared by only 21.1% of British respondents (Mann Whitney = 29893.5, significance = 0.000).

**Other Effects of Environmental Taxes: Revenue Hypothecation**

While both the direct and indirect incentive effects of packaging-waste charges have been identified as relatively marginal, it was noted earlier that the primary function of the Green Dot and PRN mechanisms is the generation of finance for investment in recycling infrastructure (Eichstädt *et al.*, 1999; Bailey 1999b). Initially it seems that both schemes have stimulated significant increases in reprocessing capacity. Of the DSD’s sales income of 4.17 billion Deutschmarks (DM) in 1998 (£1.46 billion), DM3.90 billion (£1.37 billion) was spent on waste management and recycling services (DSD, 1999a). In 1998, an estimated £56 million of PRN revenue was made available to UK reprocessors, though the amount actually spent on developing recycling networks remains a matter of conjecture (Advisory Committee on Packaging, 1998). As a result, recycling capacity is predicted to increase in Britain by 21.8% between 1998 and 2001, and by 27.2% when Energy from Waste is taken into account (Table 4). More tangibly, recycling in Germany increased six-fold between 1992 and 1998, with an overall recycling rate in 1997 of 64% (Table 5).

Table 4. Predictions of UK recovery and recycling 1998-2001

<table>
<thead>
<tr>
<th></th>
<th>Expected performance (thousands tonnes)</th>
<th>% increase</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1998 actual</td>
<td>2001 estimate</td>
</tr>
<tr>
<td>Paper</td>
<td>1,888</td>
<td>1,921</td>
</tr>
<tr>
<td>Glass</td>
<td>658</td>
<td>730</td>
</tr>
<tr>
<td>Aluminium(^a)</td>
<td>15</td>
<td>53</td>
</tr>
<tr>
<td>Steel(^a)</td>
<td>183</td>
<td>235</td>
</tr>
<tr>
<td>Plastics</td>
<td>126</td>
<td>212</td>
</tr>
<tr>
<td>Wood</td>
<td>n/a</td>
<td>350</td>
</tr>
<tr>
<td>Total recycling</td>
<td>2,870</td>
<td>3,501</td>
</tr>
<tr>
<td>EFW and composting</td>
<td>448</td>
<td>726</td>
</tr>
<tr>
<td>Total recovery</td>
<td>3,328</td>
<td>4,227</td>
</tr>
</tbody>
</table>

\(^a\) Aluminium and steel reprocessors can reprocess 375,000 and 6 million tonnes annually respectively should materials become available. Currently 75,000 tonnes is reserved for aluminium packaging and 144,000 for steel.

\(^7\) Calculated on the number of PRNs traded in 1998 and average PRN prices over the period (DETR, 1999a).
Both hypothecation schemes have experienced major difficulties, however. In 1993, the Dual System faced financial crisis; these were attributed to the following factors:

- Large numbers of firms using the Green Dot without paying licence fees (free-riding);
- Disposal firms taking advantage of the pressure to set up the Dual System by charging exorbitant collection fees;
- Green Dot fees were too low to finance the development of both reprocessing and collection; this led to the mass export of waste to other Member States, often for landfilling (Michaelis, 1995);
- The collapse of the plastics reprocessing guarantor because of lack of finance.

To avert the imminent collapse of the DSD, a consolidation plan was developed by the federal Environment Minister. This converted the DSD’s debts into long-term loans, created new materials guarantors to replace those that had folded, and increased Green Dot fees substantially (Haverland, 1999), actions which allowed the DSD to stabilise its finances and to invest in new facilities such that 81% of reprocessing requirements for packaging waste can now be met without the need for exports (DSD, 1998). One consequence of the federal government’s desire to protect its ambitious environmental programme, however, has been that the DSD is now the most expensive recycling system in Europe, a situation critics have condemned as unjustified in relation to the environmental benefits gained (Staudt, 1997; Flanderka, 1998). Even so, the main forces behind the DSD’s ability to reconsolidate were the immense political capital vested in it, which made its demise politically unpalatable, the readiness of the federal government to intervene directly in pursuit of environmental targets, and the spur to action that threats to reinstate the Ordinance’s take-back and deposit requirements gave to industry (Eichstädt et al., 1999).

At the time of writing, it remains unclear whether the PRN system’s difficulties have been satisfactorily resolved. These stemmed principally from the scheme’s market-led structure.
and, in particular, from market distortions exploited by some reprocessors. Three key issues were identified in the first review of the Regulations in 1998 (DETR, 1998). The first was that free trading in PRNs allowed organisations unconnected with the Regulations to acquire certificates so as to speculate on their value in the knowledge they were needed by packaging producers to prove their compliance with the Regulations. Second, many reprocessors were charging obligated businesses for PRNs even where producers collected and delivered packaging waste themselves. This discouraged voluntary recovery and recycling systems, as the more economic option was simply for businesses to acquire PRNs on the open market. However, the biggest concern was that PRN revenue was not being invested in recycling infrastructure as the government intended, but had instead become a source of windfall profit for some reprocessors (Advisory Committee on Packaging, 1998).

In response to these concerns and evidence that reprocessing capacity was not expanding quickly enough to meet EU targets (Table 6), the UK government has imposed the following requirements:

- Reprocessors must provide annual returns to the Environment Agency detailing the proportion of PRN revenue spent on infrastructure and end-use markets;
- The issue of PRNs must be restricted to obligated parties or those representing them;
- Businesses collecting waste for reprocessing must have first refusal on the PRNs produced (Institute of Wastes Management, 1999).

