CHAPTER

5

Pollution taxes and tradable emission permits: Theory into practice

5.1 Introduction

In this chapter, we will be analysing two economic instruments for pollution control: pollution taxes and tradeable permits. We do this in the context of the practical application of these instruments to real-world pollution control problems. To be more precise, we assume that society has set some specified target for pollution reduction. This target will be assumed to be different from the optimal level of pollution which, as we argued in the previous chapter, is virtually impossible to identify in any case. Targets instead will be assumed to have been set through the political process, using scientific inputs on likely damages, and economic inputs on both damage costs and control costs. Such targets are typically of two types. The first is a target reduction in emissions output, across a specified set of dischargers. Examples of such targets include the US government's target reduction of 10 million tons, over 1980 levels, of SO_2 emissions from power stations, and a reduction of 2 million tons in NO_x emissions, under the 1990 Clean Air Act amendments. Another example of a load reduction target is the national reductions in greenhouse gas (GHG) emissions agreed to under the Kyoto protocol.

The second type is a target improvement in ambient environmental quality. An example here is the range of target improvements in ambient water quality parameters adopted as 'environmental quality standards' by the Environment Agency in England. Thus, the Agency may have an objective of increasing dissolved oxygen levels in an estuary up to 8 mg/l, through a policy of reducing discharges of substances exerting a biological oxygen demand (BOD) in the estuary. Other examples of ambient environmental targets ~~include the~~ following:

- upper limits for substances in drinking water (e.g. for lead or nitrates)
- for bacteriological contaminants in bathing water (set as maximum levels of coliforms per litre)
- for the European Union, a target of 'good ecological status' for all waterbodies, under the Water Framework Directive.

With respect to any of these environmental objectives, an Environmental Protection Agency (EPA) has many potential policy tools at its disposal. As noted in the previous chapter, these can be classified into regulatory mechanisms such as design and performance standards; economic instruments; and voluntary approaches. In this chapter, the focus is on the practical application of economic instruments. As also set out in the previous chapter, there are many criteria against which an EPA can judge the performance of these alternatives. Here, the focus will be on efficiency, or social cost minimisation, as the main criterion of interest, although we also comment on distributional effects of taxes and permits and on political acceptability and environmental performance.

5.2 Efficiency properties of pollution taxes

In order to present the most fundamental result in efficient pollution control (the Baumol and Oates least cost tax theorem), we shall initially assume that efficiency is the sole criterion used in deciding policy choice. In deriving the result, much use is made of the notion of pollution abatement costs, and in particular the marginal abatement cost (MAC) function.

For a firm, an abatement cost function describes the cost of reducing the output of an emission. In general, firms have a number of options open to them to reduce emissions. First, they can reduce output of their product. So, if a coal-fired power station wishes to cut its output of waste gases such as SO_2, it can reduce the number of hours that its furnaces run. Electricity output falls, but so does the output of SO_2. Second, a firm may change its production process. Thus the power station could switch to a combustion process that produces less waste gases per kwh of electricity, or else substitute lower-sulphur coal for its existing coal input. Finally, the power station can install a filter on the end of its chimney to remove SO_2 from the waste gas stream (a process known as flue gas desulphurisation). This 'end-of-pipe' technology is available for many production processes: paper mills, for instance, are able to install settlement ponds and centrifuges to reduce the sediment content of liquid effluent before it is discharged.

Our assumption will be that firms will always seek the lowest cost method of pollution control available to them. This may involve a combination of approaches: input substitution up to a certain level of emission reduction, output reduction and then end-of-pipe treatment; or it may involve the use of two or all three approaches simultaneously. We shall also assume that, as a general principle, each firm is better informed about the most efficient manner for reducing its own emissions than is the regulator. Empirical evidence (e.g. Bergman, 1991) and theory both suggest that marginal abatement costs, defined as the change in the lowest-cost way of reducing emissions for a change in emission reduction, are increasing with the level of emission reduction, as is shown in Figure 5.1.

