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Environmental taxes

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Abstract

This paper provides an overview of key economic issues in the use of taxation as an instrument of environmental policy. Part A reviews the efficiency gains and other arguments for using taxes and other market mechanisms in environmental policy, discusses the choice between basing taxes directly on measured emissions and using other tax bases less-directly related to emissions, and considers the value of the revenue contribution from environmental taxes. It is argued that environmental tax revenues are unlikely to significantly alter the economic constraints on tax policy, and that environmental taxes need to be designed and justified primarily with the cost-effective achievement of environmental goals in mind. Part B discusses key areas where environmental taxes appear to have significant potential - including general taxes on energy used by industry and households, various road transport taxes, and taxes on aviation and on waste. In a number of these areas, efficient environmental tax design will make use of a number of taxes in combination - a "multi-part instrument".

Keywords
Tax reform, environmental taxation, pollution taxation, energy taxes, Pigouvian taxes.

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1. Introduction

'Environmental tax reform' has, over the course of the last decade, moved from theoretical discussion to the practical policy agenda in many European countries (OECD, 1995; EU, 2000). In the UK, a number of tax measures have been implemented primarily with environmental objectives in mind (Table 1). They have included three new national environmental taxes, on landfill, industrial energy use (the Climate Change Levy) and the extraction of aggregates. 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 change 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 – for example, reductions in the rate of VAT on energy-saving materials installed in the home, and taxes on aviation.

The growing attention given to environmental aspects of tax policy partly reflects the higher profile of environmental issues in public, political and policy debate more generally. Additional impetus for environmental tax reform has come from the recognition of the limitations of environmental policies pursued solely through conventional regulatory instruments. Over a number of years, there has been a growing awareness that some environmental problems cannot be tackled purely as technical issues, to be resolved straightforwardly through regulations requiring the use of appropriate abatement technologies. To make any serious impact on some of the major environmental problems now facing policy-makers—acid rain, global warming, traffic congestion—environmental policies will need to achieve extensive and far-reaching changes to existing patterns of production and consumption. These changes inevitably entail substantial economic costs. The search for instruments capable of minimising these costs, and capable of achieving behavioural changes across all sectors, has led policy-makers in the last decade to pay much closer attention to the potential for incentive-based environmental regulation, through taxes, charges, tradable permits, and other 'economic instruments'.

In parallel with growing awareness of the potential for "economic instruments" such as environmental taxes to improve the efficiency of environmental policy, there has been interest in the scope for tax reform using the revenues raised from environmental taxes. Some countries that have been concerned about the impact—either economic or political—of high taxes on labour income, have used environmental tax revenues to reduce tax rates on labour incomes. Sweden's 1991 reforms used revenues from new environmental taxes on energy to finance cuts in labour income taxes. Similarly, a number of the UK's environmental tax measures have been accompanied by provisions to return the revenues through a reduction in the payroll taxes paid by employers. The political attractions of 'packaging' environmental taxes and tax reform in this way are, perhaps, obvious. The environmental gains, too, are relatively clear-cut, 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 an instrument of environmental policy. Following this introduction there are two main parts. Part A discusses economic principles of environmental taxation. Section 2 reviews the arguments for using taxes and other market mechanisms in environmental policy. These arguments form the backdrop to any discussion of environmental taxes, and neglecting the basic principles they embody could lead to inefficiently designed instruments, or excessive long-run cost. Section 3 discusses the likelihood that the environmental tax bases available in practice will limit us to environmental taxes which can only approximate the ideal "Pigouvian" environmental incentive, targeted directly to polluting emissions. What does this real-world compromise imply for the use of environmental taxes? Section 4 considers the value of the revenue contribution from environmental taxes. How far would environmental tax reform alter the constraints and problems which tax policy currently faces? 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 discusses three key areas of application of the principles outlined in part A. Section 5 examines energy or carbon taxes. Section 6 focuses on road transport taxes (car taxes, fuel taxes and road pricing) and section 7 examines aviation taxes. We also consider, in Section 8, taxes on household waste. The first two of these - general taxes on energy and taxes on road transport - have the potential to generate substantial revenues, while the revenues that could be derived from waste taxes are probably more limited. In all three areas, however, there are a range of different ways in which taxes, or other similar instruments, could make a significant contribution to efficiency in environmental policy. Aviation represents a key area for the future development of environmental taxes, and the most obvious example of a sector that will require international co-operation for long-term effective environmental solutions.

In each of these sections, our main concern is UK experience and policy, but we draw on international evidence and experience where relevant. We look at the extent to which reforms in each area may be possible and desirable, drawing on economic evidence and analysis of the externalities involved to assess what appropriate tax rates might be, the revenue potential in each area, and the key considerations involved in tax design. Our aim is to outline a framework, based on sound theoretical foundations, for evaluating the economic aspects of a range of possible environmental tax reforms in the UK.

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. This is, however, an area where 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 our analysis by our own personal speculations about what measures would be publicly or politically acceptable in current circumstances.

Secondly, technology is a key issue in determining the types of environmental taxes that are practicable, and, again, is developing rapidly. Technological advances may fundamentally change the nature of optimal policymaking in the future. As carbon capture and storage becomes an option for industry, it is important to consider how this affects the choice of instrument. Taxes levied on energy inputs, for example, would discourage firms from investing in carbon capture technology whilst taxes levied on emissions directly would not. Technological advances that make it easier and faster to directly measure emissions may therefore shift optimal policy away from input-based taxes and towards more direct, targeted emissions taxes.[1]

Thirdly, the terms of the environmental policy debate in the UK have been sharply redrawn by the publication in October 2006 of the *Stern Review of the Economics of Climate Change*, commissioned by the Treasury. The *Review* concluded that there was a strong need for immediate and urgent action to mitigate the potential costs of global warming, and that the costs of action were significantly outweighed by the benefits. The *Review's* main conclusions were quickly endorsed by all the major political parties and the document received considerable international attention. However the methodology and conclusions of the *Review* have also attracted much controversy. Stern's estimates of the scale of the damage from global warming, for example, and the social costs of carbon emissions were much higher than the vast majority of earlier economic estimates. Whether Stern's estimates are right or wrong will have hugely important implications in terms of the optimal level any carbon or energy taxes might take, as well as any possible concerns regarding the distributional or competitiveness effects of energy taxation. The long-term policy impact of the Stern Review will be a major determinant of how far and how fast environmental and energy policies need to take action to

[1] Technology may also change the scope for non-tax approaches to certain problems. For example technological developments make it easier for consumers and businesses to purchase carbon offsets (such as at the time of purchasing an airline ticket online) and it may be important to consider the extent to which government intervention to encourage offsetting is justified.

restrain fossil fuel energy use, and how much policy interest there is in the potential of taxation in this area.

Finally, while the primary focus of the paper is on national tax policy, there is a key international dimension to some of the major areas of environmental policy-making which cannot be neglected. For energy and carbon, in particular, the relevant externalities are largely global in their impact – greenhouse gases emitted in one country have similar global effects to those emitted elsewhere. This means that effective policy cannot be implemented by a single country, and that national policies will have to be formulated in the context of wider international policy developments. Should the UK decide to implement a carbon tax without similar policies being developed in other countries, there would be an obvious concern that carbon emissions would simply be exported from the UK to zero-tax economies, undermining the effectiveness of national policy.

Table 1. Environmental taxes in the UK, April 2007

Name	Description	Rates (at April 2007)	Revenue[2]
Landfill Tax	Introduced in 1996, at a standard rate of £7 per tonne on waste delivered to landfill sites. A lower rate of £2 per tonne applies to inactive waste (building rubble, etc). The standard rate was raised to £10 in 1999, and an annual escalation of £1 per tonne applied for five years from April 2000, leading to a level of £15 per tonne in 2004. The escalator was then raised to £3 per tonne from 2005, with the a medium term objective set to raise the rate to £35 per tonne following a Cabinet Office Strategy Unit paper in 2002. The December 2006 Pre-Budget Report indicated an intention to raise either the rate of acceleration or the medium-term target. The initial tax rate was determined with reference to landfill externality estimates; subsequent increases have been designed to achieve greater behavioural change. Revenues in 2005/06 were £733 million (net of contributions to the accompanying voluntary Landfill Tax Credit Scheme), used to finance a 0.2 percentage point cut in employers' National Insurance contributions. The scheme is estimated to have led to a 60% reduction in the volume of inactive waste sent to landfill, but to have had negligible impact on landfilling of other wastes. In 1996/7, 84% of municipal waste was sent to landfill and 7% recycled or composted; by 2004/5 these figures had become 67% and 24% respectively. Total landfill volumes started to decline in 2002/3 though this may also have been due to changes in council waste disposal policies as a result of the 1999 European Landfill Directive which set a target to reduce the proportion of UK biodegradable municipal waste sent to landfill by half between 1995 and 2013.	£24/tonne (standard waste), to rise to £32/tonne from April 2008 and £40/tonne from April 2009. £2/tonne (inert waste), to rise to £2.50/tonne from April 2008.	2001/2: £502m 2002/3: £541m 2003/4: £607m 2004/5: £672m 2005/6: £733m 2006/7: £800m 2007/8: £900m
Climate Change Levy	A tax on energy use by business and industry, introduced in April 2001 as part of the government's Climate Change Programme. The tax does not apply to domestic energy use, or to energy used in the transport sector. Exemptions include fuels used in electricity generation, fuels used for non-energy purposes, and electricity generated from new renewable energy (eg solar and wind power). An 80% discount from the levy applies to energy-intensive sectors that negotiate Climate Change Agreements with DEFRA. Revenues are partly used to finance energy efficiency schemes, and the remainder returned to business through an offsetting 0.3 percentage point reduction in employers' National Insurance contributions. A 2005 report by Cambridge Econometrics and the Policy Studies Institute suggested that by 2005, the Climate Change Levy had contributed a cumulative emissions reduction of 16.5 million tonnes of carbon and that by 2010, the annual reduction will be around 3.7 million tonnes of carbon per year, or 2.3% of total estimated emissions that year.	0.154p / kWh (gas) 0.441p / kWh (electricity) 0.985p / kg (LPG) From April 2008: 0.159p / kWh (gas) 0.456p / kWh (electricity) 1.018p / kg (LPG)	2001/2: £555m 2002/3: £829m 2003/4: £832m 2004/5: £764m 2005/6: £743m 2006/7: £700m 2007/8: £700m
Aggregates Levy	Introduced in April 2002. Intended to reflect the environmental costs associated with quarrying. Applies to sand, gravel and rock subject to commercial exploitation in the UK and UK territorial waters. Exemptions for coal and metal ore mining, for materials used to produce lime and cement, for industrial spoil and waste, highway evacuation and marine dredging. Export tax relief, and corresponding taxation of imports. The levy is normally payable by the quarry operator. Revenues are partly used to finance a	£1.60 / tonne. To rise to £1.95 / tonne in April 2008.	2002/3: £247m 2003/4: £339m 2004/5: £334m 2005/6: £326m 2006/7: £300m 2007/8: £300m

[2] Up to 2005/6, figures are out-turns from HM Treasury Public Sector Finances Databank (as at 21 September 2006); 2006/7 and 2007/8 figures are estimates from the March 2007 Budget. Figures for Congestion Charge come from Transport for London.

	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. Budget 2006 noted that primary aggregates sales fell by 8% between 2001 and 2003 despite a buoyant construction industry, though the precise contribution of the Levy to this is hard to discern.		
Motor Fuel Differentiation	Tax differential in favour of ***unleaded petrol*** introduced in 1987 at 0.96 pence per litre, subsequently widened to 4.8 pence per litre by 1995. Initial aim was to offset the higher cost of unleaded petrol, and subsequently to provide an incentive for fuel switching. Leaded petrol was removed from normal sale in the UK market in 2000. Tax differential between ***petrol and diesel fuel*** was eliminated in November 1994 Budget, which raised the excise duty on diesel to equal that on unleaded petrol. This followed a 1993 report of the Quality of Urban Air Review Group, which drew attention to the rapidly-growing problem of particulates emissions from diesel vehicles in urban areas. Diesel duty (conventional diesel) is now taxed more heavily than conventional unleaded petrol. ***Ultra-low sulphur*** petrol and diesel are taxed more lightly, and have virtually superseded conventional petrol and diesel in sales terms. Lower rates of duty apply to ***alternative road fuels***, including biofuels, gas used as a road fuel, and bioethanol. However very little is sold – in 2005/6 23.5bn litres of diesel, 25.5bn litres of petrol and just 0.3bn litres of alternative fuels were sold.	*Petrol (pence per litre)* ULSP 48.35 Aviation 28.84 Other 51.52 *Diesel (pence per litre)* ULSD 48.35 Red Diesel 7.69 Conventional 54.68 *Alternative Fuels (pence per litre)* Biodiesel 28.35 Bioethanol 28.35 *(pence per kilogram)* Natural Gas 10.81 LPG 12.21 From October 2007: Main rates will rise by 2p per litre.	2001/2: £21.9bn 2002/3: £22.1bn 2003/4: £22.8bn 2004/5: £23.3bn 2005/6: £23.4bn 2006/7: £23.6bn 2007/8: £25.1bn NB revenue from taxation of hydrocarbon oils (including 'red diesel' for off-road vehicles, aviation fuel etc.) VAT is charged on top of fuel duty which effectively raises revenues by an additional 17.5%
Vehicle Excise Duty Differentiation	From June 1999, an annual vehicle excise duty differential was introduced to favour cars with small engines; this now applies only to vehicles registered before March 2001. For cars first registered after 1 March 2001 VED has been graduated according to the CO_2 emissions performance of the vehicle model. Charges are made in seven bands. The lowest band (A) is zero and payable for vehicles with an emissions performance of less than 100g CO_2/km whilst the highest band (G) is payable by vehicles with an emissions performance of more than 225g CO_2/km. In 2005, only six vehicles were sold in the lowest emissions band though sales of alternative fuel and hybrid vehicles were over 6,000 in total.	*Vehicles registered before March 2001* Engine size < 1,550cc: £115pa Engine size ≥ 1,550cc: £180pa *Vehicles registered in or after March 2001* £0 - £300 per year (top rate to rise to £400 in April 2008 - NB top rate payable only by vehicles registered on or after 23 March 2006; earlier registrations pay a top rate of £205 rising to £210 in April 2008).	2001/2: £4.3bn 2002/3: £4.3bn 2003/4: £4.7bn 2004/5: £4.7bn 2005/6: £5.0bn 2006/7: £5.1bn 2007/8: £5.6bn
Central London Congestion Charge	Introduced February 2003, by the Greater London Authority. Exemptions include taxis, buses, emergency services vehicles, vehicles used by the registered disabled, etc. Vehicles using gas, electric, fuel cells and bi/dual fuel sources are exempt. The charge was originally set at £5 per day, for driving within a 21-square kilometre area of central London, and was raised in July 2005. In February 2007, the zone expanded in size towards the West of London. The charge is monitored and enforced through cameras able to read vehicle registration numbers, sited at entry points to the zone, and at other locations within the zone. Annual revenues net of running costs are reinvested in transport in London. The estimated effect on congestion is a reduction of around 25 – 30% relative to a pre-charge baseline.	£8/day (residents of the zone are eligible to receive a 90% discount)	*Estimates net of running costs* 2003/4: £80m (?) 2004/5: £97m 2005/6: £122m

Part A: Principles

2. Environmental regulation: instrument choice

There is now an extensive literature on the potential for taxes to contribute to more efficient and more effective environmental policy (e.g. Smith, 1992; OECD, 1993, 1996; Bovenberg and Cnossen, 1995). In comparison with 'conventional' regulatory policies based on technology mandates or emissions standards, environmental taxes 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). However, environmental taxes are far from being a panacea for all environmental problems. Consideration of their advantages and disadvantages, as compared with other instruments, suggests that there is a group of environmental problems for which taxes, of various sorts, may be the best instrument available. There are other environmental problems which would be better tackled by other approaches, including other 'economic instruments', such as tradable permits, or various forms of command-and-control regulation.[3]

2.1 Advantages of environmental taxes

From the perspective of environmental policy, environmental taxes have attractions for a number of reasons.

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

Where the costs of pollution abatement vary across firms or individuals, environmental taxes have the potential to minimise costs, for one of two possible reasons. Other policy instruments cannot fully differentiate between polluters with different marginal costs of abatement, and thus may require some forms of abatement with high costs, but taxes provide each polluter with incentive to abate in all of the least-expensive ways, and thus can achieve a given level of abatement at lower total abatement cost. Second, taxes can side-step the need for the regulatory authority to acquire detailed information on individual sources' abatement costs, and can thus lower the public sector's costs of regulation.

(ii) 'Static' efficiency gains through performance incentives

In many cases, use of a given abatement technology does not guarantee a precise emissions level; instead, much depends on *how* the technology is used. Taxes levied on emissions provide an incentive for care and attention in the operation of mandated technologies. Providing businesses with an incentive to cut emissions can be translated into providing individuals within the business with similar incentives, and some individuals may be in a position to take actions which greatly affect the emissions performance of a given technology. Lövgren (1993) shows that a substantial part of the gains from the Swedish nitrogen-oxides (NOx) charge came from this, perhaps unexpected, source.

(iii) Dynamic Innovation incentive

Regulatory policies, stipulating that polluters must use particular technologies, or maintain emissions below a specified limit, do not provide polluters with any encouragement to make reductions in pollution beyond what the regulations require. Indeed, where regulations are negotiated on a case-by-case basis, polluters may fear that any willingness to go beyond what is strictly required by the regulations will simply lead to the regulator assigning the firm a tougher limit in future. Environmental taxes, on the other hand, provide a continuing incentive for polluters to seek ways to reduce emissions, even below the current cost-effective level. This incentive arises because of the tax

[3] 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, mandates, or other policies.

payments made on each unit of residual emissions, which create an incentive to develop new technologies, permitting further abatement at a marginal cost below the tax rate.

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

An important consideration in choosing between different strategies for environmental regulation is the extent to which efficient implementation of the policy requires firm-by-firm negotiation of individual abatement or technology requirements. As noted above, command-and-control regulatory policies could be operated in a way which requires different amounts of pollution abatement from different firms, in order to achieve a more cost-effective pattern of abatement than under a uniform abatement rule. However, the regulator is dependent on the regulated firms for information about their abatement costs, and to obtain this information is liable to be drawn into dialogue and negotiation with the regulated firms. 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 which work to their advantage.

One key difference between regulatory policies which set differentiated firm-by-firm targets and the use of environmental taxes is that environmental taxes achieve a cost-effective distribution of abatement, taking account of the abatement costs of individual firms, while taking a robust, non-negotiated form. All firms face the same pollution tax rate. There is no need for the regulator to consider the circumstances of individual firms, and there is thus little scope for individual polluters to attempt to negotiate more favourable terms with the regulator. The risk that this process of negotiation would erode the environmental effectiveness of the policy is thus substantially reduced.

(v) Cost-limiting properties

As compared with policy instruments which operate by defining a quantitative limit on pollution, environmental taxes have the attraction that they insulate polluters from the risk that regulatory requirements might involve excessive abatement costs. The tax rate per unit of emissions places an upper limit on the unit abatement cost to be incurred. If abatement turns out to be more costly per unit than the tax per unit, firms will simply pollute and pay the tax, rather than paying for costly abatement. By contrast, regulatory policies which set a quantitative limit on emissions may risk requiring that abatement measures are undertaken which are far more costly than the resulting environmental benefits.

2.2 The Limitations of environmental taxes

Likewise, and again from the perspective of environmental policy, environmental taxes have a number of identifiable drawbacks and limitations, which will in some cases be sufficiently important to rule out their use in particular applications.

(i) Uncertain environmental impact

The level of pollution abatement achieved by an environmental tax depends on individual polluters' responses to the abatement incentive that the tax creates. It is not possible to guarantee that an environmental tax will achieve 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. For example, some pollution problems may exhibit threshold effects, where environmental damage per unit of emissions rises sharply beyond a certain level of emissions. On the other hand, many pollution problems do not involve an abrupt boundary between acceptable and damaging emissions, and precise achievement of an emissions target may be relatively unimportant. In these circumstances, environmental taxes may be more attractive.

