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Don Fullerton
Andrew Leicester
Stephen Smith

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ABSTRACT

This chapter provides an overview of key economic issues in the use of taxation as an instrument of environmental policy in the UK. It first reviews economic arguments for using taxes and other market mechanisms in environmental policy, discusses the choice of tax base, and considers the value of the revenue from environmental taxes. It is argued that environmental tax revenues do not significantly alter economic constraints on tax policy, and that environmental taxes need to be justified primarily by the cost-effective achievement of environmental goals. The chapter then assesses key areas where environmental taxes appear to have significant potential – including taxes on energy used by industry and households, road transport, aviation, and waste. In some of these areas, efficient environmental tax design needs to make use of a number of taxes in combination – a "multi-part instrument".

Don Fullerton
Department of Finance
University of Illinois
340 Wohlers Hall
1206 South Sixth Street
Champaign, IL 61820
and NBER
dfullert@illinois.edu

Andrew Leicester
Institute for Fiscal Studies
7 Ridgmount Steet
London WC1E 7AE, UK
andrew_l@ifs.org.uk

Stephen Smith
Department of Economics
University College London
Gower Street
London WC1E 6BT, UK
stephen.smith@ucl.ac.uk

1. Introduction

Growing concern about climate change has brought environmental issues to the forefront of the policy agenda in many European countries. In addition to the substantial scientific literature assembled under the auspices of the Intergovernmental Panel on Climate Change, the October 2006 *Stern Review of the Economics of Climate Change* argued strongly for immediate and urgent action to mitigate the potential costs of global climate change. Taxes, charges, tradable permits and other economic instruments can play an important role in achieving cost-effective control of greenhouse gas emissions, but their potential scale and revenue contribution raise many wider economic and fiscal policy implications. A number of European countries introduced carbon taxes during the 1990s, though a proposal for an EU-wide carbon-energy tax was ultimately unsuccessful. More recently, attention has shifted to emissions trading, and the EU Emissions Trading Scheme, introduced in 2005, is the most substantial application to date of this approach.

In the UK, a number of tax measures have been implemented primarily with environmental objectives in mind. They have included three new national environmental taxes, on landfill, industrial energy use (the climate change levy) and the extraction of aggregates (quarry products). Taxes on motor fuels and the annual vehicle excise duty have both been restructured, with differential rates reflecting the different environmental attributes of fuels and vehicles. In London, the transport authority has introduced a congestion charge for vehicle use in the central area. In addition to these explicitly environmental tax measures, a wider range of areas of tax policy-making routinely include some discussion of environmental issues.

The increasing use of environmental taxes, emissions trading and other economic instruments has been partly driven by a recognition of the limitations of conventional environmental regulation. To make any serious impact on some of the major environmental problems now facing policy-makers – acid rain, global warming, traffic congestion – environmental policy cannot be approached purely as a technical issue, to be resolved merely by requiring the use of specified abatement technologies and setting emissions limits on large firms. Extensive and far-reaching changes to existing patterns of production and consumption will be needed, and these changes will inevitably entail substantial economic costs. The search for instruments capable of minimising these costs, and of achieving behavioural changes across all sectors, has led policy-makers to pay much closer attention to the potential for incentive-based environmental regulation, that is, through economic instruments.

This approach to environmental policy has the potential to generate additional government revenues – in the form of environmental tax receipts, or the proceeds of auctioned emissions trading allowances. This calls for a much closer interaction between environmental policy and tax policy than in the past. At one level, the new government revenues that could be generated may provide an opportunity for tax reform. At a deeper level is an issue about how far the availability of environmental taxes alters the constraints and costs of current tax policy, in terms of the distortionary impact of existing taxes on labour and capital markets. Here, the issues are more complex. ‘Packaging’ environmental tax reform with offsetting reductions in taxes on labour income or the payroll taxes paid by employersmay have political attractions, but the fiscal benefits of this type of tax substitution are much more contentious.

This paper provides an overview of key economic issues in the use of taxation as environmental policy. Following this introduction, the paper has two main parts. Part A discusses economic principles of environmental taxation, reviewing the arguments for using taxes and other market mechanisms in environmental policy, the efficient design of environmental taxes, and the fiscal value of the revenue contribution from environmental taxes. In what sense – if at all – would an environmental tax reform provide a ‘double dividend’, in the form of a less distortionary fiscal system as well as a cleaner environment? Part B of this paper then applies these basic principles to four specific environmental tax areas - energy, road transport, aviation and household waste. The first two of these – general taxes on energy and taxes on road transport – perhaps have the greatest revenue potential, but in all four areas taxes or other similar instruments could make a significant contribution to efficiency in environmental policy.

Before embarking on the main analysis of the paper, we have some preliminary observations of a general nature about this field of tax policy, and about the approach we have adopted:

Firstly, the focus of the paper is primarily on the economic aspects of environmental taxes. In addition to economic considerations, however, both politics and public opinion will play a crucial role in determining the scale of action needed, and the range of acceptable measures. This is a fast-changing landscape, and we have tried, as far as possible, to avoid constraining the analysis by our own personal speculations about what measures would be publicly or politically acceptable in current circumstances.

Secondly, technology is developing rapidly, and is a key issue in determining the types of environmental taxes that are practicable. For example, technological advances that make it easier and cheaper to measure emissions directly may open up new possibilities for direct, targeted emissions taxes, based on measured emissions. Also, as viable technologies are developed for large-scale carbon capture and storage, it may be necessary to replace straightforward taxes on energy use with more complex and targeted taxes that provide appropriate incentives for the use of carbon capture.

Thirdly, environmental policy choices depend on some key value judgements as well as objective data. For example, a central issue in deciding whether the costs of action to curb greenhouse gas emissions are justified by the environmental benefits is the weight to be given to the interests of future generations. The *Stern Review's* conclusions on the scale of the damages from global warming, which are much higher than many earlier economic estimates, reflect not only the accumulating scientific evidence about the severity of climate change, but also a judgement that the interests of future generations should be weighted more heavily than in much of the literature.

Finally, while the primary focus of the paper is on national tax policy, a key international dimension to some major areas of environmental policy-making cannot be neglected. For energy and carbon, in particular, the relevant externalities are global in their impact – all greenhouse gases emitted in any country have similar global effects. This means that effective policy cannot be implemented by a single country, and that national policies have to be formulated in the context of wider international policy developments.

Part A: Principles

2. Environmental regulation: instrument choice

From the perspective of environmental policy, the case for using environmental taxes, emissions trading and other economic instruments is primarily a matter of efficiency.[1] In comparison with 'conventional' regulatory policies based on technology mandates or emissions standards, economic instruments may be able to reduce the costs of achieving a given level of environmental protection (or, alternatively, can achieve a greater environmental impact for a given economic cost). Not all environmental problems, however, are best tackled in this way, and other approaches, including various forms of command-and-control (CAC) regulation, may be preferable in some cases.[2] Likewise, different economic instruments have various advantages and disadvantages, and the balance between these will vary from case to case.

2.1 Advantages of environmental taxes and other economic instruments

[1] The potential use of environmental taxes is assessed by, among others, Smith (1992), OECD (1993, 1996), Bovenberg and Cnossen (1995), Fullerton (2001), Bovenberg and Goulder (2002), Stavins (2003), and Newell and Stavins (2003). The seminal work is Pigou (1920).

[2] Bohm and Russell (1985) and Fullerton (2001) also review the goals and objectives of environmental policy, and they discuss how the tradeoffs among these goals might imply when to use incentives, direct regulation, or other policies.

(i) 'Static' efficiency gains through reallocation of abatement

Where the costs of pollution abatement vary across firms or individuals, economic instruments such as environmental taxes and emissions trading have the potential to minimise costs, as discussed in Box 2.1, for two reasons. First, other policy instruments cannot fully differentiate between polluters with different marginal costs of abatement, and thus may require some to undertake abatement with high costs. Economic instruments provide each polluter with incentive to abate in all of the least-expensive ways, thereby achieving a given level of abatement at lower total abatement cost. Second, economic instruments can side-step the need for the regulatory authority to acquire detailed information on individual sources' abatement costs, which lowers the authority's administrative costs. Newell and Stavins (2003) find that the cost of abatement using command-and-control regulation can be several times the minimum cost achieved by using an emissions tax.

(ii) Dynamic innovation incentive

Regulatory policies which stipulate that polluters must use particular technologies or maintain emissions below a specified limit may achieve compliance but do not encourage polluters to make further reductions below this specified limit. Indeed, where regulations are negotiated on a case-by-case basis, polluters may fear that any willingness to exceed requirements may simply lead the regulator to assign the firm a tougher limit in future. By contrast, environmental taxes provide an ongoing incentive for polluters to seek to reduce emissions, even below the current cost-effective level, since the tax applies to each unit of residual emissions, creating an incentive to develop new technologies that have marginal cost below the tax rate (see e.g. Fischer *et al*, 2003).

(iii) Robustness to negotiated erosion ('regulatory capture')

Efficient implementation of regulations requires firm-by-firm negotiation of individual abatement or technology requirements. As noted above, CAC regulatory policies should require different amounts of pollution abatement from different firms, to minimise total abatement costs. The regulator is dependent on the regulated firms for information about their abatement costs, however, and is liable to be drawn into dialogue and negotiation with the firm. The regulated firms, in turn, then control a key element in the process by which regulatory policies are set, and may be able to extract a price from the regulator for their co-operation, in the form of less stringent abatement targets, or other changes that work to their advantage.

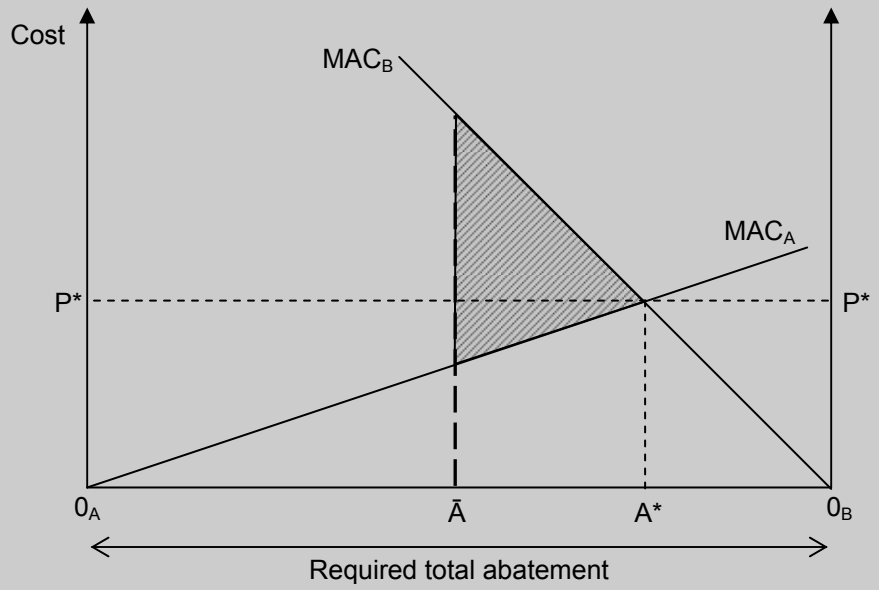
In contrast, uniform environmental taxes achieve a cost-effective distribution of abatement, taking account of the different abatement costs of individual firms, while taking a robust, non-negotiated form. All firms face the same pollution tax rate. The regulator has no need to consider the circumstances of individual firms, and thus individual polluters have little scope to negotiate more favourable terms. The risk is thus substantially reduced that this process of negotiation would erode the environmental effectiveness of the policy.

(iv) Revenue potential

Environmental taxes and auctioned tradable permits raise revenues, as a result of the payments made on each unit of residual emissions. The extent to which these revenues should really count as a further ***benefit*** of the use of environmental taxes or emissions trading has been controversial, and we defer discussion of this so-called "double dividend" to section 4.

From a fiscal policy perspective concerns are also sometimes raised about the stability and predictability of environmental tax revenues, and, in particular, their erosion as a result of the behavioural responses of polluters. We suspect this problem has been greatly overstated. The revenue from all taxes is affected by behavioural responses, and environmental taxes based on inelastically-demanded commodities such as energy might well be less affected by behavioural responses than other tax bases.

Box 2.1 The static efficiency gain from the least-cost pattern of abatement compared with uniform abatement when two types of polluter differ in abatement costs



Point A* represents the least-cost division of a given total abatement requirement between two groups of polluters with different marginal abatement costs, represented by the schedules MAC$_A$ and MAC$_B$, measured from the origins 0$_A$ and 0$_B$ respectively. The pollutant is assumed to be "uniformly mixed", so that the environmental benefits are a function only of the total abatement achieved, and not of how this is divided between the sources. Economic instruments such as taxes or trading should achieve point A* (eg through emissions trading in a competitive market, with equilibrium allowance price equal to P*, or through an emissions tax set at a rate P* per unit of emissions). If, instead the informational limitations of command-and-control regulation compel the regulator to give the two types of polluter equal abatement requirements (point Ā in the diagram), higher total abatement costs will be incurred, shown by the shaded area.

A large number of empirical studies have used data on marginal abatement costs for a range of different sources to compare the costs of achieving a given abatement outcome using uniform and least-cost regulation. The cost savings are a function of differences in marginal abatement costs between sources. Where these are large, the efficiency saving from the least-cost pattern of abatement is correspondingly large (Tietenberg, 1991; Newell and Stavins, 2003).

2.2 Disadvantages of environmental taxes and other economic instruments

Economic instruments such as environmental taxes have, however, a number of identifiable drawbacks and limitations that may be sufficiently important to rule out their use in particular applications.

(v) Geographically-varying damage

If pollution damage varies with the source of emissions, then a uniform pollution tax is liable to result in inefficiency, and source-by-source regulation may be needed to achieve a more efficient outcome (see, e.g. Helfand *et al*, 2003). In principle an environmental tax need not be constrained to apply the same rate to all sources, and could thus achieve the efficient outcome through appropriately differentiated tax rates. However, once the tax rate has to be set individually for each source, the tax may become exposed to lobbying influence from the regulated firms. Also, some forms of environmental tax may have to apply at uniform rates, even where damage is known to differ between locations. Thus, for example, environmental taxes on pollution-related inputs may be unable to differentiate between sources, because of the difficulty of preventing resale of inputs to firms with more-damaging emissions.

(vi) Incompatibility with firm decision-making structures

Except in very small firms, many business decisions may be efficiently decentralised. Specialised divisions of the firm may be given responsibility for decisions requiring particular expertise or detailed information, subject only to general instructions or guidelines from the centre. This decentralisation represents an efficient division of labour, but it implies that all aspects of the firm's operations are not necessarily taken into account. The internal organisation of the firm needs to be designed so that related decisions are grouped together, while unrelated decisions are separated.

For environmental taxes to induce efficient polluter responses, firms must draw together information on both technology choice and tax payments. Firms considering whether to undertake more pollution abatement need to balance the marginal tax savings against the marginal costs of abatement. This type of interaction may not otherwise be a high priority in the internal organisation of the firm, and may require significant changes to the decision-making structure of the firm so that tax and pollution-control technology decisions are taken together. Restructuring the firm so that such interactions can take place may be costly, and may well not be worth doing if the tax at stake is small. Firms may not, therefore, respond at all to 'small' environmental taxes, and conventional regulatory measures may be more effective in terms of both abatement costs and decision-making costs.

(vii) Damaging avoidance activities

Sometimes the consequences of an environmental tax may be adverse, if those subject to the tax respond in a way that is more damaging to the environment than the taxed emissions. For example, a tax on toxic waste may provide a powerful incentive to reduce waste, but it may also induce illegal dumping or burning.[3] Even if the overall amount of such dumping is "small", any amounts of toxic waste may be dangerous. Per unit, this waste can have much higher social costs when dumped than when taken to a proper disposal facility, and the net environmental effect of a policy that reduces total waste but leads to some dumping may be negative. In Section 3 we discuss alternative incentive-based methods, such as a "two-part instrument", that may be able to avoid these undesirable side effects.

(viii) Distributional effects

As described throughout this chapter, environmental taxes may apply to transport, carbon content of fuels, or energy generally. Yet a high fraction of low-income household budgets are spent on electricity, heating fuel, and transportation. Thus environmental taxes are often regressive. To make matters worse, the gains from environmental protection may accrue to high-income households who have the most "willingness to pay" for that public protection. A clean environment may be a luxury good. Environmental policy reforms must be careful to use a package of changes that account for and offset these distributional effects. However, this distributional problem is not specific to environmental taxes; the same problem arises with mandates that require generators to add expensive scrubbers, or car manufacturers to add expensive pollution control equipment.

(ix) Concerns about international competitiveness

Taxes on industrial inputs increase the costs of production. Where domestic output competes with products of foreign producers not subject to similar environmental taxes, the impact on the competitive position of domestic firms may be a concern. These issues have been particularly prominent in discussions of taxes on industrial energy inputs, as discussed in section 5 below.

2.3 Other relevant considerations in instrument choice

(x) Administration and enforcement costs

[3] Government monitoring and enforcement activity is quite low. For instance, the U.S. EPA fined only 200 firms in 1995. Estimates of firm compliance vary widely: Magat and Viscusi (1990) find that, despite low enforcement activity, pulp and paper mills complied with environmental regulations about 75 percent of the time between 1982 and 1985. The U.S. General Accounting Office (1990) finds that only 200 of 921 polluters thought to be in compliance actually were (Cohen, 1999).

Both environmental taxes and conventional regulations require mechanisms for administration and enforcement. The relative costs of these arrangements should be taken into account in choosing between the different instruments (Bovenberg and Goulder, 2002). Generalisations are difficult, but a few points are worth noting.

First, a pollution tax may require counting tons of emissions, whereas a design standard simply requires authorities to confirm the use of a particular kind of pollution control equipment. Government inspectors can easily check that the plant has a working scrubber, but for some kinds of emissions, they may have much difficulty trying to confirm the exact number of tons to be able to collect a tax or permit price. In some cases, the goal of monitoring and enforcement might be met more easily by some kinds of CAC regulations.

Second, a general principle of taxation is that a tax can readily be imposed upon any market transaction such as the sale of a final good or service, because the invoice can be verified by the other party to the transaction. Similarly, eligibility for a subsidy can be verified for clean market inputs such as the use of labour, capital, or legal disposal, or the purchase of forest-conserving technologies or abatement technologies. Problems arise with an environmental tax because the producer enters no market transaction for deforestation, dumping, or emissions. Trees can be cut without any record that they ever existed. Illegal waste can be dumped at midnight. Emissions are self-reported. Without expensive audits, they are relatively easy to hide.

Third, however, excise taxes on inputs may be an inexpensive way of regulating polluting processes that use these inputs. Unlike other forms of environmental regulation, input taxes do not require direct contact between the regulator and polluters. The number of polluting sources does not, therefore, affect the costs of administration and enforcement. The incentive is transmitted through the excise tax levied on the production or sale of the input. With few producers, this tax will be comparatively cheap to operate. The excise duties levied on mineral oils are a case in point; the number of petrol companies is small, and their activities are tightly controlled and well documented.

(xi) Effect on attitudes and perceptions
Environmental taxes may have effects on individuals' attitudes and perceptions that may affect the environmental outcome either positively or negatively. For example, it is sometimes suggested that imposing an environmental tax may have a particularly large effect on taxpayer behaviour, because it "signals" and encourages "green" behaviour. On the other hand, the effect of a small environmental tax could be adverse, if taxpayers believe that paying the tax legitimises their polluting behaviour.

2.4 Environmental taxes versus *emissions trading*

Under conditions of certainty, the economic properties of emissions taxes and tradable emissions permits are very similar, and from a broad fiscal policy perspective the two instruments are largely equivalent. If an environmental tax set at rate per unit of emissions T leads to an emissions level Q, then alternatively regulating the same problem by issuing a quantity Q of tradable emissions permits will lead to a permit price per unit of emissions T, if the permit market is competitive. The level and pattern of pollution abatement will be the same under the two instruments, and the abatement cost incurred by firms will be the same.

These equivalences hold, regardless of whether the permits are distributed free, or sold (for example through an auction). In either case, the value of the last permit used is given by the abatement cost that would otherwise be incurred, and this is given by the marginal abatement cost at emission level Q, which is T per unit. The value of tradable emissions permits, therefore, is independent of the way in which the permits are distributed, again so long as the permit market is competitive. In addition, where permits are sold in a competitive auction, the revenue yield will be the same as the tax revenue that would be collected from the equivalent environmental tax.

The implication of the above discussion is that, under conditions of certainty, emissions tax and tradable emissions permits are close substitutes as policy instruments. They have broadly the same environmental and fiscal properties, and the policy choice between the two instruments can be made on the basis of other considerations, such as the administrative cost of the two forms of regulation or the competitiveness of the permit market.

This equivalence does not however hold where the regulator faces uncertainty about polluters' abatement costs and has to determine the tax rate or the quantity of permits to be issued, without accurate knowledge of the abatement costs. Neither instrument is unambiguously superior in this situation. Taxes that set a price for emissions and trading schemes that set a quantity have opposing strengths and weaknesses.

An environmental tax cannot guarantee a particular environmental impact; polluters' behavioural responses may be less, or more, than expected. In cases where the precise achievement of an environmental target is a high priority, this may be an important drawback of environmental taxes, and quantity instruments such as emissions trading may be preferred. For example, some pollution problems may exhibit a threshold beyond which environmental damage per unit of emissions rises sharply. On the other hand, while emissions trading guarantees that emissions will not exceed the quantity cap, it does so at uncertain abatement cost. Some abatement measures might be much more costly than the resulting environmental benefits. In such cases, environmental taxes can insulate polluters from the risk of excessive abatement costs. The tax rate per unit of emissions places an upper limit on the unit abatement cost to be incurred, and if abatement turns out to be more costly per unit than the tax, firms can simply pollute and pay the tax rather than paying for costly abatement. For this reason, Pizer (2002) finds that a carbon tax might be preferred to quantity regulation of carbon emissions.

Weitzman (1974) compares the relative merits of "price" and "quantity" regulation under uncertainty, and concludes that the one likely to perform better will depend on the relative slopes of the marginal abatement cost and marginal damage cost schedules (i.e. the rates at which marginal abatement costs and marginal pollution damage change when emissions differ from the optimum). This is an empirical matter, and will vary from case to case. Emissions taxes (or other instruments that involve the authorities setting a price for emissions) will on average get closer to the optimal outcome if marginal abatement costs increase with extra abatement more rapidly than marginal damage costs increase with extra emissions. Quantity instruments such as emissions trading will perform better if the reverse is true (i.e. if marginal damage costs are more steeply-rising than the marginal costs of pollution abatement).