In a free market system, with no government control on emissions and no altruism on the part of the firm (we also assume that emissions from the firm do no damage to that firm itself), the firm will locate at e^f, spending no money on emissions control. As

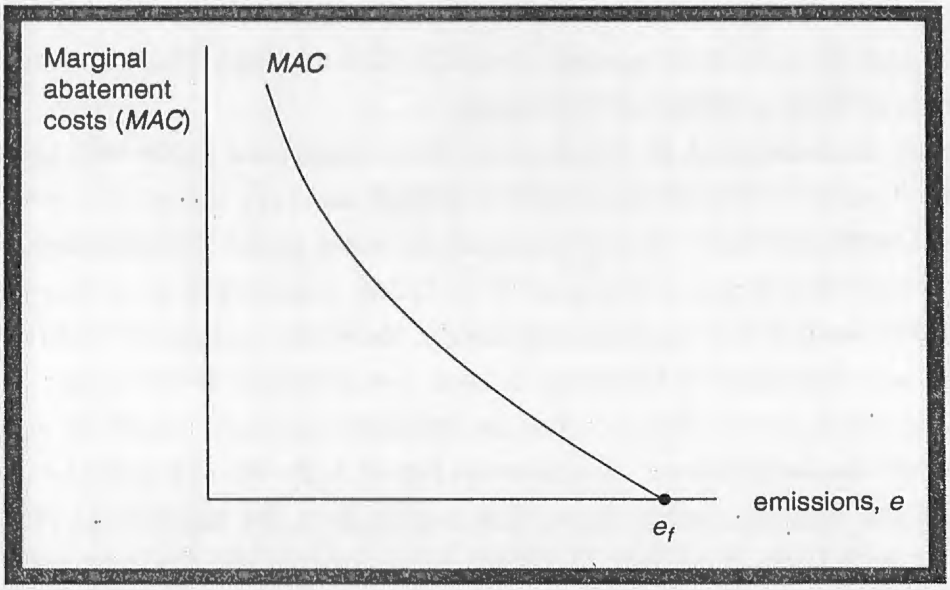

Figure 5.1 Marginal abatement costs for a firm

emissions are reduced, abatement costs rise at an increasing rate. Specifying a continuously increasing MAC function is convenient analytically, since it implies that local and global cost-minimising solutions will coincide. Some researchers have found that, for some discharges, economies of scale are present in emissions treatment (e.g. Rowley *et al.* 1979). For the remainder of this chapter, however, we will assume continuously increasing MAC functions.

It is also to be expected that MAC functions will vary across sources, for a given pollutant. This means that some sources of, for example, BOD will find incremental reductions in BOD output (much) less expensive than others, owing to differences in plant location, age and design; different production processes (distilling, paper making, oil refining); differing levels of current emissions reduction, and differing levels of managerial knowledge and ability. For example, Hanley and Moffatt (1993) found that MACs for direct discharges of BOD to the Forth Estuary in Scotland varied by as much as thirtyfold across polluters.

The observation that MACs vary across sources is a key insight into why the cost-minimising means of securing a target reduction in aggregate emissions will involve different amounts of emission reduction across sources. Assume for the present that a uniformly mixed pollutant, such as a volatile organic compound (VOC), is the object of control. Uniform mixing means that the target reduction in emissions is independent of the source of emission, since a tonne less of discharge from any source in the control area is equally effective in meeting a pollution reduction target as the same reduction from any other source. It would seem sensible, in this situation, for high abatement cost sources to reduce emissions by less than low abatement cost sources. In fact, a necessary condition for an efficient solution in this case is that abatement costs, at the margin, are equalised across all sources. This is proved formally below, but the intuition is clear enough: if at the current allocation of emission reduction responsibility source A can achieve a one-unit cut in VOCs at a cost of £100/unit, and source B faces a cost of £500/unit at the margin, then a unit of emission reduction responsibility can be reallocated from B to A for a net saving of (£500–100) or £400. These cost savings will remain possible so long as MACs are not equal.

This point is demonstrated in Figure 5.2, where emissions from two sources with varying MACs, source A (low cost) and source B (high cost) are shown. For convenience, both MAC functions are shown as originating at the same point. A performance standard designed to achieve the target emission level of $1/2(e^f_A + e^f_B)$ might set a maximum limit on each firm emissions of $\bar{e}$ (a 'uniform standard'). However, this results in firm B having a higher MAC at $\bar{e}$ than firm A: efficiency is thus not achieved in this case.