It will be noted that this uncertain environmental effect is the counterpart of the cost-limiting property of environmental taxes noted above. Quantitative instruments like direct regulation (or tradable permits) guarantee a particular impact on pollution, but at uncertain abatement cost, while environmental taxes guarantee an upper bound on marginal abatement costs, but have an uncertain pollution outcome. Which matters more will depend on the environmental problem under consideration, and on whether society would prefer to take risks on environmental quality or on the costs of environmental policy.

(ii) Compatibility with firm decision-making structures

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

For environmental taxes to lead to efficient polluter responses, it is necessary for firms to draw together information relating both to technology choice and to tax payments. Firms considering whether to undertake more pollution abatement need to balance the gains, at the margin, in reduced tax payments, against the marginal costs of abatement. This requires a type of interaction that 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, and—taking decision-making costs into account—more cost-effective, too.

(iii) Lack of experience

In the past, the lack of experience with environmental taxes may have been a significant obstacle to their adoption in any particular practical context. A novel policy instrument is only likely to be employed in preference to one which is familiar where the conventional instrument has clear, and widely recognised, defects. Increasingly, the objection that environmental taxes are untried and untested is untrue. Many studies, such as those of the OECD, document the extensive international experience with environmental taxes which is now available, and there is increasing evidence evaluating their effectiveness.

(iv) Administration and enforcement costs

Both environmental taxes and conventional command-and-control regulation require mechanisms for administration and enforcement. The relative costs of these arrangements should be taken into account in choosing between the different instruments available. It is difficult to generalise, 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. Thus, 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 labor, 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 which use these inputs. Unlike other forms of environmental regulation, there is no need for direct contact between the regulator and polluters, and 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. If there is only a small number of producers, this will be comparatively cheap to operate. The excise duties levied on mineral oils are a case in point; there is a small number of petrol companies, and their activities are tightly controlled and well documented.

(v) Geographical differences

If pollution damage varies depending on the source of the emissions, policy based on a uniform pollution tax applying to different sources will be liable to result in inefficiency, and source-by-source regulation may be able to achieve a more efficient outcome. In principle, of course, an environmental tax need not be constrained to applying the same tax 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 more exposed to lobbying influence from the regulated firms. Also, some possible forms of environmental tax may be constrained to set uniform tax rates, even where damage is known to differ between locations. Thus, for example, environmental taxes based on the taxation of pollution-related inputs to a polluting production process may be unable to differentiate between sources, because of the difficulty (or the costs) of preventing resale of inputs taxed at a low rate to polluters with more-damaging emissions.

(vi) Damaging avoidance activities

Sometimes the environmental 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 unregulated emissions. For example, a tax on toxic waste may provide a powerful incentive to reduce as much waste as possible, but it may also induce illegal dumping or burning.[4] Even if the overall amount of such dumping is "small", the small 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.

Fortunately, alternative incentive-based methods are available that retain many of the properties of a direct tax while minimizing the need for monitoring and enforcement. Note that a direct tax on waste would induce the producer to change to a cleaner process and reduce the waste per unit of output (a "substitution effect"), and it also increases the cost of production and reduces demand (an "output effect"). An equivalent alternative is to use a combination of instruments that *do* apply to market transactions. First, a subsidy can be provided for cleaner technology used in production, or for recycling the final good. This subsidy to clean inputs makes dirty inputs relatively more expensive, causing substitution away from the dirty input. Then a simple output tax can achieve the proper reduction in demand for the good. Fullerton and Wolverton (1999) call this combination a two-part instrument, and they find a first-best closed-form solution that exactly matches the incentive effects of a direct tax on pollution or waste.[5] Since the correctly set combination is equivalent to a tax on pollution, it also raises the same revenue as a tax on pollution. In other words, the output tax revenue

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[4] 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 (1979) finds that only 200 of 921 polluters thought to be in compliance actually were (Cohen, 1999).

[5] Eskeland (1994) and Palmer and Walls (1997) also discuss combinations of instruments, and Walls and Palmer (2001) also provide closed-form solutions for the first-best two-part instrument.

exceeds the cost of the clean-input subsidy, by exactly the amount that the pollution tax would have collected.

A practical example is a deposit-refund system on items such as glass bottles or aluminum cans. The two-part instrument (2PI) is a generalization of a deposit-refund system, however, because it can be used to address other types of pollution. The tax and subsidy do not have to apply at the same rate, to the same commodity, or even to the same economic agent. The point is to avoid the enforceability or measurement problems of a tax on pollution by applying the tax to observable market transactions such as the purchase of an output by the consumer and simultaneously to subsidize other market transactions such as the purchase of clean inputs by the polluting firm.

(vii) Political considerations

In the current political climate, a new "tax" might or might not be politically feasible. The Pigouvian subsidy might have similar difficulties if it costs revenue that must be covered by raising any existing tax. Instead, various sorts of CAC regulations have been popular, perhaps because costs to consumers are not so explicit. Using a regulatory mandate, legislators can "guarantee" to their constituents that pollution will be controlled, whereas a tax must rely on the theory that firms will be induced to cut pollution. Also, existing firms may provide more support for a plan to allocate tradeable permits -- at no cost to existing firms -- than for a plan to tax all emissions.

(viii) Distributional effects

As described throughout this chapter, environmental taxes may apply to transportation, 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. Thus environmental policy reforms must be careful to use a package of changes that account for and offset these distributional effects. Yet 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) 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.

2.3 The Balance between Costs and Benefits of Using Environmental Taxes

The implications of the above are 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 will be large, and in some cases prohibitive. In addition, where the activities to be regulated are highly diverse, it is likely that considerable gains could be made from allowing the required changes in environmentally damaging activities to be achieved in the most cost-effective manner.

Private-sector energy use cannot realistically be regulated through source-by-source regulation. There are too many energy users—both businesses and individuals—and their opportunities for abatement are too diverse. The available options boil down to indirect techniques—either those that operate by restricting or widening the range of available technologies (e.g. regulations requiring standards of energy efficiency from appliances, or subsidies to promote the introduction of low-energy technologies), or incentive mechanisms such as energy taxes. In the long term, given the scale of changes that would be needed in household and business energy consumption to maintain or reduce global energy-related greenhouse-gas emissions (despite rapid industrialisation outside the OECD area), it is almost inconceivable that an effective climate-change policy could be pursued without significant use of energy pricing measures, such as carbon or energy taxes or tradable permits.

On the other hand, the analysis above suggests plenty of areas of environmental policy where advocacy of environmental taxes would be misguided. There is little to 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 costly and socially wasteful litigation, and consequent erosion. Moreover, insufficiently large tax incentives may achieve little change in behaviour. As argued above, it may not be worthwhile for firms to take account of tax incentives in 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.

2.4 Decision-making Obstacles

In the policy-making process, factors other than objective costs and benefits may well play a major role in the choice between environmental policy instruments. Different actors in the process may have divergent interests, and these may affect the policy finally chosen.

For example, it is often suggested that environmental tax policies may encounter opposition from some of the 'stakeholders' in the existing regulatory process who would be liable to lose some of the control and influence that they exert within the current regime. This is in effect the counterpart of the 'regulatory capture' argument above. Negotiated firm-by-firm regulation gives significant influence to the regulatory agency and firms, and this would be simply bypassed with a policy based on a uniform environmental tax.

A second possible influence on instrument choice is that some participants in the decision-making process may perceive costs and benefits which differ from the overall costs and benefits to society.

From the perspective of firms, an environmental tax by itself may impose costs in terms of tax payments that may outweigh the efficiency savings achieved through a more efficient pattern of pollution abatement. These tax payments are not, of course, costs to the economy as a whole, but simply transfer payments, having as a counterpart the revenue flow to government. Nevertheless, taking firms as a group, and assuming that the revenues are not channelled back exclusively to the tax-paying firms, this environmental tax may result in substantial burdens. Only if the efficiency gains from a more efficient distribution of abatement across polluters are very large would it be possible for environmental taxes to result in net gains to polluters as a group. This implies polluters will not willingly add an environmental tax if given the choice, and policy-making mechanisms that accord a significant or dominant voice to polluting firms in the choice of policy instrument are unlikely to result in tax-based policies being adopted. If the addition of the tax is combined with the removal of some costly command and control mandate, however, then firms' costs may indeed fall. Newell and Stavins (2003) find that the cost of abatement using such mandates can be several times the minimum cost achieved by using an emissions tax.

In addition to these possible influences on the decision whether or not to employ environmental taxes, both objective considerations and political pressures may influence the scale of environmental taxes that can be introduced. As discussed by Rajah and Smith (1993), there may be a number of restraints on the rates of environmental tax that can in practice be applied, that may lead policy-makers to combine environmental taxes, set below the first-best level, with other, ‘second-best’, policy instruments (such as, for example, abatement subsidies or quantitative regulation). The first-best rate of an environmental tax may, for example, present an excessive incentive for evasion, and may consequently involve excessive costs of administration and enforcement; setting a lower rate for the tax may keep the level of evasion, and the costs of enforcement, within more acceptable bounds. Another possibility is that the first-best environmental tax rate may have undesirable consequences for distributional objectives; if it is impracticable to offset this impact fully through adjustments elsewhere in the fiscal system, it may then be preferable to levy a lower environmental tax, and to supplement it with other measures (such as subsidies to increase the elasticity of behavioural responses). Third, the tax rate that can be set may be constrained by the perception of adverse effects on the international competitiveness of industry. The adverse impact of energy taxes on energy-intensive industry (although balanced by corresponding gains elsewhere) attracts enormous policy attention; the European Commission’s proposed carbon/energy tax attempted to ‘buy off’ these objections through sectoral exemptions, the Swedish carbon tax was drastically revised under pressure from energy-intensive industry, and the UK Climate Change Levy is charged at a much lower rate on energy-intensive sectors that have concluded "Climate Change Agreements" with the government. Whether justified or not, concerns about competitiveness are likely to place severe restraints on the ability of governments to set energy taxes at the first-best level, and other instruments will need to be employed in parallel.

3. Designing environmental taxes

The key to achieving the potential gains from the use of taxation as an instrument of environmental policy 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 which policy seeks to influence. Poorly targeted environmental taxes may increase the economic costs of taxation, while offering little in the way of compensating environmental gains. Indeed, badly-designed environmental taxes may be much worse than no environmental taxes at all.

A key design issue is highlighted by the contrast between two different types of environmental tax:

- ***Measured emission taxes***. This group of market-based instruments are those which involve tax payments which are directly related to metered or measured quantities of polluting effluent. Practical examples of measured emissions taxes include Sweden's tax on Nitrogen Oxides emissions, and emission charges for water pollution in the Netherlands. In both cases, large emissions sources are charged an amount based on measured total emissions from that particular source.

- ***Use of other taxes to approximate a tax on emissions***. Changes in the rates of indirect taxes (excise duties, sales taxes, or value-added taxes) may be used as an indirect alternative to the explicit taxation of measured emissions. Goods and services which are associated with environmental damage in production or consumption may be taxed more heavily (e.g. carbon taxes, and taxes on batteries and fertilisers) while goods which are believed to benefit the environment may be taxed less heavily than their substitutes (e.g. reduced taxes on lead-free petrol).

  It may be possible to achieve a close approximation to the effects of a tax on measured emissions through artful combination of indirect taxes in a "multi-part instrument". 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 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.

Both forms of environmental tax may be appropriate in particular circumstances. The choice between them, as we discuss below, will need 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.

In addition to the use of environmental taxes to provide incentives for changes in behaviour, many of the so-called "environmental taxes" introduced in practice been used primarily for revenue-raising (Opschoor and Vos, 1989). Where environmental taxes have been employed in this way, it has generally been to raise earmarked revenues for particular public expenditures related to environmental protection - for example, to recover the costs of administering a system of environmental monitoring or regulation, or to pay for public or private expenditures on pollution abatement measures. The environmental effects of the taxes themselves may be limited - indeed, in some cases, their link to the environment is solely through the use to which the revenues are put.

3.1 Measured-emissions taxation

The two types of environmental tax identified above both employ the tax system as an incentive instrument, but they achieve this in different ways. In the first case, tax payments are directly related to polluting emissions; in the second, the environmental incentive is based on an indirect relationship between the tax base and emissions. The choice between a tax that is directly related to emission quantities, and a tax that is more indirectly linked to the pollution it aims to control will depend on considerations of two sorts, administrative cost and "linkage" (meaning *efficient targeting* of the incentive). Often there will be a trade-off between lower administrative cost and better linkage. In many cases, environmental taxes based on measured emissions will have higher administrative costs than taxes which are levied on some other base, but will be better linked to the amount of pollution caused, and will thus provide a more precisely-targeted incentive to reduce pollution. The balance between these two considerations is, however, likely to differ from case to case.

New environmental taxes based on measured emissions quantities will require, as a minimum, the additional costs to be borne of a system for the assessment or measurement of the emission quantities on which the tax is to be levied. These costs will depend on:

(i) ***Measurement costs per source***. This will vary depending on the technical characteristics of the emissions (flow, concentration, stability, etc), the substances involved, and the range of currently-available measurement technologies. Recent scientific and commercial developments in measurement and control have substantially widened the range of technologies available for monitoring the concentrations and flows of particular substances in effluent discharges. Such technologies are increasingly accurate, and in many cases are now able to take reliable automated readings on a continuous basis. Technological change has therefore expanded the range of pollution problems for which charging on the basis of direct measurement could be a feasible and cost-effective option. It is also probable that the future pace of development and commercialisation of such technologies will in part be stimulated by a greater use of direct emissions charging.

(ii) ***The number of emissions sources***. Direct charging for measured emissions quantities will be less worthwhile, the more separate emission sources there are. An extreme case of this is the case of non-point-source pollution - in other words, where no identifiable pipe, outlet or chimney provides a "point source" at which emissions can be measured. The leaching of agricultural fertilisers and pesticides into the water system are examples of non-point-source pollution. For such pollution problems, direct measurement is likely to be costly and/or highly imprecise.

(iii) ***Scope for integration with normal commercial activities***. The costs of a system of emissions measurement will generally be reduced, if the measurement of emissions can be integrated with activities that would naturally take place for normal commercial reasons. Not merely does this reduce the additional costs of measurement for tax purposes, but it also tends to reduce the risk of false or misleading information being provided, since there are non-tax reasons for accurate measurement.

The administrative costs involved in new taxes on measured emissions affect the optimal environmental tax policy in three ways. First, if these costs are high, they may rule this option out entirely. Second, even where the additional administrative costs of new emissions taxes are sufficiently low to be acceptable, they may still affect the efficient design of a system of environmental taxes. Taking account of the administrative costs of taxation would be likely to mean that the range of tax instruments employed will be more restricted than would be the case if tax administration were costless; policy-makers will need to weigh up the costs and benefits, at the margin, of adding an extra emissions tax, which may improve efficiency, but at the cost of higher administration "deadweight". If different emissions are in any way correlated, an emissions tax that would be worthwhile on its own, may not be worthwhile if a number of taxes on "similar" emissions already exist. Third, in some circumstances it will be desirable to take administrative costs into account in choosing the appropriate rate at which emission taxes should be levied. Polinsky and Shavell (1982) demonstrate that the relevance of administrative costs to the efficient emissions tax rate depends on the relationship between administrative costs and the number of polluters, and on where the costs of administration are borne. For example, where the administrative costs are a fixed amount per polluter and are borne by government, it may be appropriate to set the emission tax rate above the level of the external cost, so as reduce the administrative cost, but this rule will not be appropriate where the costs of administration are borne by the polluters themselves.

As far as the issue of "linkage" is concerned, environmental taxes based directly on measured emissions can, in principle, be very precisely targeted to the environmental objectives underlying policy. 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, the only actions which the polluter can take to reduce their tax liability are actions which also reduce emissions.

3.2 Indirect taxes on pollution through restructuring existing taxes

The administrative costs of any new tax will normally depend on how much scope there is for the tax to be incorporated in existing systems of administration and control. Where the additional administrative costs of an environmental tax system based on direct charging for measured emissions are high, restructuring of the existing tax system may provide an alternative way of introducing fiscal incentives to reduce environmental damage. If the assessment, collection or enforcement of environmental taxes can be "piggy-backed" on to corresponding operations already undertaken for existing taxes, the costs of environmental tax measures may be significantly less than where wholly-new administrative apparatus and procedures are required.

The vast majority of existing taxes in the fiscal system are levied on transactions – for instance, the value of goods and services sold, the value of incomes paid or received, etc. The scale economies that can be achieved from administrative integration of environmental taxes are likely to be greatest where environmental taxes, too, are levied in a form based on transaction values. Thus, for example, the differentiation of the rates of existing taxes (the tax reform most closely compatible with existing definitions of the tax base) may gain considerably from combined administration. On the other hand, there are likely to be few gains from combining the administration of a tax on measured emission quantities with existing transaction-based taxes.

It is necessary to bear in mind that most administrative piggy-backing is unlikely to be wholly costless from the point of view of the administration of existing taxes. Greater complexity is likely to increase administrative costs in all areas - though the extent of this will depend on the existing degree of complexity in the tax structure.

How effective changes in the existing tax system can be in achieving an efficient pattern of pollution abatement will depend on the amount of “linkage” – in other words, the degree to which the taxation is closely linked to the pollution which it aims to control. If the tax rises, does it encourage taxpayers to seek to reduce this tax burden by reducing the processes or activities which give rise to polluting emissions? Or are they, instead, just as likely to find ways to reduce their tax payments that do not change their level of pollution?

For "indirect" environmental tax policies to achieve efficient polluter responses, the relationship between the tax base and pollution needs to be stable. Unfortunately, relationships which are observed to be stable in the absence of policy measures can turn out to be unstable once a tax is introduced. (Sandmo, 1976)

This issue of linkage is central to any case for or against using fiscal instruments other than those based on direct charging for measured emissions. Where the linkage between tax base and pollution is weak, the tax may fail to have the desired impact on pollution, and may, at the same time, introduce unnecessary and costly distortions into production and consumption decisions.

Where there is a wide range of available techniques which differ widely from one another in the relationship between tax base and pollution, linkage is likely to be more of a problem than where the range of technologies is small, and the relationship between tax base and pollution broadly stable across production techniques. Technical data about the range of available production techniques and their environmental attributes will thus help to assess the practical relevance of linkage problems for any particular environmental tax.

A particularly severe problem of linkage arises where it is sought to influence pollution emissions from a production process through taxes on inputs, and where significant scope exists for pollution abatement through effluent "cleaning" at the end of the production process. One case in point is the scope for cleaning the sulphur dioxide emissions of coal-fired power stations by fitting "scrubbers" (flue gas desulphurisation equipment, or FGDs). Where effluents can be cleaned in this way, taxes on production inputs will not be an effective way of encouraging an efficient pattern of pollution abatement. Such a tax – for example, a tax on sulphurous coal - may discourage the use of polluting materials in production, but will provide no incentive to clean up effluents from the process. Although pollution may be reduced, the way in which pollution reductions are achieved will not necessarily be the most efficient.

Environmental taxes on fuel inputs may be more appropriate to deal with carbon dioxide emissions, where effluent cleaning is not currently a commercially-viable option, than with sulphur emissions, where important effluent-cleaning technologies are available. However, it should be noted that what is at issue is not merely the existence of (commercially-viable) alternative technologies, but also the potential for them to be developed, since an efficient pollution tax will create an incentive for new technologies, involving less pollution, to be developed. The acceptability of a carbon tax on fuel inputs instead of a tax on measured carbon emission quantities depends in part on a judgement about how rapidly such technological developments are likely to take place, and about how far their future development might be inhibited by the choice of a tax on inputs rather than on measured emissions.