Developing this line of argument further, Roberts and Spence (1976) observe that a combination of price and quantity regulation may perform better under uncertainty than reliance solely on one or other approach. An emissions trading system with upper and lower "safety valves" (setting a high price at which the authorities would be willing to issue additional permits and a low price at which the authorities would buy back permits) might perform better than a single fixed quantity cap on emissions. This could be implemented in various ways, and one possibility is that a (small) emissions tax could be used to set a floor to the marginal incentive for abatement.

Market efficiency

Where markets are created (as with emissions trading), these need to be low-cost and competitive. If pollution abatement is to be allocated efficiently between firms, all should face the same marginal incentive for abatement. If transactions costs or monopoly power in allowance markets drive a wedge between the marginal abatement cost of allowance buyers and sellers, some potential efficiency gains will be foregone.

Allowance allocation

The allocation of emissions trading permits may appear to offer an attractive degree of flexibility in policy implementation. In broad terms, a choice is between free distribution of allowances to firms,

based typically on their past output or emissions levels (referred to as “grandfathering”) or some form of sale or auction. In formal terms, this issue is equivalent to the choice between two ways in which the revenue from an environmental tax could be employed – either returned to polluters on a basis unrelated to their current emissions (which may be seen as equivalent to a form of grandfathering), or used as revenue contributions to the general exchequer (corresponding to the auction case). In practice, emissions trading systems have generally involved grandfathering, and have made little use of the potential for auctioning, despite the economic case for doing so (see section 4.2). By contrast, governments appear to have been able to use environmental tax revenues flexibly. Only in certain examples such as the NOx tax in Sweden have tax revenues been returned to firms in a way very similar to grandfathering.

Price versus quantity regulation of exhaustible resource depletion
A further consideration in choosing between environmental taxes and emissions trading arises in the case of climate change policies. It has long been recognised that using price-based instruments to regulate environmental problems resulting from the use of an exhaustible resource (e.g. fossil fuels) involves complications arising from the interaction between the regulation and the time profile of the price for the resource itself. Energy tax policy may, for example, accelerate the depletion of fossil fuel energy resources if it raises expected future energy prices by more than current prices (Sinclair, 1992). Revenue-maximising owners of the resource may wish to exploit the resource sooner rather than later, if the tax reduces the future net revenue. As Sinn (2007) has recently argued, these effects on resource pricing may operate distinctly differently if regulation takes the form of emissions trading instead of taxes. Emissions trading could be used to place quantity constraints on the use of the resource in each period, so that the inter-temporal shift in energy consumption is avoided.

2.5 The balance between costs and benefits of using environmental taxes

The considerations above imply that environmental taxes are likely to be particularly valuable where wide-ranging changes in behaviour are needed across a large number of production and consumption activities. The costs of direct regulation in these cases are large, and in some cases prohibitive. In addition, where the activities to be regulated are highly diverse, society may gain substantially from changing these damaging activities in the most cost-effective manner.

In other areas, market instruments may work less well. In the next section we discuss the high costs that may sometimes be incurred in operating well-targeted environmental taxes. In other cases, an outright ban might be substantially easier to implement and enforce than a tax rate that requires fine measurement, or where avoidance activities are costly or dangerous.

Little can be gained from over-sophistication in the tax structure through the introduction of finely-graded tax differentials to reflect the environmental characteristics of commodities with little environmental significance. Complex tax structures are liable to be costly to operate, and the tax ‘boundaries’ between products subject to higher and lower rates of tax are always open to socially wasteful litigation, and consequent erosion. Moreover, insufficiently large tax incentives may achieve little change in behaviour. As argued above, firms may not take account of tax incentives when making environmental technology decisions if the tax incentives are too small to justify the costs of changing established decision-making structures. It is perhaps an over-generalisation to suggest that environmental taxes should be large, or not be imposed at all. However, the costs of complexity and the risk that minor environmental taxes will simply be ignored should both caution against too much environmental fine-tuning of the fiscal system.

3. Designing environmental taxes

The key to achieving the potential gains from environmental taxes does not lie in the indiscriminate introduction of taxes with a vaguely-defined environmental justification. Rather, it lies in the effective targeting of incentives to the pollution or other environmental problems that policy seeks to influence.

Poorly targeted environmental taxes may increase the economic costs of taxation, while offering little in the way of environmental gains.

This issue is highlighted by the contrast between different types of environmental tax:

- ***Taxes on measured emissions.*** Practical examples of such taxes include Sweden's tax on nitrogen oxides emissions (Millock and Sterner, 2004), and emission charges for water pollution in the Netherlands (Bressers and Lulofs, 2004). Environmental taxes based directly on measured emissions can, in principle, be very precisely targeted to the policy's environmental objectives. When polluting emissions rise, the polluter's tax base rises, and the polluter pays additional tax directly in proportion to the rise in emissions. Likewise, any actions the polluter can take to reduce their tax liability also reduce emissions. The costs of measuring individual emissions may deter widespread use of environmental taxes of this form[4], except where small numbers of emissions sources are involved, or where it is important to relate the incentive precisely to the amount of pollution emitted rather than basing the tax on some more easily-assessed proxy for emissions.

- ***Tax on a market good that is related to emissions***. An alternative to direct taxation of emissions is to set or modify a rate of indirect tax (excise duty, sales tax, or value-added tax), or to introduce an environmental tax based on the sale of polluting goods or production inputs. Goods and services associated with environmental damage in production or consumption may be taxed more heavily (e.g. tax on batteries and fertilisers). Goods believed to benefit the environment may be taxed less heavily than their substitutes, as with reduced tax on lead-free petrol (Hammar and Löfgren, 2004). Environmental taxes of this sort may have lower administrative costs than taxes based on measured emissions. In some cases it may be possible to use existing taxes (for example by differentiating VAT), and this may be less costly than wholly-new administrative apparatus and procedures[5]. Even where this is not possible, a separate environmental tax levied on transaction values may have lower administrative cost than one levied on measured emissions, especially if it can be operated in a way compatible with existing definitions of the tax base. The drawback of such taxes is that they are less-precisely targeted to emissions than measured-emissions taxes, and they may therefore encourage an inefficient pattern of polluter responses (Sandmo, 1976). Some of the responses of polluters to the tax may seek to reduce tax payments in ways that do not lead to any environmental benefit.

- ***Multi-part instrument***. Using a tax on a proxy for emissions can thus economise on the administrative costs of directly measuring emissions, but risks behavioural responses that do not always achieve the most efficient patterns of pollution abatement. In some cases, a more efficiently-targeted environmental incentive can be created through artful *combination* of indirect taxes – a "multi-part instrument" – to approximate more closely the effects of a tax on measured emissions. An excise tax on the sale of a commodity plus a subsidy for clean technology together can provide the desired substitution and output effects, and may be better than either on its own. A tax directly based on motor vehicle emissions may not be feasible, but it may be approximated by the combination of instruments such as a tax on petrol, a subsidy to new car purchases, or tax on older cars, and a tax on cars with low fuel-efficiency or high emission rates (Fullerton and West, 2002). Likewise, efficient incentives to reduce consumption of waste-intensive products and to dispose of waste properly, approximating the effects of an otherwise infeasible Pigouvian tax on waste disposal, may be achieved by a combination of an "advance disposal fee" (an excise tax on sales, based on the product's

[4] The existence of administrative costs may affect the optimal structure and level of environmental taxes (Polinsky and Shavell, 1982; Cremer and Gahvari, 2002).
[5] "Piggy-backing" environmental taxes onto existing tax systems such as VAT is unlikely to be wholly costless, however. As Crawford Keen and Smith (2008) discuss, the administrative complexity of multiple-rate VAT systems is a strong reason to minimise the extent of VAT rate differentiation.

waste content) and a subsidy for proper disposal (Fullerton and Wolverton, 1999). One example of this is a simple deposit on glass bottles with a refund for recycling, but the idea can be applied much more widely. It can even apply to industrial pollutants. A targeted tax on emissions may be difficult if the emissions cannot be measured, especially since the tax does not apply to a market transaction. But the same effects can be achieved by the combination of a tax on the output of the firm and a subsidy to the purchase of pollution abatement technology. Since both the sale of output and the purchase of clean inputs are market transactions, these two instruments together may cost less to administer than the single ideal tax on emissions.

Each such form of environmental tax may be appropriate in particular circumstances. The choice among them needs to take account both of the administrative costs of different tax options and the extent to which different tax designs can achieve effective targeting of the environmental incentive. The institutional assignment of responsibility for tax-setting and the allocation of the revenues may also affect the efficiency of the outcome.

A particularly severe problem of linkage arises for indirect environmental taxes on inputs when pollution abatement can efficiently be achieved through effluent "cleaning" at the end of the production process. This would significantly limit the scope to regulate sulphur dioxide emissions from power stations by taxing the sulphur content of input fuels, since end-of-pipe effluent cleaning technologies in the form of flue gas desulphurisation (FGD) equipment are one of the principal abatement options; a tax on input fuels would encourage abatement through fuel switching, but would not ensure an efficient balance between fuel switching and FGDs.[6] By contrast, a tax on carbon content of fuels would lead to efficient abatement of carbon dioxide emissions, where effluent cleaning is not currently a commercially-viable option. In a dynamic context, however, it would provide no incentive to develop new end-of-pipe technologies, unlike a direct tax on carbon dioxide emissions; and as such technologies (eg for carbon capture and storage) are developed it would require increasing adaptation, to ensure efficient incentives for their use. The acceptability of a carbon tax on fuel inputs rather than on measured carbon emissions therefore depends on the likely speed of these technological developments and on the extent that their development might be inhibited by the choice of a tax on inputs rather than on measured emissions.

Many of the so-called "environmental taxes" introduced in practice have been used primarily for revenue-raising (Opschoor and Vos, 1989), sometimes to raise earmarked revenues for particular public expenditures related to environmental protection. "Environmental taxes" of this sort have been used to recover the costs of administering environmental regulation, to pay for public or private expenditures on pollution abatement, and, in the US, to pay for Superfund clean-up of contaminated waste sites. The environmental effects of these taxes themselves may be limited. In some cases, their link to the environment is solely through the use of their revenues.

4. Revenue aspects of environmental taxes: a 'double dividend'?

Some commentators such as Pearce (1991) and Oates (1991) have drawn attention to a potential 'double dividend' from environmental taxes – the possibility that an environmental tax might both improve the environment *and* provide revenue that can be used to reduce other distorting taxes on labour supply, investment, or consumption. This argument[7] has a number of implications for two important kinds of policy decisions. First, in the tax policy choice about how to raise a given revenue, some have argued for a switch from conventional distorting taxes to environmental taxes. Second, for the environmental policy choice about how to control pollution, some have argued for a switch from non-revenue-raising instruments (quotas or grandfathered permits) to revenue-raising instruments (environmental taxes or auctioned permits). We look at each such choice in turn.

[6] A fuel tax based on sulphur content can be combined with a subsidy for installing FGD, which is an example of a two-part instrument.
[7] A recent review is in Bovenberg and Goulder (2002).

(i) Tax policy choice: the switch from distorting taxes to environmental taxes

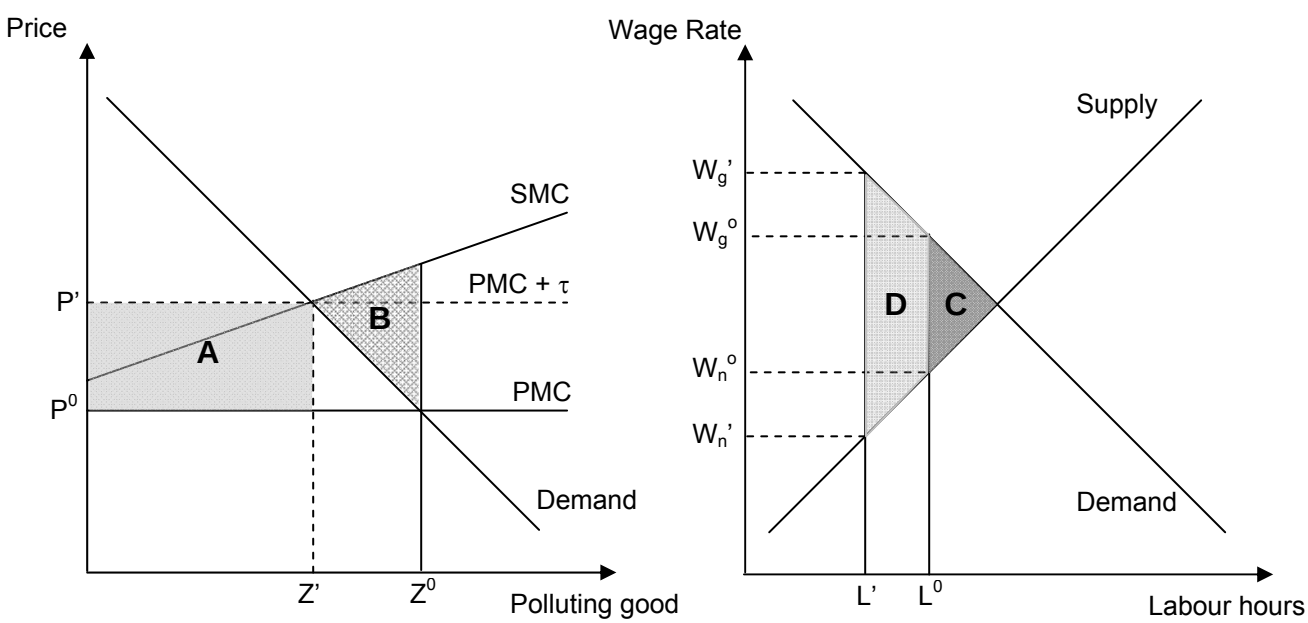
Most taxes induce undesirable behavioural adjustments that reduce labour supply or investment. These taxes create "excess burden", meaning that they reduce individual welfare by more than the actual tax payment. Raising the rate of conventional taxes typically increases these distortionary costs, by an amount called "marginal excess burden". For existing taxes, empirical estimates of these marginal distortionary costs are appreciable. Bovenberg and Goulder (2002) review several estimates (such as in Ballard *et al* 1985), and they find that marginal excess burden is 20–50 cents for each extra dollar of tax revenue. However, environmental taxes induce desirable behavioural adjustments that reduce emissions. In these circumstances, making use of environmental taxes would appear distinctly preferable to relying on conventional taxes. Isn't it better to raise revenues from taxes that correct distortions rather than from taxes that create distortions?

This case for environmental taxation is very important, but ambiguous, so some simple analytics are worthwhile. To understand the pros and cons most clearly, start with the market for a polluting good on the left-hand side of Figure 4.1, and suppose that the original equilibrium has no policy to control pollution. The normal downward-sloping demand curve reflects marginal benefits to consumers; it crosses the flat private marginal cost (PMC) curve at the original quantity (Z^o) and at the original low price (P^o). However, the social marginal cost (SMC) is higher than the cost faced by firms and consumers, because pollution imposes costs on others. In this diagram, an ideal Pigouvian tax at rate τ would raise the private marginal costs enough for the consumers to face a new price (P'), so that they reduce purchases to Z'. At the new equilibrium, revenue from the tax is area A. In addition, the welfare gain from controlling pollution is area B – the extent to which the social costs exceed the marginal benefits to consumers for all those purchases from Z' to Z^o.

To show the value of collecting revenue in that way, consider the supply and demand for labour in the right-hand side of Figure 4.1. In this diagram, a pre-existing tax on wage income means that the old net wage (W_n^o) is less than the old gross wage (W_g^o). Since the old quantity of labour is L^o, the excess burden is area C. If the government needed more revenue and increased the tax rate, this excess burden could increase to include both area C and area D. Thus D is marginal excess burden.

The simplest form of the double dividend hypothesis is the claim that the addition of the environmental tax would provide two benefits: it would provide the welfare gain B by fixing the pollution problem, and its revenue would allow the government to reduce the wage tax, which would raise the net wage, raise labour supply, and reduce the welfare cost C.

Figure 4.1: Tax on a Polluting Good, with Revenue used to Cut the Labour Tax Rate



An extensive academic literature has focused on the general validity of this proposition. Most importantly, Bovenberg and de Mooij (1994) and Parry (1995) have shown that the analysis above is missing one key element: the environmental tax has its own distorting effects on labour supply and therefore can have more or less excess burden than the labour tax itself! The second dividend might not arise.

This new point can also be seen in Figure 4.1, again by using both diagrams. First, on the left-hand side, note that the environmental tax raises the price of the polluting good (from P^o to P'). Since this polluting good is one of the goods in the basket of consumer goods, this tax also raises the overall price of consumption goods (relative to the wage rate). This effect *reduces* the *real* net wage (the bundle of goods that can be purchased with an hour of labour). Bovenberg and de Mooij (1994) remind us that labour supply depends on this *real* net wage. If the higher price P' reduces the real net wage (to $W_n^{'}$), it might *reduce* labour supply, adding to excess burden. However, if the environmental tax revenue is used to cut the labour tax rate, this would *raise* the real net wage and *raise* labour supply, reducing excess burden. Either effect might dominate, and so the real net wage might rise or fall. The second dividend is positive if labour supply rises, but it is negative if labour supply falls. In other words, the environmental tax by itself might have welfare gain area B, but it has ambiguous effects on the real net wage, labour supply, and excess burden area C.[8]

Another way to see this ambiguity is to remember that the first dividend from the environmental gain must be set aside in order to consider the second dividend from improving the tax system. Aside from environmental considerations, a given amount of revenue may be raised most efficiently with a particular "optimal" set of tax rates on different goods. If one such good is taxed at less than its optimal rate, *and* this good is associated with uncontrolled pollution, then indeed, an increase in that rate may provide two dividends – an environmental gain and a more efficient tax system. If the good is taxed at exactly its optimal rate already, however, then an increase in that rate may provide only the environmental dividend. If the pollutant is already controlled in some other way such as through command-and-control regulation, then an increase in the tax on this good may not even provide an environmental dividend. Finally, if the good is already taxed at a rate that is higher than optimal, an increase in the environmental tax may reduce welfare. Clearly, the possibility of a 'double dividend' cannot be ruled in or out; it must depend on the circumstances.

In summary, starting from a position in which the system of taxes has been designed to minimise excess burden without any concern for the environmental implications of the tax structure, usually welfare would indeed improve by shifting the balance of revenue-raising towards greater reliance on environmental taxes – from the environmental dividend alone. In this sense, the tax system is more efficient if environmental taxes are used than if they are neglected. Yet this improvement could only hold up to a certain point. As the environmental tax rate is increased, the excess burden costs of behavioural changes rise more than proportionately, eventually overtaking the additional environmental benefits.

(ii) Environmental policy choice: The switch from quotas to revenue-raising taxes or permits

A second strand in the 'double dividend' literature concerns the choice of environmental policy instrument. If policy employs a revenue-raising environmental policy instrument, such as an environmental tax or auctioned tradable permits, do the revenues collected as a 'by-product' of its environmental effects provide a more efficient fiscal policy, compared with the use of an equivalent non-revenue-raising instrument?

[8] This definition of 'excess burden' ignores any environmental gain. Environmental taxes could be said to have negative excess burden if the definition of excess burden were to include environmental benefits, but this definition would risk double counting (if one were to say that the tax has negative excess burden in addition to environmental benefits).

This point is important but ambiguous, so we turn again to some simple analytics. On the left-hand side of Figure 4.1, the "optimal" quantity of the polluting good is *Z'*, where marginal benefits to consumers are exactly offset by the social marginal costs of production. Policy makers have several ways to achieve this quantity (all of which are equivalent in this simple model):[9]

(a) Impose a tax at rate τ, which raises the price to *P'* and reduces purchases to *Z'*. This policy raises revenue, area A in the figure.

(b) Impose *Z'* as a simple legal limit on the total quantity of production. This mandate or non-tradable quota is a type of command and control (CAC) policy that does not raise revenue.

(c) Set a fixed number *Z'* of tradable permits and hand them out for free to existing firms. These permits are "grandfathered" in the sense that each firm is given permits in proportion to their emissions in some prior period. This policy does not raise revenue.

(d) Set a fixed number *Z'* of tradable permits and sell them at auction. This policy raises revenue.

The number of permits is the same for either (c) or (d), so the permits are equally valuable. Since consumers are willing to pay *P'* for that limited output, and production costs are only P^o, firms are willing to pay the difference ($P'–P^o$) as the price to buy a permit. Because the number of permits is limited, this value is called a "scarcity rent". As can be seen in the figure, this price of a permit is exactly equal to the tax rate τ. Thus, the total "scarcity rent" is exactly the rectangle, area A.

All four policies can be viewed in the left-hand side of Figure 4.1: all raise the consumer's price to *P'* and reduce purchases to *Z'*, so all reduce pollution and achieve the environmental gain (area B). All four policies make output scarcer, and thus generate scarcity rents. The key difference is that the tax and auctioned permits ((a) and (d)) allow the government to "capture" the scarcity rents as revenue, while the quotas and grandfathered permits ((b) and (c)) allow *firms* to capture the rents. Thus the grandfathered case is equivalent to the case where permits are auctioned, but with the revenues transferred back to firms through lump-sum transfers.[10]

Even though firms are competitive in this model, they make pure profits! How can competitive firms make pure profits? Normally, antitrust authorities do not let firms collude, act like a monopoly, restrict output, raise prices, and make profits. With quotas or grandfathered permits, however, the environmental authority essentially *requires* firms to restrict output! The policy erects an "entry barrier", because new firms would have to buy permits to sell their product in this market.

In contrast, the tax and the auctioned permits capture the scarcity rents, as revenue. In this simple competitive model, the policy has no long-run effect on firms: they had zero profits before the policy, and they still have zero profits after the tax or auctioned permit policy. Output shrinks, as is necessary to reduce pollution. If this industry is not too large, and if labour and capital are mobile, then these factors can be re-employed elsewhere – at the same wage or rate of return they earned before. If not, then clearly the industry may suffer some transition costs such as temporary unemployment. Yet all four policies above shrink the industry the same amount, and they thus have similar transition costs.

If the government captures those scarcity rents, then it can use the revenue to reduce other distorting taxes, such as the labour tax on the right-hand side of Figure 4.1. It *might* even be able to raise the net

[9] In particular, this model assumes constant costs, competitive markets, no uncertainty, and a fixed amount of pollution per unit of output. These assumptions can be relaxed without changing the basic intuition of this section, but the model would become unnecessarily complicated.

[10] In a one-time, unanticipated, permit allocation, the transfers are lump-sum in the sense that they cannot be influenced by any current decision of the firms. In a repeated, or anticipated, allocation, firms may realise that their current decisions could influence future permit allocations, and grandfathering could be distortionary rather than equivalent to a lump-sum transfer.

wage and reduce excess burden (area C). But remember the problem above: the environmental policy itself raises the price of the polluting good (to *P'*). That effect *reduces* the real net wage. Thus labour supply may rise or fall, so the excess burden from labour taxes may rise or fall.