Baumol and Oates (1971) showed that an efficient outcome could be achieved by setting a per-unit tax on emissions. As shown in Figure 5.2b, this tax rate, t^*, is calculated as the MAC of the industry (MAC_I; here, firm A plus B) at the target level of emissions. Faced with t^*, each firm, as shown in Figure 5.2a, equates the tax rate with its MAC schedule by varying its level of emissions. This is its cost-minimising reaction. For firm A, emitting more than e^t_A is inefficient, since the marginal benefits of reducing emissions (avoided tax payments on the marginal unit, t^*) exceed the marginal costs, as shown by MAC_A. Similarly, cutting emissions below e^t_A is inefficient, as the marginal costs exceed the marginal benefits. Note that, through self-interest alone, the desirable pattern of emission reduction has been achieved, since firm B (the high abatement cost source) has reduced emissions by less, relative to its no intervention level of e^f_B, and we have the result that $MAC_A = t^* = MAC_B$. To state the theorem as Baumol and Oates put it, 'A tax rate set at a level that achieves the desired reduction in the total emission of pollutants will satisfy the necessary conditions for the minimisation of the programme's cost to society' (Baumol and Oates, 1988).

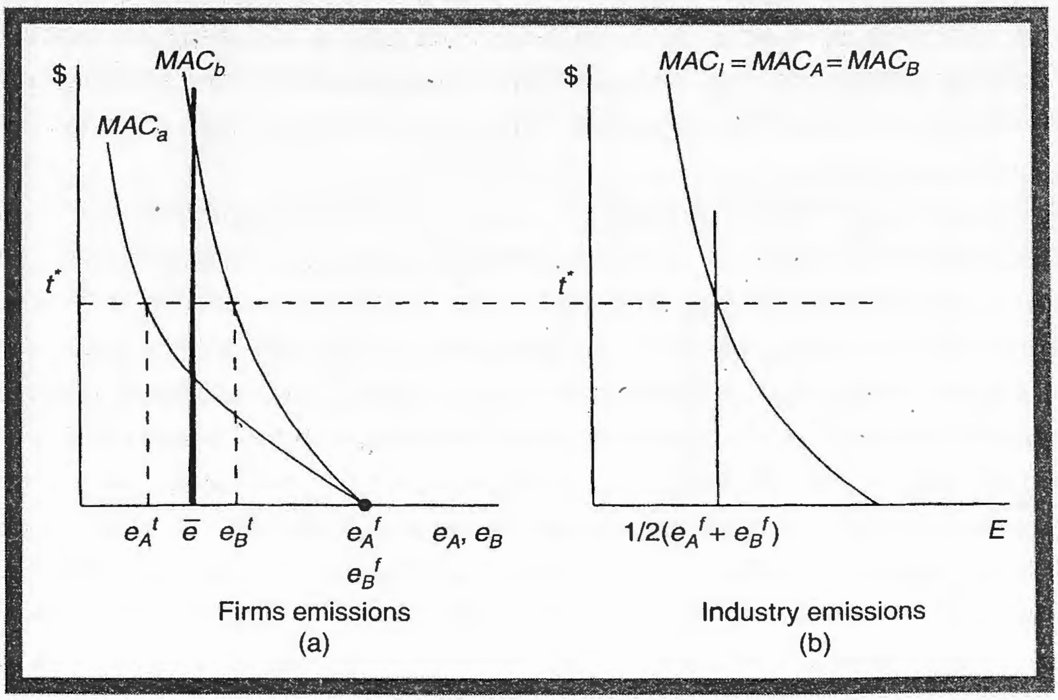

Figure 5.2 An efficient tax on emissions

One important caveat to state here is that it is the resource costs to society that we seek to minimise: the solution to this problem will only coincide with firms' reactions to a tax if the costs that firms face in controlling pollution are identical to social costs – it rules out, for example, the case where pollution reduction processes actually increase emissions of a second pollutant at no cost to the discharger. In this case, private resource cost minimisation will not coincide with social resource cost minimisation, since the marginal private costs of inputs will not equal their marginal social costs, unless all pollutants are subject to Baumol and Oates-type taxes. We should also stress that the present analysis relates solely to a uniformly mixed pollutant, that is, one where the environmental damage done does not depend on the spatial location of the pollution event: allowing for non-uniform mixing, where the spatial location of discharges partly determines environmental impact, complicates the tax policy option, as will be discussed later.