It is clear that technology will have a vital role in determining the nature of environmental taxation in the future, as already discussed in terms of direct emissions measurement and the scope for technological advance reducing the costs and improving the accuracy of this form of taxation. Environmental policy by its very nature is long-term and one consideration of any economic instrument is the extent to which it may be flexibly adapted to take account of, perhaps unanticipated, technological advancement which make new forms of policy feasible. One example is the scope for carbon capture and storage (CCS) – the idea that carbon emissions (particularly from large point sources such as power stations) can be removed and stored away rather than released into the atmosphere. An IPCC (2005) report argued that CCS could play a significant role in climate change policy in the future. At the moment, the costs of carbon capture are relatively high, and if the costs of avoiding a tonne of CO2 emissions exceed the economic damage they generate when released it would not be economically desirable to do so. The IPCC report estimated the cost of avoiding a tonne of emissions at present for a coal-fired station at around $30 - $70 compared to a coal-fired station without CCS. Clearly, though, as the technology matures, costs fall and the stock of power generation facilities around the world is upgraded or replaced it may well become a much more attractive option. Current policy design could influence the extent to which investment in such technology is worthwhile: if taxes are largely piggybacked onto existing input taxes, there is no incentive for the installation and use of CCS technology.

3.3 Multi-part Instruments

Using tax bases which "proxy" emissions, rather than basing environmental tax payments on directly-measured emissions, can thus economise on administrative costs, but risks behavioural responses which do not always achieve the most efficient patterns of pollution abatement. In some cases, a more efficiently-targeted environmental incentive can be created by using multiple proxy taxes in combination - a "multi-part instrument".

For example, a simply-administered system of charging for household refuse collections could be based on charges for special refuse sacks which householders must use. Such schemes, however, run the risk that some households respond by dumping refuse, causing environmental problems elsewhere. This dumping problem can be corrected by use of a two-part instrument in the form of a deposit-refund system. The consumer pays a disposal tax at the time of purchase, and then gets a refund or subsidy – perhaps in the form of free curbside collection of garbage and recycling. There remains the tax on any units dumped.

A number of papers have considered the efficient pattern of tax rates where the only administratively feasible policies involve some degree of approximation in the linkage between tax base and pollution damage, and have demonstrated that where linkage is poor, it may be possible to improve matters by going beyond taxation of the good associated with the externality; complements or substitutes to the externality causing good may be taxed or subsidised. Sandmo (1976), Green and Sheshinksi (1976) and Balcer (1980) consider cases where individual externality effects differ, and where the optimal tax on the good causing the externality would therefore be at different rates. Where goods exist that are complements (or substitutes) to the externality-generating good, this may allow a better package of tax measures to be designed, including changes to the taxation of these related goods. Depending on the interaction between demand for the complement (or substitute) and the externality, it may be possible to improve on the taxation of the externality-causing good alone, by taxing or subsidising the complementary or substitute good. For example, if the objective is to deal with urban congestion, high petrol taxes might be supplemented by subsidies to urban public transport and taxes on urban parking spaces[6].

[6] Unfortunately the appropriate policy rules are not always straightforward. Balcer (1980) shows a simple case where a subsidy to a complementary good would be appropriate, and Wijkander (1985) discusses cases where the policy rule is complicated by the cross-effects between various complements or substitutes to the externality-causing good.

4. Revenue aspects of environmental taxes

What is the revenue potential of environmental taxes, and how much difference could they make to the balance of the fiscal system, and to the scope for tax reform? Are the potential revenues large enough to significantly alter the constraints and opportunities in tax policy-making?

Many environmentally-motivated tax measures that might be contemplated do not have the potential to raise significant tax revenues. The tax bases involved are insufficiently large to yield revenues that significantly alter the overall tax structure. Taxes on certain types of battery, for example, or on plastic carrier bags, or on household purchases of garden fertilisers and pesticides, have all been employed in some European countries. In each case, they may have appreciable merit as an instrument of environmental policy, but individually, their revenues are negligible in the context of the overall public finances. Even the substantial tax introduced by the UK on the use of landfill sites for dumping waste, which currently yields about £800 million per year, contributes less than 0.2 per cent of total revenues. In addition, of course, some environmental taxes may achieve highly elastic polluter responses, eroding the revenue yield. Thus, for example, the tax differential introduced between leaded and unleaded petrol in many European countries was followed by fuel substitution, as consumers shifted to the lower-taxed fuel.

Significant scope for major tax reform financed by the revenues derived from environmental taxes is only likely to arise in the case of two potential environmental tax bases—taxes on road transport, and on energy. Congestion charges on private motoring could, for example, be a major source of tax revenues, if levied at a rate reflecting the congestion externality imposed by each individual motorist on other road users. For example, Newbery (1990) estimated that the congestion cost per vehicle-kilometre averaged some 3.4 pence across the UK. If this was fully reflected in a UK-wide congestion charge, it would imply revenues of £20 billion annually at current values, about 5 per cent of total fiscal receipts. The more limited London congestion charge, which takes the form of a £8.00 flat-rate daily fee for access to the central London area, raises £120 million annually, a significant amount in relation to the total budget of the sponsoring authority.

The taxation of energy, to reflect the environmental externalities involved in energy use, is the other area with potential to raise major tax revenues. The European Commission’s proposal for a carbon-energy tax in the early 1990s would have generated substantial revenues for member states, of between 1 and 3 per cent of total fiscal receipts, despite exempting the most energy-intensive sectors. In 1994, the UK government ended VAT zero-rating on domestic energy as one of a package of revenue-raising measures. The initial proposal to tax energy at the standard VAT rate of 17.5 per cent would have raised some £3 billion of revenues annually, some 1 per cent of total fiscal receipts, but subsequent modifications have reduced the tax rate to 5 per cent, and annual revenues to around one-third of the original projection. Still larger revenues could be raised through carbon tax schemes designed to use carbon pricing to halt the growth in carbon dioxide emissions. The recent Stern Review of the Economics of Climate Change concluded that the social cost of carbon was of the order of $300 (£160) per tonne of carbon in current prices. A carbon tax at this level applied uniformly to all UK energy use, would raise revenues of some £24 billion, equivalent to about 5.5 per cent of total UK tax receipts. However, a more limited carbon tax, at one tenth of this rate, would be implied by Tol's (2005) "consensus" estimates of the social cost of carbon, and would make a correspondingly smaller impact on overall revenues. Moreover, the additional revenue contribution, over and above the £0.7 bn revenue already contributed by the existing “Climate Change Levy” (a tax on industrial and commercial energy use, introduced in 2001), would be smaller still - less than half of one per cent of total UK tax receipts.

The observation that it may be hard to justify the retention of some existing taxes in parallel with new environmental taxes is particularly relevant to the case of road transport taxes. Once congestion externalities and greenhouse gas emissions are separately taxed, this would substantially undermine the case for motor fuel excises and other taxes on motoring at the current very high levels. At present these contribute substantial revenues - the excise duty on motor fuels raises about £24 billion annually and Vehicle excise Duty about £5bn. If the remaining motoring externalities did not justify retention of the existing high taxes on motor fuels, the net revenue gain from a congestion tax and carbon tax would be substantially lower.

4.1 Revenue Sustainability

The revenues that would be raised from environmental taxes on particular raw materials or products associated with pollution will be a function of the responsiveness of demand and supply to price. The more effective the tax is in restraining production and use of the taxed good, the lower will be the revenue derived from the tax. In some sense, therefore, revenue issues arise in inverse proportion to the environmental effectiveness of an environmental tax; the tax is paid and revenues obtained only where the good continues to be produced and consumed.

The effects on revenues of an environmental tax are likely to change over time. Since, in general, supply and demand responses to the imposition of an environmental tax are likely to be rather greater in the long run (when taxpayers' patterns of production and consumption can be freely adjusted), than in the short run (when taxpayers' production and consumption decisions may be constrained by existing capital equipment), there may be circumstances where the revenues to be obtained from the environmental tax could decline over time. Where long-run supply and demand responses to the environmental tax are large, reflecting the existence of close substitutes which are less heavily taxed, the opportunities and problems posed by the tax revenues and the burden of additional tax payments will be short-lived.

In practice, forecasting the long-run revenue effect of environmental taxes is unlikely to be a precise matter. Not only are there likely to be important uncertainties regarding the size and timing of the effects of the tax on production or consumption of the good in question, but also demands and hence revenues will be a function of the overall economic climate and level of economic activity. Economic growth may increase demands for the polluting good, partly (or fully) offsetting the effects of the environmental tax. Where the price elasticity of demand for the taxed good is low, and the income elasticity is high, the increases in demand due to growth are likely to be large relative to the reductions in demand due to the environmental tax. Thus, one concern in considering the use of tax on energy to control environmental problems associated with energy use is that the price elasticity of energy demand is so low, that a steeply rising energy tax would be needed merely to keep energy demand constant in the face of rising incomes.

4.2 A 'Double Dividend'?

From the perspective of fiscal policy, what are the gains from using environmental taxes? Do they have the potential to reduce the overall costs involved in raising fiscal revenues? Some commentators (e.g. Pearce, 1991; Oates, 1991) have drawn attention to a potential 'double dividend' from environmental taxes—the possibility that, in addition to their merits as instruments of environmental policy, they have a second benefit in that the revenue raised from the environmental taxes allows other taxes, with possible distortionary effects on labour supply, investment, or consumption, to be reduced. There are a number of strands to this argument.

(i) 'Distortion-correcting taxes are better'

Empirical studies of the marginal distortionary costs (the marginal excess burden) of existing taxes show that these costs can be appreciable. For example, Ballard et al. (1985) estimate the marginal excess burden of public revenues in the USA at 20–30 cents for each extra dollar of tax revenue. These costs reflect the fact that most taxes (apart from lump-sum taxes) lead to behavioural adjustments which reduce individual welfare, over and above the value of the actual tax payment by the private sector. Raising the rate of conventional taxes will typically increase these distortionary costs. However, the behavioural adjustments that arise from environmental taxes include some which are positively desirable, reflecting changes in private-sector activities that reduce emissions. In these circumstances, making use of environmental taxes to raise revenues would appear distinctly preferable to relying on conventional taxes, which generate undesirable distortions in activity. Surely it must be better to raise revenues from taxes that correct distortions, rather than create them?

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, there would, indeed, usually be gains from shifting the balance of revenue-raising towards greater reliance on environmental taxes. In this sense, the tax system will be more efficient if environmental taxes are used, than if they are neglected. However, there are two key observations.

First, the particular meaning of 'excess burden' in this argument should be noted. Environmental taxes on energy can be said to have negative excess burden (at least over some range) if we include within the definition of the excess burden the environmental benefits from the induced behavioural changes. Environmental taxes, on this argument, may have negative excess burden, but to say that they have this desirable property in addition to their environmental benefits involves double counting.

Second, although there will almost certainly be gains from some shift to distortion-correcting taxes, this will be true only up to a certain point. The purely fiscal component of the excess burden, in the form of such things as the reductions in energy consumption and expenditures on energy-saving technologies, will have costs that rise more than proportionately with the rate of tax. Raising the tax rate on energy will initially confer benefits, in the sense that the environmental gains offset the costs of these behavioural adjustments, but as the energy tax rate is further increased, the costs of these behavioural changes will rise more than proportionately, eventually overtaking the additional environmental benefits. As Oates (1991) observes, economic efficiency in raising public revenues requires that the marginal deadweight burden from each revenue source be equal; in other words, that there should not be scope to raise the same revenues at lower deadweight cost by changing the pattern of public revenues. This will imply shifting the pattern of revenue-raising towards environmental taxes, up until the point where the marginal excess burden of each environmental tax has risen to equal the marginal excess burden from other taxes.

The above form of the argument that introducing environmental taxes will lead to more efficient fiscal policy does not establish clear and separate 'environmental' and 'fiscal' dividends from the use of environmental taxes. The environmental benefits form part of the claim that revenues can be raised at lower cost through environmental taxes; there are not two separate 'dividends'.

(ii) 'Using environmental tax revenue to reduce other tax rates reduces excess burdens'

A second strand in the 'double dividend' literature concerns the significance, or value, of the revenues raised from environmental taxes. What, if any, are the benefits from choosing an environmental policy instrument which raises revenues, in preference to one which has similar environmental effects but raises no revenues? If we employ 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 give us a more efficient fiscal policy, compared with the use of an equivalent non-revenue-raising instrument?

The closest comparison that can be made between a revenue-raising environmental policy instrument and one that is otherwise identical but raises no revenues, is that between auctioned and 'grand-fathered' tradable permits. Auctioned tradable permits trading at a price x per unit of emissions would provide the same incentive for pollution abatement, and raise the same revenues, as an emissions tax set at the same rate x per unit. The comparison between auctioned and grandfathered tradable permits thus provides a way of thinking through the consequences of two equivalent instruments, differing only in the fact that one raises revenues.

Under a 'grandfathered' scheme, permits are distributed free of charge, according to some system of distribution. Polluting firms may, for example, be allocated permits in proportion to their emissions levels in some past period. The free allocation of permits on the basis of historic emissions levels has a clear opportunity cost to the government. Revenues could have been raised by auctioning the permits, since they are of value to the firms that receive them, and, instead, the government is making a transfer to the firms, of a value equal to the number of permits allocated, times the price at which they subsequently trade. Aside from the possibility that the number of firms in the industry might be affected if permits are auctioned rather than distributed free, it would be expected that the market price at which permits trade would be identical under the two regimes, and the level and pattern of pollution abatement would be identical. The only difference between the two regimes is, then, that one raises revenues, while the other forgoes the opportunity to raise revenues. The latter case may be seen as equivalent to the case where permits are auctioned, and the revenues raised then transferred back to firms through lump-sum transfers.[7]

Looking at the above comparison between auctioned and grandfathered tradable permits, the environmental effects are held constant, and the only difference is that revenues are obtained under the former, but not under the latter. Any fiscal policy benefit from the revenues raised can be clearly distinguished from the environmental benefits from the instrument, which are identical across the two cases.

The revenues raised from the revenue-raising instrument do clearly have a benefit in that they reduce the need to raise revenues from other taxes, and reduce the need to incur the distortionary costs involved in raising revenues through these taxes.

In his discussion of the double-dividend debate, Goulder (1995) refers to this case as that of a 'weak' double dividend. Cost savings are made by using environmental tax revenues to reduce distortionary taxes, rather than returning tax revenues to taxpayers through lump-sum payments. He points out that the existence of a double dividend, in this sense of the term, 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'.

The claim of a double dividend in this form is undramatic, but not without policy significance. In making a choice between 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 created, through taxes or the auction of a fixed number of permits, then the scarcity rents need to be captured by government and used to reduce the rates of existing distortionary taxes. There are significant costs if the potential revenues from environmental taxes are dissipated or forgone.

Two examples can be drawn from the recent discussion of the possible use of economic instruments to reduce business use of energy in the Marshall Report (1998). First, the Marshall Report pays considerable attention to the idea that the tax revenues derived from energy taxes levied on industry should be returned to firms, rather than used in other ways. The economic arguments which would support this recommendation are considerably less clear-cut than Marshall appears to think. In the long run, it is not at all obvious that reducing taxes on firms would enhance UK firms' competitiveness by more than if taxes on individuals were reduced. However, if we accept Marshall's recommendation that the revenues should be returned to industry, the implication of the analysis above is that it matters how this is done. Efficiency gains will be made if this is done by reducing the marginal rates of other

[7] 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.

taxes, and arrangements for revenue return which have a more lump-sum character (as with the return of revenues from the Swedish NOx charge described earlier) will forgo these fiscal gains.

Second, the Marshall Report discusses the relative merits of grandfathered and auctioned tradable permits. The report rightly observes some significant difficulties with grandfathered permits. The basis for allocation may well be controversial, and allocation on the basis of historic emissions would tend to disadvantage firms which had already reduced emissions in the past, and give greatest benefit to the least-dynamic firms. Also, grandfathering may discourage competition, because new entrants to an industry (which have to buy permits) do not compete on equal terms with existing firms (which receive an allocation of free permits). However, the report omits to mention probably the most serious disadvantage of grandfathering, which is that it simply forgoes the chance of capturing scarcity rents. If the estimates of marginal excess burden from Ballard et al. (1985) are taken, the cost of grandfathering, in terms of the forgone fiscal gains from recycling the revenues through cuts in the marginal rates of distortionary taxes, is of the order of 20–50 per cent of the forgone revenues.

A corollary of the double-dividend argument in this form is that the optimal level of pollution abatement will not be independent of the environmental policy instrument used (Lee and Misiolek, 1986). Where increasing the rate of the environmental tax increases tax revenue, instruments such as regulation or grandfathered tradable permits which forgo revenue will have a higher total marginal abatement cost (taking into account the marginal deadweight burden of raising public revenues as well as the conventional marginal abatement costs) than environmental tax instruments, which can use the extra revenue raised to reduce the distortionary costs of other taxes. In this case, an efficient policy will set a higher level of pollution abatement if the tax instrument is used than if an environmental policy instrument is employed which does not create scarcity rents and fail to capture them.

(iii) 'Switching to environmental taxes reduces excess burdens'

There has recently been a spate of theoretical papers which have modelled the conditions under which a double dividend would arise, in a more demanding sense of the term, defined by Goulder (1995) as a 'strong' double dividend. He defines this as the claim that a tax switch which increases taxes on energy and reduces existing non-environmental taxes would have negative 'gross costs'. In gross costs he includes all the welfare costs of all behavioural changes from the tax switch, but excluding the environmental benefits. This is an extremely demanding criterion, and will be seen to be a substantially different claim from that being made at the start of this section. A shift towards environmental taxes would have a double dividend only if there were environmental gains (the first dividend), and if the total deadweight costs of revenue raising (excluding the environmental benefits) are negative. The double-dividend argument in this 'strong' sense becomes a 'no regrets' argument; even if the changes in energy use turn out to have no environmental benefit, achieving them has been costless because the overall fiscal costs of the tax change are negative.

An extensive academic literature has focused on the general validity of such a proposition. Bovenberg and de Mooij (1994) and others have shown that the environmental tax has its own distorting effects on labor supply and therefore can have the same excess burden as a tax on labor income. Thus the double-dividend hypothesis is said to fail.

In this section, we make four main points. First, the validity of the double-dividend hypothesis cannot logically be settled as a general matter. Clearly, under some conditions, a particular reform might be able to improve the environment and improve the tax system by reducing some particularly egregious existing tax. Equally clear is that some other misguided reforms would not. Each proposal must be evaluated individually. The important point is that this evaluation must *fully specify the policies already in place as well as the reform under consideration*. If this polluting activity is already taxed at a rate higher than the "optimal" rate, taking all considerations into account, then any suggested increase is not warranted. Even if it is taxed at a low rate, or not at all, the polluting activity might already be subject to other regulatory restrictions. Existing policies are crucial to understanding the benefits of any proposed reform. Moreover, the reform itself needs to be fully specified: is this tax added on top of existing regulatory restrictions, or does it replace those restrictions? And how will the revenue be used? In this regard, an important contribution of the double-dividend debate is that the proposal to add an environmental tax is only half of a proposal, since the reform must also specify whether the revenue goes to deficit reduction, a specific spending program, or a specific tax reduction.

Second, the double-dividend literature relates to another earlier literature about how some types of environmental policies generate "scarcity rents" by restricting the amount of pollution. To at least some extent, a restriction on the amount of pollution is a restriction on the amount of output, which enables firms in equilibrium to charge a higher price for their output. Given this higher price of output, the right to pollute is more valuable. The "scarcity rent" is the increase in the value of the right to pollute one unit. It is reflected, for example, in the price of a tradeable permit for one ton of sulfur-dioxide emissions under the Clean Air Act Amendments of 1990. These permits are sold on the Chicago Board of Trade for about $150 each. Other command-and-control restrictions create similar scarcity rents, even if pollution rights are not tradable. Consider the simple case where the production technology requires a fixed amount of pollution per unit of output, and where the government requires every firm to cut pollution to 90 percent of last year's level. Then firms must cut production to 90 percent of last year's level. The price of output must rise, for the market to clear, but actual production costs have not changed. Normally firms are prohibited from agreements to restrict output, but this kind of regulation essentially requires them to restrict output. The result is super-normal profits.