Now we are in a position to restate the importance of raising revenue. It is not that the revenue allows the government to reduce excess burden from labour taxes. Rather, the environmental policy inherently raises the cost of production and exacerbates labour tax distortions, *unless* the government captures the scarcity rents and uses this revenue to offset that effect by cutting the labour tax rate.

This point has clear and important implications for environmental policy choices. Handout of permits may be a way for government to "buy" the cooperation and agreement of industry to enact the new environmental policy, but those firms are indeed paid for their acquiescence. That payment is delivered to firms in the form of being able to charge higher prices. If these higher prices reduce the real net wage enough, the exacerbation of labour supply distortions (extra area "D") could completely offset the environmental gain (Goulder *et al*, 1997, and Fullerton and Metcalf, 2001).

Coordinating tax and environmental policy can be treacherous. Even starting with an uncorrected pollution problem, the introduction of a pollution quota or grandfathered permits may raise prices, reduce the real net wage, and exacerbate labour supply distortions – enough to exceed the environmental gain and provide a net negative effect on welfare.

(*iii)* *Is the double dividend weak, strong – or irrelevant?*

The environmental gain is the first dividend from any of the environmental policies listed above. When does the second dividend arise? In his discussion of the double-dividend debate, Goulder (1995) defines a 'weak' double dividend as the case with a positive welfare gain from using the environmental tax revenues to reduce distortionary taxes instead of returning those tax revenues to taxpayers through lump-sum payments. This weak double dividend also arises by using auctioned permits, rather than grandfathered permits, if the auction revenue is used to cut distorting labour taxes. He points out that the existence of this weak double dividend is uncontroversial, because 'the idea that swapping a distortionary tax for a lump sum tax has a positive welfare cost is part of the usual definition of distortionary' (Goulder, 1995, p160). Note that this weak double dividend is related to subsection (*i*) above, the environmental policy choice between quotas or grandfathered permits on the one hand, versus an environmental tax or auctioned permits (with revenue that can be used to cut distorting taxes).

The claim of a double dividend in this form is undramatic, but not without policy significance. In making a choice between environmental policy instruments, it implies that – other things being equal – a substantial premium should be placed on selecting instruments that do not create scarcity rents and leave them in the hands of private parties. If scarcity rents are captured, through taxes or the auction of a fixed number of permits, then the scarcity rents can be used by government to reduce the rates of existing distortionary taxes. Significant costs are incurred if the potential revenues from environmental taxes or permit auctions are dissipated or forgone.

In a more demanding sense of the term, Goulder (1995) defines a 'strong' double dividend as the case where raising an environmental tax and reducing a distorting tax has not only the environmental gain (first dividend), but also reduces the overall distortionary costs of taxation. This case is primarily about subsection (*ii*) above, the tax policy choice. If a strong double dividend does arise, it means that the environmental tax reform has negative 'gross costs' (defined to include all the welfare costs of behavioural changes from the tax switch, but to exclude environmental benefits). A tax reform with a strong double dividend is an attractive policy because it has 'no regrets'; even if the changes in energy use turn out to have no environmental benefit (no area B), it has been costless because the overall fiscal costs of the tax change are negative (a reduction in excess burden area C). The environmental tax itself distorts labour supply decisions, however, because it reduces the real net wage. This effect could outweigh the reduction in the labour tax. Thus, the 'strong' double dividend may be attractive, but it is far from guaranteed.

The double dividend debate points to the importance of thinking about tax and environmental policy simultaneously (Bovenberg and Goulder, 2002). The number of dividends, however, is not relevant in itself. Once we integrate tax and environmental policy reforms properly, all that really matters is whether the net effect is positive or negative on overall welfare.

Part B: Applications

5. Environmental taxes on energy

5.1 The policy context

At the Earth Summit in Rio in June 1992, more than 150 countries signed the UN Framework Convention on Climate Change, making a collective commitment to action to avert dangerous man-made effects on the global climate. This commitment responded to the accumulating scientific evidence from the Intergovernmental Panel on Climate Change (IPCC) that the increasing concentration of greenhouse gases in the atmosphere, arising from human activity, was causing discernible and pervasive changes in global climate. Subsequent negotiations led to the Kyoto Protocol, agreed in 1997, under which a number of industrial countries took on binding commitments to reduce their emissions of a basket of the principal greenhouse gases.

Under the Kyoto Protocol the EU is committed to reduce greenhouse gas emissions by 8 per cent by 2008-2012, measured against a baseline of the 1990 emissions level. Within this overall EU target, the UK is required to achieve a 12.5 per cent emissions reduction. In addition, however, the UK has unilaterally stated a policy goal of reducing emissions of carbon dioxide, the principal greenhouse gas, to 20% below 1990 levels by 2010. The 2003 Energy White Paper stated a further ambition to achieve a 60% cut in CO_2 emissions by 2050, “with significant progress by 2020”.

Current international discussions are considering the form of a further agreement, to follow the Kyoto commitments. These discussions have been given added impetus by the increasing strength of the IPCC’s concern about climate change, and, in the UK, by the publication in 2006 of the *Stern Review on the Economics of Climate Change*. This *Review* analysed the economic and environmental costs of climate change, and the costs and benefits of policy action. It makes a strong economic case for urgent and significant action to reduce greenhouse gas emissions, to stabilise the concentration of greenhouse gases in the atmosphere, with the aim of limiting the rise in global temperatures, and reducing the risks of catastrophic changes to the global climate.
Given the global nature of the climate change problem, effective policy needs to involve international coordination. The impact that an individual country can make on climate change through independent action is negligible, while this action incurs appreciable domestic costs of abatement. Sufficient international cooperation will not be straightforward to achieve[11], and UK policy-making will be influenced by the nature of whatever international policy framework can be concluded. However, it is difficult to imagine that any substantial reduction in the UK’s emissions can be achieved without according a significant role to energy pricing measures, in some form, whether through taxes or emissions trading. The extensive and far-reaching changes that would be needed to existing patterns of production and consumption across a wide range of economic activities can be stimulated by general price signals more efficiently than by detailed regulatory intervention. Equally, it is it is unlikely to be possible to tackle the problem using simple pricing instruments alone, because of the range of both real and perceived obstacles to setting pricing instruments at first-best levels.

[11] An extensive economic literature involves achieving efficient bargains for the control of international environmental problems, including the distribution of the costs of CO_2 control and the achievement of a stable coalition of signatories to an international agreement. See for example Barrett (2003).

In this section we discuss how tax instruments, and the closely-related approach based on emissions trading, could be used to establish price signals to encourage reduced emissions of carbon dioxide. In 5.2 we begin with some observations on the scale of the carbon price that would be justified by current evidence on the costs and risks of climate change. We then consider in 5.3 the range of available pricing instruments, including existing taxes on energy, carbon pricing though EU emissions trading, and possible new taxes on carbon-based fuels. In 5.4 we note the potential scale of the revenues that could be raised from these instruments, and emphasise the significant opportunity cost if these revenues are foregone by distributing emissions trading allowances without charge to incumbent firms ("grandfathering"). Finally, in 5.5, we look at the difficulties that would be encountered in setting carbon prices at first-best levels, especially in terms of perceived effects on industrial competitiveness and distribution, and set out strategies to overcome these obstacles.

5.2 How high should the carbon price be?

A carbon tax or emissions trading would aim to set a price on the use of fossil fuels (and more generally on activities that generate greenhouse gas emissions) that would reflect the otherwise-unpriced social costs of their use. The "price of carbon" at which policy should aim – in other words, the appropriate rate of carbon tax, or the emissions trading price from a chosen quantity constraint on emissions – should in principle reflect an assessment of the climate change consequences of the marginal tonne of carbon dioxide emitted, at the socially-optimal level of abatement. As with any other externality tax, the aim would be to ensure that private decisions that result – directly or indirectly – in additional greenhouse gas emissions take account of the costs imposed on the global climate. These costs will be spread over a considerable period of time, and will include costs of adapting to sea level rise and changed temperature and weather patterns, changes in agricultural productivity, health effects, damage caused by a greater frequency of extreme climate events such as storms and floods, and – with more severe climate change – the costs of population displacement and conflict caused by rapid changes in climate and living conditions in different parts of the world.

Two broad approaches could be taken to assessing the carbon price at which policy should aim. The first is to build up a picture of the economic costs of climate change from models reflecting the various effects, including both "predictable" effects such as regional changes in agricultural productivity resulting from changes in mean temperature and rainfall, and also, where possible, assigning values for "unpredictable" catastrophes and irreversible changes. The resulting estimates could then, in principle, be used to describe a marginal damage cost function, and compared with the corresponding marginal abatement costs for reduced CO_2 emissions. The level of abatement at which policy should aim, and the carbon price or Pigouvian carbon tax needed to achieve this outcome, would then be identified by the point where marginal climate-change damage equals the marginal cost of reducing CO_2 emissions.

The difficulties in such an approach are formidable. Any assessment of the economic effects of climate change must begin from scientific assessments of the underlying physical/environmental processes which are in themselves surrounded by considerable uncertainty and enormous margins for error. The science relating to the risks of major threshold effects (such as the reversal of deep ocean currents) and the consequences of greenhouse gas accumulation at higher concentrations is uncertain, and cannot be modelled in terms of precise trajectories with clearly-defined probabilities. Nevertheless, this has to form the staring point for economic assessments, with further uncertainties – and plenty of scope for disagreement – in translating the scientific projections into economic values.

The difficulties in assessing the value of changes in CO_2 emissions are complicated by the unprecedented length of time over which the effects of emissions are felt. Since the rate of decay of any addition to the stock of atmospheric CO_2 is slow, current emissions have an effect that extends into many future periods. Likewise, policy measures taken now potentially confer benefits to future generations as well as to the current one. Given the length of the time horizon involved, balancing the interests of present and future generations in climate change policy raises unusually difficult

philosophical issues (Broome, 1992). These issues concern the treatment of large gains and losses in the distant future, which conventional discount rates could render of negligible current value.

An approach of this sort underlies many estimates of the "social cost of carbon" – the monetary value of worldwide damage caused by marginal emissions in the current year. Pearce (2005) exemplifies this approach, reviewing the evidence on the appropriate value for the social cost of carbon to be used in UK policy appraisal. His conclusions suggest a range for the social cost of carbon of £0.82–£1.64 per tonne of carbon dioxide (/tCO_2), assuming a 3 per cent discount rate. Incorporating equity weighting and time-varying discount rates could increase the range to £1.09–£7.36/tCO_2. Both ranges lie well below the UK government's central estimate of the marginal social cost of carbon of £19 /tCO_2 (Clarkson and Deyes, 2002).[12] These figures are likely to overstate the corresponding optimal carbon price, to the extenet that marginal damage costs rise with emissions, because the optimal carbon price will be lower than marginal damage costs at unconstrained emissions.

The recent *Stern Review of the Economics of Climate Change* bases its principal recommendations for emission-reduction targets on a different approach altogether, but supports this approach with estimates of the marginal damage cost based on model simulations similar to the above, albeit with some critical differences in assumptions and methodology.

The key policy recommendations of the *Stern Review* are based on its assessment of the target that should be adopted for stabilisation of the concentration of greenhouse gases in the atmosphere. The review argues that policy should aim at stabilising this concentration at a maximum of 550 parts per million of carbon dioxide equivalent (ppm CO_2e). A higher concentration would involve substantial risk of temperature rises above 5°C, which the *Review* argues could involve dangerous and potentially disastrous changes to the planet. Even stabilisation at 550 ppm CO_2e involves a 7% chance of temperatures rising by more than 5°C, and a 24% chance of more than 4°C. The *Review* argues that a path stabilising at 550 ppm CO_2e would require global emissions to reach their peak within 20 years, and reductions in emissions against business as usual of around 30% by 2050 compared with 2000. Drawing on data on the costs of emissions abatement, Stern (2008) argues that this implies a global CO_2 price of around €30/tCO_2 (roughly £75/tC).

In its modelling of the economic effects of emissions trends and policy intervention, the *Review* argues that "business as usual" would generate a 50-50 chance of warming by around 5°C relative to pre-industrial temperatures[13]. A key feature of the *Review*'s approach has been explicit modelling of the risks and their associated costs, rather than simply focusing on an average trajectory across the range of possible outcomes. Incorporating the risk of very high costs into its estimates, the *Review* finds that the costs of unchecked climate change are around 5–20% of global output, and estimates the current social cost of carbon at $85/tCO2e (about $300/tC). But if emissions were restricted to a level consistent with long-term stabilisation of greenhouse gas concentrations of 550 ppm/CO_2e, the stabilisation goal recommended by the *Review*, the efficient carbon price would fall to $30/t$CO_2$.

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[12] Equivalent to £70 per tonne of carbon. As with the other figures quoted from Pearce (2005) the estimates in the text are shown in terms of £ per tonne of CO_2, to assist comparison with EU ETS prices which are quoted on this basis. A price of £1 per tonne of CO_2 is equivalent to a price of £3.67 per tonne of carbon.

[13] The *Review*'s estimates of the external cost of carbon emissions reflect particular assumptions about the relative weighting of the interests of present and future generations which have attracted some criticism (e.g. Dasgupta, 2006; Nordhaus, 2007). These estimates reflect a low value of 0.1% for the rate of 'pure time preference' (δ), a parameter reflecting the extent to which wellbeing in future periods should be discounted relative to today, and justifies this low value on the grounds that the only ethical reason to weight differently the interests of people living at different times is the small probability of future planetary annihilation, some small risk that future generations will not exist. Secondly, the elasticity of the marginal utility of consumption (η), which describes how to value income of the rich compared to income of the poor (regardless of when they may exist) is set equal to 1. Higher values imply that an extra pound of income is valued much less for someone with high income than low income; if future generations are richer than the present, this reduces the weight given to future effects. The Review's conclusions are relatively sensitive to the choices of these two parameters, both of which imply much lower discounting of future costs and benefits than in most earlier research in this field. The central conclusion that the cost of 'business as usual' emissions amounts to about 5% of global output is reduced to 2.9% if η is increased from 1 to 1.5, and to 2.3% if δ is increased from 0.1% to 1%.

5.3 Energy taxes and emissions trading

(a) Existing energy taxes in the UK

The current tax treatment of energy in the UK has three main components. In quantitative terms the most significant taxes levied on energy in the UK are the excise taxes on mineral oils, in particular motor fuels, which raise some £25 billion in revenue. Ultra low sulphur petrol and diesel are currently subject to an excise of 50.35 pence per litre[14]. Lower rates of duty are applied to some alternative fuels such as LPG and biofuels.

Domestic energy is subject to VAT at a rate of 5%. Before 1994 domestic energy had been zero-rated (i.e. untaxed) in the UK's VAT system. In 1993 the government proposed extending standard-rate VAT to domestic energy, primarily for revenue reasons, but also recognising the growing environmental concerns about fossil fuel use. The measure proved highly controversial, and the planned two-stage transition to the standard rate stalled at the first stage, with the rate at 8%. This rate was subsequently reduced to 5% by the incoming Labour government in 1997. Compared with uniform taxation of all consumption at the standard VAT rate, the UK effectively subsidises domestic energy at 12.5%, at an annual revenue cost of almost £3 billion.

The third element of the current energy tax regime is the climate change levy (CCL), a single-stage excise tax imposed since 2001 on industrial and commercial energy use. The full rates of the levy are 0.154p/kWh on gas, 0.441p/kWh on electricity, and 1.201p/kg on coal; the tax is not applied to renewables. Firms in some 45 energy-intensive sectors that have concluded "climate change agreements" with the government, making commitments to legally-binding targets for reduced energy use or reduced emissions, are entitled to an 80% reduction in the rate of the levy, which raises around £0.7 billion annually.

The CCL has been criticised, by Helm (20xx) amongst others, for its failure to tax fuels in proportion to carbon content. To the extent that electricity is taxed at a single rate, regardless of the fuel mix in generation, the tax simply raises the cost of energy to users, and provides no incentive to switch the fuel mix in generation to lower-carbon inputs. Also, the if the rates of the levy are expressed as an implicit tax per tonne of CO_2, the tax on coal is considerably less (£4.30 per tonne of CO_2) than on electricity and gas (both approx £8.10 per tonne of CO_2). The lower tax on coal appears to have reflected an explicit political decision to avoid adverse effects on the mining industry, but its unfortunate impact is to penalise switching from coal to lower-carbon fuels.

In addition to taxes on energy, some of the regulatory obligations placed on the power sector and the introduction of the EU emissions trading scheme (ETS) for CO_2 both have some quasi-fiscal effects.

Power generators are subject to a Renewables Obligation, obliging them to obtain a given proportion of their electricity from renewable sources. Compliance with these obligations is verified by Renewables Obligation Certificates (ROCs), which are tradable, allowing flexibility in compliance. The cost of meeting the targets is, however, significant, and may have raised electricity prices by something on the order of xx per cent.

The EU ETS covers the power sector and CO_2-intensive industries (such as iron and steel, cement, pulp and paper), a total of around 1,000 plants in the UK. It began operation in 2005, for a first three-year trading phase (2005–07). Phase two (2008–12) covers the Kyoto commitment period, and is now underway. Allowances can be traded within phases, but not between phases, and the cost of allowances thus reflects the cap on emissions set in each phase. For phase one the allowance cap is widely regarded as having been too permissive, and outturn CO_2 emissions were well within the cap

[14] VAT is also levied on motor fuels, charged at the standard VAT rate of 17.5 per cent. . However we regard VAT as a general tax on consumption, and the VAT charged on motor fuels should not be counted in any comparison of the level of motor fuel externalities and taxes.

such that the allowance price dropped to zero. The phase two cap, however, is more restrictive, and the market price of phase two allowances is currently around €20 (£14)/tCO_2. Although allowances have been distributed free to existing firms in phase one, and only some 7% of allowances are scheduled to be auctioned by the UK in phase two, the allowance price has much the same effect on abatement incentives and on marginal costs as a tax.

(b) Design of a carbon tax

Ideally, a tax to control atmospheric emissions of CO_2 would be levied directly on the individuals or firms who are responsible for the emissions, and based directly on the amounts of CO_2 emitted. In practice, sources of emissions are too numerous and varied for direct measurement of emissions. Carbon taxes therefore normally take the form of a tax on the carbon content of fuels, intended to proxy for the carbon emission that result from combustion. The relationship between carbon content and eventual carbon emissions is very close, because no viable end-of-pipe emissions cleaning technologies are generally available. And while technologies for carbon capture and storage are developing rapidly, they generally will be implemented on a scale sufficiently large that appropriate credit for these activities could be given – for example, by refunding carbon tax – without extensive and costly administrative complications.

Nevertheless, some practical issues about the structure and administration of an energy tax could affect how efficiently it reduces CO_2 emissions. Some of these issues have a counterpart in the design of emissions trading arrangements for carbon dioxide.

In the European countries that have actually introduced carbon taxes (Sweden, Norway, Finland, the Netherlands, and Denmark), these taxes take the form of extended systems of fuel excises. Rates of tax are defined separately for each fuel, and relative tax levels on different fuels are set so as to equate the implicit rate of tax per unit of carbon. This requirement is not, however, always observed; in Denmark and Norway, for example, some fuels are not subject to the carbon tax. Also, the tax can vary across types of energy user; in Sweden and the Netherlands, much lower rates of tax apply to industrial energy users than to private households. Most of the carbon taxes actually implemented in these countries have provisions exempting firms or sectors that are particularly exposed to international competition.

The presumption that a carbon tax should naturally be implemented as an extension of existing fuel excises has been questioned by Pearson and Smith (1991). A “primary” carbon tax, levied on primary fuels where they are mined, extracted or imported (e.g. crude oil, coal, and gas) would have some advantages compared to excises levied on final fuel products sold to industrial users or households (such as coke, anthracite, and petrol). It would involve fewer payers of tax than would a “final” tax, and it would have no need for fiscal supervision of the energy chain beyond the first point; administrative costs would be expected to be low, and tight supervision could prevent evasion.

Applying the tax at an earlier stage in the production chain does not necessarily imply any difference in the economic incidence of the tax or its environmental effects. The prices paid by industry and consumers would be much the same as with an equivalent set of excises. However, the primary carbon tax, because it taxes carbon at the earliest possible stage, accurately reflects the carbon emissions during processing as well as in the final product, and encourages lower processing emissions, as well as the use of fuel products containing less carbon. An exact equivalence could only be achieved with fuel excises if these could somehow be differentiated according to the carbon emissions associated with the processing of each product – its carbon “history” – as well as its actual carbon content. Where different processing technologies are used, with different emissions during processing, a final carbon tax levied on the basis of average carbon emissions during processing is liable to lead to inefficient technology choices, providing poorer incentives to adopt low-emissions processes. In principle, therefore, a primary carbon tax might be more efficient, in both static and dynamic terms, although the quantitative significance of this issue remains unclear. It may be more significant with greater variation in intermediate emissions across different fuel processing technologies.

A somewhat similar issue is discussed by Poterba and Rotemberg (1995) who consider the case with joint production of final fuel products, where the output mix is a choice variable. In other words, they consider the case where a single primary fuel is processed into more than one final fuel product, and where the mix of final fuels produced can be varied. They show that no objective basis can be used to estimate the intermediate carbon emissions associated with the production of particular final fuels.

One implication of these arguments is that environmental and economic efficiency is unlikely to be fully attainable with a carbon tax levied on final fuel products (such as an excise tax). Broadly similar issues apply in defining the point at which an emissions trading system should operate. In practical terms, the most significant choice has been whether to apply emissions trading to power station input fuels or electricity outputs. The EU emissions trading scheme has chosen to take the former route and provides better incentives to reduce carbon "wastage" in the course of generation than would permits or taxes applied (like the UK's climate change levy) to electricity outputs.

(c) Carbon tax in parallel with EU emissions trading

Given the establishment of the EU ETS in 2005, the context for policy-making is one in which a substantial part of carbon emissions are already priced. Taxes on carbon or energy may have a role to play in regulating the energy use of activities outside the scope of the EU ETS. But does taxation play a role in sectors already covered by the EU ETS?

Since emissions taxes and emissions trading have such similar economic properties, the market prices of emissions allowances will be directly affected by environmental taxes applied to the same emissions or to closely-related transactions. In the case where the same emissions regulated by emissions trading are also subject to an emissions tax, the effect of a tax will generally be to reduce the value of emissions allowances by the amount of the tax. The allowance price is determined by the value of the marginal allowance at the constraint – the "cap" – set by the ETS, and in the absence of an emissions tax, holding this allowance has a value given by the marginal abatement cost that would otherwise have to be incurred. With an emissions tax, however, the value of an allowance is lower, because the saving that can be made by holding an allowance is now the difference between the marginal abatement cost and the tax per unit of emissions.