Formal proofs of the efficiency properties of a tax on emissions have been provided by Baumol and Oates, and by Fisher (1980). Our proof of the theorem is adapted from this latter source. Suppose there is some uniformly mixed pollutant, defined at any point in time as a flow E. Total emissions are given by the sum of individual discharges across all sources k,

$$E = \sum_k e_k \tag{5.1}$$

Firms produce output y_k using inputs r_{ik} (so that input r_{1k} is the amount of input 1 used by firm k), according to a production function $y_k = f_k(r_{1k}, \ldots, r_{nk})$. Firms make use of an end-of-pipe technology to reduce polluting emissions which involves use of abatement inputs v_k, so that there is an emissions function $b_k(y_k^*, v_k) = e_k$. The cost of a unit of abatement is given as p_v, the price of inputs (assumed exogenous to the firm) is p_i. The social planners' problem is to

$$\underset{(r_{ik}, v_{ik})}{\text{Min}} \quad \sum_i \sum_k p_i r_{ik} + \sum_k p_v v_k \tag{5.2}$$

$$\text{subject to} \quad y_k^* = f_k\ (r_{1k}, \ldots, r_{nk}) \tag{5.3}$$

$$\text{and} \quad e_k = b^k\ (y_k^*, v_k) \tag{5.4}$$

$$\sum_k e_k \leq \overline{E} \tag{5.5}$$

$$e_k \geq 0, \quad \text{all } k = 1, \ldots, K \tag{5.6}$$

The planner thus seeks to minimise the sum of input costs (the first term in Equation 5.2) and pollution abatement costs (the second term in 5.2), subject to production being equal to some exogenous, specified level for each firm (y_k^*) determined by demand and the firm's objectives – since otherwise minimising pollution emissions and abatement costs comes about by minimising output – given the production function in (5.3), the emissions production function (5.4) and a constraint on the maximum permitted level of emissions (5.5). Equation (5.6) just says that emissions from any firm cannot be negative. Substituting for the actual level of emissions using the emissions production function, and forming the Lagrangian, we have

$$L = \sum_i \sum_k p_i r_{ik} + \sum_k p_v v_k + \sum_k \lambda_k [y_k^* - f_k(.)] + \mu \left[\sum_k b^k(.) - \overline{E}\right] \tag{5.7}$$

where λ_k and μ are Lagrangian multipliers. The first-order conditions for a minimum of (5.7) with respect to input use r_i and pollution abatement v_k are:

$$\partial L/\partial r_{ik} = p_i - \lambda_k (\partial f_k/\partial r_{ik}) = 0, \qquad \text{all } i \text{ and } k \tag{5.8a}$$

and

$$\partial L/\partial v_i = p_v + \mu (\partial b^k/\partial v_k), \qquad \text{all } k. \tag{5.8b}$$

These conditions say that inputs, and pollution abatement, should be employed up to the point where their prices are equal to the value of their marginal products, where the marginal products are shown by the partial derivatives, which are then converted into value terms by the Lagrangian multipliers.

Let us suppose that the planner decides to achieve the target emission level $\overline{E}$ by setting a per unit tax on emissions of t^*. Clearly, this must be of a particular value to achieve $\overline{E}$, given firms' abatement costs: from the earlier graphical analysis, we know that t^* must be set equal to aggregate MACs at $\overline{E}$. Taking the problem faced by a representative, cost-minimising firm facing an emissions tax set at t^*, firm k will want to achieve

$$\underset{(r_{ik}, v_{ik})}{\text{Min}} \sum_i p_i r_{ik} + p_v v_k + t^* e_k \tag{5.9}$$

subject to Equations (5.3), (5.4) and (5.6). Again, substituting for e_k using the emissions production function $b^k(.)$ and forming the Lagrangian

$$L^k = \sum_i p_i r_{ik} + p_v v_k + t^* b^k(.) + \beta^k [y_k^* - f^k(.)] \tag{5.10}$$