The point is that environmental protection can raise the price of output for two very different reasons. Prices may rise because of the necessary costs of environmental technologies such as switching to low-sulfur fuel, switching output to a plant with lower emission rates, or installing flue-gas desulfurization units (scrubbers). And, output prices may rise further, to cover scarcity rents. The former costs can be minimized by well-designed policies, but they must ultimately offset some of the benefits of environmental improvement. The latter type of cost is not essential to environmental protection. Scarcity rents raise the cost of output unnecessarily, which offsets more of the benefits of environmental improvements. These costs may exceed the benefits of environmental improvements, which would turn the whole reform into a net losing proposition.

Moreover, this point about scarcity rents helps explain what was missing in the debate about the double-dividend hypothesis. When a tax on pollution raises revenue, the government is merely capturing the scarcity rent associated with restricting that pollutant. In a sense, the tax raises the cost of production by more than the minimum necessary. It requires the firm not only to install scrubbers or undertake other expensive pollution abatement, but also to pay the tax. The revenue is the scarcity rent. The difference is that the government can use those revenues to *offset* the increased costs of production, by reducing other existing taxes on production. This point can be clarified by comparing the handout of tradable permits, as under the Clean Air Act Amendments (CAAA), with an alternative government sale of permits at auction. In both cases the permit requirement increases the cost of production, but only in the latter case does the government capture the scarcity rent in a way that allows it to reduce other tax-related costs of production.

The third major point is a critical distinction between two types of command-and-control (CAC) regulations. Some regulations restrict pollution, and create scarcity rents, without capturing those scarcity rents in a way that would allow government to offset the extra costs of production (by reducing some other tax). The CAAA falls in this category. The only advantage of this type of regulation is political feasibility, since the prospects for private profits may induce firms to support the proposal. A different kind of command-and-control regulation does not create scarcity rents in the first place. Examples are policies that require all firms to use the latest technology, or otherwise to reduce pollution *per unit* of output. If properly designed, this kind of requirement can improve the environment at minimum cost, but firms are still allowed to pollute and produce as much as desired. Thus it does not restrict entry and create scarcity rents. This kind of CAC policy can have the same effects as a policy shift to a pollution tax (which further raises costs of production) while using the pollution tax revenues to cut another tax (which reduces costs of production).

Recent contributions to the double dividend literature emphasize that environmental policy tends to raise output prices in a way that reduces the real net wage (Bovenberg and de Mooij, 1994). If pollution tax revenue is used to reduce the labor tax, it can offset the effect on the real net wage – with no net effects on distortions from the labor tax. The only dividend is the environmental dividend. Thus, much of the literature has emphasized the importance of raising revenue in order to obtain this second dividend.

Our fourth main point, however, is that the focus on revenue-raising is misplaced. The key is for policy to avoid creating scarcity rents, or to capture those scarcity rents. In the tax example just stated, the scarcity rent is captured and used to reduce labor taxes, with no adverse effects on the real net wage. Yet Fullerton and Metcalf (2001) demonstrate that two other *non*-revenue-raising policies have exactly the same effect. One is a restriction on pollution per unit of output that applies to all firms in the industry. Any new firm can enter, so the policy does not create entry barriers or generate scarcity rents. For firms that were already operating at the privately optimal pollution/output ratio, a small restriction on that ratio has negligible effects on costs because of the envelope theorem. Yet it does reduce pollution. In other words, using this policy, the marginal cost of abatement (MCA) emerges from the origin. Further restrictions may be costly, but the initial restriction has MCA=0. It does not raise product prices, and therefore does not reduce the real net wage nor exacerbate labor tax distortions.

Even more counterintuitive is that a revenue-losing subsidy can have the same effects. Consider a subsidy to abatement of all firms, which also does not restrict any output nor create scarcity rents. All firms may be induced at the margin to cut pollution, but this subsidy to abatement may also reduce costs of production. The equilibrium output price falls, which tends to *raise* the real net wage. Therefore, even if the subsidy needs to be financed by *raising* the labor tax, the effects on the real net wage offset.

A well-designed reform may generate environmental benefits, and it may reduce other existing distortions, but those outcomes are entirely unrelated to whether it raises revenue. A non-revenue-raising type of command-and-control regulation can have identical economic effects to the combination of an environmental tax increase and income tax reduction. A revenue-losing environmental subsidy (financed by an increase in the income tax) that has identical economic effects to a revenue-raising environmental tax (with revenue used to reduce the income tax). If designed to affect behavior in the same way, all three have identical economic effects. The choice among these three policies then depends on considerations *other* than revenue, such as which policy is easier to administer, easier to enforce, or easier to enact.

Part B: Applications

5. Carbon taxes on energy use by industry and households

At the Earth Summit in Rio in June 1992 more than 150 countries signed the UN Framework Convention on Climate Change, which established a collective commitment to take action to ensure that the level of greenhouse gases in the atmosphere does not give rise to dangerous man-made effects on the global climate. This commitment responded to the accumulating scientific evidence, drawn together by the Intergovernmental Panel on Climate Change, that human activity has been responsible for a substantial rise in the concentration of greenhouse gases in the atmosphere (IPCC First Assessment Report, 1990), and that this has had a discernible influence on global climate (IPCC Second Assessment Report, 1995). Subsequent negotiations led to the Kyoto Protocol, agreed in 1997 and ratified in 2005, under which a number of industrial countries took on binding quantitative targets to reduce their emissions of a basket of the principal greenhouse gases, measured against a baseline of the 1990 emissions level. Within the Kyoto Protocol different countries took on different targets. For the EU countries as a whole a reduction of emissions of 8% is required by 2008-2012, compared to 1990 levels. A subsequent "burden-sharing" agreement within the EU established targets for individual member states; for the UK the commitment is a 12.5% reduction over this period.

The policy instruments employed by countries in taking measures to meet these international commitments have been extremely varied:

- Regulations on fuel efficiency of vehicles, appliances, etc, and other policy measures to stimulate greater energy efficiency.
- Energy pricing measures, including carbon taxes in at least five European countries, Finland, Sweden, Norway, Denmark and the Netherlands, and energy taxes in others (including the UK Climate Change Levy).
- Emissions trading schemes, in the UK, Denmark, Canada and the EU
- Voluntary agreements between industry and government (including the UK's Climate Change Agreements).
- Policies to stimulate research, development and diffusion of low-carbon energy technologies (through R&D subsidies for renewables, etc).
- Joint implementation deals, to support emissions reductions in other countries where abatement is less costly.

There are good reasons for this "portfolio" approach to policy, making use of a variety of different instruments. Unlike the road transport externalities which we discuss in the next section, the reason for this is not the presence of multiple externalities, and the difficulty of designing instruments precisely targeting the marginal externalities associated with individual decisions. With global climate change, the externality involved is driven by an unusually straightforward and single-dimensional variable - the global stock of greenhouse gases - and the relevant emissions contributing to the problem can be clearly defined, and straightforward pricing instruments can be readily envisaged. Nevertheless, it is unlikely to be possible to tackle the problem using simple pricing instruments alone, because of the scale of the adjustments required, and a range of both real and perceived obstacles to setting pricing instruments at first-best levels.

This section considers the issues involved in using taxes - a carbon tax, based on the carbon content of fossil fuels, or other general taxes on household and industrial energy use - to reduce emissions of carbon dioxide, the principal greenhouse gas. The analysis is in four main parts. The first provides the backdrop to the analysis, by reviewing the nature of the global warming externality, and recent assessments of its scale. A key focus of this discussion is the contribution made by the recent Stern

Review on the Economics of Climate Change which has aimed to provide a framework for assessing the economic case for policy action on global climate change, through an assessment of the economic impact of the climate change externality and the costs of intervention. This discussion indicates the potentially substantial scale of intervention required, and the substantial carbon tax that would be required if Stern's assessment of the social cost of carbon emissions were to be fully reflected in the pricing of fossil fuels.

Translating an estimate of the social cost of current carbon dioxide emissions into the appropriate rate of carbon tax is complicated by a number of considerations. We discuss two of these in section 5.2. First we consider dynamic considerations bearing on the optimal time path of a carbon tax: should the tax be set at a high level immediately, or phased in over time? Second we consider issues relating to the international context of policy. Since control of CO_2 emissions is designed to deal with a global environmental problem, the policies appropriate in individual countries will depend on what other countries are prepared to do. A key issue discussed in the literature is whether the form of international policy coordination which is required should continue to be based on quantity targets, or whether the carbon tax rate should form the focus for international coordination.

We then turn to more detailed issues of carbon tax design and implementation. Section 5.3 defines precisely what would be involved in using carbon or energy taxes to control CO_2 emissions, and discusses the possible ways in which a carbon tax might be implemented. Section 5.4 considers the possibility that efficient policy could involve high carbon tax rates, and looks at issues of adjustment efficiency, international competitiveness and distributional effects which may arise when energy is subject to heavy taxation.

5.1. What is the externality, and how large is it?

The *Stern Review of the Economics of Climate Change*[8] was commissioned by the UK Treasury in summer 2005 and reported in October 2006. Its remit was to examine the economic and environmental costs of climate change, and the costs and benefits of policy to reduce greenhouse gas emissions.

The *Review* concluded that there was a strong and urgent case to reduce greenhouse gas emissions, stabilise the concentration of GHGs in the atmosphere and limit any potential increase in global temperatures. Under the *Review's* modelling estimates, “business as usual” would generate a 50-50 chance of warming relative to pre-industrial temperatures of around 5 degrees, compared to 2 or 3 degrees if GHG concentrations could be stabilised. To do so would require cuts in global emissions amounting to about 25% by 2050, and 80% in the very long run.

The *Review's* modelling led to the conclusion that the costs of mitigating climate change now were substantially less than the possible costs of doing nothing. Explicitly charged with incorporating the risk of very high costs into its estimates, the estimates of the costs of unchecked climate change were variously reported at around 5 – 20% of global output whilst the costs of mitigation by 2050 were estimated at equivalent to 1% of global output. In terms of policy, the *Review* argued that international solutions were most desirable, in particular the establishing of a global carbon price either through taxation, trading or regulation (or some combination of all three) but also argued there was scope for subsidy for low-carbon technology and some role for adaptation to climate change such as investment in coastal protection. In particular, the *Review* estimated the current social cost of a tonne of CO_2 emissions at around \$85, or in carbon rather than carbon dioxide terms, around \$300.

The *Review* was largely welcomed by all the main UK political parties. However amongst the wider economic and scientific community, whilst there have been many voices welcoming the analysis of the *Review*, there have also been many critical analyses published. These have tended not to critique the

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[8] http://www.hm-treasury.gov.uk/independent_reviews/stern_review_economics_climate_change/sternreview_index.cfm

spirit of the *Review*, but instead to focus on particular assumptions, modelling methods or presentational aspects of the *Review's* findings[9].

Of particular concern for attempting to analyse optimal environmental policies is the estimate of the external costs of carbon emissions. The Stern estimate is considerably higher than most earlier economic analyses – a meta-analysis by Tol (2005), for example, suggested a central figure of around $29/tC, an order of magnitude smaller than the Stern figure. Which is 'right' has important implications for the optimal size of a global carbon tax, for example, or even what level UK fuel duty should be set at (see discussion in section 5).

A response published on the UK Treasury website to some of the criticism of the *Review* argued that the Stern estimate of the cost of carbon was higher "… because of our approach to discounting, risk aversion, and the latest science." In particular, it seems that the *Review* attempted to explicitly model the risk of extremely catastrophic events occurring as a result of climate change, and factor this into the estimated costs. Whilst these events may be extremely unlikely even under unchanged environmental policy, it can certainly be argued that some concern should be paid to them in terms of attempting to calculate the cost of carbon emissions.

In terms of discounting, some criticisms of the *Review* focused on its choice of two key variables in estimating the costs of climate change and the costs of mitigation. First, the rate of 'pure time preference' (δ), which is a parameter reflecting the extent to which future periods should be discounted relative to today. The *Review* uses a figure of 0.1% per year, effectively meaning that the future is valued almost entirely as much as the present in calculating costs that may only occur or may persist many decades ahead. Nordhaus (2006) argues that this extremely low discount rate can produce what appear to be positive benefits in a net present value sense of a large income sacrifice today for a very small income benefit many years ahead. The second parameter, the elasticity of the marginal utility of consumption (η), describes how the income of the rich is valued in welfare comparisons *vis-á-vis* the income of the poor (regardless of when they may exist). Higher values imply that an extra pound of income is valued much more strongly for someone with low income than high income. The *Review* selects a value of 1 for this parameter which implies relative inegalitarianism – a pound of income for the rich is valued about the same as that for the poor. Dasgupta (2006) argues that the combination of low δ and low η is inconsistent since it implies much richer future generations are to be given almost equal weight in welfare evaluations as current generations.

Indeed, it does seem that some of the *Review's* conclusions are quite sensitive to the choices of these parameters. The central conclusion of a cost of 'business as usual' emissions amounting 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%. [presumably also implications for cost of carbon but not sure if they are detailed?]

Ultimately, of course, the questions over what these parameter values should be and the extent to which highly unlikely but extremely costly outcomes should form part of the analysis of the costs of carbon emissions are matters for individual judgement. There is obviously considerable uncertainty and huge controversy around these issues and taking values from one extreme of estimates or the other will clearly have massive policy implications. [AL: I want to say something here about how this shouldn't distract from the need to do something, take some central estimate, can always adjust if seems policy wrong – perhaps something a bit less woolly than this. But I can't think of any particularly good way to write it.]

5.2 When should action be taken, and by whom?

5.2.1 Dynamics.

[9] In particular, the authors cited here as "critics" of the *Review* should not be interpreted as critics of the idea that there is a case for governments to intervene to mitigate the effects of climate change.

In the most straightforward formulation of the Pigouvian tax problem, that of a single-period externality, the efficient externality tax (in the absence of any wider fiscal considerations) would be set equal to the level of marginal pollution damage at the socially-optimal level of emissions. Determining an appropriate rate for a carbon tax is however, considerably more complex than in this simple case, because the CO_2 control problem involves significant multi-period, or dynamic, elements. These make the assessment of marginal pollution damage more difficult, and also mean that the policy-maker may need to consider the optimal time path of carbon tax rates, rather than a single optimal tax rate.

(a) The damage from global warming is a function of the accumulated stock of CO_2 and other greenhouse gases in the atmosphere. Since the rate of decay of any addition to the stock of atmospheric CO_2 is slow, current emissions have an effect which extends into many future periods. Likewise, policy measures taken now potentially confer benefits on future generations as well as 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) concerning the treatment of large gains and losses in the distant future, which conventional discount rates could render of negligible current value.

(b) There may be a substantial adjustment lag to any climate change policy based on energy pricing measures, since the level of energy use in the economy will be partly governed by the speed at which the existing capital stock is replaced. The scope for changing the energy intensity of production using existing capital equipment is likely to be much lower than when existing machinery is being replaced. As Ingham, Maw and Ulph (1990) show, the average life of plant and machinery may be of the order of 15 years. Even if the introduction of a heavy new tax on energy led to some acceleration of the replacement of the most energy inefficient capital equipment, a considerable proportion of existing capital would be likely to continue in use for a decade or more, and full adjustment of energy use to higher energy prices would thus take many years to complete. At the same time, there may be scope for policy to exploit announcement effects, in which the announcement of future carbon taxes may affect current investment decisions and lead to the adoption of low-emission technologies, even before the carbon tax is actually imposed.

(c) Using taxes on fossil fuels to control CO_2 emissions has a further dynamic aspect, due to the fact that the tax would be levied on a non-renewable natural resource with a finite stock. If a tax is levied permanently at a constant *ad valorem* rate (ie a constant percentage of price) on an exhaustible resource (with a zero cost of extraction), it will have no impact on the time profile of resource extraction and use (Dasgupta and Heal, 1979). Following this line of argument, Sinclair (1992) suggests that a high carbon tax would be liable to be ineffective if it is constant over time, and a rising carbon tax over time would be even less attractive, because it would accelerate the depletion of the energy stock, and hence lead to higher carbon emissions in the initial years of the policy than if no carbon tax is levied. To reduce current carbon emissions it is necessary to shift depletion of the finite energy stock to future periods, which would require that the carbon tax should fall as a percentage of the energy price over time.

The implications of the above for the carbon tax rate that should be levied per unit of carbon are, as Ulph, Ulph and Pezzey (1991) note, not immediately apparent. The Hotelling rule would imply that the price of the finite energy resource would be rising, and though the optimal ad valorem tax rate might be falling, the tax rate per unit of carbon need not necessarily also be falling. In addition, there are other factors which may also be relevant in assessing the optimal dynamic profile of tax rates, including the effect of natural decay in the stock of atmospheric CO_2, and of discounting of future costs. Ulph, Ulph and Pezzey (1991) show that the optimal time profile for a carbon tax may be quite complex, and the optimal tax per unit of carbon may initially be low, and might subsequently rise. For a period, the optimal tax may even rise in ad valorem terms.

5.2.2 International bargaining and co-ordination of carbon tax policies

Given the global nature of the climate change problem, effective policy will need to involve international coordination. The impact that an individual country can make on climate change through independent action is negligible, whilst national policies incur appreciable costs of abatement. Other literature has discussed many of the issues in achieving efficient bargains for the control of international environmental phenomena, including the distribution of the costs of CO_2 control, the achievement of a stable coalition of signatories to an international agreement, and so on.

There are some interesting interactions between the issue of instrument choice and the feasibility, credibility and/or efficiency of international agreements. In particular, is bargaining over tax rates likely to lead to a better or worse outcome than bargaining over quantitative emissions reductions? And if countries agree on coordinated tax measures, how far should carbon tax rates be harmonised?

In some circumstances it would, in principle, be desirable for international negotiations to go further than simply specifying quantitative targets for emissions reductions in individual countries, and to determine the form of policy measures to be taken.

One is that negotiating over tax rates rather than over quantitative emissions targets may in certain circumstances achieve a more efficient pattern of abatement across countries. For example, where global pollutants are concerned, agreement on a uniform tax rate to apply in all countries may be more efficient than agreeing that all countries implement the same percentage reduction in pollution. Indeed, in some circumstances involving uniformly-mixed global pollutants a uniform tax rate on emissions across countries would constitute the optimal policy. Hoel (1994) considers whether a carbon tax, levied to control climate change problems arising from the accumulation of global carbon dioxide emissions, should be uniform across countries. Ideally, he finds, the tax should be uniform. But there are two sets of circumstances in which non-uniformity across countries could be appropriate. One is where side-payments are ruled out, a case which Hoel, however, argues is unlikely to be a major constraint in practice. In this case, agreements must confer net benefits on all signatories, and non-uniform carbon taxes could then be Pareto-optimal. Another group of circumstances concerns the availability of other tax instruments which can be deployed to tackle other energy-related externalities, and to tax energy within a structure of optimal revenue-raising taxes on commodities. Typically, the efficient rates to be set for these energy taxes will vary across countries; in the first case to reflect differences in abatement costs, the assimilative capacity of the national environment, and citizens' preferences for environmental quality, and in the second case to reflect the range of factors underlying optimal tax structures. If separate taxes on energy were not available, and the carbon tax had to perform these functions as well as reducing carbon emissions, uniformity in the carbon tax rate across countries would be unlikely to be optimal. However, since all countries already levy substantial taxes on at least some types of energy, Hoel argues that the case for uniformity across countries in a new coordinated carbon tax would appear to be strong.[10]

As Smith (1995) discusses, another reason for specifying the form of policy, rather than simply emission reduction targets, may be the credibility of any agreement. Agreeing to introduce a tax at a particular rate may make it easier for countries to verify that the bargain was being implemented, than where the agreement was simply to undertake unspecified measures to achieve a quantitative target for emissions reduction at some future date. It may be particularly difficult for other countries to judge whether a package of non-fiscal measures is sufficient to achieve a country's commitments to carbon abatement, or whether it is simply an attempt at window-dressing. It may then be relatively easy for countries to free-ride without detection, and this will reduce the incentive for any country to undertake costly abatement measures.