Where the tax is less than the allowance price in the absence of tax, the tax would reduce the allowance price pound for pound, but would have no effect on the level of emissions. If the tax is set higher than the no-tax allowance price, emissions will fall below the cap, and the permit price will fall to zero. In effect, the emissions cap no longer binds, and allowances have no value. What determines the level of emissions is simply polluter responses to the tax, and the ETS is superfluous.

It is well-known that where abatement costs are uncertain, a better outcome can be achieved by a mixture of price and quantity regulation than by reliance on tradable permits or an emissions tax alone (Roberts and Spence, 1976). Price "safety valves" could be set to limit the range of feasible variation in allowance prices, avoiding the costs and inefficiency of the extreme outcomes that could arise if policy was based solely on a fixed emissions cap. An emissions tax, set at a low-ish level, might be a way of instituting a welfare-improving floor to the price incentive for abatement. If the emissions tax applies to all firms in the emissions trading system, it will reduce the allowance price by the amount of the tax; the lower threshold thus comes into play when the allowance price reaches zero.

Less clear-cut relationships between taxes and emissions trading arise where some, but not all, of the emissions covered by a trading scheme are also subject to an environmental tax. An example would be a national carbon tax, covering activities already regulated by EU emissions trading; the tax would only apply to the emissions the country concerned, and not to all emissions covered by the ETS. In this case, if the tax rate is low, and the proportion of EU-wide emissions covered by the tax is small, the tax will have a correspondingly small effect on permit prices (though perhaps not wholly negligible in the case of a significant carbon tax imposed by one of the larger EU countries). Its principal effect

would be to induce an inefficient pattern of abatement across countries, since the abatement incentive in the country imposing the tax would be higher than elsewhere.

As discussed below, one of the effects of free distribution of EU ETS allowances has been to confer windfall profits on the firms to which allowances are allocated, since their selling prices are expected to rise to reflect the impact of allowances on the marginal cost of production. The effect of a carbon tax on allowance values suggests that it would be possible to recover some of the excess profits given to firms through allowance grandfathering by introducing an emissions tax on a base that closely proxies the use of allowances. Then the value of allowances will fall by the amount of the tax. However, except where the tax is imposed by all countries in concert, it would lead to abatement inefficiency as well as recovering windfall profits. And if all countries could act in concert to impose a tax, a similar outcome could have been achieved more directly by auctioning the allowances.

5.4 Revenues from energy taxes and emissions trading

(a) The scale of potential revenues

Total UK emissions of the six groups of greenhouse gases regulated by the Kyoto Protocol were 652 million tonnes CO_2e in 2006[15]. If all UK greenhouse gas emissions were to be taxed (or covered by auctioned tradeable permits) at the price of carbon implied by the *Stern Review* (€30 / tonne CO_2, or roughly £20), then the aggregate revenue would be about £13 billion, equivalent to some 2.6% of total receipts from taxes and national insurance contributions. However, this represents an upper bound to the net revenue potential of carbon taxes or emissions trading, and assumes systematic taxation of all emissions sources. If some sectors are exempted, or if some energy users receive grandfathered allowances, revenues would be reduced. For example, exemption of residential sector energy use would reduce the revenue potential by about £1.7 billion. Also, at least some existing taxes on energy should probably be deducted from this sum, including the £0.7 billion currently raised from the climate change levy. Rather more significantly, existing taxes on road transport fuels might arguably already be justified partly as a tax on greenhouse gas emissions, in which case the £2.4 billion that would raised by carbon tax on road transport should be offset by a corresponding reduction in motor fuel tax.

(b) The case for auctioning ETS allowances

Auctioning tradable emissions allowances has significant economic advantages over “grandfathering” free allowances to existing polluters. Moreover, as Hepburn *et al* (2006) describe, plenty of practical experience has been gathered in regular auctions of government securities, which provide a model on which allowance auctions can draw. So far, however, auctioning of EU ETS emissions allowances has been very limited, and initially confined to the “new entrant reserve” in some member states; the rules of the ETS restrict member states to auctioning no more than 10 per cent of allowances in Phase II. The free allocation of EU ETS allowances has mainly been a reaction to concerns about adverse effects of auctioned allowances on competitiveness, and we argue below that this response is misplaced.

The economic case for auctioning emissions trading allowances is argued clearly by Fullerton and Metcalf (2001), Cramton and Kerr (2002), and Hepburn *et al* (2006). The arguments fall into three principal groups – those concerning the auction revenues, those concerning the windfall gains that grandfathering typically confers on polluters, and those concerning the better dynamic properties of an industry where new firms and existing firms are treated on an equal basis.

The principal argument for auctioning is the value of the revenues. These can be used to reduce other taxes, with consequent gains in the overall efficiency of the economy compared with non-revenue-raising forms of allowance allocation. This efficiency argument is the well-established and uncontroversial “weak” double dividend hypothesis discussed in section 4, which asserts that an

[15] Some 85% of these emissions were of carbon dioxide (of which 40% was emitted by the power sector, 17% by other industry and business, 22% by road transport, and 15% by the residential sector), 7.5% methane (principally from landfill decomposition and agriculture) and 6% nitrogen oxides (largely from agriculture). See http://www.defra.gov.uk/environment/statistics/globatmos/index.htm for latest UK emissions data.

efficiency gain is achieved if environmental tax revenues are used to reduce the rates of distortionary taxes in the economy, rather than being returned on a lump-sum basis (Goulder, 1995). In the emissions trading context this means that an efficiency gain is made by auctioning and using the auction revenues to reduce the rates of distortionary taxes in the economy, rather than foregoing the revenue through free distribution (the counterpart to taxing with lump-sum return of the revenues).

The aggregate asset value created by the EU ETS is substantial. Allowances have been issued for each year corresponding to 2 billion tonnes of CO_2 EU-wide, with an aggregate annual value of some €40 billion. Within this total, the allowances allocated within the UK (245 million tonnes) have an aggregate annual value of some €5 billion (£3.3 billion). The market value of these allowances should not be affected by the method of distribution – whether auctioned or grandfathered – because the price of allowances in a competitive market is determined by the marginal abatement cost at the emissions constraint set by the quantity issued. Full, competitive, auctioning of allowances in the ETS would therefore have generated annual revenues broadly equivalent to these asset values.

Equivalently, grandfathering foregoes the potential to raise these revenues. If the marginal excess burden of raising tax revenue is assumed to be of the order of 20–50 pence per pound raised (the range of US estimates surveyed by Bovenberg and Goulder, 2002), then as a first approximation, the aggregate economic cost to the UK economy of <u>not</u> auctioning £3.3 billion of EU ETS allowances is some £0.7 to £1.7 billion.

Parry presents some comparative simulations of the net benefits of policies to achieve a reduction in US greenhouse gas emissions by 150 million tonnes, based on an assumption of a $20 per tonne environmental benefit from reducing carbon dioxide emissions. Ignoring the fiscal consequences of raising the price of carbon, such a policy would generate benefits of $3 billion, and net benefits (after abatement costs) of $1.5 billion per year. Taking into account the fiscal interactions between policy measures that raise the price of carbon and the distortionary cost of the existing tax system, Parry calculates that the net benefit from implementing this policy using a carbon tax (or, equivalently, auctioned permits) would be $1.13 billion per year. By contrast, implementing the policy using grandfathered permits incurs a welfare loss of $6 billion. In other words, the choice between auctioning and grandfathering makes a difference of over $7 billion to the net benefit of controlling emissions, a huge figure compared with the gross environmental gain ($3 billion) or the abatement costs incurred by polluters ($1.5 billion). The comparison between the benefit of auctioning and the other costs and benefits of environmental policy interventions depends on the particular policy measures being studied, and is not always as dramatic as in the calculation described. Nonetheless, the point remains that the choice of grandfathering over auctioning entails substantial economic costs in an economy where existing taxes are high.

The counterpart to the foregone revenues if allowances are grandfathered rather that auctioned is that substantial and arbitrary windfall profits are conferred on polluters receiving free allowance allocations. Free allowance allocations do not simply compensate firms for the costs of an emissions trading system. Instead, an emissions trading system acts to raise the marginal costs of output, because additional output requires additional costly allowances, which have to be bought, or, if already held, could otherwise have been sold. The effect is to raise product prices, so that allowance costs are passed on to customers, regardless of whether allowances were auctioned or distributed free.[16] The windfall gains made by polluters are essentially an arbitrary redistribution within the economy. Auctioning would avoid this redistribution, and, to the extent that firms or industries do experience adverse effects from emissions trading, the revenues gained by auctioning can be partly deployed in targeted measures to offset undesired distributional effects.

[16] These product price effects are not confined to emissions trading, or to economic instruments; they arise with many other environmental policy measures including some forms of command-and-control regulation, as noted by Fullerton and Metcalf, 2001.

The windfall profits conferred by grandfathering can generate costly political lobbying and “baseline inflation” to influence the pattern of free distribution. A further advantage of auctioning is that it avoids giving rise to these potentially wasteful activities.

Auctioning allowances also has better effects on industry dynamics, promoting a more efficient long-term evolution of the regulated industries, avoiding the adverse effects on new entry and exit that can arise when allowances are grandfathering to existing firms. Auctioning ensures that existing firms and new entrants are treated the same, obtaining allowances in the same way, facing the same allowance cost per unit of emissions. Regular auctioning will also tend to increase market liquidity, ensuring that potential purchasers have the opportunity to buy; new entrants cannot be excluded by the unwillingness of incumbent firms to release allowances for sale. Likewise, auctioning tends to promote efficient decisions about whether to cease production and leave the industry. Firms will choose to do this if, when the full costs of their pollution are taken into account, they cannot earn profits. Exit in these circumstances is one of the ways of achieving cost-effective pollution reductions. Allowance grandfathering can tend to inhibit exit, especially if “use it or lose it” allocation rules are applied, so firms that leave the industry forego their allowance allocation.

If auctioning ETS allowances has such substantial economic benefits, what explains the choice of grandfathering, not only of EU ETS allowances, but also of allowances in the large-scale US Acid Rain Programme, and in nearly all other emissions trading applications? Grubb *et al* (2005) discuss two main arguments for grandfathering emissions trading allowances, in terms of compensation and competition respectively.

In many areas, environmental policy makers are reluctant to impose regulation on existing plants as stringent as that applied to new facilities. Regulation is frequently “vintage differentiated” (Stavins, 2005), to avoid imposing unforeseen regulatory costs on the owners of sunk assets. Grandfathering can be seen as an alternative approach to this issue, compensating holders of existing assets for the effects of environmental regulation that was not foreseen at the time of the initial investment. This argument does not necessarily entail grandfathering all allowances; nor does it justify grandfathering in perpetuity. To the extent that a case can be made for compensation, it should presumably be transitional, for the foreseeable lifetime of existing assets; moreover it is clear that any free distribution should not apply to new entrants. Johnston (2006) assesses the legal issues, and contends that any compensation made through grandfathering needs to be proportionate to the profit foregone as a result of the unanticipated additional regulation, if grandfathering is not to be vulnerable to EU state aids legislation. If firms pass forward part of the cost into higher prices, and if they can do substantial abatement at low cost, their additional costs may be very much lower than the benefit they would receive from a completely free distribution of allowances.

Grandfathering is also widely seen as a way of offsetting the impact of competition from foreign producers not subject to similar regulation. Again, the proportion of allowances that need to be grandfathered to offset this effect may be less than 100%. Grubb and Neuhoff (2006) point out that most industries covered by the EU emissions trading system would need to increase prices only a little to cover their net exposure to the costs of emissions trading. In practice, many have increased prices by more than this, passing the opportunity costs of allowances into product prices. This has allowed firms to earn higher profits than in the absence of regulation, even at the costs of sometimes significant reductions in international market share.

5.5 Barriers to carbon pricing: competitiveness and distribution

Two widely-perceived obstacles to more extensive use of carbon taxes or emissions trading are the effects on international industrial competitiveness and the income distribution. Concerns about the (perceived) additional business costs imposed by carbon taxes or emissions trading, and the effects of these costs on the competitive position of businesses in international markets have been a source of political opposition in many countries. In Sweden, this quickly prompted changes to its carbon tax regime in the early 1990s, sharply reducing the additional tax paid by many firms. The issues are, in

principle, most significant where countries introduce a carbon tax unilaterally. It is possible to identify strategies to offset these effects (for example, by using the revenue raised from environmental taxes to reduce other taxes), but concerns remain, especially about the impact of energy taxes on internationally-exposed energy-intensive sectors. These concerns explain the widespread use of sectoral exemptions from environmental taxes and free allocation of emissions trading allowances.

Similarly, others are concerned about the regressive distributional impact of environmental taxes or permits, to the extent that they would raise the prices of goods that form a higher proportion of the spending of poorer households (domestic energy in particular). Again, the revenue raised from environmental taxes or permits auctions provides scope for compensating measures to offset these effects, but they are likely to remain an obstacle of considerable political significance in the UK.

(a) Competitiveness

It is important to recognise that the effects of a carbon tax or emissions trading on the international competitiveness of a country's industry are not uniform across all sectors: the overall impact on competitiveness could be offset, on average, by exchange rate movements. Much the same effect could be achieved by returning the revenues from the tax to the industrial sector, through reductions in other taxes (such as corporate profit taxes or payroll taxes). In each case, the net impact of the carbon tax would be to worsen the relative position of carbon-intensive sectors, whilst improving the competitiveness of sectors of industry with low carbon intensity.

In the long run, some contraction of carbon intensive sectors might be one of the desired outcomes from policies to reduce carbon emissions. If other countries do not impose the tax, however, then these sectors may contract too much, relative to the ideal where all countries impose similar carbon taxes. Part of this contraction may represent "carbon leakage" – international displacement of carbon intensive production. This leakage may impose adjustment costs and loss of profits, without any corresponding environmental gain.

Exemptions were proposed for the six most energy-intensive sectors in the European Commission's 1991 plans for a carbon tax (Commission of the European Communities, 1991). Exemptions have been a feature for most countries introducing carbon taxes, and the 80% reduction in the UK's climate change levy for firms in energy-intensive sectors performs a similar function.[17] These exemptions raise some difficult issues, however. First, they are liable to lead to inefficiency in abatement, because the incentive for abatement differs across sectors. Second, they typically exempt the sectors with the greatest emissions, and hence the greatest scope for abatement; for the same overall effect, more abatement is then required elsewhere in the economy. Third, as with the free allocation of emissions trading allowances, significant resources are likely to be consumed in socially-unproductive but privately-profitable lobbying. The same processes are likely to make it very difficult to discontinue the arrangements. Trade policy shows many examples of the extraordinary durability of sectoral trade protection measures, even when they were originally introduced on a temporary basis, and the same is likely to apply here.

Border tax adjustments provide an alternative way of neutralising the competitiveness effects of carbon pricing policies. The effects of a carbon tax or allowance costs on the relative competitive position of producers could, in principle, be neutralised by levying a tax on competing imports and rebating carbon tax or allowance costs on exports. Imports would need to be charged a tax equal to the carbon tax or allowance costs that would be borne by an equivalent domestic producer, while refunds to exports should reflect the costs of tax or allowances incurred during domestic manufacture. The introduction of border tax adjustments has significant legal impediments, as they might not be compatible with WTO rules (OECD, 2006). They also involve difficult practical and economic issues. First, border tax adjustments may erode incentives for cost-effective pollution abatement, because of

[17] As discussed above, the 80% reduction is conditional on reaching a climate change agreement with the government, in which the sector commits to a legally-binding target for emissions reductions or improved energy efficiency. All the main energy-intensive sectors have concluded such an agreement.

the rebating of the cost of carbon used in producing exported goods. Second, a decision would have to be made whether border tax adjustments would apply to all imports and exports, or only to trade with countries that did not pursue broadly-equivalent environmental policies; the latter case raises considerable difficulties in defining policies of equivalent stringency. Third, it may be impossible to define the appropriate rate of tax adjustment, where domestic firms have a choice of production techniques involving different levels of pollution (Poterba and Rotemberg, 1995). If the border tax adjustments are calculated on the basis of average pollution characteristics, then they will overcompensate some firms and undercompensate others.

Hoel (1996) discusses the relative efficiency of border tax adjustments and sectoral differentiation of carbon tax rates (either in the form of exemptions for some sectors, or differential tax rates). He observes that if countries are not able to levy tariffs on trade with non-signatories to an international agreement to restrain carbon dioxide emissions, then differentiated taxes across sectors may be used to offset the competitive advantage that energy intensive sectors receive in non-signatory countries. However, if countries are able to set tariff rates without restriction, then tariffs should be employed for this purpose, and the optimal pattern of tax rates across sectors is uniform. Wider international agreement on carbon pricing, however, would be preferable to either border tax adjustments or sectoral differentiation, which are only really justifiable as temporary measures while international agreement remains partial.

Within the context of emissions trading, similar issues arise, but have typically been approached differently by distributing allowances for free. While free distribution could, in principle, be confined to internationally-exposed sectors (corresponding to exemption of these sectors from a carbon tax), the EU ETS and most other emissions trading systems have distributed free allowances to all existing producers, and not only those exposed to pressure from international competitors not subject to similar regimes. The effect, as already noted, has been to confer considerable windfall profits on those firms – including the power producers – able to raise output prices to reflect the increased opportunity cost of marginal output. Firms in sectors exposed to more intense international price competition do not have the same opportunity to raise prices, but nonetheless experience higher marginal costs of production – regardless of whether they have been granted the allowances for free or have had to pay for them. Free distribution would thus be wholly ineffective in preventing output reductions and closures by marginal firms, unless rules withdraw allowances from firms that close down. The net effect is messy: firms in internationally exposed sectors face higher costs because of the effect of carbon pricing on opportunity costs of marginal output, but some remain in operation only because closure would lead to the forfeit of future allowance allocations. Emissions trading with grandfathering has an advantage over a carbon tax with sectoral exemption in that it preserves a more uniform incentive for carbon abatement across all sectors, but the complex distortions to business decision-making in exposed sectors need to be weighed against this advantage.

(b) Distribution

The distributional impact across household groups of a carbon tax or other measures that raise energy prices will reflect the impact of the carbon tax on the prices of household electricity, motor fuels, and other goods and services (through the higher cost of energy inputs to production)[18]. The distributional issues are most acute in the case of the additional tax on domestic energy, which in advanced economies has the character of a necessity, forming a much larger part of the budgets of poor households than of the population as a whole.

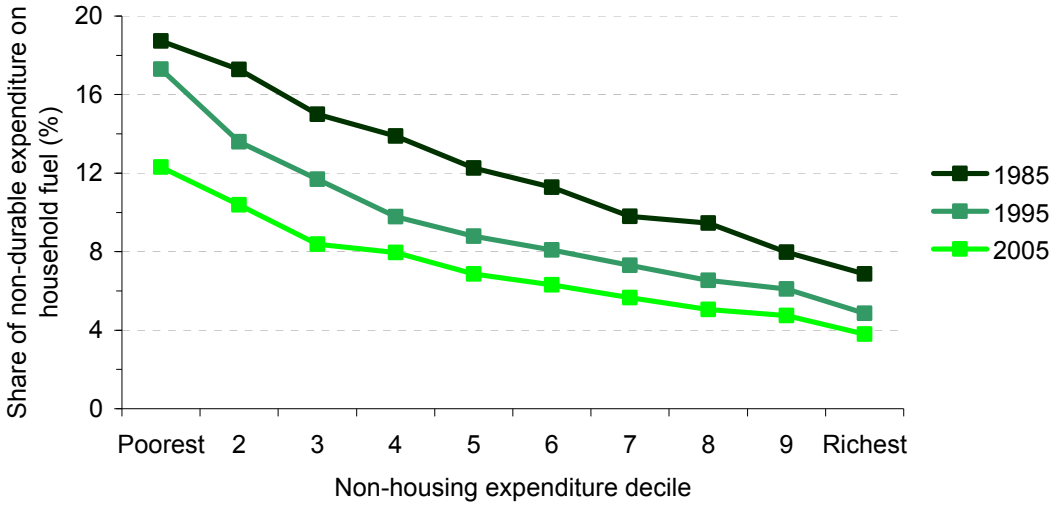
Figure 5.1 shows the proportion of non-durable expenditures devoted to domestic fuel for UK households across the (non-housing) expenditure distribution in the mid-1980s, 1990s and 2000s. In each period, the share of non-durable budget devoted to fuel for the poorest households is around three times larger than for the richest households. Over time, the non-durable budget share of fuel has fallen for all deciles, but at a similar rate, so the relative differences have remained largely unchanged.

[18] Poterba (1991), Symons, Proops and Gay (1994), and Cornwell and Creedy (1996).

By 2005, the lowest spending households devoted around 12% of their budget to fuel, compared to just under 4% for the highest spenders (and 7% across all households).

Taxes on domestic energy thus tend to be regressive, in the sense that extra energy tax payments represent a higher percentage of income (or of total spending) for poorer households than for the better-off. Additional revenues from energy taxation can be used in a revenue-neutral package to provide higher transfers to poorer households, however, and the package can be designed to at least leave poorer no worse off (OECD, 1996; Metcalf, 1999; Dresner and Ekins, 2006).

Figure 5.1: Share of non-durable expenditure devoted to domestic fuel by expenditure decile



Source: Authors' calculations from UK Family Expenditure Survey and Expenditure and Food Survey, various years.
Notes: Deciles are calculated from household expenditures corrected by the OECD equivalence scale. Those spending less than £5 per week on average are excluded, as are households with recorded incomes below £5 per week.

Overall, a wide-ranging carbon tax that does not apply to all emissions may be more or less regressive than a domestic fuel tax alone, depending on the distributional effect of price rises of other products resulting from the tax. Smith (1992) showed that for the UK, a carbon tax would have an overall regressive distributional, because the effect of higher taxation of domestic energy outweighs the progressive effect of higher taxes on motor fuels, but that in many other EU countries this might not be the case.

Another issue is the distribution of the burden of reductions in energy consumption in response to higher energy prices. In the case of the UK, the reduction in energy consumption induced by the imposition of higher taxes on domestic energy appears to be greater amongst poorer households; Pearson and Smith (1991) estimate that the energy spending of the bottom quintile falls in response to a $10 per barrel carbon tax by 12 per cent in volume terms, whilst the average reduction in the volume of household energy consumption is around 7 per cent. Similarly for the US, West (2004) finds that greater price responsiveness of poor households generally reduces the regressivity of petrol taxes.