Differentiating L^k with respect to input and abatement use and assuming no boundary solutions, the first-order conditions for a minimum are

$$p_i - \beta^k (\partial f^k/\partial r_{ik}) = 0 \qquad \text{for all } i \tag{5.11a}$$

and

$$p_v + t_k^* (\partial b^k/\partial v_k) = 0 \qquad \text{for all } k \tag{5.11b}$$

Comparing Equations (5.11) with Equations (5.8), it can be seen that the firm's optimum will coincide with the social optimum when

1. Input prices faced by the firm, p_l and the pollution abatement price, p_v correspond to their competitive levels: that is, the firm has no price-setting power in the input or pollution abatement markets.
2. The tax rate t^* is equal to μ, the shadow price of pollution reduction in the social planners' problem. Note that this is just what was said above: the least-cost tax is equal to the marginal (shadow) cost of abatement at the target level of emissions, E^*. This can be seen more clearly if condition (5.11) is rearranged, giving $t^* = -p_v/b_v^k$ (where $b_v^k \equiv \delta b^k/\delta v_k$), since the expression $-p_v/b_v^k$ is the marginal abatement cost for firm k. Note that this also implies, for a given t^*, that MACs across all firms must be equal under the cost-minimising solution, which is the conclusion we reached earlier by an intuitive route.

What happens if we are trying to use a tax to control a non-uniformly mixed pollutant instead? For many potentially polluting substances, ambient concentrations at a given monitoring point are dependent not just on the total amount of emissions (E in the preceding model), but also on their spatial location. A good example is dissolved oxygen (DO) levels at a particular point j in a river. For given flow and temperature conditions, the DO level will be a function of both the total amount of BOD discharges *upstream* of point j and their location, and discharges at point j itself. This is because 2000 kg/day of BOD discharged one mile upriver of point j will have a bigger impact on the DO level than the same quantity discharged five miles upriver since, in this latter case, natural degradation and re-aeration processes will have had longer to 'work' on the effluent than in the former case. This spatial relationship is also true for many air pollutants: acid deposition (from SO_2, NO_x and ammonia discharges) in a particular lake will depend on prevailing wind directions and distance from major discharge points. Targets set for such pollution problems are likely to be in terms of maximum deposition rates in certain geographic areas for acidity, or maximum hourly concentration levels in a city for a pollutant such as NO_x.

What are the implications for the Baumol and Oates theorem of a non-uniformly mixed pollutant? Basically, that a single tax rate will no longer be efficient, since the tax rate should vary across sources according to their marginal impacts on ambient air or water quality levels. Suppose that the ambient level of pollution at any monitoring point j, a_j, is a function of emissions from all sources:

$$a_j = \sum_k d_{jk} e_k \tag{5.12}$$

The d_{jk} coefficients are often referred to as 'transfer coefficients' and form a $(k \times j)$ matrix, where there are $k = 1, \ldots, K$ sources and $j = 1, \ldots, J$, monitoring points. Any particular transfer coefficient, such as d_{23}, shows the impact of discharges from source 2 on water quality (for example) at monitoring point 3. These d_{jk} terms will vary, for a river or estuary, according to the time of year and consequent variations in temperature and flow rate. They are often measured under worse-case conditions (for a river, these are known as dry weather flow, DWF). For an air shed, transfer coefficients may be calculated as an average across all windspeed/direction conditions recorded in some time period. In all cases, the transfer coefficient matrix is generated from some model of the environmental system of interest, for example a river, the air shed over a city. An excellent account of such a process is given in O'Neil *et al.* (1983).

For non-uniformly mixed pollutants, the control agency's target might be specified as seeking to reduce ambient concentrations to some target ambient level (such as 7 mg/l of DO in a river under low flow conditions). This can be written as

$$\sum_k d_{jk} e_k \leq a_j^* \tag{5.13}$$

where a_j^* is the ambient target at monitoring point j. The planners' problem is now to minimise (5.2) subject to (5.3), (5.4), (5.6) and (5.13). The Lagrangian becomes

$$L = \sum_i \sum_k p_i r_{ik} + \sum_k p_v v_k + \sum_k \lambda_k \left[y_k^* - f^k(.)\right] + \sum_j \mu_j \left(\sum_k d_{jk}\left[b^k(.)\right] - a^*\right) \tag{5.14}$$