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[10]There would, of course, be a risk that countries might seek to avoid their obligations to contribute to international action to control carbon dioxide emissions by, at the same time, reducing their other taxes on energy to offset the effects of the agreed carbon tax. Essentially, the test of whether countries are making their contribution to global climate change policy requires a comparison with what they would have done otherwise, and this is impossible to observe.

5.3 Practical design of a carbon tax

5.3.1 Efficient specification of the carbon tax base

Ideally, a tax to control atmospheric emissions of carbon dioxide would be levied directly on the individuals or firms who are resonsible for the emissions, and would be based directly on the amounts of carbon dioxide emitted. In practice, there are too many sources of emissions for direct measurement of emissions to be practicable. In practice, therefore, carbon taxes take the form of a tax on the carbon content of fuels, intended to proxy for the carbon emission which result from the combustion of these fuels. The relationship between carbon content and eventual carbon emissions is very close, partly because no viable end-of-pipe emissions cleaning technologies are available. Many of the problems that might, in principle, arise from the taxation of inputs as a proxy for emissions (Holterman, 1976, Sandmo, 1976) are not likely to be of great significance for the carbon tax case.

Nevertheless, there are some practical issues about how a carbon or carbon/energy tax should be structured and administered which might have implications for economic efficiency.

In the European countries (Sweden, Norway, Finland, the Netherlands and Denmark) which have actually introduced carbon taxes these have taken the form of extended systems of fuel excises. Rates of tax are defined separately for each fuel, in terms of fuel quantities, and relative tax levels on different fuels are set so as to equate the implicit rate of tax per unit of carbon across fuels. This requirement is not, however, always observed; in Denmark and Norway, for example, some fuels are not subject to the carbon tax. Also, the level of tax can vary across types of energy user; in Sweden and the Netherlands, for example, much lower rates of tax apply to industrial energy users than to energy use by private households. Most of the carbon taxes actually implemented in these countries have provisions which exempt firms or sectors which 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). They contrast two schemes - a "primary" carbon tax, levied on primary fuels (eg crude oil, coal, and gas) where they are mined, extracted or imported, and a "final" carbon tax, levied on final fuel products (such as coke, anthracite, four star petrol) sold to industrial users or households. The latter corresponds to the current approach of extending existing fuel excises. They argue that, although there are advantages and disadvantages associated with each, a primary carbon tax would have some advantages compared to the excise duty route.

A "primary" carbon tax would involve fewer taxable individuals than a "final" tax, and no need for fiscal supervision of the energy chain beyond the first point; administrative costs would be expected to be low, and there would be scope for tight supervision to prevent evasion.

The fact that a primary carbon tax would be applied at an earlier stage in the production chain would not necessarily imply that it would have different economic or environmental effects from an equivalent tax levied on final fuel products. The incidence, for example, of the carbon tax on fuel consumers could be largely invariant to the stage at which tax is formally incident; some part of the burden of a primary carbon tax would be passed on in the prices of fuel products according to their carbon content, so that the prices of fuels purchased by industry and consumers would be much the same as if an equivalent final carbon tax had been levied.

However, Pearson and Smith point out that this equivalence can only be achieved if the tax authorities have comprehensive information about the carbon "history" of final fuel products. To calculate the carbon tax to be applied to a final fuel product requires information not only about the actual carbon content of the fuel, but also about the carbon emissions associated with its processing. This means that the amount of tax to be applied to a particular final fuel product can no longer be determined simply by reference to its physical characteristics (which would provide a straightforward and uncontroversial basis for administration of the tax) but requires in addition that these measurements be supplemented by assumptions about the carbon emissions associated with its past history. 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 will be liable to lead to inefficient technology choices. In principle, therefore, a primary carbon tax might be expected to be more efficient, in both static and dynamic terms (though this greater efficiency of a primary tax may be partly undermined if carbon tax refunds then have to be made on processed energy which is exported or used as inputs to exempt activities). Whether there is any significant efficiency gain from using a primary carbon tax is essentially an empirical matter; it will be greater the more variation there is in intermediate emissions across different technologies of fuel processing. The quantitative significance of these issues for the choice of carbon tax type have not, to date, been assessed.

Developing this line of argument, Poterba and Rotemberg (1995) have considered the case where joint production of final fuel products take place, and where the output mix is a choice variable. In other words, they consider a process of fuel refining or processing where a single primary fuel is processed into more than one final fuel product, and where the mix of final fuels produced can, to some extent, be varied. In this case they show that it may be impossible to define any objective basis 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 (ie an excise tax). Further implications, however, are that arrangements for refund of past carbon tax on fuel exports, and on fuel supplies to any exempt sectors and non-fuel uses, cannot be done in a way which accurately reflects their total (potential and past) carbon emissions; likewise, levying the carbon tax on imported processed fuels can only approximate the past carbon emissions involved in their processing.

5.3.2 Interaction between CO_2 taxes and other taxes.

A further source of complexity in determining the optimal carbon tax is the existence of other energy taxes, and other energy-related pollution problems which might be tackled using taxation. How should these be taken into account in setting the carbon tax rate? There are a number of separate issues:

First, what in principle should be the relationship between a carbon tax and other taxes levied on energy, either for revenue-raising purposes or to correct other externalities? Newbery (1992) considers the relationship between the optimal rate of a carbon tax and the rates of any existing externality taxes on energy. He concludes that in some circumstances the relationship may not simply be additive. For example, introduction of a carbon tax may increase the optimal rate of a tax on motor fuel levied to reflect other traffic externalities. The reason for this is that in the long run the carbon tax will increase fuel efficiency of motor vehicles, and hence will increase the non-carbon externality costs per litre of fuel.

Second, how should interactions between different pollution problems (eg CO_2 and acid rain) be treated in the design of the tax structure? Pearce (1991) has argued that carbon taxes in excess of the marginal damage from CO_2 emissions may be appropriate to reflect the "secondary benefits" of carbon abatement in the form of reductions in other sulphur dioxide and other energy-related emissions, if other instruments do not provide appropriate and more specific incentives for control of these pollutants.

Third, in international discussion of coordinated carbon taxes, how much "credit" - if any - should be given to countries which already levy high rates of tax on energy? Also, should any restriction be placed on their ability to reduce other taxes on energy when the carbon tax is imposed by international agreement? It is possible to interpret existing taxes on energy as implicit carbon taxes (Hoeller and

Wallin, 1991, Hoeller and Coppel, 1992), and it may then be argued that any new carbon tax should be imposed at different rates, in order to equalise the overall tax burden on carbon in different countries (see the discussion in Cnossen and Vollebergh, 1992, for example). The issue has a number of subtleties - how to avoid discouraging countries from introducing carbon taxes in advance of any agreement, and how to avoid countries gaining creidt for simply renaming existing energy taxes. It also relates to the question, discussed at more length below, of whether the efficient pattern of carbon taxes across countries would be uniform, or whether optimal carbon tax rates would vary across countries.

5.4. Policy implications of high carbon tax rates

It is possible that first-best carbon tax rates could be high, leading to a substantial increase in energy prices, a heavy carbon tax burden on energy-intensive activities, and significant carbon tax revenues. The scale of taxation required may lead to a number of problems which have concerned policy-makers. Three are discussed below: the requirements for efficient adjustment to increased energy prices, and the impact of the carbon tax burden on industrial competitiveness, and income distribution.

5.4.1 Adjustment efficiency

If high carbon taxes are imposed, the efficiency of market adjustment to higher energy prices becomes a matter of considerable importance.

The aggregate economic cost of adjustment to higher energy prices will be higher, where energy consumers are prevented by market failures from making optimal adjustments in energy use. An efficient pattern of adjustment to higher energy prices might include both reductions in energy consumption, and also greater levels of investment by both household and industrial energy users in various measures to increase the efficiency with which energy is used. In the domestic sector, measures which households can take to improve domestic energy efficiency may include such things as loft insulation, double glazing, and wall insulation. Similar measures may be undertaken by industrial energy users, who may also seek to develop new, more energy-efficient, products and technologies. It has been suggested that markets for these investments may be subject to various forms of market failure, possibly including credit market failures, informational failures, and certain market failures related to housing tenure (Sutherland 1991, Brechling and Smith 1994, Levene et al, 1995). Where these market failures prevent efficient adjustment to higher energy prices, reductions in energy consumption in response to higher energy prices will tend to be smaller, and more "painful" in terms of their welfare cost.

5.4.2 Impact of a carbon tax on "competitiveness"

The impact of carbon taxes on the international competitiveness of industry has, in practice, been a major source of political opposition in many of the countries which have introduced carbon taxes. In Sweden, in particular, it has led to substantial modifications of the initial carbon tax system, which had the effect of reducing sharply the level of carbon tax on industry. The issues are most significant where countries introduce a carbon tax through unilateral action.

The impact on competition could, of course, be offset, on average, by exchange rate movements; a devaluation by a country imposing a carbon tax could offset the impact of the higher taxation on industrial competitiveness. 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 profits 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. However, whilst other countries do not impose the tax, these sectors may be liable to contract too much, in the countries which do impose the tax, relative to the final desired equilibrium where all countries impose similar carbon taxes. Part of this contraction may represent "carbon leakage" - international displacement of carbon-intensive production when a carbon tax is implemented without full international coordination - and this may impose adjustment costs and loss of profits, without any corresponding environmental gain.

One possible way of limiting this would be by exempting particular sectors in the tax structure. This was 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) and has been a feature in most of the countries which have so far introduced carbon taxes.

An alternative approach which might reduce the extent of international displacement to countries which do not impose the carbon tax would be to make compensatory border tax adjustments on traded goods to reflect the tax treatment of the carbon used in their manufacture. Thus it would be possible in principle (though perhaps less straightforward in practice) to levy tariffs on goods imported from countries which do not impose a carbon tax, at a level reflecting the carbon used in their manufacture, and to make corresponding refunds of tax to reflect the carbon embodied in exported goods. There has been some debate amongst policy-makers as to whether such border tax adjustments would be compatible with WTO rules.

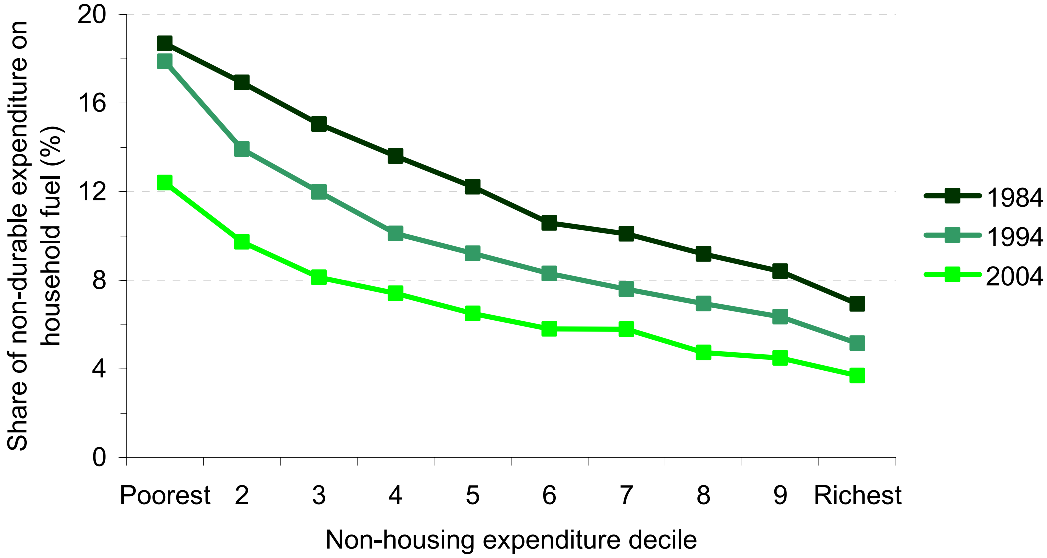
Hoel (1995) discusses the circumstances in which sectoral differentiation of carbon tax rates (either in the form of exemptions for some sectors, or differential tax rates) would be efficient. He observes that if countries are not able to levy tariffs on trade with non-signatories to an international agreement to restrain carbon dioxide emissions, it may be appropriate to levy differentiated taxes across sectors. Informally, these taxes would aim to offset the competitive advantage that firms in energy-intensive sectors would receive in non-signatory countries, although the efficient pattern of carbon tax differentiation across sectors would be complex. However, if countries are able to determine tariff rates without restriction, then tariffs should be employed for this purpose, and the optimal pattern of tax rates across sectors will be uniform.

5.4.3 Distributional impact, and possible offsetting policy measures

The distributional impact across household groups of a carbon tax has been a focus of policy concern in some countries, and is discussed, for example, by Poterba (1991), Smith (1992), OECD (1995, 1996), and Cornwell and Creedy (1996). The distributional impact will reflect the impact of the carbon tax on the prices of household energy (for heating, light, etc), motor fuels, and other industrial goods and services (through the higher cost of energy inputs to production). The distributional issues are most acute in the case of the additional tax on domestic energy, which in Northern Europe at least has the character of a necessity, forming a much larger part of the budgets of poorer households than of 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 lowest spending households devote a share of their non-durable budget to fuel that is around three times larger than the highest spending households. Over time, the non-durable budget share of fuel has fallen for all deciles but at a similar rate such that the relative differences have remained largely unchanged. By 2004, the lowest spending households devoted around 12% of their non-durable budget to fuel compared to just under 4% for the highest spenders (and 7% on average across all households).

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: Household expenditure deciles are calculated from expenditures equivalised by the OECD equivalence scale. Households that spend less than £5 per week on average are excluded as are households with recorded incomes below £5 per week.

Additional taxes on domestic energy will thus tend to have a regressive distributional incidence, in the sense that the extra energy tax payments will be a higher percentage of income (or of total spending) for poorer households than for the better-off. However, once the use of the additional revenues from energy taxation is considered, a revenue-neutral package of measures, including higher energy taxes combined with higher transfers to poorer households, could be designed which, overall, could leave poorer households better off (or at least no worse off) on average.

Smith (1992) shows that in the UK the EC's proposed carbon tax would have been likely to have a regressive distributional impact too; the regressive effect of higher taxation of domestic energy would have outweighed the progressive distributional incidence of higher taxes on motor fuels. However, in many other EC countries this would not have been the case, because the domestic energy component of the tax is either smaller relative to the motor fuels component, or less regressive, than in the UK.

In addition to the burden of extra taxation, a second issue has been the distribution of the burden of reductions in energy consumption in response to higher energy prices. In the case of the UK it appears likely that the reduction in energy consumption induced by the imposition of higher taxes on domestic energy is greater amongst poorer households; Pearson and Smith (1991) estimate that the energy spending of the bottom quintile would have fallen 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 would be of the order of 7 per cent.

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 (Smith, 1992). Thus, for example, income related market failures, such as those related to the credit market, or to housing tenure, may tend to amplify the distributional cost of reducing energy consumption through pricing instruments. Measures (such as building regulations or home energy audits) to rectify the underlying market failures would then have the twin merits that they would tend to reduce the aggregate economic cost of achieving a given reduction in consumption, and at the same time would also help to reduce the social and distributional cost of higher energy taxation.

6. Road transport externalities and the tax system

Using taxes to address road transport externalities is complicated by the presence of a number of different externalities, each of which varies greatly between different road users and journeys, in a way that cannot be precisely reflected in the various tax instruments available. Optimal tax policy for road transport is likely to require a combination of taxes, to achieve the closest possible approximation to the pattern of externalities - in other words, a well-targeted policy requires the "multi-part instrument" approach we described in Section 3. It is also likely to require non-tax instruments too, because even a sophisticated combination of tax instruments is unlikely to be able to reflect all of the relevant externalities.

Among the significant externalities road use Sansom et al (2001) identify the following: operating costs, the cost of accidents, air pollution, climate change, noise and congestion. They present estimates of the scale of each of these externalities for the average motorist, expressed as the marginal externality in pence per km (Table 6.1).

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

Externality	Low estimate	High Estimate
Operating costs	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

It will be seen that congestion costs are by far the largest element in overall marginal road costs, accounting for about three quarters of the overall external cost of 15 pence/km. Congestion externalities, however, vary hugely according to time and location – Sansom et al (2001) estimate marginal externalities of around 86p/km for central London peak time roads but just 3p/km for non-major rural roads. Other components of the external costs also vary with a variety of factors. For example, air pollution costs, noise costs, climate change costs and accident risks all vary according to the style of driving, with more-aggressive driving styles generating higher-than-average external costs per unit distance.

In some cases the variation in costs per km will be closely proxied by one of the available tax bases. Thus, for example, climate change externalities may be closely proxied by fuel consumption, and so will be ideally suited to an indirect tax on fuel. However fuel duty alone will be inadequate to reflect the range of externalities involved: it can capture some of the externalities quite well but others, such as congestion costs, rather inefficiently. To be sure that the range of decisions that individuals make are appropriately guided by incentives which accurately reflect the social costs of the choices they make - about whether to own a car and of what type, about when and where to drive, and so on - fuel taxes will need to be supplemented by other measures to reflect externalities which are not closely related to fuel consumption.

6.1 UK road transport taxes

UK policy has typically relied quite heavily on petrol taxation as the major 'environmental tax' and as a significant source of Exchequer revenues[11]. Duties on hydrocarbon oils are expected to generate

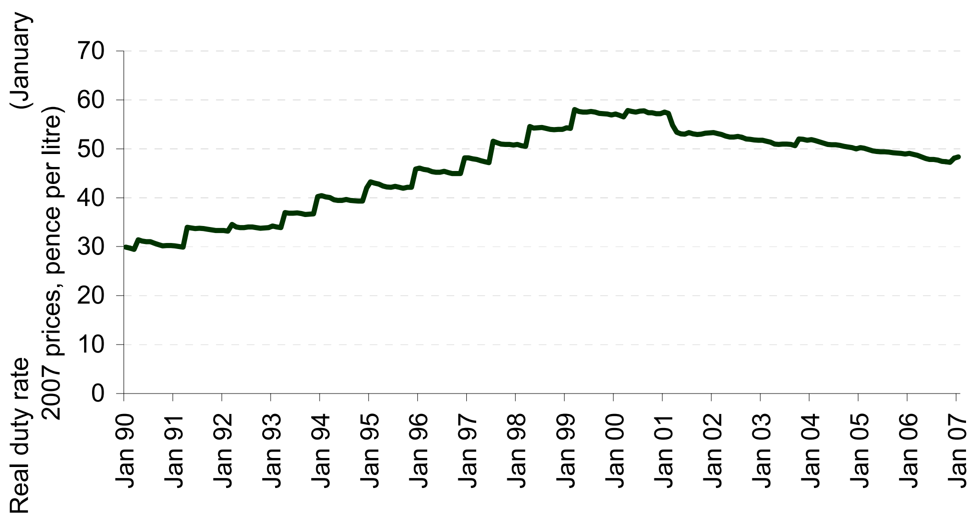
[11] See section 5 of Leicester (2006) for more details of UK fuel taxation

receipts of just under £24bn in 2006/7, just over 4½% of total revenue. Only income tax, National Insurance contributions, VAT and corporation tax raise more revenue in the UK tax system.

Most road fuel sold in the UK is either ultra low sulphur petrol (ULSP) or diesel (ULSD). Both are taxed at a rate of 48.35p per litre (around £1.83 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 – this was known as the "fuel price accelerator". 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 2006 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. Real rates of duty have not increased since 1999, and nominal rates have increased only three times, though Budget 2007 pre-announced inflation-linked increases that would be enacted in 2007, 2008 and 2009. This has been justified by the Government as a response to the high price of oil, though in principle there should be no economic reason to adjust fuel duty rates (or withhold increases) in response to changes in the pre-tax price.

Even after the inflation-adjustment to fuel duty rates in the December 2006 Pre-Budget Report, real duty rates are now around their lowest in almost a decade as figure 5.1 highlights. The duty escalator period of 1993 to 1999 is clearly visible from the 'saw-tooth' pattern of annual real-terms rises being gradually eroded by inflation within-year.