The social and distributional costs of higher energy prices may be exacerbated if market failures in energy efficiency investment are particularly concentrated amongst low-income households or other vulnerable groups (Brechling and Smith, 1994). Thus, for example, income-related market failures such as those in the credit or housing markets may tend to amplify the distributional cost of a carbon tax. Measures to rectify the underlying market failures, such as building regulations, or home energy audits, would then have the twin merits of reducing the aggregate economic cost of achieving energy efficiency, and helping to reduce the social and distributional cost of higher energy taxation.

6. Road transport externalities and the tax system

This section examines the use of taxes and economic instruments to correct for the external costs generated by road transport. Many different externalities that may vary by time and location are involved, which makes the situation very complicated. Whilst these costs are not always 'environmental' in a strict sense – most obviously congestion – any green policies on road transport will have to consider how best to take into account the various externalities involved and so it is appropriate to consider the full range of the problem in this section.

We begin with an assessment of the scale of the different externalities involved and then examine the recent history of UK tax policy towards road transport. We consider the options for transport tax reform, starting with a discussion of 'first-best' policies that would separately and accurately price each of the externalities involved. The focus here is on the prospects for an explicit congestion charge introduced at a national level and on the appropriate design of such transport taxes. We then consider how tax policy might look, were such a charge either technologically or politically constrained, and whether existing fuel taxes are set at the right rate. Drawing on evidence from the US, we consider the extent to which multi-part tax and non-tax instruments could be used to approximate the first best outcome.

6.1 The external costs of road use

A study by Sansom *et al* (2001) identified the key externalities of road use as operating costs, accident costs, air pollution, climate change, noise, and congestion. Their range of estimates for the scale of the marginal externality in each case for the average motorist is presented in table 6.1 below.

Table 6.1: Estimated marginal external road costs (pence/km), 1998 estimates

Externality	Low estimate	High Estimate
Operating costs[19]	0.42	0.54
Accidents	0.82	1.40
Air pollution	0.34	1.70
Noise	0.02	0.05
Climate change	0.15	0.62
Congestion	9.71	11.16

Source: Sansom *et al* (2001).

The total external cost of an extra kilometre travelled is about 11.5–15.5 pence, of which congestion costs are by far the largest element, at around three-quarters of the total. Congestion externalities, however, vary hugely according to time and location –marginal externalities were estimated at around 86p/km for central London peak time roads but just 3p/km for non-major rural roads. A more recent study by the Department for Transport (2006) estimated the marginal external congestion costs in 2003 across the British road network and again found considerable variation by location: almost 900 roads in and around major conurbations had a cost in excess of 56p/km but almost 6,000 roads in more rural areas had costs of less than 2p/km.

Other components of the external costs also vary with a variety of factors, such as vehicle and fuel type, engine size, location, vehicle maintenance and driving style. The key question for this section is to consider what policy or range of policies may be best suited to take account of this range of externalities and how far existing policy succeeds in doing so. To pre-empt our conclusions somewhat, the broad story is that whilst in some cases the variation in costs per kilometre are closely proxied by one of the available tax bases – climate change externalities are strongly related to fuel consumption, for example – in other cases, notably congestion, existing taxes are inadequate to reflect the externalities involved. The range of decisions that individuals make need to be appropriately guided by

[19] This mainly refers to road wear and tear costs resulting from a marginal increase in vehicle kilometres

social costs of choices such as whether to own a car, what type of car, when to drive, and where to drive. Thus, fuel taxes need to be supplemented by other measures to reflect externalities that are not closely related to fuel consumption. Congestion charging is the obvious approach, but formidable costs and political constraints may preclude establishing a wide-ranging national scheme at present. Nevertheless, artful combinations of existing tax bases can better approximate an ideal outcome than at present.

6.2 Current UK road transport taxes

Motor fuel taxation is a significant source of Exchequer revenues.[20] Duties on hydrocarbon oils are expected to generate receipts of around £24.9 billion in 2007/8, or 4.5% of total revenue.[21] In the UK tax system, more revenues are raised only by the income tax, National Insurance contributions, VAT, and corporation tax.

Most road fuel sold in the UK is either ultra low sulphur petrol (ULSP) or diesel (ULSD). Both are taxed at a rate of 50.35p per litre (around £1.91 per US gallon), though lower rates apply for vehicles powered by biofuels or LPG. Between 1993 and 1999, real rates of duty were increased as the default option at each Budget – known as the "fuel duty escalator". Between 1997 and 1999, the accelerator increased fuel duties by 6% above inflation each year. By mid-1999, real duties were around 55p–60p/litre (in 2007 prices). The accelerator was abandoned in the pre-Budget Report of November 1999, and the high price of petrol sparked protests and blockades of oil refineries in Autumn 2000. Nominal duty rates were increased only once between April 2000 and November 2006, leading to a significant real-terms decline in duties, though Budget 2007 pre-announced inflation-linked increases that would be enacted in 2007, 2008 and 2009.[22]

Real duty rates are now around their lowest in almost a decade, as highlighted in figure 6.1. The duty escalator period of 1993 to 1999 is clearly visible from the 'saw-tooth' pattern of annual real duty rate increases that are gradually eroded by inflation within each year.

This decline in real duties has corresponded with a decline in fuel tax revenues as a share of national income since 2000. This decline has been largely responsible for the falling share of total green tax revenues in GDP, which in 2006 fell to the lowest level since the late 1980s[23]. Figure 6.2 shows the receipts from fuel duty (both including and excluding the VAT charged on top) as a share of national income since the mid-1970s. In 2006, the figure fell to the lowest level since the start of the escalator period.

The decline in revenues relative to national income is also due in part to the significant switch towards diesel fuel that has taken place in recent years as car manufacturers have developed diesel engines for domestic car owners. Diesel engines have a much higher fuel efficiency than petrol engines, and so fuel purchases can be reduced even as total distance driven rises. In 1997/8, 52% of all fuel released for consumption was petrol compared to diesel's 32% share. By 2006/7, it is estimated that the petrol share had fallen to 44%, the same as diesel. This trend looks set to continue.

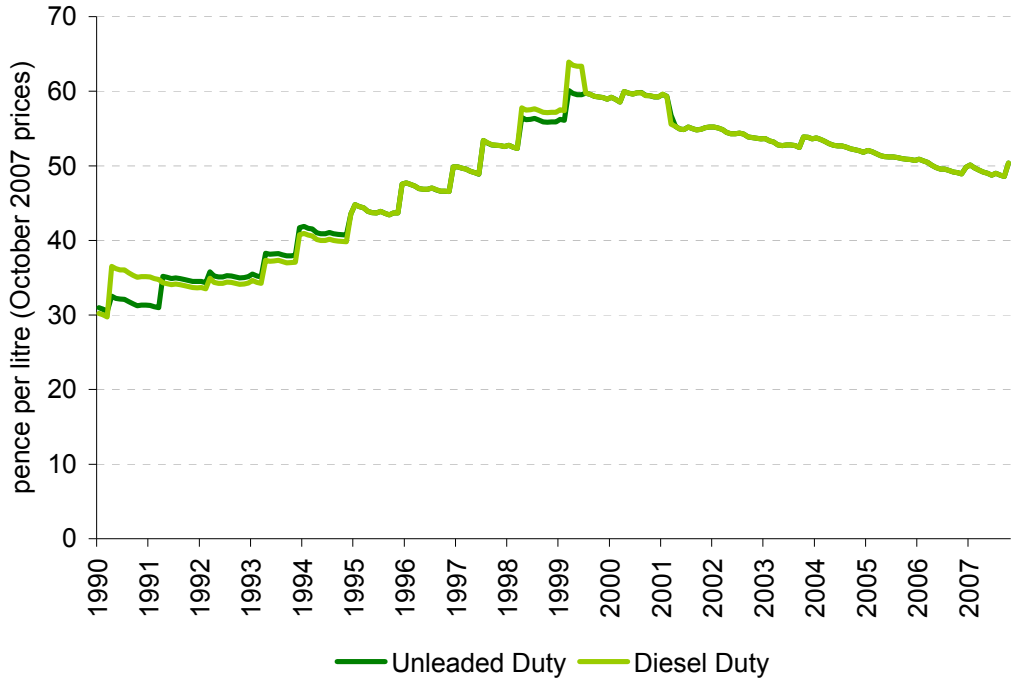
Despite the real decline in fuel duties over the last few years, the UK pump price of petrol and diesel has a higher tax component than any other EU country. Figures from the Department for Business, Enterprise and Regulatory Reform (DBERR, formerly the DTI) for April 2007 show that taxes (including duties and VAT) made up 67% of the pump price of a litre of unleaded fuel in the UK, compared to an (unweighted) EU-25 average of 57% and an (unweighted) EU-15 average of 60%. For

[20] See section 5 of Leicester (2006) for more details of UK fuel taxation
[21] Estimate from October 2007 Pre-Budget Report

[22] The real decline in duties since 1999 was justified in part by the higher crude oil prices that raised the pre-tax fuel price. However we see no real economic justification for taxes to adjust to the pre-tax price: the tax per litre should vary with inflationary increases in monetary *damages* from fuel use, not with increases in fuel prices per se.
[23] See Etheridge and Leicester (2007) for more on total green tax revenues.

diesel, the differences are even greater: the tax component in the UK is 66% compared to EU-25 and EU-15 averages of 51 and 53%, respectively. Interestingly, *pre-tax* fuel prices in the UK are amongst the lowest in the EU, though the variation across the 25 member states is relatively small compared to the variation in post-tax prices. Old EU member states (i.e. those that joined before 2004) typically have higher fuel prices than new member states, and this effect is is tax-driven, as pre-tax prices are similar across most countries. Importantly, however, the UK is alone in the EU-25 countries in making no tax differential between unleaded and diesel fuels; all other member states have substantially lower taxes on diesel than on unleaded (we discuss this point further in section 6.4.1).

Figure 6.1: Real rates of duty for most commonly purchased unleaded and diesel fuels, 1990 – 2007



Notes and source: Calculated from DBERR data; duty rates are deflated to October 2007 prices using the all-items RPI index. This figure updates figure 5.5 of Leicester (2006).

Aside from fuel duty, the other major road transport tax in the UK is Vehicle Excise Duty (VED), an annual per-vehicle tax that varies according to the type of vehicle, the age of the vehicle, engine size and (since 2001) the vehicle's CO_2 emissions. For new cars, the top rate payable by the most polluting cars will be £400/year, whilst the least polluting vehicles will be exempted altogether. In their most recent annual review of the new car market, however, the Society for Motor Manufacturers and Traders (2007) note that less than 300 vehicles in the exempted "A" class for VED were registered in 2006. Nevertheless it appears that the graduation of VED by CO_2 emissions has coincided with a decline in the average emissions of new cars, though the extent to which this is directly attributable to the VED system is unclear. In 2000, around 34% of new cars sold emitted more than 186g of CO_2/km compared to 21% in 2006; the proportion emitting less than 150g of CO_2/km rose from 19% to 37% over the same period.

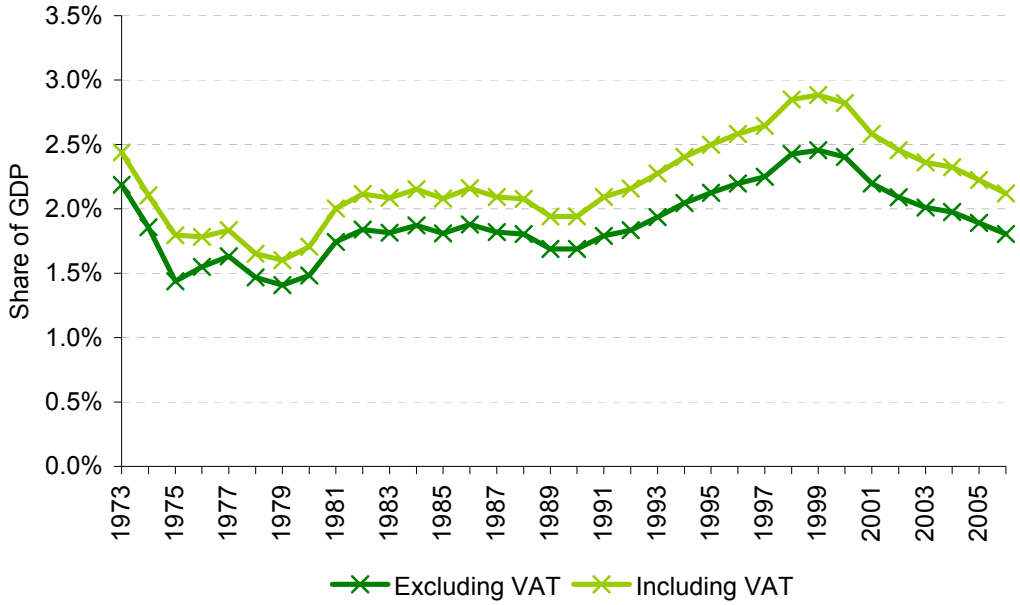
Company car and fuel taxes were also reformed in 2002 and 2003, respectively, to make the liability reflect the emissions rating of the vehicle supplied by the employer. As company car fleet vehicles make up more than half of new vehicles purchased each year, obvious benefits arise from providing incentives to firms to select lower emissions vehicles. An estimated value of the benefit in kind from the vehicle and fuel is derived by multiplying the list price by a percentage that varies from 10–35% for petrol cars (13–35% for diesel cars), with a lower multiplier for lower emissions cars. The higher

multiplier for diesel cars presumably reflects some belief about additional externalities from diesel fuels, as discussed in more detail below, though it is strange that the 'diesel supplement' should be a fixed 3 percentage points up to a cap of 35%. The result is that a low emissions diesel vehicle with the same list price as a corresponding petrol vehicle will attract an extra 30% tax, whereas a high emissions diesel vehicle will not attract any extra tax. More sensible, perhaps, would be an additional percentage increase for diesel vehicles. It is also not clear why diesel cars would be more heavily taxed, while diesel fuel itself is typically taxed at the same rate as petrol.

6.3 A 'first-best' system of road transport taxes

The key feature of a tax system designed to ensure that individual decisions properly take account of external costs is that the taxes should impact on the marginal decisions regarding vehicle purchase and fuel use at a level that reflects the *marginal* externalities generated. It is not that taxes should reflect *average* externalities from motoring or that total tax receipts from road users should cover the total costs.

Figure 6.2 Fuel duty receipts as a share of national income, 1973 – 2006



Source: Calculated from ONS figures for fuel duty receipts (series GTAP), VAT on duty (CMYA) and GDP (YMHA).

Clearly the existing system of road transport taxes cannot adequately take account of the range of externalities associated with road use, which may vary by time and location. A litre of fuel is taxed at the same rate no matter where it is purchased, and VED is a lump-sum tax that does not vary at all with distance driven.

Fuel duties *are* effective at capturing the externalities relating to climate change: the cost of a tonne of CO_2 emissions is the same no matter where and when it is released, and emissions are closely related to fuel consumption. Further, variation in fuel duty by fuel type has been successful in influencing consumers and manufacturers to switch to lower-emissions fuel types (for example, unleaded petrol enjoyed a significant tax advantage relative to unleaded petrol in the 1980s and 1990s). Current tax reductions for alternative fuels may help further fuel switching in the future, though considerable debate concerns the environmental benefits of alternative fuels (see section 6.4.1 below).

VED may help capture the external costs of road damage, and it may also help influence consumers to purchase less polluting vehicles (particularly since its reform to be based on emissions in 2001).

It is clear, however, that an optimal system of road transport taxes would require taxes that could be precisely targeted against the various externalities involved. In particular, road pricing should charge drivers according to the distance driven, location and time. If so, then prices would vary to take account of congestion and noise externalities, leaving fuel duties to capture environmental externalities. It is not clear that a restructuring of the road transport tax system along these lines would raise any significant extra revenues, but it would send a much more precise signal to motorists and would significantly affect patterns of traffic and perhaps overall traffic levels. This section considers how such a pricing system could be enacted and possible difficulties in doing so.

6.3.1 Congestion Pricing

In 2006, a report for the Department for Transport headed by Sir Ron Eddington came out strongly in favour of a national system of road pricing where the charge varied by location, time, and distance driven. A scheme would require vehicles to be fitted with on-board units (OBUs) that could monitor their location; drivers would be sent bills over a set period for their driving. Flexibly designed, such a scheme could closely capture the marginal external costs on a road-by-road basis and so in principle induce the most efficient use of the road network and substantially reduce the need to invest in extra capacity. Eddington's projections, drawing on work carried out as part of a feasibility study into road pricing by the Department for Transport (2004), suggested that a complex pricing scheme with 75 different pricing levels could reduce congestion by 50% by 2025 relative to a no-charging baseline and generate gross welfare benefits in the order of £25 billion per year. Revenues were estimated at around £8 billion per year.

Of course implementing such a scheme has enormous practical difficulties. Identifying the correct price to charge for each road at each time would be extremely difficult. The marginal externalities in the Sansom and Department for Transport studies highlighted above assume no road pricing; in a world where roads are priced, the congestion externality from an additional kilometre driven would be considerably lower as congestion levels fall. Nash *et al* (2004) suggested that model estimates showed that marginal external congestion costs in a post-pricing world could be, on average, just 20% of the costs before pricing. One key problem with identifying the appropriate prices to charge is the lack of practical experience on which to draw, with most estimates coming from traffic models. Estimates of the response to the central London congestion charge proved ultimately too pessimistic – the charge reduced traffic levels and congestion by more than was predicted (and hence the scheme generated lower net revenues than had been forecast).

Another problem would be ensuring that the pricing scheme is transparent and well understood by motorists. With a large number of price bands that vary by time as well as location this would obviously be difficult, with an obvious trade off between simplicity versus precise targeting of the marginal externality. Modelling results in Eddington (2006) and the Department for Transport (2004) study suggest that simpler schemes with fewer pricing bands could still generate substantial benefits and revenues – for example, by having only ten price bands or by targeting the scheme on urban areas where the congestion costs are greatest.

A severe initial obstacle to a complex road pricing scheme would be technological – the costs of implementation and annual running would be extremely high. The Department for Transport feasibility study produced a wide range of estimates for the initial set up costs of £10–£62 billion. This range reflects huge uncertainties over the scope of the charge (such as the complexity and gradations of the prices by time and location). It also reflects technological costs and 'optimism bias' (the idea that initial cost estimates are almost always revised up). The main costs would be fitting each existing vehicle in the fleet with an on board unit and the costs to develop and procure the administrative side of the system in the first place. Annual running costs could also be high – Eddington suggested a figure of £2–£5 billion per year (largely for administration and access to telecommunications networks for location monitoring of vehicles). The costs of compliance and enforcement would also need to be considered.

Whilst costs and uncertainties constrain the ability of governments to implement a national road pricing scheme, an additional political problem arises. If the perception is that road pricing would be introduced on top of existing motoring taxes, any such proposal is likely to face substantial opposition (as evident in online petitions that arose in 2006/7 against any such scheme). A key question is therefore to what extent current taxes could be reduced or replaced by a road pricing scheme.

Box 6.1 The London Congestion Charge

An explicit congestion charge was introduced in central London in 2003.[24] The intitial charging zone covered 21 square kilometres mainly in the City and the City of Westminster. Originally set at £5 per day, the charge rose to £8 per day from July 2005 for any vehicle entering or parking in the zone between 7am and 6.30pm on a weekday (the charge now ends at 6pm). Exemptions are provided for taxis, motorcycles, pedal cycles, buses, emergency service vehicles, those holding a disabled person's badge, and some alternative fuel vehicles. Residents of the zone are also entitled to a 90% discount. From February 2007, the zone was extended towards West London[25].

The latest impacts monitoring report (Transport for London, 2007) suggests that traffic within the original zone was around 20% lower than pre-charging periods in 2006, and congestion levels were around 8% lower. Earlier reports had suggested much larger falls in congestion in the order of 25–30%; the 2007 report notes that a large increase in roadworks within the zone in 2006 had contributed to a rise in congestion and that the charge was still reducing congestion levels by around 30% relative to a world with no charge. Revenue from the charge for 2006/7 amounted to £123 million net of running costs. These revenues are hypothecated towards funding public transport in London. The congestion charge is also cited as one of the major factors contributing towards reduced emissions from transport in London, by allowing traffic to flow more freely and by reducing time spent idling in traffic queues. Latest estimates suggest the charge has reduced road traffic emissions of CO_2 by around 16% within the charging zone.

Sansom *et al* (2001) estimate the external costs from congestion in central London during peak hours at around 86 pence per vehicle kilometre driven. The London charge does not vary according to time spent or distance travelled within the zone, nor with the time at which the vehicle first arrives in the zone; payment of the £8 entitles the driver to full access for the day. Thus, it does not represent an attempt to capture the marginal external costs of congestion directly, but monitoring the distance driven inside the zone would be difficult. In addition, the exemptions and discounts suggest that environmental, political and social considerations have also been built into the charge, further removing it from an explicit congestion payment.

6.3.2 Fuel duty after congestion charging

Assuming a national system of road pricing is established that accurately captures congestion (and possibly noise) externalities, what role would remain for fuel duty? As discussed earlier, fuel taxes are good at capturing the carbon externality from motoring and may be the best current way of capturing other environmental and road damage costs.

Currently, fuel duty for most fuel types is 50.35p per litre. Taking a 'typical' fuel efficiency of around 40 miles per gallon, a litre of fuel will allow a vehicle to travel around 14.2 kilometres[26]. Thus 'per kilometre' duty is around 3.5 pence. Taking the estimates from table 6.1, excluding congestion and noise, the external costs of a marginal kilometre are around 1.7–4.2 pence per kilometre (if we also exclude accident costs the range is around 0.9–2.8 pence per kilometre). This calculation suggests that in a world with a first-best congestion charge, current rates of duty would be towards the upper end of levels that might be justified by other motoring externalities.

Modelling results carried out by the Department for Transport as part of their 2004 feasibility study showed that with a 'revenue neutral' scheme whereby revenues from road pricing were recycled in the

[24] Blow *et al* (2003) discuss the workings of the scheme and background to it in depth.
[25] A study by Santos and Fraser (2006) argued that the benefit to cost ratio of the extension would be in the range of around 0.74 to 0.9.
[26] 40 miles per gallon = 64.4 kilometres per gallon (1 mile = 1.61km) = 14.2 km per litre (1 litre = 0.22 UK gallons).

form of reduced fuel duty, net total benefits would be around £8 billion per year by 2010, similar to the £10 billion or so from a scheme where revenues were not recycled. The difference arises largely because estimated combined revenues from fuel duty and congestion charges actually fall, by around £2 billion per year, if the revenue recycling occurs[27]. This is due to traffic flowing more efficiently after the charge, improving fuel efficiency and thus depressing fuel purchases relative to a no-charge environment. What seems clear is that the scope for introducing significant revenue-raising reforms to road transport policy through congestion charging is limited, since it would be likely to be accompanied by reductions in existing transport taxes.