### Table 6. UK recovery and recycling 2001, thousands tonnes

<table>
<thead>
<tr>
<th></th>
<th>Estimated packaging 2001</th>
<th>Reprocessing capacity 2001</th>
<th>Capacity required 50% recovery</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Recycling</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Paper</td>
<td>4,308</td>
<td>1,921</td>
<td>2,154</td>
</tr>
<tr>
<td>Glass</td>
<td>2,200</td>
<td>730</td>
<td>1,100</td>
</tr>
<tr>
<td>Aluminium</td>
<td>111</td>
<td>53</td>
<td>56</td>
</tr>
<tr>
<td>Steel</td>
<td>735</td>
<td>235</td>
<td>368</td>
</tr>
<tr>
<td>Plastics</td>
<td>1,912</td>
<td>212</td>
<td>956</td>
</tr>
<tr>
<td>Wood</td>
<td>1,300</td>
<td>350</td>
<td>650</td>
</tr>
<tr>
<td>EfW Recovery</td>
<td></td>
<td>726</td>
<td></td>
</tr>
<tr>
<td><strong>Total packaging</strong></td>
<td><strong>10,566</strong></td>
<td><strong>4,227</strong></td>
<td><strong>5,284</strong></td>
</tr>
<tr>
<td><strong>Shortfall</strong></td>
<td></td>
<td></td>
<td><strong>1,057</strong></td>
</tr>
</tbody>
</table>

*Source: DETR (1999, p.21).*

Whilst the market-led approach has not been abandoned, the British government clearly deemed it necessary to increase market regulation in order to ensure PRN trading operates towards policy objectives as well as private-sector profit. The British and German
experiences with packaging-waste charges therefore demonstrate some of the practical benefits and pitfalls associated with the hypothecation of environmental charges. Although both schemes have stimulated major expansions in recycling, the British government’s arguably excessive faith in the compatibility of market-led hypothecation and environmental protection allowed distortions to develop within the PRN system. The practical lessons in terms of refining the blend of market incentives and constraints are especially pertinent since similar mechanisms are proposed for implementing the End-of-life Vehicles, Electrical and Electronic Equipment, and Landfill directives (Cooper, 1999). Equally, the German model demonstrates the advantages of co-ordinated hypothecation and the economic inefficiencies that can result from the exclusion of market forces from environmental-charge systems.

The policy challenge therefore lies in devising arrangements that combine environmental and economic prerogatives within a coherent system of hypothecation. Equally, the adoption of industry-led schemes, where environmental charges are allocated between economic actors without the need for potentially inflexible government taxes, is a promising model for the future development of price-based regulation. Neither the Dual nor PRN system has proven ideal in achieving their objectives but this reflects the complexities of market-based forms of environmental charge and the fact that knowledge of the technique is still evolving. Finally, whilst hypothecation has produced more readily identifiable environmental impacts in this instance, the British and German packaging-waste systems have both employed hypothecation as an end-of-pipe measure which is reliant on pollution occurring to release revenue for environmental projects. In order further to enhance the environmental potential of hypothecation within industry-led schemes, policies should also incorporate mechanisms for re-directing revenue towards activities that challenge the production of targeted pollutants, rather than focusing exclusively on remedial measures or matters of social redistribution. A variant of this approach seems to have been adopted as part of the UK’s Climate Change Levy strategy, where energy intensive industry sectors will be offered 80% reductions in the charge in return for signing up to negotiated agreements for setting energy targets that meet government criteria. The use of hypothecation to assist this process may serve as an important element in developing positive incentives for improved environmental protection.

Conclusions

Ten years after the first of their Blueprint series, Pearce and Barbier (2000) reflect on the changes that have occurred in environmental policy since its publication. They conclude that environmental economics has become a common language for scientists and policy-makers and that its opponents are fewer because many have realised some virtue in the economic approach to environmental policy. However, they concede that putting the economic message into practice has been more difficult than anticipated in many cases, as it has necessitated changing (often unsustainable) institutions, cultures and policies that have been built up over many years. As an addendum they warn that analysis should focus on applying environmental economics rather than on reconstituting the problem.

Whilst generally supporting this position, this paper has highlighted some of the practical challenges involved in putting environmental economics into practice. Recognising the obvious dangers of over-generalising from one study, and particularly where multiple legislative and economic instruments have been employed, the implementation of the Packaging Waste Directive reveals some broader lessons about the relationships between
environmental charges and polluter behaviour. In particular, it has been argued that the incentive effect of economic instruments should not be over-stylised in situations where polluter responses are guided by multiple market factors, especially in the case of price-inelastic products or processes. The sustained decline in packaging consumption in Germany therefore appears to be explained by the use of multiple policy instruments, rather than by the application of environmental charges.

At the same time, the hypothesisation of packaging-waste charges was found to have generated significant revenues for re-investment in the development of recycling infrastructure. In both the German and British schemes, this has been achieved without the need for direct environmental taxes and their attendant restrictions on government expenditure priorities. That said, the study has identified areas where market measures and regulatory constraints must be carefully balanced in order to achieve both environmentally and economically efficient policy outcomes. However, in order further to enhance the sustainability benefits gained from industry-led hypothesisation, it is suggested that schemes should also incorporate measures to prevent the production of targeted pollutants alongside those ameliorating their effects. The proposed rebates under the UK’s Climate Change Levy for industries committing to energy saving targets certainly appears to adopt this latter model. But even where hypothesisation arrangements are aimed mainly towards remedial measures, their prudent utilisation could provide a useful addition to the incentive benefits accruing from the use of environmental charges or taxes.

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References