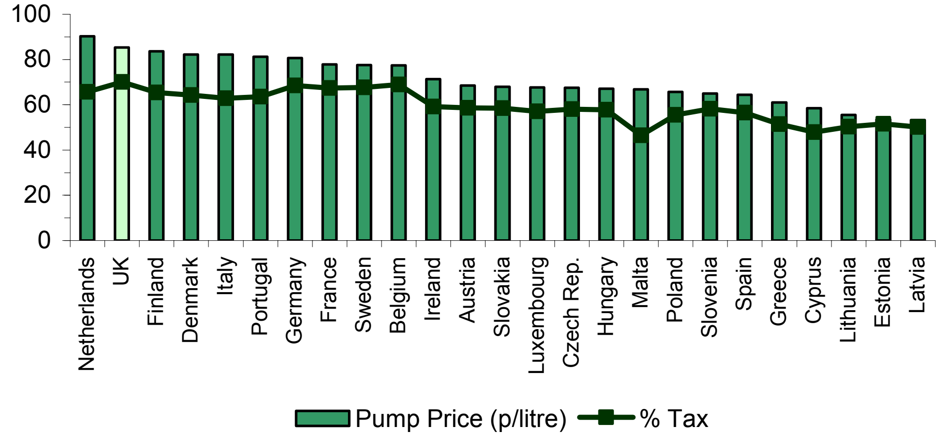
Figure 5.1: Real rates of duty for most commonly purchased unleaded fuel, 1990 – 2007



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

By January 2007, total taxes (duty plus VAT) accounted for around 71% of the pump price of a litre of unleaded fuel, much lower than the peak value of 86% in March 1999 at similar to the levels last seen in around 1994. However, DTI figures for November 2006 show that the in the UK, a greater proportion of the pump price is accounted for by tax than in any other EU country at 70% for unleaded and petrol, though many other countries (particularly older EU member states) have similar ratios as illustrated in figure 5.2. The total UK pump price, averaging 85.4p/litre, was the second-highest in the EU with only the Netherlands, at 90.2p/litre, being higher. This was due to a much higher pre-tax price in the Netherlands – indeed, excluding tax and duty, prices in the UK were the fifth-lowest in the EU-15.

Figure 5.2: Unleaded pump price and proportion accounted for by taxes, EU 25, November 2006



Source: DTI

6.2 What is the optimal level of fuel duty?

Assuming that fuel duty rates can only vary according to the type of fuel rather than to the type of vehicle, an obvious question is what the desirable rate of duty should be, to reflect, on average the principal external effects.

Parry and Small (2004) calculate the optimal level of fuel taxes in the UK and the US based on estimates of the marginal externalities from motoring, allowing for the fact that a large proportion of the response to fuel duty operates through improved fuel efficiency, and also allowing for interactions between fuel taxes and the wider tax system. Under their central modelling assumptions, they derived an optimal fuel tax for the UK of $1.34/gallon. The current UK tax rate of 48.35p/litre equates to roughly $3.57/gallon (using a market exchange rate of around £1 = $1.95 and a conversion of 1 litre = 0.2642 US gallons), considerably higher than this estimate. Of this $1.34, around five cents is accounted for by the fuel-related pollution caused by carbon emissions, and around 61 cents through congestion externalities. However, in calculating their fuel-related pollution, they choose a value for the marginal damage caused by a tonne of carbon emissions at $25, drawing on various estimates from the international literature. The UK Stern Review (2006) estimated the damage cost at closer to $300/tC (see section 6 below). Even at this level, however, the optimal UK tax, assuming everything else unchanged, would probably rise only by around 65 cents or so to close to $2. Even allowing for the current relative weakness of the dollar and choosing a currency conversion of around £1 = $1.50, the UK tax rate is still higher than this at around $2.70, though it may be that using the Stern estimates for climate change costs coupled with the high-range estimates of other externalities (particularly congestion) from Parry and Small would bring existing UK tax rates close to the 'optimal' under their modelling. What these results do show, though, is that given the variation in estimates of externalities, even with careful modelling determining "the" correct fuel tax rate is extremely difficult.

A feature of UK fuel taxation policy has been differentiation of tax rates to encourage fuel substitution: unleaded petrol is taxed more lightly than leaded petrol, for example, and 'alternative' fuels such as bio-ethanol are also favoured in their tax treatment. In terms of the major fuels sold for private motoring, Ultra Low Sulphur Petrol (ULSP) and Ultra Low Sulphur Diesel (ULSD), both are taxed at the same rate, 48.35p/litre. Diesel has often been more heavily taxed, however, because of its emissions of particulates which are associated with respiratory illnesses, particularly in built-up urban areas. In part this less favourable treatment was removed because diesel engines are less harmful in terms of their carbon emissions. Having some assessment of the relative damage caused by climate change and through particulates is therefore important in terms of estimating whether differential tax rates for petrol and diesel would be justified. Using Stern estimates of the costs of carbon would point to a more favourable tax treatment for diesel whereas using the lower estimates from other literature may point to the opposite.

6.3 Other road transport externalities

As already noted, motor fuel taxes are a very blunt instrument to reflect the congestion costs generated by marginal road use. These costs vary widely between location and by time of day, and are very much higher at peak times in central urban areas than for other journeys. Explicit charging for congestion is likely to achieve much more efficiently-targeted incentives to reduce congestion than a fuel tax set at a level that reflects the congestion cost generated by the average journey.

Such an explicit Congestion Charge was introduced in central London in 2003, one of the largest and most ambitious congestion charging schemes in the world[12]. The charging zone covers 22 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. However there are exemptions 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 discount of 90%.

The latest impacts monitoring report (Transport for London, 2006) suggests that congestion has fallen by around 25 – 30% relative to pre-charging baselines and that revenue from the charge for 2005/6 amounted to £122 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 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) estimated of 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, or 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 clearly there would be practical difficulties in monitoring the distance driven once inside the zone.

Newbery (1990) estimated that the congestion cost per vehicle-kilometre averaged some 3.4 pence across the UK and as detailed in table 5.1, the Sansom estimates average at around 10 pence per kilometre for 1998. Figures from the British Transport Statistics (DfT, 2006) show that in 2005 just under 500 billion kilometres were travelled by motor vehicles on the road; an average tax rate of, say, 5 pence per kilometre applied nationally to all road vehicles would therefore have generated revenues of some £25bn (assuming no behavioural response). This is equivalent to just under 5% of total receipts, or about the same amount of revenue as generated by fuel duty at the moment.

6.4 Designing an "optimal" vehicle emissions tax

So far, policymakers in most nations have addressed vehicle emission problems with a variety of mandates and restrictions. "Certification standards" require that all vehicles have emission rates lower than specified levels, while "fleet composition" standards mandate that each manufacturer's total sales of new cars each year must include a certain percentage that are low-emission vehicles. "Inspection and Maintenance" programs require that each motorist pass a smog check every two years, and

[12] Blow *et al* (2003) discuss the workings of the scheme and background to it in depth.

"reformulated petrol" requirements can ensure that all petrol meets eight specifications for cleaner-burning fuel.

These command and control (CAC) regulations can guarantee vehicle emission reductions, but they do not provide much flexibility. In contrast, the standard case for market-based incentives requires a tax or price on each unit of emissions. Each form of abatement is then pursued until the marginal cost of reducing pollution matches the tax per unit of pollution, and the resulting combination of abatement technologies minimizes social costs (Pigou, 1920). If a driver has to pay the price of a permit or a tax per unit of emissions, then that individual has the incentive to find all of the cheapest and most convenient ways to reduce emissions. Rather than assuming that "one size fits all," a system of incentives might allow each driver to choose the extent to which to save tax. For vehicles, a tax on emissions could induce drivers to: (1) buy a newer, cleaner car, (2) buy a smaller, more fuel efficient car, (3) fix their broken pollution control equipment, (4) buy cleaner petrol, (5) drive less, (6) drive less aggressively, and (7) avoid cold start-ups.[13] Moreover, economic efficiency requires different combinations of these methods for different consumers: some lose little by switching to a smaller car, some could easily walk, and some just pay the tax. With heterogeneity, the least-cost solution involves greater pollution reduction for some than for others.

Yet the implementation of a tax on emissions would require the measurement of each car's emissions. Such a tax would be difficult to implement. It is not a tax on a market transaction, like the purchase of labor services or the sale of a product, with an invoice confirmed by two parties to the transaction. To the contrary, emissions are hard to measure and easy to hide. It is therefore important to look for alternative incentive instruments that apply to market transactions *rather* than to emissions.

Technological advances might soon make it feasible to levy a tax directly on emissions (Harrington and McConnell, 2003). Three such methods can be discussed briefly, but each has problems. First, the most direct method would install on-board devices to measure the tailpipe emissions of each vehicle, but this method would be expensive — particularly to retrofit millions of existing vehicles. Also, this method misses evaporative emissions, and it is subject to tampering. Moreover, it may not satisfy legal restrictions against the search of a private vehicle. Second, authorities could simply measure each vehicle's rate of emissions per mile *(EPM)* once each year and multiply by the number of miles driven since the last reading. This method is subject to evasion, however, if drivers can roll back their odometers. And even with accurate mileage, the emission rate cannot be measured accurately because it depends on *how* the car is driven. A third approach discussed by Harrington et al. (1994) would use remote sensing at selected locations: "As vehicles pass the sensor, a tailpipe emission reading is taken and the license plate is identified electronically" (p. 24). If enough monitoring stations are set up frequently and moved randomly, authorities could approximate the total emissions of each vehicle and send the owner a monthly tax statement.[14] With over-sampling during high-ozone periods, the tax bill could reflect the social cost of those emissions. Still, however, each driver's emissions are not exact. This method is expensive, and it misses evaporative emissions. And some drivers may disproportionately miss or intentionally avoid the sensor locations.

For these reasons, recent research considers alternative scenarios about 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 behaviors that would minimize 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-

[13] Heeb et al (2003) find that cold start emissions rates (in g/km traveled) exceed stabilized emissions rates by a factor of two to five, depending on the pollutant. Sierra Research (1994) finds that a car driven aggressively has CO emissions that are almost 20 times higher than when driven normally.

[14] See Sierra Research (1994). Remote sensing in Texas (http://www.tnrcc.state.tx.us/air/ms/vim.html#im3) and Albuquerque NM (http://www.cabq.gov/aircare/rst.html) is used in 2005 to identify polluting vehicles.

reducing characteristics of incentive instruments, policymakers can consider alternative taxes and subsidies on various market transactions that reflect choices affecting emissions.

The main advantage of such taxes and subsidies is that each applies to a market transaction: a purchase with an invoice and a seller who can help collect the tax. This reduces the cost of measuring the taxed activity, and it helps with enforcement of the tax. The tax on engine size can be collected at the time of sale by the manufacturer; subsidies to pollution control equipment and to newer cars can be paid upon annual inspection; and a petrol tax can be collected at lower rates on cleaner fuels.

To analyze these questions, 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. Thus, they capture most of the important determinants of emissions. They also capture heterogeneity. In this model, individuals differ by income and tastes for engine size and miles. Moreover, by using a general equilibrium model, they capture the simultaneity of those choices by consumers. All markets clear simultaneously.

First, using this model, they confirm that a single rate of tax on emissions of all different consumers will minimize the total cost of pollution abatement, even as it induces each consumer to change behavior 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).

Second, they show that a vehicle-specific petrol tax can attain the same efficient outcome. To do so, it must affect all the same behaviors in the same way as the emissions tax. The petrol pump would need to be able to "read" a computer chip in the car, so that the petrol tax rate can depend on the characteristics of the vehicle at the pump. If purchasers realize *how* their payments depend on these choices, then the petrol tax itself can present them with incentives to buy smaller cars, newer cars, more pollution control equipment, and less petrol. On the other hand, for this tax to be assessed, cars would need to be equipped with tamper-resistant computer chips.

Third, a tax on the vehicle that depends upon the characteristics of the vehicle and on the miles driven each year also achieves the same efficient outcome. If vehicle size and age were the determinants of emissions per mile *(EPM)*, then a tax rate per mile for that vehicle could be calculated on the basis of its size and age, and then multiplied by the year's miles to calculate the tax due. On the other hand, this policy requires yearly odometer readings, and it is thus subject to tampering.

More-realistic 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. Therefore Fullerton and West derive formulas for the second-best optimal unifirm tax rate on petrol, and on observable car characteristics. Policymakers could just insert into those formulas the average engine size, average vehicle age, and average mileage. This single set of tax rates based on those average characteristics could then be applied to everybody's engine size, vehicle age, and use of petrol. This procedure does not take advantage of available information *other than* those simple averages.

In general, available data can be used to calculate not only average size, age, and mileage, but also the correlations among these variables. If individuals with bigger cars also tend to choose more than average mileage, or conversely, then that information can be used to adjust the tax rates in a way that improves their effectiveness, even while each of those tax rates is still limited to be uniform across all consumers. Therefore, finally, they consider a set of tax rates that use all available information, but which are still limited to uniform rates across all consumers. This policy does not perform as well as the first-best emissions tax, but it is the "second-best" as 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. They start with data on household incomes, petrol purchases, other expenditures, and automobile ownership (including make, model, and year). They then link that auto type to a different data set to find that car's engine size, estimated miles per gallon *(MPG)*, and estimated emissions per mile *(EPM)*. They can also multiply *MPG* times gallons of petrol to get an estimate of miles driven. They specify the price paid to acquire one more gallon of petrol, the price paid to get a car that is one year newer, and the price paid to get a car that has one more unit of engine size. Each such price could be affected by a tax or subsidy. Each household gets utility from miles driven, engine size, vehicle "newness," and other commodities. Maximizing this utility subject to a budget yields demand functions that recognize how the price of petrol affects *all* choices: demand for petrol, demand for larger cars with lower fuel efficiency, *and* demand for newer cars with higher fuel efficiency. Similarly, any change in the effective price of buying a newer car or a larger car affects demands for those characteristics and the demand for petrol. Given an outcome for each household's chosen engine size, and vehicle age, they can calculate that household's *MPG* and *EPM*. They multiply *EPM* by miles to get the household's emissions, and they add over all households to get total emissions. They can also calculate each household's utility, and the overall gain or loss in the welfare of all households.

Using this model, they evaluate different combinations of tax rates. As a basis for comparison, they calculate the effect of an ideal Pigovian emissions tax. This tax raises the effective price of engine size and lowers the effective price of getting a newer vehicle. All consumers change behavior in various ways, and they calculate the reduction in emissions and the increase in total welfare. They then suppose that the ideal emissions tax is not available and instead consider tax rates on engine size, vehicle newness, and petrol. Assuming all three of these instruments *are* available, they use the computer to search over combinations of tax rates to find the one set of rates that maximize the gain in welfare. This set of tax rates is the "second-best" policy, given the constraint that the first-best emissions tax is not available.

The main result from this model is that the second-best combination of tax rates achieves a welfare gain that is 71 percent of the maximum gain obtained by the ideal-but-unavailable tax on emissions. A petrol tax reduces demand for petrol by inducing people to drive fewer miles *and* to buy smaller more fuel efficient cars. An additional subsidy to newness helps induce them to buy newer cars with lower emission rates. The petrol tax is large relative to the newness subsidy. The overall message here is that the petrol tax is the single most effective tool to reduce emissions.

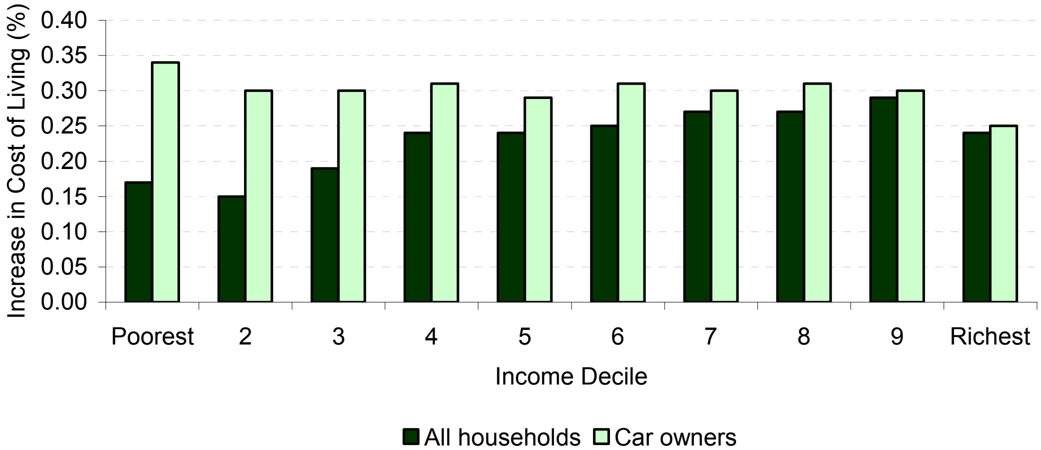
They then consider situations in which policymakers are limited to just one or two of those three tax rates. With just a petrol tax, the welfare gain is 62 percent of the gain from the ideal emissions tax. Without the petrol tax, the other two instruments can only achieve about 20 percent of the gain of the ideal emissions tax. Thus a petrol tax is the key ingredient of any market-based incentive policy – or at least of one that cannot employ the ideal emissions tax.

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 would not hurt the poorest families. Thus the petrol tax is not regressive at the very poorest levels, but it is regressive across most of the rest of the income spectrum. This is demonstrated for the UK below. Using data from the 2004/5 Expenditure and Food Survey (EFS), households are broken into ten decile groups based on their income. Amongst the poorest 10% of households, only 44% are car owners. Three-quarters of households in the middle of the income distribution are car owners, and 94% of the richest decile own a car. If we look at multiple car ownership, the rates are 11%, 26% and 57% respectively.

Figure 5.3 shows the impact of a 5% rise in petrol prices across the income 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 poorer households. Amongst car-owners only, poorer households are hit slightly harder and richer households slightly less (though even here, across the majority of the range of incomes the effect is broadly similar).

Figure 5.3: Distributional impact of 5% petrol price increase, all households and car-owning households only (2004/5)



Source: Calculated from Expenditure and Food Survey 2004/5. Note: Assumes no behavioural response to price rise.

Other road transport instruments may also have distributional effects. For example, a subsidy to vehicle newness would be regressive. Though it increases the number of newer, cleaner cars, this subsidy to “newness” mostly helps those with high incomes. Even if a petrol tax and newness subsidy are regressive, however, they could be combined with a change in transfer programs, other aid to low-income families, or a change to the overall progressivity of the income tax. Exempting low-income families from a petrol tax or other emissions policy reduces its effectiveness. If instead all households were ensured adequate incomes through other policies, then the most effective emissions policy would tilt all households away from driving and polluting by presenting all of them with incentives such as the petrol tax and newness subsidy.

7. Aviation Emissions

Aviation represents one of the fastest-growing sources of greenhouse gas emissions in the UK and other developed economies. There is some confusion as to exactly how much aviation contributes to national CO_2 emissions levels. Figures released by DEFRA[15] under the system of emissions accounting agreed by the IPCC suggest that in 2004, aviation accounted for around 1.8m tonnes of carbon equivalent (mtce) emissions, around 1.2% of the national total and up from 1.1mtce and 0.7% in 1990. However this is domestic aviation only; emissions from international aviation not included in the totals reported to the IPCC or those used for Kyoto and domestic emissions targets. Figures are available, however, estimated from refueling at UK bunkers and suggest total aviation emissions for 2004 of 10.8mtce, 6.6% of total emissions. This compares to 5.3mtce (3.2%) in 1990. This latter estimate suggests emissions from aviation, both in absolute terms and as a proportion of all CO_2 emissions, have more than doubled in the UK since 1990. Taking a broader view of all greenhouse gases, figures from the UK Environmental Accounts[16] suggest that in 2004, air transport accounted for 5.3% of emissions, compared to 2.5% in 1990, again more than doubling. All of these figures are clearly based on slightly different interpretations of what should and should not be included in national emissions totals, but all give a very similar picture as to the growing contribution of aviation emissions.