6.4 'Second-best' road transport taxation

Given the significant costs, technological and perhaps political constraints that may prevent congestion pricing schemes being established, it is worth considering how policy could develop without it. Specifically, we consider the extent to which existing tax bases can approximate the first-best outcome.

6.4.1 Optimal fuel duties

We start by considering the case where the only available tax the authorities can use is fuel duty, and where duty rates can only vary according to the type of fuel and not the type of vehicle. VED and local road pricing and toll schemes aside, this case fairly close approximates the current UK road transport taxation situation. What rate of fuel duty should be set to reflect the external costs of motoring?

Parry and Small (2005) calculate the optimal level of fuel taxes in the UK and in the US based on estimates of the marginal externalities from motoring, allowing for the fact that the fuel duty induces improved fuel efficiency, and allowing for interactions between fuel taxes and other taxes. Under their central modelling assumptions, they derive an optimal fuel tax for the UK of $1.34/US gallon, which equates to around 18 pence per litre (significantly below the current rate in the UK of 50.35 pence).[28] Their estimate for the fuel-related pollution caused by carbon emissions is less than 1 penny per litre, and for congestion around 8 pence. In calculating their fuel-related pollution costs, however, they choose $25 for the marginal damage caused by a tonne of carbon emissions, drawing on various estimates from the international literature. The UK *Stern Review* estimates the damage cost at closer to $100/tC (and perhaps considerably higher under 'business as usual' emissions). Even at this level, assuming everything else unchanged, the optimal UK tax would rise only by around 3–4 pence per litre. Their central estimate of the marginal external congestion cost is only around 2.2p/km, only around one quarter of the low-end estimate given by Sansom in table 6.1[29]. Using *Stern* estimates of climate change costs and much higher marginal congestion costs, the Parry and Small approach could lead to a fuel tax rate much closer to the existing UK figure. Clearly, estimating optimal fuel taxes is extremely complicated and sensitive to parameter estimates.

A key feature of UK policy is differentiation of fuel tax rates to encourage fuel substitution: 'alternative' fuels such as bioethanol are also favoured in their tax treatment, for example, and unleaded petrol is taxed more lightly than leaded petrol (which is now virtually never bought). The major fuels sold for private motoring are Ultra Low Sulphur Petrol (ULSP) and Ultra Low Sulphur Diesel (ULSD), both of which are taxed at the same rate, 50.35p/litre. In the past, petrol and diesel have been taxed differently; typically diesel attracted a lower rate of duty because carbon emissions from diesel are less than those from petrol. However, a 1993 report by the Quality of Urban Air Group (QUARG)

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[27] Clearly this implies a distinction between 'revenue neutral' at the time of implementation and how revenues may evolve over time.

[28] We use a market exchange rate of around £1 = $1.95 and a conversion of 1 litre = 0.2642 US gallons.

[29] The Parry and Small estimates of the marginal congestion cost are derived from an uprated estimate made by Newbery (1990) of costs of around 3.4p/km that are then reduced because of the way the congestion estimates enter into their specification as adjusted by fuel price demand elasticities. Sansom *et al* (2001) discuss their congestion estimates relative to Newbery's and argue that the Newbery figures are based on estimates using data from 1985, at which time traffic volumes were considerably lower.

suggested that diesel cars have higher particulates emissions that affect respiratory illness, with a severe impact on urban air quality. The differential for diesel was subsequently removed.

As figure 6.3 made clear, other European countries have considerably lower taxes on diesel than on petrol, which may represent an attempt to reduce the tax on fuel for commercial vehicles from a Diamond/Mirrlees efficiency perspective (reducing the tax on intermediate inputs). This perspective suggests that diesel fuel duty should be lower if diesel fuel is a proxy for commercial as opposed to residential motor fuel and if fuel taxes exceed the marginal external costs in order to raise revenue. Clearly the growth of diesel engines in non-commerical cars makes diesel fuel an increasingly poor proxy for a commercial input, and we see no *environmental* reason why commercial road use should be tax favoured.

From a purely environmental perspective, having some assessment of the relative damage caused by climate change and through particulates is crucial for estimating appropriate differential tax rates for petrol and diesel. Using *Stern* estimates of the costs of carbon points to a more favourable tax treatment for diesel, whereas using the lower estimates from other literature may point to the opposite.

A non-trivial issue for diesel fuels relates to 'cross-border' shopping and might explain why diesel duty rates are often lower in other EU countries: commercial enterprises that operate abroad can refuel vehicles in low-duty countries rather than at home. The most obvious examples are commercial vehicles crossing between Northern Ireland and Eire and those using the Channel Tunnel. The marginal additional revenue to be gained from such firms from small changes to diesel duty rates is likely to be close to zero. HMRC estimates for 2005/6 suggested that the non-UK duty paid market share of diesel fuels in Northern Ireland is around 40–50%, with associated revenue losses of around £190m–£230m. For petrol, the corresponding figures are around 6–24% and £20m–£80m[30].

A more recent focus has been on alternative fuels. Particularly in the US, considerable resources are now being devoted to bringing them onto a more commercial footing, especially biofuels. These alternative fuels, either alone or blended with traditional fuels, may emit less greenhouse gases in combustion. Over the lifecycle, however, taking into account emissions associated with biofuel production, the environmental impact may be considerably more than that of more traditional fuels. More evidence is urgently needed to determine the extent to which such fuels should be tax favoured. At the moment, however, they account for an extremely small share of the UK market: in 2006/7, biodiesel and bioethanol combined made up around 0.6% of fuel consumed. However the Renewable Transport Fuel Obligation (see below) will require this proportion to rise substantially in the next few years.

6.4.2 A Multi-part instrument

Clearly policymakers face huge difficulties both in implementing an optimal road transport taxation policy and in relying on fuel duty alone as the dominant transport tax. Road transport taxes have enormous technological, cost and political opposition barriers, while fuel duty alone is inadequately differentiated to deal with the range of externalities involved. However, a range of instruments together can approximate the optimal outcome using existing tax bases and technologies.

So far, policymakers in most nations have addressed vehicle emission problems with a variety of mandates and restrictions. In the UK, passenger cars are required under European legislation to be sold with information about their fuel consumption and CO_2 emissions levels, and the latest "Euro IV" emissions standards limit the emissions of carbon, nitrogen oxides and particulates from petrol and diesel engines in passenger cars and commercial vehicles. From 2008, fuel suppliers will be required to ensure that 5% of their sales will be biofuels by 2010 under the Renewable Transport Fuel Obligation. Suppliers will be issued with certificates according to their sales of biofuels. These certificates can be traded. So the plan will have the flavour of a "cap and trade" scheme.

[30] See http://www.hmrc.gov.uk/pbr2007/mitl.pdf, tables 3.8 and 3.9.

Such regulations can guarantee vehicle emission reductions, but they do not provide much flexibility or incentives to go beyond them. As an alternative, a direct tax on vehicle emissions would provide these ongoing incentives and would minimise vehicle emissions at least cost, but it would require complex measurement of individual vehicle emissions. Technological advances may make such a tax feasible in the future: Harrington and McConnell (2003) discuss ways in which this might be achieved, though each method has problems.

Alternative incentive instruments that apply to market transactions *rather* than to emissions may therefore be needed. One possibility would be to bring fuel suppliers into the European Emissions Trading Scheme, such that emissions would be priced into the final pump price. This scheme covers carbon emissions only, however, and the existing fuel tax can already be used effectively to cover carbon emissions. Another alternative would be to introduce a general carbon tax that covers all emissions, as discussed in the previous section; this tax would then apply to the carbon content of vehicle fuels, again as a replacement for existing duties. However, the carbon tax does not solve the problem of pricing other transport emissions and externalities.

Recent research considers the availability of alternative instruments. As a benchmark for comparison, consider a world where the emissions tax is perfectly available and enforceable, and use a model to calculate the theoretically-ideal set of driving behaviours that would minimise the costs of achieving a given air quality. Then suppose that the ideal emissions tax is *not* available, and consider alternative instruments. In order to take advantage of the cost-reducing characteristics of incentive instruments, policymakers can consider alternative taxes and subsidies on various market transactions that reflect choices affecting emissions.

Fullerton and West (2002) build a simple theoretical model in which many different consumers buy cars with different characteristics and fuels of different types. They specifically model the consumer's choice of engine size, pollution control equipment, vehicle age, fuel cleanliness, and amount of fuel, capturing the most important determinants of emissions other than driving style. They also capture consumer heterogeneity: individuals differ by income and tastes for engine size and miles. Using this model, they confirm that a single rate of tax on emissions of all different consumers minimises the total cost of pollution abatement, by inducing each consumer to change their behaviour to a different extent for each method of pollution abatement (such as buying a smaller car, newer car, better pollution control equipment, cleaner petrol, or less petrol).

Given the difficulties of direct emissions taxation, alternatives might be limited to charging the same uniform rate for all consumers – one tax rate per unit of engine size, one tax rate that depends on vehicle age, and one tax rate on each grade of petrol, no matter who buys it. A set of uniform tax rates can use all available information (including how these various vehicle characteristics are correlated). If it is still limited to uniform rates across all consumers, it does not perform as well as the emissions tax, but it out-performs all other *available* incentive-based policies.

In a computational model, Fullerton and West (2000) employ actual data for more than a thousand individual cars and their owners to assess the potential welfare gains of these second-best policies relative to an ideal emissions tax. The main result from this model is that the second-best combination of tax rates on engine size, vehicle age and fuel type achieves a welfare gain that is 71% of the maximum gain obtained by the ideal-but-unavailable tax on emissions. If only a petrol tax can be employed, the welfare gain is 62% of the gain from the ideal emissions tax. Thus a petrol tax is the key ingredient of any market-based incentive emissions policy where the ideal tax is unavailable.

The focus of existing research has been approximating an optimal emissions tax; less has been said about approximating an optimal congestion charge, though some of the research examined in section 6.3.1 above suggested that simpler congestion schemes targeted on urban areas may still achieve substantial welfare benefits. Another key question concerns policy towards public transport; subsidies here may be a crucial component of any multi-part incentive structure – the London congestion charge

has been introduced at the same time as a wide-ranging investment in the capital's bus services. Over the last 20 years, the price of private transport has been constant compared to other prices (and falling over the last five or six years). Yet the price of public transport has risen substantially, in the order of 25%. Clearly public transport cannot be neglected in the consideration of transport tax policy as a whole.

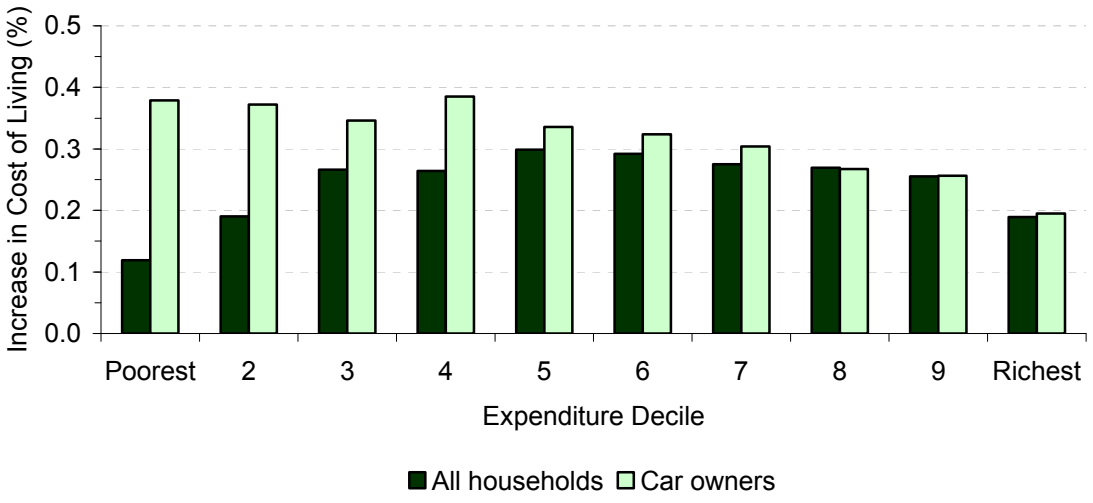
6.5 Distributional aspects of road transport taxes

The very poor do not own cars and do not buy petrol, so a tax on petrol does not hurt the poorest families. Using data from the 2005/6 Expenditure and Food Survey (EFS), households are broken into ten decile groups based on their expenditure[31]. Amongst the lowest-spending 10% of households, only 29% are car owners, compared to 77% in the middle of the expenditure distribution and 93% for the highest spending households. Only 4% of the poorest households own more than one car, compared to 28% of the middle decile and 50% of the richest.

Figure 6.3 shows the impact of a 5% rise in petrol prices across the expenditure distribution. The bars show the average increase in the cost of living that results from the price rise – darker bars represent the increase across the whole population, and the lighter bars the increase across just the population of car-owning households. Over the whole population, the impact is lowest for the very poorest and the richest households, with the biggest impact on those in the middle of the spending distribution. Amongst car-owners only, poorer deciles are hit slightly harder than those in the middle, whilst the richest car owners have the smallest increase in the cost of living.

Even if a petrol tax is regressive, the existing tax and benefits system can be used to compensate. Low income drivers cannot in a practical way be exempted from fuel taxes altogether (or from congestion charges). Lower tax rates in rural areas would also have adverse effects, as people drive from urban areas to take advantage of the reduced rates.

Figure 6.3: Distributional impact of 5% petrol price increase, 2005/6



Source: Calculated from Expenditure and Food Survey 2005/6. Note: Assumes no behavioural response to price rise.

A move towards congestion charging would have substantial redistributive effects that would need to be modelled alongside concurrent reforms to other road transport taxes. The most obvious effect is that urban commuters would be negatively affected, whilst rural car owners would benefit substantially.

[31] We present results based on household expenditure rather than income, though the story is similar for both measures of well-being. Expenditure may better capture household's long-run living standards than does a snapshot measure of current annual income.

6.6 Conclusions

Conceptually at least, the principles of road transport taxation are quite well understood, and a growing body of evidence covers the extent to which different approaches would have considerable benefits relative to the existing reliance on fuel duties. However,any wholesale reform of the tax structure will depend crucially on public support as well as on technological developments that may reduce the cost of initial investments and ongoing financial commitments. Part of any public concern may be the perception that tax reform would have adverse distributional consequences, though this effect can be modified by offsets. In any event, a move towards congestion charging would be extremely hard to justify without commensurate reductions in fuel duty, which would leave overall revenues unchanged or possibly even slightly reduced.

7. Aviation Taxation

Aviation represents one of the fastest-growing sources of greenhouse gas emissions in the UK and other developed economies. Total CO_2 emissions from domestic and international aviation have increased from around 7.29 million tonnes (1.0% of the national total) in 1970, to 16.95 million tonnes (2.8% of the total) in 1990, and further to 37.47 million tonnes (6.3% of the total) in 2005[32].

The 'greenhouse effect' caused by aviation is greater than that caused by carbon alone. Additional effects are attributable to emissions of water vapour, nitric oxide (NO) nitrogen dioxide (NO_2), sulphur oxides (SO_x) and soot (Penner *et al*, 2003). The IPPC estimated in 1999 that the total effect that can currently be quantified is between 2 and 4 times the effect of carbon dioxide alone. Much of the uncertainty surrounding this estimate has to do with the unknown effect of the formation of aviation-induced cirrus clouds. Sausan *et al* (2004) also investigated the relative effects of these gases and came to a qualitatively similar conclusion to the IPCC. Taking a broader view of all greenhouse gases, figures from the UK Environmental Accounts suggest that in 2005, air transport accounted for 5.8% of emissions, compared to 2.5% in 1990.[33]

Aviation taxation is enormously complicated by international considerations. International aviation is governed by the International Civil Aviation Organization (ICAO), developed as a result of the 1944 Chicago Convention, whose resolution on the taxation of aviation fuel, for example, states that:

> "… the fuel, lubricants and other consumable technical supplies contained in the tanks or other receptacles on the aircraft shall be exempt from customs and other duties."[34]

Further, European rules on aviation as well as bilateral Aviation Service Agreements (ASAs) between different countries can act as constraints on government policy towards airlines. For example, Norway introduced a CO_2 fuel tax on all flights in January 1999, but by May that year had to abandon the tax for international flights because it violated the rules of ASAs between Norway and other countries.[35] In 1999, Sweden had to remove a domestic aviation fuel tax, because it conflicted with the EU Mineral Oil Directive. As a result, new rules were drawn up that allow domestic flights to be subject to fuel taxes and that will allow intra-EU flights to be similarly taxed.

The key instrument for aviation taxation in the UK has been Air Passenger Duty (APD), a tax levied on airlines on a per-passenger basis, introduced in the November 1993 Budget and implemented in November 1994. It was not introduced with explicitly environmental incentives in mind, and it has often been seen as a revenue-raising measure at a time when the Government finances were in relatively poor shape. Given that air tickets are exempt from VAT, however, and that airline fuel is exempt from

[32] http://www.defra.gov.uk/environment/statistics/globatmos/download/xls/gatb05.xls
[33] See table 2.3 of http://www.statistics.gov.uk/downloads/theme_environment/EADec2007.pdf
[34] See ICAO (2006).
[35] See ECON Analyse (2005) for more details.

duty, aviation might be relatively under-taxed compared to its increasing contribution to emissions. APD represents one relatively straightforward way in which aviation can be taxed.

APD varies according to the destination and the class of the flight. Broadly speaking, European destinations are taxed at £10 for standard-class flights and £20 for other classes, whilst those rates for non-European destinations are £40 and £80, respectively. These rates apply from February 2007, having been doubled in the December 2006 Pre-Budget Report. APD raised around £1 billion in 2006–07 before the doubling and are forecast to raise £2 billion in 2007–08 after the doubling of the tax rates.

Pearce and Pearce (2000) estimate the marginal external costs from pollution and noise at Heathrow airport for different models of aircraft, and argue that the per-passenger tax should be £3 on a short-haul flight on a Boeing 747–400, and £15 on a long haul. Even after uprating to current values, these figures are below current APD rates, though APD clearly also has a significant revenue-raising component given the absence of other forms of aviation taxation.

APD can be considered an environmental tax to the extent that it reduces the demand for flights (and that the demand is not instead taken up by more polluting forms of transport). The Department for Transport works on the assumption that a 10% rise in the price of flights reduces demand by 10% (i.e. the own-price elasticity is –1.0). This elasticity could vary according to the purpose of the flight (business flights are much less price elastic than pleasure flights).[36] The rates do not vary, however, according to the emissions of the aircraft or the total distance travelled within Europe or beyond. As a per-passenger tax, it provides no incentives to airlines to ensure that planes depart as fully loaded as possible, and it is not applicable to freight flights. In the October 2007 Pre-Budget Report, the Government announced that from 2009, the basis of the tax would change from the passenger to the aircraft, a reform previously advocated by both the Liberal Democrats and the Conservatives. No final details have been announced though an initial consultation document[37] suggested that the tax rate could vary with the maximum take-off weight of the aircraft (which is argued to correlate closely with environmental emissions) and with three distance bands rather than two as at present.

A per plane tax would encourage fuller flights (as presumably the airlines would have to absorb the tax for unfilled seats). It may be more easily applied to freight flights (though freight-only flights account for only a very small fraction of total departures from UK airports). To the extent that the reformed tax is passed on to passengers, those likely to benefit (relative to APD) will be those flying on fully-loaded aircraft, on smaller, less polluting aircraft, and those flying relatively short distances. However, any reforms that set the tax rates finely according to a wide range of aircraft and destination characteristics will make the tax substantially more complex to administer[38].

An alternative approach to aviation taxation might be to bring aviation into an international system of emissions trading. In principle, such a scheme could be managed at a global level, though in practice regional variants are likely to emerge first. In December 2007, European environmental ministers drew up plans to include aviation in the EU Emissions Trading Scheme (ETS) from 2012[39], covering all flights departing from or arriving into an EU airport even for non-EU airlines. The number of permits available to airlines will be capped at the average level of EU aviation emissions between 2004 and 2006. Any increases in aviation emissions above this level will have to be matched by emissions reductions elsewhere in the scheme. In the first year of aviation's inclusion, only 10% of allowances are planned to be auctioned with the remaining 90% allocated for free according to the tonnage-kilometres flown by each airline.

[36] See Gillen, Morrison and Stewart (2004).
[37] See http://www.hm-treasury.gov.uk/media/E/2/consult_aviation310108.pdf
[38] A full discussion of the issues around this tax reform can be found in Leicester and O'Dea (2008).
[39] See http://www.consilium.europa.eu/ueDocs/cms_Data/docs/pressData/en/envir/97858.pdf

To the extent that aviation is included in the ETS, a domestic aviation tax can hardly target CO_2 emissions. However, aviation also produces considerable noise externalities that vary according to airport of departure, as well as other non-CO_2 environmental emissions, and contributes to congestion both in the skies and around airports. All may provide ongoing justifications for a domestic aviation tax.

A recent paper by Keen and Strand (2007) examines an optimal system of international aviation taxation and summarises the current practice of domestic aviation taxation in different countries. Many EU nations currently charge VAT on domestic tickets, though only Germany and the Netherlands do so at the full rate. Ireland, Denmark and the UK are the only countries to zero-rate domestic aviation tickets. The US does not charge a sales tax on domestic aviation tickets, but it does impose a 7.5% ticket tax as a 'security charge'. Trip-based charges are more popular, whether as an 'airport tax' that accrues to the airport authority or as a departure tax that accrues to the government. Keen and Strand estimate that the average international passenger in the UK pays between 27 and 109 US dollars in such charges per trip, compared to $34 for US passengers and around $9–$16 in France. The paper argues that taxation should take the form of a combination of internationally co-ordinated fuel and ticket taxes that apply as non-refundable excise taxes rather than a VAT: the former would approximate an emissions tax, and the latter could be used both to internalise noise externalities and to raise revenue. Given the constraints imposed by international agreements, however, such a system could be extremely difficult to implement. Fuel excise and ticket taxes on domestic aviation coupled with suitably-varying departure taxes for international aviation may be the best feasible policy.

8. Taxes and Waste Management

Considerable public concern about the environment and the environmental "sustainability" of current patterns of production and consumption focuses on the generation and disposal of waste. Many households devote time and effort to recycling, and many resist excessive product packaging. One prominent anxiety has been that society is using up the earth's finite, non-renewable, resources of raw materials at an excessive rate, so that "not enough" will be left for future generations. This concern raises questions of efficient inter-temporal resource pricing that are important, but beyond the environmental-policy focus of this chapter. However, important environmental externalities are associated with waste generation and management that require attention. Moreover, this field has considerable scope for taxes and other incentive mechanisms to be employed as part of an efficient policy package.

The environmental issues concerning waste largely have to do with the environmental consequences of different methods of waste disposal – the pollution and disamenity costs of landfill disposal, incineration, illegal dumping, etc. Other forms of waste management besides disposal may also involve pollution and amenity costs. The energy used in glass recycling, for example, adds to CO_2 emissions, and bottle banks and other collection methods could impose disamenity costs on some local residents.

The generation of waste involves a series of decision-makers, and inter-related decisions. Product manufacturers determine product design and packaging, both of which influence subsequent waste management costs. Consumers decide which products to purchase, and when and how to dispose of waste. Households also play a key role in initiating recycling in many systems, by separating recyclables from other household wastes. The public authorities or private firms who collect household and industrial waste decide what disposal option to employ – landfill, incineration, or recycling. The environmental problems of waste management arise because at each of these stages the decision-makers concerned do not face costs that reflect the full social costs of the choices they make. In some cases decision-makers face no costs at all – households, for example, face a zero marginal cost for waste disposal in the UK. In other cases, while decision-makers may face some costs such as the charges by operators of landfill sites or incinerators, these costs do not reflect the social and environmental costs of choosing a particular course of action. In short, therefore, price signals may be wrong, or non-existent.

Clearly taxes on various elements of the waste process can be used to correct these faulty price signals, so that decision-makers face the full social costs of their actions. Externality taxes on disposal options can, for example, be used to ensure local authorities and others involved in waste disposal take environmental costs into account in their choice of disposal option. Likewise, unit charging for household waste disposal could encourage households to minimise waste and to increase their use of recycling. We discuss these approaches in sections 8.1 and 8.2.

The difficulty for policy, however, is that it is unlikely to be practicable to levy appropriate taxes on all disposal options, or to ensure that the financial incentives link all of the relevant decision-makers. Illegal disposal (such as fly-tipping) remains, by definition, uncharged, and raising the costs of legal disposal options may encourage greater use of illegal routes, which could have significant environmental costs. Transmitting the financial signals back may be difficult, too. Even where charges for the collection of household refuse seek to provide households with an incentive to minimise waste volumes, they rarely distinguish between different categories of waste according to their costs of disposal. Also, if increased household waste disposal costs are to provide an incentive for manufacturers to change the design of products and packaging, this requires not only on the existence of shifts in consumer demand towards products with lower disposal costs, but also on the ability of firms to identify correctly the shifts in demand, and their reason. For certain products a different approach may be necessary – for example, advance disposal fees in the form of taxes levied when products are sold, deposit-refund systems (to encourage proper disposal), and other policy interventions to encourage product manufacturers to take end-of-life product disposal costs into account. We discuss these approaches in sections 8.3 to 8.5.

8.1 Pricing the external costs of landfill disposal

Three main methods can be used for the disposal of waste: landfill, incineration or recycling. Historically the UK and the US have made extensive use of landfill, a sharp contrast with Japan and some northern European countries including Denmark, Belgium and the Netherlands, where incineration plays a much greater role. The European Union has actively discouraged landfill disposal in member states, and the 1999 European Landfill Directive commits EU countries to a timetable of demanding quantitative targets for reducing the amount of biodegradable municipal waste sent to landfill by 2020. For the UK the Landfill Directive requires reductions in landfilled biodegradable municipal waste to 75% of the 1995 level by 2010, 50% by 2013, and 35% by 2020.

To begin with, what are the grounds for policy intervention to reduce landfilled waste? In many countries, landfill sites for waste disposal are becoming increasingly scarce, as existing sites are exhausted, and as planning obstacles limit the development of new sites. This trend is forcing many countries to reappraise waste management strategies, to reduce reliance on landfill disposal, and to increase the proportions of waste reused and recovered. However, it does not involve any obvious externality, and any role for environmental taxation. Where waste management is operated by decentralised agencies of government and by private sector firms paying the full market rate for the landfill facilities they use, the central government has no obvious need for intervention to discourage the use of landfill disposal on grounds of future scarcity. Scarcity of landfill sites will be reflected in higher charges levied for their use by private owners and operators, reflecting the opportunity cost of current landfill use, in terms of the loss of future landfill capacity. In areas where landfill is becoming scarce, market forces should ensure that disposal of waste to landfill is correspondingly expensive.

Government intervention in waste management is, however, needed to regulate the externalities from landfill and other waste disposal options that are not reflected in the charges levied by operators. Three principal externalities may be relevant:

- current environmental externalities from landfill sites, which may include disamenity costs to local residents, emissions of greenhouse gases and “conventional” air pollutants, leachate seeping into water systems, and environmental costs of transporting waste to the landfill;

- future social costs, which may arise if landfill operators make inadequate provision for the long-run costs of managing landfill sites and are able to avoid liability for these costs through bankruptcy or other means;
- lost social benefits from alternatives to landfill, which may arise if disposal through incineration leads to energy production that substitutes for more-polluting forms of energy supply.

For the UK, studies commissioned in the 1990s by the Department of the Environment provide quantitative evidence on these landfill externalities, for different types of landfill, differentiating between urban and rural locations, and landfill with and without energy recovery. They also estimated externalities from incineration with energy recovery, the most likely long-term possibility for diversion of landfilled wastes. The external costs of landfill were estimated to lie in the range between –£1 and £9 per tonne of waste, depending on the type of landfill, while incineration with energy recovery had a net external benefit of £2–£4 per tonne of waste, reflecting the greenhouse gas and other emissions of the power generation that would be displaced (CSERGE, 1993). The largest external cost element of landfill disposal was the climate change externality from methane emissions from landfill sites, valued at £0.86–£5.40 per tonne; in addition, the climate change impact of carbon dioxide emissions was valued at some £0.08–£1.27 per tonne. The external costs of leaching accidents were estimated at some £0.90 per tonne from exiting landfill sites (while the regulatory conditions attached to new sites were assumed to internalise the cost of leaching risks). Transport externalities were estimated to be less that £1 per tonne, and disamenity costs were approximately £2 per tonne (inferred mainly from US evidence). Averaging across the whole waste stream and the various types of landfill, the externality was about £5 per tonne, or approximately £7 for 'active' and £2 for 'inactive' waste (where inactive wastes are those that do not biodegrade).

The UK's ***landfill tax*** is charged per tonne of commercial, industrial and municipal (household) waste delivered to landfill sites. Two different components of the waste stream are taxed at different rates. The standard rate applies to "active" (biodegradable) waste, and a reduced rate applies to "inert" waste, such as building rubble. When first introduced in 1996, the rates applied were £7 per tonne for standard waste and £2 per tonne for inert waste, in line with the above estimates of the external costs of landfill (Davies and Doble, 2004). On introduction, the projected revenues were used to finance a cut of 0.2% in the rate of employers' National Insurance contributions (a payroll tax), with the declared aim of ensuring that the tax would not lead to a net increase in overall business costs. It was also possible for landfill operators to make contributions to registered environmental trusts in lieu of paying the tax (the "Landfill Tax Credit Scheme").

Prior to the introduction of the landfill tax, some consideration of the relative merits of weight or volume-related taxes on the one hand, and ad valorem taxes on the charges levied by landfill operators on the other was made. The initial proposal announced in the 1995 Budget was for an ad valorem tax, which attracted considerable criticism. One relevant consideration was the documentation needed to levy taxes on the different possible bases; since ad valorem taxes are based on the value of transactions, they would be likely be more straightforward to operate than taxes based on physical attributes requiring measurement. The choice of a weight-related tax requires records to be kept for tax purposes of the weight of deliveries to landfill sites, but the additional costs of this documentation were in most cases believed to be small, as most landfill operators were charging waste disposal authorities by weight and had suitable weighbridges already in place at many sites. A second consideration in the choice between possible bases was how well each related to the various external costs generated by landfill use – including transport-related externalities, local disamenity through noise, dust and smell, the leaching of dumped materials into groundwater and rivers. While some of these external costs may be broadly proportional to the weight of materials dumped, an ad valorem tariff would charge more for wastes that required more costly management, which could be a better proxy than weight for some of the leaching externalities. However, a strong argument was that ad valorem taxation would tend to penalise the operators of higher-quality facilities, operating to more stringent – and more costly – environmental standards. The choice of a weight-related tariff with two charging categories was seen as a reasonable compromise between

differentiation to reflect the external costs of different components of the waste stream, and administrative practicality and cost.

In a review of the tax, HM Customs and Excise (1998) observed that the it had led to a significant reduction in the volume of inactive waste sent to landfill (paradoxically the least-taxed component), but that it also had negligible impact on landfilling of other wastes. Landfilling of inactive waste fell from 35.4 million tonnes in 1997–98 to 15.8 million tonnes in 2001–02, a 55% reduction, while landfilling of waste subject to the standard rate of tax actually increased slightly over the same period, from 50.4 million tonnes to 50.9 million. remained static at some 50m tonnes over the same period.

The standard rate of the landfill tax remained unchanged until 1999, when it was raised to £10 per tonne, and a commitment made to an annual £1 in the rate over the five years 2000–2004, so that the rate reached £15 per tonne in 2004. Noting that the tax had, so far, been ineffective in reducing the amount of non-inert waste landfilled, a Cabinet Office Strategy Unit paper in 2002 concluded that "a rise to £35 a tonne is required over the medium term". Accordingly, the annual escalator was then raised to £3 per tonne, with the aim of raising the rate eventually to the £35 per tonne level. By April 2007 the rate had reached £24 per tonne, and rises to £32 per tonne from April 2008 and £40 per tonne from April 2009 have been announced. In contrast to this succession of increases in the standard rate, the lower rate of landfill tax for inert waste has so far remained unchanged from the start of the system, but is scheduled to rise to £2.50 per tonne from April 2008.

Recent trends suggest a substantial decline in active landfill waste volumes, but the decline in inactive landfill waste has slowed down and possibly halted altogether. By 2006–07, inactive waste landfill volumes had fallen only slightly further from their 2001–02 volumes, to 13.1 million tonnes, whilst active waste landfill volumes were down to 40.8 million tonnes, a fall of around 20% since 2001–02.

The steady acceleration in the standard rate of the landfill tax reflects increasing concern about the inability of the UK to reduce its use of landfill as required under the EU Landfill Directive. Failure to meet these mandatory EU targets will subject the UK to substantial penalties for non-compliance. What has driven the acceleration in UK landfill tax rates is not an upward revision in estimates of landfill externalities, but the overriding priority that has been given to attainment of the EU landfill targets. The setting of these targets appears not to have been based on quantitative assessment of landfill externalities, nor on the relative external costs of different disposal options, and measures to achieve these targets therefore imply tax rates well in excess of marginal external costs. Even raising the landfill tax to very high levels cannot however guarantee compliance with the quantity targets set by the Landfill Directive. The UK therefore has turned to a tradable permit system, to operate in parallel with the existing landfill tax, as a mechanism intended to achieve a predictable quantity outcome in the target years specified in the Landfill Directive.

The ***Landfill Allowance Trading Scheme*** (LATS) is designed to restrict landfill disposal of biodegradable municipal waste (BMW), in order to ensure UK compliance with EU targets (Salmons, 2002; Barrow 2003). The scheme started in April 2005. Permits relate to a particular target year and are allocated without charge to Waste Disposal Authorities (local governments) according to a formula based on current landfilling and total current waste quantities. The first allocation was for the period up to 2010, by which date the UK must reduce its landfilling of BMW to 75% of the 1995 level.

Unsurprisingly, given the purpose of the scheme, the compliance obligations for local waste disposal authorities have a structure that mirrors the timing of the UK's targets under the EU Landfill Directive, for years 2009–10, 2012–13, and 2019–20. Waste Disposal Authorities are required to meet targets in those years without recourse to borrowing or banking, that sum to the relevant global target (75% of 1995 levels in 2010, 50% in 2012–13 and 35% in 2019–20). For these target years, too, substantial penalties for non-compliance with the scheme will apply: any penalties imposed on the UK by the EU (up to approximately £0.5 million per day) would be passed on to non-compliant authorities. For the years between Landfill Directive targets, local waste disposal authorities are also assigned targets, implying a broadly linear adjustment to each successive EU target year. Greater inter-temporal

flexibility is allowed in compliance with these intermediate targets: during the period between each EU target year, waste disposal authorities can bank allowances and can anticipate ("borrow") a small percentage of future allowance allocations (currently limited to 5% of the next year's allocation). This inter-temporal flexibility does not apply in the target years, or across target years, meaning that the trading system effectively consists of six separate sub-periods (the three target years, and the intermediate years before each target).

The interaction between price determination in LATS and the landfill tax should be noted. Since the biodegradable municipal waste regulated by LATS is also covered by the landfill tax, the value of allowances will be given by the marginal cost of diverting biodegradable municipal waste from landfill, at the quantity constraint given by LATS, minus the landfill tax paid on each tonne of waste. The average landfill tax rate applicable over the LATS period prior to 2010 will be £32 per tonne. If LATS allowances are trading at approximately £20 per tonne, this implies that the marginal cost of achieving the constraint on landfilling set by the first period of LATS is of the order of £20 + £ 32 = £52 per tonne. Future movements in the LATS allowance price should then reflect changing expectations about the marginal cost of diverting sufficient waste to achieve the aggregate quantity cap set by LATS, and would offset pound for pound any further changes in the rate at which the landfill tax is charged.

As far as BMW is concerned, the introduction of LATS has made the landfill tax effectively redundant in environmental terms, in the sense that it is the quantity constraint under LATS that will determine the use made of landfill, and not the rate at which the landfill tax is charged.[40] The higher rates of landfill tax planned for future years mainly have the effect of depressing the landfill allowance price, pound for pound with the landfill tax rate (although they also act to recover some of the rents distributed to local authorities through free allocation of LATS allowances). Yet some incentive roles remain for the landfill tax. First, it is a floor price, in the event that the LATS targets prove so easy to achieve that the value of LATS allowances falls. Second, it regulates those components of the waste stream, including industrial wastes, not covered by LATS (which only regulates landfilling of BMW).

While this interaction between the LATS price and the landfill tax illustrates the similarity between environmental tax rates and emissions trading prices, the comparison highlights the key difference in terms of abatement outcomes and abatement cost uncertainty. LATS is being employed because of the overriding priority that has been given to achieving quantity targets for reduced landfill use – targets that appear to have been based on no objective assessment of costs, benefits and risks. The cost of achieving the Landfill Directive targets is unknown, though it will be revealed in due course in the LATS allowance price. Whereas a case can be made in climate change policy for giving high priority to achieving a particular quantity outcome, it is hard to find any corresponding environmental justification for the Landfill Directive's rigid targets. A price-based approach, giving greater weight to environmental taxes than quantity targets, would have been a preferable basis for long-term waste management policy.

The ***aggregates levy***, introduced in April 2002, intended to reflect the environmental costs associated with quarrying sand, gravel and rock. The levy is charged at £1.60 per tonne, normally payable by the quarry operator. It further reinforced incentives to avoid landfilling of inert waste, by stimulating demand for recycled materials to replace virgin aggregates in road-building and other applications. Full-year revenues are of the order of £300 million, part of which is used to finance a Sustainability Fund (to promote local environmental benefits in areas affected by quarrying), and the remainder to finance a 0.1 percentage point cut in employer NICs.

[40] In economic terms, the quantity constraint on the number of permits is binding, and the permit price is the cost of meeting the constraint. That does not mean the tax is totally irrelevant. It usurps some portion of the value of permits, and so it acts like a windfall profits tax on the value of permits handed to Waste Disposal Authorities. In this respect, it is similar to the U.S. tax on chlorofluorocarbons that took part of the windfall profits associated with quantity constraints of the 1989 Montreal Protocol.

8.2 Household waste charges

Current arrangements for the collection and disposal of household waste in the UK provide households with no individual financial incentive to change the amount of waste they throw away. Household refuse collection and disposal is provided by local authorities, financed through the council tax. Whilst council tax rates may be affected by the cost of household waste disposal, it provides no incentive for individual householders to reduce the amount of waste requiring disposal.

In the UK, as elsewhere, a charge per bag or for volume of waste (a "unit pricing scheme" or UPS) has been the subject of considerable recent speculation. The possible systems include:

- weighing individual bins;
- "subscription" programmes, where households pay a monthly fee that depends on their chosen size and/or number of bins;
- revenue stickers, where households purchase stickers that have to be attached to each bag or bin to be emptied.

Many countries now have experience with systems of this sort, and a growing body of research studies their impact on household waste behaviour. The results of a number of studies for various communities in the US are summarised in Box 8.1. The estimated impact on the quantity of household waste varies between studies, and much appears to depend on the design of the UPS, and on parallel policy measures.

In any event, the social benefit of unit charging is not measured by the impact on waste quantities, but the welfare gain from efficient pricing of a service previously supplied at zero marginal cost (Figure 8.5). In this figure, the social marginal cost (SMC) of excess waste includes both operational costs incurred by municipalities and environmental externalities. The social marginal benefit (SMB) is the amount that consumers are willing to pay for one more unit of disposal. The optimal amount of disposal, Q*, is where SMC=SMB. Any quantities in excess of Q* cause a net loss to the extent that the social costs exceed the marginal social benefits of those units – the grey triangle in Figure 8.1.

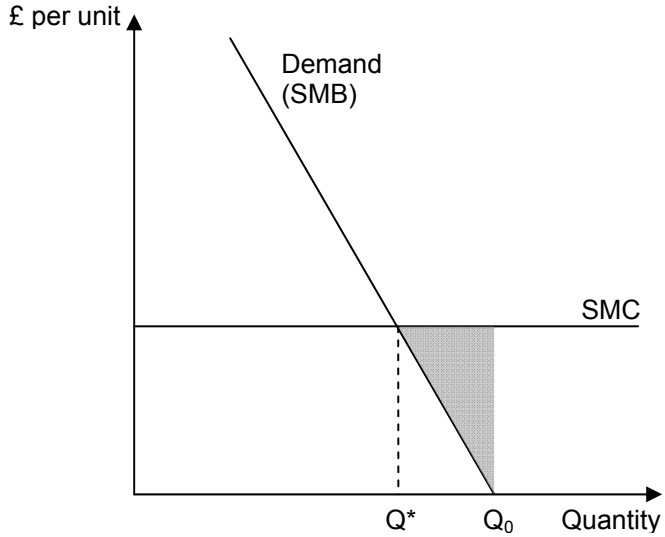
Jenkins (1993) and Repetto *et al* (1992) use their estimates of price effects on waste demand to estimate this welfare cost to be some $650 million per year in the U.S., roughly $3 per person per year. Fullerton and Kinnaman (1996) use household data and also estimate the potential benefits of marginal cost pricing to be in the neighbourhood of $3 per person per year.

Even this small welfare gain from unit charging for waste is not necessarily available, however, for a number of reasons:

- First, the administrative costs of implementing the waste-pricing program may exceed the social benefits. Fullerton and Kinnaman (1996) estimate that the administrative costs of printing, distributing, and accounting for waste stickers in Charlottesville, Virginia, could exceed the $3 per person per year benefits mentioned above.
- Second, some of the available unit charging mechanisms generate only weak incentives to reduce waste; charging per bag, for example, can encourage households to cram as much as possible into a small number of bags – the so-called "Seattle stomp" – which economises on bags, but not on disposal costs. Unit charging mechanisms with better incentive properties, such as individual weighing, may be more costly to operate.
- Third, a uniform tax on all types of waste may be inefficient if materials within the waste stream produce different social costs (Dinan, 1993). If the social cost of disposal of batteries is greater than that of old newspapers, for example, then the disposal tax on batteries should exceed that on old newspapers, and unit charging cannot achieve this precise differentiation.
- Fourth, the welfare calculation neglects the costs of any adverse side-effects (possible littering and other avoidance activities). The partial equilibrium method reflected in Figure 8.1 does not

consider other disposal methods. It does not convey *why* demand for waste disposal slopes down, that is, what substitutes are available. The welfare gain calculation is correct if recycling is the only alternative, but not if more costly alternatives such as illegal dumping are possible. In this case, it would be better to offer free rubbish collection than to implement a pricing policy that leads to widespread dumping.

Figure 8.1: The Optimal Amount of Disposal, and the Welfare Cost of Excess Waste



Available data rarely allow for direct comparisons of illegal dumping before and after implementation of unit pricing. Many economists have asked town officials whether they believe illegal dumping has increased following the introduction of unit charging, and many have stated that it has, but many more have stated otherwise. Reschovsky and Stone (1994) and Fullerton and Kinnaman (1996) asked individual households whether they observed any change. In the former study, 51% of respondents reported an increase in dumping. The most popular method was household use of commercial skips. 20% admitted to burning rubbish, though it was not possible to determine whether this was in response to the charge. Roughly 40% of the respondents to the Fullerton and Kinnaman survey said that they thought illegal dumping had increased in response to pricing. Those authors also use survey responses with direct household waste observations to estimate that 28% of the reduction of waste observed at the kerb was redirected to illicit forms of disposal. Nonetheless, Miranda and Bynum (1999) estimate that more than 4,000 communities use some form of unit pricing in the US.[41] To avoid illegal dumping, some communities have chosen to provide free waste collection for the first bag of garbage, applying unit pricing only to every additional bag. This pricing system leaves some distortion in economic incentives, however, because households have no incentive to reduce their garbage generation below one bag per week.

The great merit of unit charging for household waste is the incentive it gives households to separate wastes for recycling. The problem of dumping and other illegal disposal is a dysfunctional response to the same incentive. Whether unit charging is worthwhile depends on the balance between these effects, which may reflect cultural and social pressures as much as economic incentives. However, unit charging for waste almost certainly needs to be accompanied by additional measures for particular products, tailored to the toxic content of the material for disposal, to the risks and costs of illicit methods of disposal like dumping, and to reflect monitoring capabilities. The following three sections discuss possible instruments, in the form of product charges, deposit refund systems, and the

[41] ISWA (2002) reports that a recent study for Denmark recommended against weight-based charges after finding that municipalities with such charges had more illegal disposal and less recycling than other municipalities.

assignment of producer responsibility. Each may help to resolve the inefficiencies generated by excessive reliance on unit charging to control the growth of household waste.

Box 8.1 The effect of unit pricing for household refuse: US evidence

The initial econometric study of unit pricing, Jenkins (1993), gathered monthly data over several years from 14 US towns (10 with unit-pricing). She found inelastic demand for garbage collection; a 1 percent increase in the user fee leads to a 0.12 percent decrease in quantity.

Two studies rely on self-reported garbage quantities from households (rather than as reported by municipal governments). Hong *et al* (1993) use data from 4,306 households, who indicate whether they recycle and how much they pay for waste collection. Results indicate that a UPS increases the probability that a household recycles, but does not affect the quantity of rubbish produced. Reschovsky and Stone (1994) use data from 1,422 households on recycling behaviour, income, and demographic information. The price of waste disposal alone is estimated to have no significant impact on the probability that a household recycles. When it is combined with a kerbside recycling program, however, recycling rates increase by 27 to 58%, depending on material type.