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."[17]

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 CO2 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[18]. In 1999, Sweden had to remove a domestic aviation fuel tax as 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 will eventually 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 largely revenue-raising measure at a time when the Government finances were in relatively poor shape. However, given that air tickets are exempt from VAT and that fuel used by large airlines is exempt from duty, there is a clear argument that aviation is relatively under-taxed compared to its increasing contribution to emissions.

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 non-European destinations are taxed at £40 and £80 respectively. These rates apply from February 2007, having been doubled in the December 2006 Pre-Budget Report. It is estimated that APD raised around £1bn in 2006–07, and revenues are forecast to more than double to £2.1bn in 2007–08 as a result of the doubling of the tax rates.

APD can be thought of as 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

[15] See http://www.defra.gov.uk/environment/statistics/globatmos/download/xls/gatb05.xls
[16] See table 2.3 of http://www.statistics.gov.uk/downloads/theme_environment/EANov2006.pdf
[17] See ICAO (2006).
[18] See ECON Analyse (2005) for more details.

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) though there are obvious variations in the elasticity according to the purpose of the flight (business flights are much less price elastic than pleasure flights)[19]. However, there is no variation in rates according to the emissions of the aircraft or the total distance traveled 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 is not applicable to freight flights. A sensible reform would be to change the basis of the tax to the flight rather than the passenger. This would allow simpler variation according to the age or emissions rating of the aircraft and would encourage fuller flights as presumably the airlines would have to absorb the tax for unfilled seats. A proposal of this nature has been suggested by both the Liberal Democrats and the Conservatives.

An alternative 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. The EU, supported by the British Government, has signalled an intent to include aviation in the second phase of the EU Emissions Trading Scheme (ETS), starting with intra-EU flights from 2011 and all flights from 2012. This may be subject to legal challenges from non-EU countries under the ICAO rules and there is concern from some green groups that the current plan makes it likely that any permits would be grandfathered not auctioned.

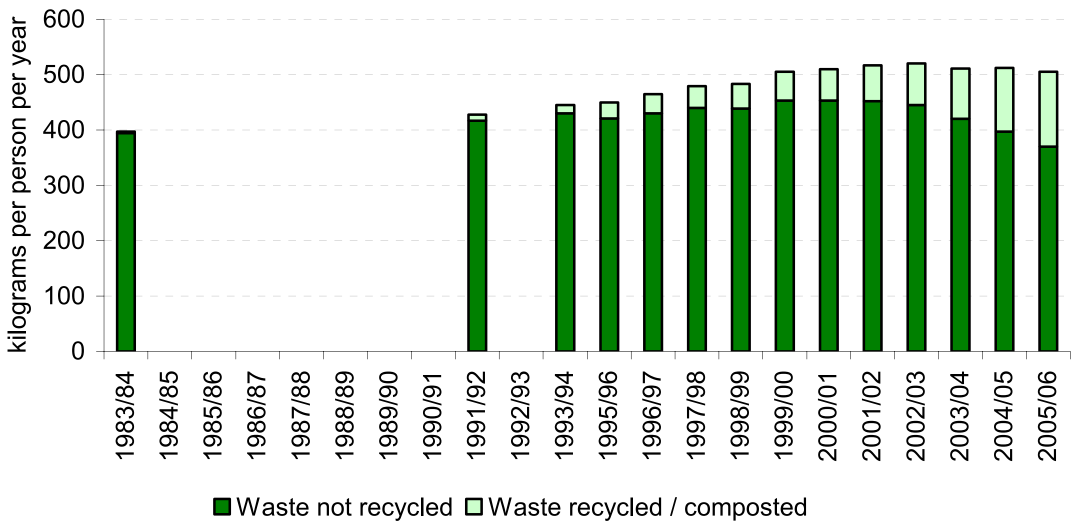
A recent paper by Keen and Strand (2007) examines an optimal system of international aviation taxation and summarises the current practice of domestic aviation taxation in different countries. Many EU nations currently charge VAT on domestic tickets, though only Germany and the Netherlands do so at the 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 does impose a 7.5% ticket tax as a ‘security charge’. More popular appear to be ‘trip-based’ charges, whether as an ‘airport tax’ (which typically accrues to the airport authority) or as a departure tax which accrues to the government. Keen and Strand estimate, for example, that in the UK, the average international passenger 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) rather than relying on trip taxes which cannot be targeted on the environmental costs of aviation particularly well. Given the constraints imposed by international agreements, however, it may be that such a system would be extremely difficult to implement; as such fuel excise and ticket taxes on domestic aviation coupled with departure taxes for international aviation may be the most desirable policy that is feasible. Pearce and Pearce (2000) estimated the marginal external costs from pollution and noise at Heathrow airport for different models of aircraft, and argued that, on a per-passenger basis, a short-haul flight on a Boeing 747-400 should be set at £3 and a long haul tax at £15.

[19] See Gillen, Morrison and Stewart (2004).

8. Taxes and Waste management

Although waste management has perhaps not been the high-profile focus of environmental issues in the way that transport or energy use has, there has been a significant change in household behaviour in recent years in the UK. Figure 8.1 shows average household waste generated per person in England between 1983/4 and 2005/6. Waste volumes increased in the 1980s and early 1990s, rising from around 400kg per person to just over 500kg, or more than 25%. However, this figure has recently stabilised. More striking is the growth in the proportion of total waste which is recycled. Less than 1% was recycled in 1983/4 and only 3% in 1993/4. This has increased rapidly over the last decade: recycled waste in 2005/6 accounted for 26% of the total, more than a quarter. In 2005/6, the total amount of *non* recycled waste generated per person was at its lowest level since this series began.

Figure 8.1: Recycled and non-recycled household waste generated per person in England, 1983/4 – 2005/6

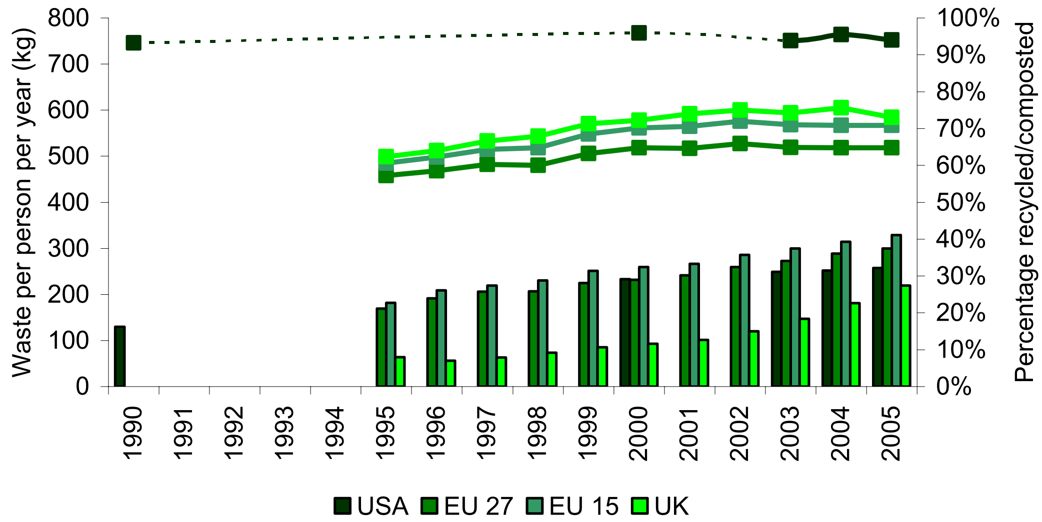


Source: DEFRA. Note: Data not available in every year.

Despite this trend, the Environment Agency estimates that waste management accounts for around a fifth of the UK's emissions of methane, a greenhouse gas, or around 2% of total greenhouse gas emissions. The main problem comes from biodegradable waste which emits methane as it decomposes.

International figures for total muncipal waste generation (household wastes plus waste generated from industry and commerce that is also disposed of by municipal authorities), illustrated in figure 8.2, show similar trends. Total waste generation per capita rises slowly in the 1990s and peaks and levels off in the early part of the 21st century across the EU and in the US, a similar trend to the UK. The UK generates more waste on average than the rest of the EU (and the 'old' member states generate more per capita than the new member states that joined in or after 2004) but substantially less than the US. However, by 2005 the UK was still recycling (or composting) only just over one quarter of total municipal waste, less than the EU 27 average (37%) and EU 15 average (41%) and, perhaps more surprisingly, less than the US average (32%). However, recycling in the UK has grown more quickly than elsewhere, rising from just 8% of the total in 1995 and 12% in 2000. Between 2000 and 2005, recycling rates in the US grew only marginally from 29% to 32% of waste whereas in the EU 15, which had the same recycling rate in 2000, the growth has been stronger, to 41%.

Figure 8.2: Municipal waste generated per person per year and percentage recycled or composted, 1990 – 2005, EU, USA and UK



Sources: US Environmental Protection Agency, Eurostat. US figures have been converted from daily waste generation in lbs per person to annual kilograms.

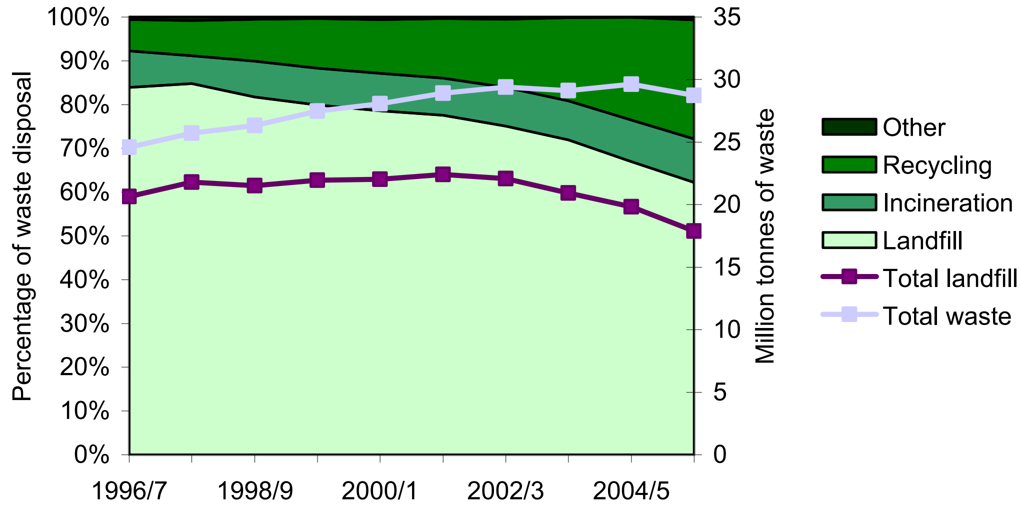
Principally, waste can be disposed of in three ways: landfill, incineration or recycling. Trends in European dealings with municipal waste have been heavily influenced by the 1999 European Landfill Directive which commits EU countries to reduce biodegradable municipal waste levels sent to landfill by 2020. The UK's commitment, for example, is to reduce it to 75% of the 1995 level by 2010, 50% by 2013 and 35% by 2020. The key policy measure in the UK used to attemp to hit this target has been the Landfill Tax, a tax on the volume of weight send to landfill. Introduced in 1996 at a rate of £7 per tonne (or £2 for inert waste such as rubble which have relatively small environmental effects), the rate has been increasing under the "Landfill Tax accelerator" since 1999 – by April 2007, the rate had reached £24/tonne (thought the reduced rate for inert waste remains at £2, rising to £2.50 from April 2008) and there is a commitment to annual increases of at least £8/tonne between 2008 and 2011.

Figure 8.3 shows modes of municipal waste disposal for England between 1996/7 and 2005/6, the period since the Landfill Tax was introduced. Total municipal waste increased from 24.6m tonnes to 28.7m tonnes. In volume terms, landfilled waste peaked in 2001/2 at 22.4m tonnes; as a percentage of total disposal, however, landfill peaked in 1997/8 at just under 85%. By 2005/6, landfill accounted for 62% of disposal. Incineration's share of disposal has increased only marginally, from 8% to 10%, over this period. The biggest increase has been in recycling (and composting), rising from 7% of total municipal waste disposal to 27% over the period, or from 1.8m tonnes to 7.8m tonnes in volume. It seems clear that there is at least some relationship between the increase in the rate of Landfill Tax and the decline in landfill both as a share of total disposal and in volume terms. Landfill volumes declined by just over 20% between 2000/1 and 2005/6 whilst total waste volumes fell by less than 1%.

Internationally, there is considerable variation in the chosen means of disposal. Typically, where land is scarce incineration is a preferred method, popular in Japan and several European nations such as Denmark and the Netherlands[20], though given the UK's high population density it is perhaps surprising that incineration accounts for such a small share of disposal. Figure 7.4 shows data from OECD countries covering the period around the turn of the century, showing total municipal waste generation by means of disposal.

[20] See Jenkins (1993) and Kinnaman and Fullerton (2000b).

Figure 8.3: Handling of municipal waste and total volumes generated/landfilled, England, 1995/6 – 2004/5

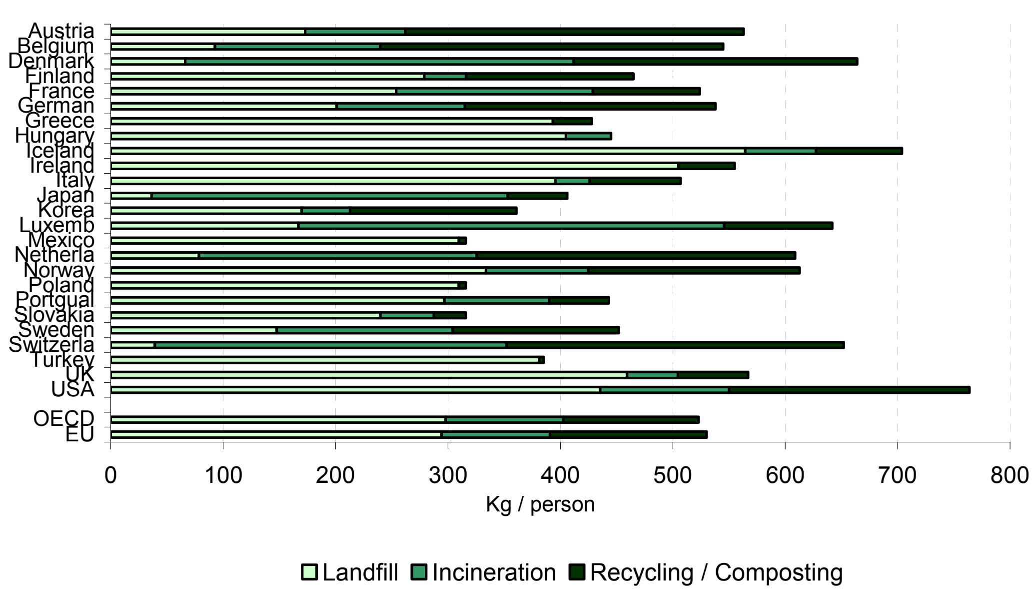


Source: DEFRA (various years)

Across the OECD and EU as a whole, landfill accounts for a little over half of total municipal waste disposal but with much variation between countries. Recycling rates also vary significantly as illustrated in figures 8.2 and 8.4. Although seven countries in the figure below generate more waste per head than the UK, only four of those (the US, Luxembourg, Ireland and Iceland) landfill or incinerate more waste per head and in only Ireland and Iceland is more waste per head sent to landfill.

Data from Eurostat suggests that across the EU 15 countries as a whole, landfill as a share of total waste disposal fell from 60% in 1995 to 39% in 2005 and in volume per capita terms by around one-quarter (more than 20% between 2000 and 2005 alone). This, coupled with the UK evidence, suggests that the direct targeting of Landfill under the Directive has had considerable impact. By comparison, per capita landfill volumes in the US fell by only 6% between 2000 and 2005.

Figure 8.4: Disposal methods for municipal waste in the OECD, c. 2000



Source: OECD

Note: Excludes countries where figures counted only domestically generated municipal waste (Spain, New Zealand). Most figures are for the period 1999 to 2000, though some countries have data from 1997 (Greece, Italy) or 1998 (Germany, Sweden, Turkey) as the most recently available. EU refers to the EU-15 member states prior to 2004's expansion.

Facing less competition from land-intensive landfills, incinerators in densely populated areas can capture the economies of scale necessary to keep down the average cost of incineration. But even though many countries rely heavily on incineration, Brisson (1997) finds that the private and external costs of incineration exceed those of landfill disposal in most European countries.

These days, large regional sanitary landfills are designed to reduce the negative externalities associated with garbage disposal. Governments impose technology-based standards on the construction, operation, and closure of solid waste landfills. Each landfill now is required to install thick plastic linings along the base, to collect and treat leachate, to monitor groundwater, and to cover garbage within hours of disposal.

8.1 Conceptual Issues in Charging for Household Waste

The optimal amount of garbage depends on estimates of the social marginal cost (SMC) and the social marginal benefits (SMB) of disposal. SMC include all costs associated with garbage disposal. Some of these costs are easy to observe, such as costs of labor and space at the landfill, while other costs are more difficult to quantify, such as external costs of leachate and methane emissions. The SMC curve depends on the waste disposal method. Landfills impose aesthetic costs on individuals through noise pollution from collection trucks and less scenic views. They may also have negative health effects from toxins in the leachate that seeps into the groundwater.[21] As the organic material in a landfill degrades, methane gas is produced. Landfills are the source of 35% of methane emissions in the U.S. and 28% of methane emissions in the E.U., or 4% of *all* greenhouse gas emissions.[22] If the methane is flared, then the combustion converts the methane into carbon dioxide, a less potent greenhouse gas. Flaring thus reduces the greenhouse effect, but it also generates other local air pollutants.

Thus, the choice of disposal method will determine the height and shape of the SMC curve. Estimates of SMC are reviewed below; these are important, because SMC is the "optimal" price to charge per unit of garbage. For present purposes, suppose this SMC curve is flat, in a simple partial equilibrium model of garbage.

The second major piece of information is the social marginal benefit of disposal, that is, the amount that consumers are willing to pay for one more unit of disposal. Empirical estimates of SMB are also reviewed below, but the general result is that this demand is fairly steep. Thus, the SMB curve is steep. The optimal quantity can be called Q*, found where SMC=SMB.

Yet most cities and towns still finance garbage collection through property taxes or monthly fees, with no price at the margin. This is at present the case in the UK, where refuse collection is financed

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

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

through the local Council Tax, though there has been considerable recent speculation about charging per bag or for volume of waste. This price of zero leads consumers down their demand curve to the horizontal axis. The welfare cost of the excess garbage is defined by Jenkins (1993) and Repetto et al. (1992) as the triangle representing the extent to which SMC exceeds SMB for each of those extra units. They use their estimate of demand to reflect social marginal benefits, and they calculate the welfare cost arising from the current under-pricing of garbage to be as much as $650 million per year in the U.S., roughly $3 per person per year. Fullerton and Kinnaman (1996) use household data and also estimate the potential benefits of marginal cost pricing to be in the neighborhood of $3 per person per year.

This partial equilibrium model considers only the externality from garbage, however, so charging a price-per-bag can reduce garbage and avoid this welfare cost. The alternative must be some clean activity like recycling. In a more general model, households may have multiple alternatives, each with its own negative externality. If households are able to avoid paying the garbage fee through illegal burning or dumping, then the calculation just described is not an accurate depiction of welfare gains. Waste policies need to account for methods of avoidance or evasion. If households are able to dump their garbage, the social cost of dumping may far exceed the social cost of additional waste disposal in a landfill. In this case, it would be better to offer free garbage collection than to implement a pricing policy that leads to widespread dumping. Policies must also reflect monitoring capabilities. Thus the goal of monitoring and enforcement might be met more easily by some kind of deposit-refund system (DRS), or if that DRS has high administrative cost, then perhaps by a simple mandate.