Miranda *et al* (1994) gather data from 21 towns with UPS programs and compare the quantity of waste and recycling over the year before implementation of unit-pricing with the year following it. Results indicate that these towns reduce waste by 17 to 74% and increase recycling by 128%. These large estimates cannot be attributed directly to pricing: in every case, kerbside recycling programs were implemented during the same year as the unit-pricing program.

Only Fullerton and Kinnaman (1996) use household data that are not based on self-reported surveys. The weight and volume of the waste and recycling of 75 households were measured by hand over four weeks before and after implementation of a price-per-bag in Charlottesville, Virginia. A kerbside recycling program had already been in operation for over a year. Results indicate a slight drop in the weight of rubbish (elasticity of –0.08) but a greater drop in waste volume (elasticity of –0.23). Indeed, the density of waste increased from 15 pounds per bag to just over 20 pounds per bag.

Since collectors and landfills compact the garbage anyway, the compacting by households does not help reduce the actual costs of disposal. Disposal costs are based on the space used in the landfill, and that is not well measured by the number of bags at the kerb, but rather the weight. These results suggest that a price per bag is not very effective at reducing that measure of the space used in the landfill.

Van Houtven and Morris (1999) look at two policy experiments in Marietta, Georgia, including a traditional “bag or tag” program and a second program that requires households to pre-commit or “subscribe” to the collection of a specific number of containers each week. The household pays for the subscribed number whether these containers are filled with rubbish or not. Direct billing may reduce administrative costs. Yet the subscription program does not effectively put a positive price on every unit of waste, since the containers may be partially empty most weeks. Indeed, they find that the bag program reduces waste by 36%, while the subscription program reduces it by only 14%.

Podolsky and Spiegel (1998) employ a 1992 cross-section of 159 towns clustered in New Jersey, twelve with UPSs. They estimate the largest price elasticity of demand in the literature (–0.39). The authors attribute this estimate to the fact that all towns in their sample had mature recycling programs in place, and no towns in their sample had implemented subscription programs. Kinnaman and Fullerton (2000) use a 1991 national cross-section of 959 towns, 114 with user fees. They find that accounting for endogeneity of the policy variables raises the demand elasticity to –0.28, but that is still not very high. They also estimate that subscription programs have less of an impact than bag/tag programs on waste and recycling. Other important studies include Hong and Adams (1999) who look at the effect of unit pricing on aggregate disposal and recycling behaviour, and Jenkins *et al* (2003) who use household level data to look at recycling behaviour by material. They find that unit pricing has no effect on recycling but that kerbside collection has a big effect on recycling of all materials.

8.3 Product charges (advance disposal fees)

In principle, charging households for waste disposal could have effects throughout the chain of production and consumption. If it encourages consumer substitution away from goods with high waste

disposal costs, it could provide incentives for manufacturers and retailers to package products so as to minimise the subsequent waste disposal costs on households. However, the impact of waste charges may be insufficient to modify households' purchasing behaviour appreciably. The signal transmitted to producers may be weak and difficult to distinguish from other factors affecting purchasing patterns.

If these signals to manufacturers are too weak, or if new charges on waste disposal would induce too much illegal dumping, then manufacturers may not get the signal to reduce packaging or to make products than can be recycled. In this case, incentives can be introduced at the "manufacturer" end of the chain, by levying taxes on products and packaging that reflect the costs of their ultimate disposal. The efficient level of the packaging tax on an individual product would be broadly the same as the waste disposal tax that would have been levied on its disposal under unit charging. Higher rates on new products that have higher costs of subsequent disposal provide incentives to switch to products and packaging with lower disposal costs. Thus, the system is compatible with the retention of tax-based financing of household waste disposal, where households face a zero marginal cost for disposal. The fact that households might seek to avoid waste charges by littering and other environmentally-damaging practices provides good reasons to collect household waste at zero marginal cost. The downside of the advance disposal fee, however, is that it does not affect household choices about disposal. It is paid by the manufacturer and may affect production decisions, but it is already paid at the time of disposal and provides no incentive or refund for preferred methods of disposal.

Defining efficient product taxes to reflect end-of-life waste disposal costs would be most straight-forward and effective where producer choices are flexible and important, and where consumer choices are inflexible or unimportant. If consumers invariably throw batteries in the landfill, for example, then producers could be induced to make the type of batteries that cause less damage once landfilled. On the other hand, if consumers can be induced to recycle batteries at much lower social cost, then a deposit refund system (see section 8.4) can work much better than advance disposal fees.

Products that are predominantly thrown away by households, ending up in landfill after one use, could be subject to a higher packaging tax than products that get recycled; both might be taxed more heavily than products packaged in re-usable containers. Packaging taxes of this form have been introduced for beverage containers and certain types of other packaging in some Scandinavian countries, and have been studied by Brisson (1992) and OECD (1993c). Brisson's estimates of the disposal costs of each type of container suggest some surprising conclusions about the relative tax rates that should be applied to drink containers made of different materials. Milk cartons, which are hardly recycled at all, would have low tax rates (about 9 pence per 100 litres in current prices), because their weight and hence disposal costs are low. The corresponding tax rate to be applied to returnable milk bottles would depend on the rates of return achieved. Non-returnable glass bottles without recycling would have tax rates that were some twelve times as high as cartons and plastic containers, but a lower tax rate could be applied to glass bottles used in a context in which re-use is sufficiently substantial and routine. Brisson's figures suggest that returnable milk bottles would need to achieve a 93 per cent rate of re-use for the tax to fall to the same level as the tax on cartons.

The difficulty with an advance disposal fee is that does not provide incentives to encourage reuse, recycling or other actions that reduce disposal costs. Combining advance disposal fees on products with a subsidy to proper disposal may, however, achieve the efficient outcome. Although neither instrument provides all the right incentives by itself, Fullerton and Wolverton (2000) show conditions under which the combination can match exactly the effects of a Pigouvian tax on waste: incentives to reduce consumption of waste-intensive products *and* to dispose of waste properly.

8.4 Deposit-Refund Systems

Several studies have favoured the use of a deposit-refund system to correct for the external costs of garbage disposal, including Bohm (1981), Dinan (1993), Fullerton and Kinnaman (1995), Palmer and

Walls (1997), and Palmer *et al* (1997). Worldwide, these programs have been successful at reducing waste and recovering recyclable materials (OECD, 1998).

To achieve the efficient allocation, the deposit for each good should be set equal to the social marginal cost of dumping the waste, and the refund on return is that deposit minus the marginal external cost of recycling. If the external cost of recycling is zero, then the refund matches the deposit. The deposit could be levied either on the production or the sale of goods. As long as transaction costs are low, the refund can be given either to households that recycle or to the firms that use the recycled materials. If the refund is given to households, then the supply increase can drive down the price of recycled materials paid by firms. If the refund is given to firms, then firms increase demand for recycled materials and drive up the price received by households. In addition, Fullerton and Wu (1998) find that the refund given under a deposit-refund system can encourage firms optimally to engineer products that are easier to recycle. Households demand such products in order to recycle and thereby to receive the refund. This result is important, since directly encouraging the recyclability of product design can be administratively difficult.[42] If the administrative cost of operating the deposit-refund system is high, then Dinan (1993) suggests that policymakers could single out products that comprise a large segment of the waste stream (newspaper) or that involve very high social marginal disposal costs (batteries).[43]

Some have suggested that a "virgin materials tax" might encourage recycling as well as internalize the environmental externalities generated by material extraction (e.g. cutting timber or strip mining). It might increase manufacturers' demand for recycled materials, driving up the price of recycled materials and thus increasing the economic benefits to households that recycle. However, both Fullerton and Kinnaman (1995) and Palmer and Walls (1997) find that as long as other policy options are available, then a tax on virgin materials (such as the UK aggregates levy) is only necessary to correct for external costs associated with extracting the virgin material. The virgin materials tax is not optimally used to correct for the marginal environmental damages of garbage disposal if a tax is available on garbage disposal.

8.5 "Producer responsibility" for waste costs

One of the most far-reaching policy innovations in waste management in recent years has been the idea of "Extended Producer Responsibility" (EPR), which makes producers responsible for the end-of-life waste management of their products. This approach was developed in the German legislation on packaging waste, and it resulted in a parallel, industry-run system of waste collection and management for packaging, operated by the industry-financed company DSD (*Duales System Deutschland*). It has underpinned much of the recent direction of EU waste policy, including directives on packaging, end of life vehicles, and waste electrical and electronic equipment.

In "conventional" waste management practices for household wastes, the collection and disposal of end-of-life products is typically the responsibility of local governments, financed through some form of

[42] This result depends on the assumption that recycling markets are complete. Calcott and Walls (2000a, 2000b) argue that imperfections in recycling markets prevent attainment of the first-best. It is costly to collect and transport recyclables, and it is difficult for recyclers to sort products according to their recyclability and pay consumers a price based on that recyclability. If so, then price signals may not be transmitted from consumers and recyclers back upstream to producers.

[43] Deposit refund systems entail their own administrative costs. Those administrative costs might be quite low if the system is implemented implicitly by the use of a sales tax on all purchased commodities at the same rate, together with a subsidy to all recycling and proper garbage disposal. That practice is currently followed in the US, at least implicitly, since cities do impose local sales taxes, and they do provide free collection of curbside recycling and garbage. If the recycling subsidy needs to be large, administrative costs can be reduced by providing a subsidy per ton, paid to recyclers, rather than providing an amount for each bottle recycled by each household. But then optimality may require a different tax and subsidy amount for each type of material – a plan that might be very costly to administer. According to www.bottlebill.org, the eleven states with current bottle bills are: California, Connecticut, Delaware, Hawaii, Iowa, Maine, Massachusetts, Michigan, New York, Oregon, Vermont. In Europe, Austria, Belgium, Denmark, Finland, Germany, the Netherlands, Norway, Sweden, and Switzerland are all listed as having beverage container deposit refund systems. Canada has also had success with their program.

general taxation. In contrast, EPR shifts responsibility for the financing and management of certain categories of wastes to a separate system, run by or financed by producers. EPR schemes vary in how they are organised, but typically include:

- Obligations on the producer concerning the collection ("take-back") of product packaging or end-of-life products;
- Responsibility for the costs (including environmental costs) of disposal or treatment of the collected products;
- Rules or targets governing the methods of waste management of recovered products, for example specifying minimum required rates of re-use or recycling.

Compared with the conventional municipal system of waste management, EPR shifts direct financial responsibility for the costs of waste management "upstream" to the producer, and away from the municipality and taxpayer. By confronting the producer with the costs of end-of-life disposal of their products, the aim is to provide incentives for the producer to take account of these costs in designing and marketing their products – so-called "design for environment" innovations. In addition, most EPR schemes set targets for higher rates of recovery and/or recycling than in conventional waste management.

EPR has the merit that it confronts producers directly with the costs of end-of-life waste management for their products, through the payments they have to make to cover the costs of collection, recycling and disposal. In principle, it thus internalises all of the costs of production and disposal to a single decision-maker – the firm. It therefore provides the incentive of a product tax or advance disposal fee to make products with less packaging and that can easily be recycled, *and* the incentives of a disposal tax or recycling subsidy to undertake the least costly form of actual subsequent disposal. In other words, as shown by Fullerton and Wu (1998), an EPR can in principle provide *all* the optimal incentives inherent in charges for post-consumer waste disposal or an optimal deposit refund system.

Where actual producer responsibility rules differ from an "optimal" disposal charge is in two respects: First, as mentioned above, actual EPR systems may specify quantity targets or particular methods of waste management. Any "minimum required rates of re-use or recycling" is either redundant with the optimal recycling incentives, or else it takes recycling away from the optimal level. Second, actual EPR systems may be difficult or costly to administer. Bringing the waste back to the responsibility of the producer is an extra step, with extra accounting and linkages, at least compared to direct taxes on each form of consumer disposal if those were feasible.

9. Conclusions

Environmental policy has been transformed over the past decade by the use of environmental taxes, emissions trading, and other economic instruments. These incentives allow stringent environmental protections to be introduced at lower economic cost than with the use of less-flexible forms of conventional regulation that dictate particular abatement technologies. The cost-reducing flexibility of economic instruments will become increasingly important when seeking higher standards of environmental protection.

For example, if the UK and other countries decide to make the drastic cuts in CO_2 emissions advocated by the recent *Stern Review of the Economics of Climate Change*, then taxes or other economic instruments such as emissions trading may need to play a central role in achieving the required extensive changes in energy use of firms and individuals. Energy-pricing measures, either in the form of energy taxes or emissions trading, would provide a common incentive signal to a wide variety of energy users with different abatement costs and opportunities. This pricing can promote cost-effective responses, reducing the cost of achieving any given level of emissions abatement. In addition, the use of taxes instead of regulation can spread the burden of adjustment efficiently across all energy users, rather than concentrating burdens on those subjected to direct regulation.

Despite these efficiency advantages of environmental taxes and other market mechanisms, many areas may still require more conventional regulatory approaches as a major part of the policy mix. In some cases, regulation may be needed to ensure minimum environmental standards, particularly where responses to economic incentives suffer from inertia or where uncertainty over responses mean that environmental damages could be significantly larger than anticipated.

In addition to the improved efficiency in environmental policy that could result from greater use of environmental taxes and other market mechanisms, could their use have a significant impact on fiscal policy?

Environmental taxes could make a significant contribution to tax revenues in two particular areas: energy taxes and road transport congestion charges. In both cases, the available tax base is broad, high tax rates may be justified by the environmental externalities, and demand is inelastic (so revenues are not greatly eroded by behavioural responses, particularly in the short-term). The potential revenues from these taxes hold out the possibility of tax reform packages that include tax reductions and reforms elsewhere in the fiscal system. The political constituency in support of environmental tax measures could create an opportunity for tax reforms that might not otherwise be politically viable.

In a more fundamental economic sense, however, environmental taxes do not necessarily alter the scope for efficient revenue-raising. A tax reform that introduces new environmental taxes *may* have a "double dividend" if it provides the dual gain of a cleaner environment and a more efficient tax system, achieved by reducing "distortionary" taxes that discourage work effort or investment. But that second dividend will not always arise. Environmental taxes raise the costs of production, and hence raise output prices, so they also reduce the net return from each hour worked – just like the taxes being replaced. Hence, a revenue-neutral shift from labour taxes to environmental taxes may or may not reduce the distortionary impact of the tax system.

An implication is that the case for environmental tax reform must be made primarily on the basis of potential environmental gains. The fiscal aspects of environmental tax reforms are still important, however, because the costs of environmental policy can be multiplied by inappropriate use of the revenues or by their unnecessary dissipation. But appeals to beneficial fiscal consequences of environmental tax reforms are unlikely to justify measures that do not pay their own way in purely environmental benefits.

Turning to our discussion of the scope for environmental taxes within each of our main environmental policy applications, these involve three common themes:

1. Empirical evidence is necessary for grounding environmental tax policy on the size of the marginal externalities involved, in order to indicate the level of environmental taxes that might be justified and the potential benefits of policy action.
2. Multi-part instrument combinations can frequently and usefully be employed when available tax instruments do not accurately target the relevant externalities.
3. Environmental taxes may form only part of a portfolio of policy measures. Political and practical constraints may prevent setting environmental taxes at the first-best level, so other measures can then help stimulate the development or use of abatement technologies. The case for such packages may be enhanced by recognising various market failures in the development or dissemination of new abatement technologies. Thus, well-targeted measures to stimulate research and development or diffusion may enhance efficiency.

Industrial and household use of energy

The *Stern Review* recommends significant abatement now, to reduce the long-term costs of climate change. A carbon tax would be an appropriate, broadly-based incentive measure to achieve the recommended changes in energy use. But the EU is now well advanced along an alternative track, as

a carbon tax has been eschewed in favour of a cap on emissions with trading of permits. Similar choices appear in carbon abatement proposals in the Northeastern US and California.

The economic properties of carbon taxes and emissions trading are quite similar. Both allow cost-reducing flexibility in polluter responses, a major improvement on conventional regulation. If tradable permits are auctioned, then the similarity between the economic effects of taxes and emissions trading is particularly close; the main economic differences then arise in the case with uncertainty about the costs of reducing emissions. *Stern* argues that their different properties under uncertainty favour the case for permits over taxes, although Pizer (2002) reaches the opposite conclusion. Most importantly, however, either auctioned permits or carbon taxes would clearly dominate a plan to distribute permits without charge to polluters (as is largely current practice in the EU ETS). Any such policy raises the cost of production and exacerbates labour supply or investment distortions, unless the scarcity rents can be captured and used to reduce existing labour or capital taxes.

If either the UK or EU introduces a carbon policy unilaterally, production may shift abroad in a way that would mute its beneficial environmental impact. An ideal world might have a common global carbon price, but that goal might well be difficult to achieve soon for various political and economic reasons.

With or without global or even national carbon abatement policy, however, a case still can be made for other measures as well to stimulate new abatement technologies. And all such measures are also likely to have distributional consequences that might need to be offset by further supplementary policies. Energy consumption constitutes a relatively high fraction of low-income budgets, and so any energy policy is likely to be regressive. New energy policy could be combined with other measures to re-establish the desired degree of redistribution within the overall tax system.

Road transport

Road transport is already heavily taxed in the UK, and environmental gains are more likely to be achieved by better targeting of road transport taxes to the externalities involved, rather than further increases in these existing taxes. In other words, for example, petrol taxes might currently account for the combination of externalities from emissions and congestion on average, but some of those petrol taxes could be replaced by charges directly on congestion.

Congestion charges on private motoring could be a major source of tax revenues, if levied at a rate reflecting the congestion externality imposed by each individual motorist. However, separate taxes on congestion externalities weaken the case for high motor fuel excises. Sansom *et al* (2001) estimate that the congestion component of motoring externalities constitutes three quarters of the total externality. If the remaining motoring externalities do not justify retention of the existing high taxes on motor fuels, the net revenue gain from a congestion tax is reduced. The interaction between a well-targeted congestion tax and other externalities requires careful consideration, and so careful modelling is required to establish the right motor fuel taxation policy once a congestion tax is introduced.

Estimates of the relative externalities can be used to make recommendations about the relative taxation of diesel fuel and petrol, though these recommendations would be sensitive to one's view about the appropriate value to place on abatement of CO_2 emissions. Adopting a high value for the social costs of CO_2 emissions would imply a strong preference for diesel, despite its higher emissions of particulates – with adverse health effects in urban areas. Also, biofuels can be promoted through reduced taxation or direct subsidy, but the strict environmental case so far seems weak (depending on further evidence on the size of the various externalities). Finally, a case can be made for policies to stimulate vehicle fuel efficiency, over and above the generalised incentives provided by high taxes on fuels. The UK and EU are considering a set of tradable fuel efficiency targets for manufacturers, and the US experience with CAFE standards is relevant in assessing this proposal.

Aviation

Even without any environmental concerns, a case can be made for reform to aviation taxes. The current UK tax on air travel (air passenger duty) is a rather crudely-designed revenue-raising tax,

intended to partly compensate for the absence of other aviation taxes, while recognising the difficulties for a single country in unilaterally taxing international travel. Better revenue-raising taxes on aviation could be designed, although this would require a significant amount of international coordination.

From the environmental point of view, the UK's current taxes are poorly designed, basing the tax on the passenger rather than a closer measure of the environmental impact of each flight. The recently-announced reform to air passenger duty may improve matters. Substantial gains could be made by aviation taxes that directly reflected the climate impact, noise and other environmental effects of individual flights, although international coordination would again be needed for optimal environmental taxation of aviation, without disruptive effects on competition and wasteful diversion and avoidance.

Waste

In the area of waste management, environmental taxes are unlikely to raise major revenues. Still, however, waste taxes can have considerable environmental and economic significance, in ensuring that waste management decisions take account of the environmental consequences of different disposal options – landfill, incineration, recycling, etc – and encouraging substitution by producers and consumers towards products and packaging that involve less waste, and more efficient recycling.

The UK's first explicit environmental tax was the landfill tax introduced in 1996. When introduced, it was set at a level based on estimates of the external costs of landfill. More recently the tax has been increased sharply, and its effects supplemented with a Landfill Allowance Trading Scheme, both with the aim of ensuring tat the UK complies with the targets for a sharp reduction in landfilling of biodegradable municipal waste set in the EU Landfill Directive. The economic case for these changes has not been properly made: it is likely that achieving the directive's targets will require diversion away from landfill, at high and unpredictable cost, probably well in excess of what could be justified in terms of the environmental externalities involved.

A second area of active controversy in waste management concerns the possible use of unit charging for collecting household waste, in place of current arrangements where refuse collection is financed from general local tax revenues. We now have considerable international experience of such systems, and the research evidence suggests that they may make some quantitative impact on the amount of household waste, although the welfare gain from this may be quite modest. One concern, however, is that unit charging may provoke increased dumping and other forms of avoidance.

Alternatives, which may reduce the risk of dumping, include advance disposal fees of products (for example packaging taxes on drinks containers, reflecting their contribution to waste disposal costs), or deposit-refund systems (which may be costly to operate, but which provide explicit incentives for proper waste disposal, re-use or recycling). In a number of areas, European policies have been built on the notion of "producer responsibility" for the costs of waste management, typically coupled with tough targets for collection and recycling. While this approach may encourage producers to take end-of-life waste costs into account in product design, it does so at unpredictable cost. The costs of waste management under producer responsibility are less-transparent than with tax-financed municipal collection or unit charging, but ultimately borne by consumers through higher product prices to much the same extent as with conventional waste management.

This area illustrates the complexities of environmental tax policy. Taxing the externalities involved in waste disposal would provide many of the right incentives, including incentives for producers to use designs that facilitate recycling, for firms to sell products with less packaging, for stores to re-use grocery bags, for consumers to buy products with less waste content, for waste processors to recycle, and for landfills to dispose of waste appropriately. Yet this approach would also face many problems of measurement, administration, enforcement, and compliance, risking dumping and other undesirable outcomes. A more complex "multi-part instrument", combining taxes, subsidies and various forms of direct regulation may be capable of achieving a better outcome. Each part of this policy can apply to a market transaction, and the appropriate combination of such instruments can reflect the complexities and help provide the right incentives to all parties involved in the generation and disposal of waste: a

direct subsidy for use of designs that facilitate recycling, a direct tax on excess packaging, a simple mandate such as for reuse of grocery bags, a tax on the waste-content of goods purchased, and a subsidy for recycling. The multiplicity of instruments might not be simple, but all such instruments together may be more efficient than relying solely on the taxation of waste.

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