A unit pricing system (UPS) requires households to purchase a sticker or special bag for every unit of garbage they generate. Instead of viewing garbage collection as free, households face a positive price for every bag. Theoretically, this policy can induce households to recycle more of their waste. Even if alternative policies have the same aggregate net welfare gain, they may differ in terms of who bears the costs and who receives the benefits. All households must pay for their waste, but only some may receive social benefits of reduced waste in landfills. Also, the government receives additional revenue that can be used to lessen other distorting taxes.

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

Available data rarely allow for direct comparisons between illegal dumping before and after the implementation of unit pricing. Many economists have asked town officials whether they believe illegal dumping has increased, and many have stated that it has, but many more have stated otherwise. Reschovsky and Stone (1994) and Fullerton and Kinnaman (1996) asked individual households whether they observed any change. In the former study, 51% of respondents reported an increase in dumping. The most popular method was household use of commercial dumpsters. For the 20% who admitted to burning trash, the authors were unable to confirm whether these burners did so in response to the program. Roughly 40% of the respondents to the Fullerton and Kinnaman (1996) survey said that they thought illegal dumping had increased in response to the UPS. Many of these lived in the more densely populated urban areas of the city. Those authors also use survey responses with direct household garbage observations to estimate that 28% of the reduction of garbage observed at the curb was redirected to illicit forms of disposal. Nonetheless, Miranda and Bynum (1999) estimate that more than 4000 communities use some form of unit pricing in the United States.[23]

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

8.2 Social Marginal Benefits of Disposal

In the initial econometric study, Jenkins (1993) gathers monthly data over several years from 14 US towns (10 with unit-pricing). She finds inelastic demand for garbage collection; a 1 percent increase in the user fee leads to a 0.12 percent decrease in the quantity of garbage. Two studies rely on self-reported garbage quantities from individual households (rather than as reported by municipal governments). Hong et al. (1993) utilize data based on 4,306 surveys. Households indicate whether they recycle and how much they pay for garbage collection. Results indicate that a UPS increases the probability that a household recycles, but does not appreciably affect the quantity of garbage produced at the curb. These households may offset increased recycling by producing more total waste. Reschovsky and Stone (1994) mailed questionnaires to 3040 households and received 1422 replies. Each household reported its recycling behavior, income, and demographic information. The price of garbage alone is estimated to have no significant impact on the probability that a household recycles. When it is combined with a curbside recycling program, however, recycling rates increase by 27 to 58%, depending on type of material.

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

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

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

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

[23] 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.

effectively put a positive price on every unit of garbage, since the can may be partially empty most weeks. Indeed, Van Houtven and Morris (1999) find that the bag program reduces garbage by 36 percent, while the subscription can program reduces it by only 14 percent.

Two studies expand on the work of Jenkins (1993) by increasing the number of towns in the sample. Podolsky and Spiegel (1998) employ a 1992 cross-section of 159 towns clustered in New Jersey, twelve with unit-based pricing programs. They estimate the largest price elasticity of demand in the literature (-0.39). The authors attribute this estimate to the fact that all towns in their sample had mature recycling programs in place, and no towns in their sample had implemented subscription programs. Kinnaman and Fullerton (2000a) use a 1991 national cross-section of 959 towns, 114 that implemented user fees. They find that accounting for endogeneity of the policy variables raises the demand elasticity to -0.28, but that is still not very high. They also estimate that subscription programs have less of an impact than bag/tag programs on garbage and recycling quantities.[24]

8.3 Social Marginal Cost of Disposal, the Optimal Fee to Charge

Beginning in the early 1990's, several studies actually try to construct estimates of the full social cost (internal plus external cost) per bag of garbage. Repetto et al (1992) found that in areas with high direct disposal costs, the external costs are almost equal to the internal costs (adding up to $1.43-1.83 per bag). They base this figure on a study conducted in two places with high disposal costs. In particular, the Tellus Institiute (1991) studies the full cost of waste disposal for the state of California, employing the "control cost" method of pollutant evaluation for the external costs. Under that method, the marginal external damage (MED) attributed to a specific pollutant is the cost of preventing that pollutant from being emitted into the environment.[25]

For the U.K., CSERGE (1993) found external costs of landfill ranging from £1.3 to £4.0 per ton, depending on whether the landfill has energy recovery from methane gas. These estimates exclude amenity effects. Total external costs of landfill were estimated at £7 per tonne for active wastes and £2 per tonne (the disamenity cost) for inert wastes – these estimates helped provide input for setting U.K. policy. The Landfill Tax, introduced in October 1996, charges those wishing to dispose of waste in landfill sites according to the weight of waste disposed of. At the time of introduction these charges matched the estimated external costs for active and inert wastes, but since 1999 the Landfill Tax rates have been incerased , largely to assist with hitting the UK's international obligations under the EU Landfill Directive to reduce the level of biodegradable municipal waste to 35% of its 1995 level by 2020. As at April 2006, the rate of Landfill Tax was £21 per tonne of active waste and £2 per tonne of inert waste. The medium term target is to raise the rate to £35 per tonne, far above the estimated external costs.

For the Netherlands, Dijhkgraaf and Vollebergh (1998) estimate that the external cost of landfill garbage is US$28 per ton (about $.28 per 32-gallon bag of garbage).

For the U.S., Miranda and Hale (1999) employ several prior studies to produce a range of estimates for marginal social cost per ton of garbage, including both global and local pollutants. These studies "use a range of techniques to quantify impacts, including direct estimates of human and environmental health impacts, cost-benefit analysis, abatement costs for specific pollutants, and contingent valuation

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

[25] Textbooks such as Baumol and Oates (1988) note that the Pigouvian tax rate ought to reflect the marginal damages on others, not the cost of controlling the pollutant. If *at* the optimum, however, the two concepts are the same: reaching the optimal control of pollution means continuing to undertake more control until the marginal cost of control (MCC) equals the marginal benefit of control (the MED from pollution). Moreover, if the MCC is constant, then the current MCC is a good estimate of the MCC at the optimum (which is the MED at the optimum).

of changes in human and environmental health" (p. 24).[26] They show that the average tipping fee in each state is a good measure of the internal cost of disposal. This tipping fee in each state may depend upon the daily quantity intake of garbage in the landfill, the number of years of capacity left before the landfill must be shut down, the initial capital costs incurred during planning and construction of the facility, and the operation and maintenance costs incurred during active use of the facility.

The external costs can include aesthetic costs incurred by individuals such as the noise pollution from the collection trucks and a less scenic view. So far, no studies have estimated these aesthetic costs per unit of solid waste. The external cost also includes the negative health effects from toxins in the leachate that seeps into the groundwater. It includes the effect of methane gas produced as organic material in the landfill degrades. Before methane was collected and vented, it could seep through crevaces underground, emerge in some neighboring house or basement, and then possibly ignite and explode. That danger has been virtually eliminated and is not part of calculations here. If the methane *is* vented and released into the atmosphere, then external cost includes the greenhouse effect on global warming. Finally, if the methane is "flared" as required on all new landfills, then the high greenhouse effect of the methane is reduced to the lower greenhouse effect of carbon dioxide (but the flaring creates other air emissions).[27]

Table 2 provides the estimates of these costs from Miranda and Hale (1999), where each pollutant has an estimated range of external costs. The first entry, under water emissions, is the range for leachate's environmental costs. Leachate is produced from the interaction between degrading landfill material and water from precipitation. It contains toxins that may contaminate groundwater. For present purposes, leachate is a local water pollutant that will be linked with the precipitation level in each state. Because of recent controls, however, the range in the table is small ($0.0-0.1 per ton of waste).

Under air emissions, methane is the biggest potential problem. In fact, landfills are the source of 35% of methane emissions in the U.S., or 4% of *all* greenhouse gas emissions (U.S. EPA, 2001). More is said about the greenhouse effect of carbon dioxide, because so much more carbon dioxide is released into the atmosphere from industries and vehicles that burn fossil fuels, but each methane gas molecule has much more greenhouse effect than a carbon dioxide molecule. Thus the row of the table for methane shows a high range of costs from the greenhouse effect ($8.8-59.5 per ton). If the methane is flared, then the combustion converts it into carbon dioxide and other emissions, so the effect of methane from the landfill is much reduced ($2.1-6.9 per ton). On the other hand, that flaring creates more carbon dioxide, and so the range of estimated costs from carbon dioxide increases (from $0.4-1.4 per ton, to $0.7-2.0 per ton).

As shown in Table 8.6, Miranda and Hale (1999) also estimate the external cost per ton of waste for vinyl chloride ($4.3-4.8/ton), benzene ($0.1-2.8/ton), and other gases ($0.3-4.8/ton).[28] These ranges are the same whether or not the landfill flares its methane. These gases are local pollutants. The range of cost for each global pollutant reflects an inherent uncertainty about the quantity and effect of those gases that are produced at any landfill. The range is widely different for the no-flaring and the flaring facilities. On the other hand, the range for the local pollutants largely follows from the differences in local demographic and weather conditions (although again some of the uncertainty is about the amount of gas actually emitted).

[26] As their title indicates, Miranda and Hale apply these external cost estimates to data on waste quantities in Puerto Rico. Since no prior study provides estimates of external cost in Puerto Rico, however, they compile prior studies on external costs in the U.S. It is this compilation for the U.S. that is employed here.

[27] Flaring of methane creates carbon dioxide, a less-potent greenhouse gas, but landfills are also considered to be carbon "sinks" because they keep the carbon in material such as wood products from escaping into the atmosphere.

[28] "Other gases" include carbon monoxide, trichloroethylene, carbon tetrachloride, 1,1,1-trichloroethane, chloroform, 1,2 – dichloroethane and methylene chloride.

Table 8.6
Environmental cost estimates for a Landfill (1997$/ton)

Cost	No methane flaring	Methane flaring
Water emissions		
Leachate	0.0 - 1.0	0.0 - 1.0
Air emissions		
Methane	8.8 - 59.5	2.1- 6.9
Carbon dioxide	0.4 - 1.4	0.7 - 2.0
Vinyl chloride	4.3 - 4.8	4.3 - 4.8
Benzene	0.1 - 2.8	0.1 - 2.8
Others	0.3 - 4.8	0.3 - 4.8
Total	13.8 - 73.4	7.5 - 22.3

Source: Miranda & Hale (1999).

8.4 Deposit-Refund Systems

The dumping problem might also be fixed by implementation of a deposit-refund system (DRS), but such systems entail their own administrative costs. Those administrative costs might be quite low if the DRS is implemented implicitly by the use of a sales tax on all purchased commodities at the same rate, together with a subsidy to all recycling and proper garbage disposal. That practice is currently followed in the U.S., at least implicitly, since local governments do impose local sales taxes and they do provide free collection of curbside recycling and garbage.[29] In order for the local sales tax to approximate the deposit portion of a DRS, it should reflect the SMC of dumping garbage. A sales tax set lower than the SMC of dumping will not encourage an efficient level of proper garbage disposal. If the recycling subsidy needs to be larger, administrative costs can be reduced by providing a subsidy per ton, paid to recyclers, rather than providing an amount for each bottle recycled by each household. But then optimality may require a different tax and subsidy amount for each type of material – a plan that might be very costly to administer.[30] Worldwide, these programs have been successful at reducing waste and recovering recyclable materials (OECD, 1998).

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

[29] Since money is fungible, it does not matter if the subsidized collection of garbage and recycling (the "refund") is financed from sales taxes (the "deposit") or from some other source like property taxes.
[30] According to www.bottlebill.org, the eleven states with current bottle bills are: California, Connecticut, Delaware, Hawaii, Iowa, Maine, Massachusetts, Michigan, New York, Oregon, Vermont. In Europe, Austria, Belgium, Denmark, Finland, Germany, the Netherlands, Norway, Sweden, and Switzerland are all listed as having beverage container DRS. Canada has also had success with their program.

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 DRS encourages firms optimally to engineer products that are easier to recycle. Households demand such products in order to recycle and thereby to receive the refund. This result is important, since directly encouraging the recyclability of product design can be administratively difficult.[31] If the administrative cost of operating the DRS is high, then Dinan (1993) suggests that policymakers could single out products that comprise a large segment of the waste stream (newspaper) or that involve very high social marginal disposal costs (batteries).

Some have suggested that a "virgin materials tax" might encourage recycling as well as internalize the MEDs of 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 is only necessary to correct for external costs associated with extracting the virgin material. The virgin materials tax is not optimally used to correct for the marginal environmental damages of garbage disposal if a tax is available on garbage disposal.

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

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

9. Conclusions

Environmental policy has been transformed over the past decade by the use of environmental taxes, emissions trading and other economic instruments. These allow stringent environmental policies to be introduced at lower economic cost than with less-flexible forms of conventional regulation, which dictate the abatement measures that firms must take. The cost-reducing flexibility in pollution abatement offered by economic instruments will become increasingly important, the higher the standards of environmental protection which are sought.

For example, if the UK and other countries decide to make drastic cuts in CO_2 emissions, as advocated by the recent Stern Review of the Economics of Climate Change, taxes or other economic instruments such as emissions trading will be needed to play an central role in achieving the extensive changes in the energy use of firms and individuals that will be required. 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 will 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 simply concentrating on those most amenable to direct regulation.

Despite these efficiency advantages of environmental taxes and other market mechanisms there are, of course, likely to remain many areas of environmental policy where more conventional regulatory approaches will still be needed as a major part of the policy mix. In some cases regulation will be needed to ensure minimum environmental standards, particularly where the response to economic incentives may suffer from inertia or where uncertainty over responses may cause significant costs should the outcome generate more environmental damage than anticipated.

While taxes and other market mechanisms have clearly led to major changes in environmental policy-making in recent years, do they have a similar potential to transform the constraints and opportunities in tax policy?

There are two areas in particular where environmental taxes could make a major contribution to tax revenues - environmental taxes on energy and congestion charges on road transport. In both cases the available tax base is broad, demand is inelastic (so revenues will not be greatly eroded by behavioural responses), and high and/or increasing rates of tax may well be warranted by the environmental externalities resulting from energy use and vehicle congestion. The revenue potential of these taxes holds out the possibility of tax reform packages, in which the introduction of significant environmental taxes on energy or congestion lubricates tax reductions and reforms elsewhere in the fiscal system. It is possible that the political constituency in support of environmental tax measures could create an opportunity for tax reforms that might not otherwise be politically viable.

But in a more fundamental economic sense, environmental taxes do not alter the scope for efficient revenue-raising. It is an illusion to think that there might be a “double dividend” from a tax reform which introduces new environmental taxes, in the sense that besides the environmental benefits there would also be fiscal gains because the tax burden is shifted from existing “distortionary” taxes on labour and/or capital to environmental taxes. By raising marginal costs of production and hence prices, environmental taxes reduce the net return from each hour worked to a similar extent as the direct taxation of labour; and a revenue-neutral shift in the tax burden from labour to environment would not reduce the distortionary impact of the tax system on labour supply.

The implication is that the case for environmental tax reform must be made primarily on the basis of the environmental gains that would result. The fiscal aspects of environmental tax reforms are important, because inappropriate use of the revenues, or their unnecessary dissipation, can greatly add to the costs of environmental policy. But an appeal to the fiscal consequences of environmental tax reform cannot justify measures that do not pay their way in purely environmental benefits.

Turning to the three main environmental policy areas where we have discussed the scope for environmental tax measures, we have general observations about three common themes in our analysis:

- Environmental tax policy needs to be grounded in empirical evidence on the scale of the marginal externalities involved, to indicate the scale of environmental taxes that might be justified and the potential scale of the benefits of policy action.
- Frequently it will be necessary to think of multi-part instrument combinations, because the available tax instruments (especially those that rely on differentiation of existing taxes) may not be very accurately targeted to the externalities which environmental policy aims to address.
- We may want to think of environmental taxes forming part of a portfolio of policy measures, including measures to stimulate responses to the price signal that environmental taxes establish. There may well be political and practical constraints on setting environmental taxes at the first-best level, and measures that stimulate the development or use of abatement technologies may then be needed, rather than simply relying on an externality tax set at the first-best level. The case for such packages may be enhanced by recognising that there may be various market failures in the development or dissemination of new abatement technologies, and that well-targeted measures to stimulate research and development or diffusion may enhance efficiency.

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.

Congestion charges on private motoring could, for example, be a major source of tax revenues, if levied at a rate reflecting the congestion externality imposed by each individual motorist on other road users. For example, Newbery (1990) estimated that the congestion cost per vehicle-kilometre averaged some 3.4 pence across the UK. If this was fully reflected in a UK-wide congestion charge, it would imply revenues of some £20 billion annually at current values, some 5 per cent of total fiscal receipts. On the other hand, once congestion externalities are separately taxed, this may weaken the case for high motor fuel excises and other taxes on motoring. Sansom *et al* (2001) estimated that the congestion component of motoring externalities made up around three quarters or more of the total externality. If the remaining motoring externalities did not justify retention of the existing high taxes on motor fuels, the net revenue gain from a congestion tax would be substantially lower. The interaction between a well-targeted congestion tax and other externalities requires careful consideration. By providing the right price signals to motorists, national congestion pricing could increase overall traffic volumes and hence environmental emissions. Establishing the right motor fuel taxation policy once a congestion tax is introduced will require careful modelling of the likely response to such a measure.

Drawing on estimates of the relative externalities involved, we are likely to want to make recommendations about the relative taxation of diesel fuel and petrol, though these will be very sensitive to the view we take about the appropriate value to place on abatement of CO_2 emissions. Adopting the high value for the social costs of CO_2 emissions would imply a strong preference for diesel, despite its higher emissions of particulates which have adverse health effects, especially in urban areas. We should also say something about the promotion of biofuels through reduced taxation or direct subsidy. The environmental case seems weak, and we should be able to assemble externality evidence on this.

We should also say something about the case for measures 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 tradeable fuel efficiency targets for manufacturers, and the US experience with CAFE standards will be relevant in assessing this.

Industrial and household use of energy

The Stern review recommends that significant abatement should be undertaken now to reduce the long-term costs of climate change.

A carbon tax would be an appropriate, broadly-based incentive measure to achieve the kinds of changes in energy use that would be required by the Stern recommendations. But the EU is now well advanced along the alternative track of emissions trading.

There are close similarities in the economic effects of taxes and emissions trading, where the tradeable permits are auctioned (although Stern argues that the different properties under uncertainty favour the case for permits over taxes). However, where permits are distributed without charge to polluters (as is largely current practice in the EU ETS), there is a significant opportunity cost (Parry, 19xx).

[[Do we want to include anything about the international scope of such policies? Presumably one concern is that if the UK or EU introduce a carbon tax unilaterally, production will shift abroad and the environmental impact is muted. In an ideal world, a common global carbon price would operate but there are obvious political and economic reasons why getting such a policy operating quickly is hard.]]

In general there will be a case for measures to stimulate new abatement technologies alongside energy or carbon pricing.

There are some particular issues about particular components of energy use where we should discuss recommendations

- The case for replacing the current fixed-rate Air Passenger Duty with a tax better related to the external costs of air transport. We might want to discuss something about the difficulties of action in this area given the Chicago Convention, EU regulations etc. – the implication to me seems to be that provided it can get through this, bringing aviation into the ETS is desirable so long as the ETS itself is reformed so that permits are auctioned and the allocations are not over generous. Discuss case of Norwegian attempts to reform its domestic air taxes?
- The need for a package of accompanying policy measures to tackle distributional issues raised by taxing domestic energy

Waste

Here, environmental taxes are unlikely to raise major revenues, but their use is of considerable environmental and economic significance

Current UK policy is introducing sharp rises in the landfill tax, to achieve policy goals relating to the reduction of landfill use which have no clear justification in terms of the scale of environmental externalities.

This is an area where multi-part instruments are likely to be essential, to reflect the complex externalities involved, and the multiple goals of waste management policy (upstream waste-reducing innovation by producers, consumer purchase, disposal and recycling decisions, downstream decision-making about waste management, etc.

We should illustrate with some suggestions based on estimates of the marginal externalities, and drawing on evidence on behavioural responses for US policy (eg on pay-as-you-throw).

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