Solving for the first-order conditions with respect to r and v, and comparing these with the decisions of firms faced with a pollution tax, it is possible to show that, in order to achieve an efficient solution, each firm must face a different tax rate t_k^*, which is determined by that firm's degradation of environmental quality at each monitoring point (given by the transfer coefficients), and by the ambient target itself: that is, is equal to $\sum_j d_{jk}\mu_j$ for firm j. Shadow prices of improving ambient quality at any monitoring point j, μ_j, are positive so long as emission reductions are necessary to meet the ambient target and where, after the imposition of the tax policy, the ambient standard is met exactly. In the language of linear programming, these shadow prices are dual values and exist only for constraints which are binding in the optimal solution. Since firms can have different transfer coefficients for different monitoring points, it might be desirable to calculate tax rates on the basis of transfer coefficients for the most polluted monitoring point, or the monitoring point where economic measures of pollution damage are highest. The alternative, as Tietenberg (1973) first proved, is to have separate tax rates for each monitoring point, which are then adjusted for each firm according to its transfer coefficient relating to that point. Firms would thus face a different tax bill for each monitoring point they affect, with the firm's total tax bill being the sum of taxes paid at each monitoring point. Thus a unique shadow price or tax rate μ_j exists at each monitoring point j, and firm k pays a tax equal to $[d_{jk}\mu_j]$ for emissions affecting point j. The total tax paid by the firm would be $\sum_j d_{jk}\mu_j$ per unit of emissions.

As Tietenberg (1974) first pointed out, '. . . forcing upwind and downwind polluters to pay the same tax will produce the desired concentration (reduction), but at a cost which exceeds the minimum cost means of achieving that concentration' (p. 464). Tietenberg goes on to point out that a perfectly differentiated tax system, with each polluter facing a unique, location-determined tax rate, would be 'administratively difficult at best and politically infeasible at worst', so that a compromise, such as a zonal tax system where tax rates vary across zones but not within zones, might be preferred. Empirical evidence on this point was provided early on by Seskin *et al.* (1983), see Box 5.1.

BOX 5.1

Pollution taxes and air quality

In a 1983 paper in the *Journal of Environmental Economics and Management*, Seskin *et al.* examine the costs of meeting a target improvement in ambient levels of nitrogen dioxide (NO_2) in Chicago. They compare a uniform standards regime with pollution taxes, using a mathematical model of air quality to estimate transfer coefficients between emission sources and 600 receptor (monitoring) points. Dischargers were divided into nine source categories, including power stations, municipal incinerators and industrial boilers. Each policy option was compared to a 'no control' baseline, under which 36 receptors were found to be in violation of the NO_2 standard. Engineering data were used to estimate marginal abatement cost functions for discharge sources, and a type of programming model (known as integer programming) used to simulate the least-cost outcome.

The control strategies modelled were the following:

- a state implementation plan (SIP) strategy, whereby uniform design standards were imposed on similar categories of sources;
- a uniform emissions tax, set at a rate high enough to ensure that those sources having the largest effect on ambient air quality per unit of discharge (i.e. those with the highest transfer coefficients) were controlled sufficiently to meet the target improvement; and
- an emission tax differentiated by source category.

Given that NO_2 is a non-uniformly mixed pollutant, none of these strategies could replicate the least-cost solution, which enables the target to be met at all points for an annualised total abatement cost of $9 million. This is because the least-cost solution would require all sources to face a unique tax rate, that is that there is a perfectly differentiated tax system. Simulation results are given in the following table.

Policy	*Number of polluting firms controlled*	*Area-wide reductions in emissions (%)*	*Annual control costs ($m)*
Least-cost	100	3	9
Uniform regulation (SIP)	472	21	130
Uniform tax	534	84	305
Source category tax	472	18	66

As may be seen, the uniform tax rate has to be set so high (high enough to sufficiently restrict emissions from the most damaging source) that this policy has a higher resource cost than the command and control option of the SIP. The uniform tax rate also gives the biggest reduction in area emissions, since this high tax rate produces too much abatement from less damaging sources (note that all policies in the table achieve the ambient target level of air quality). A source category charge, however, is more efficient than either a SIP or a uniform charge, with tax rates varying between $15,800 (per year per pound of NO_2 per hour) for industrial coal-fired boilers and $13,500 for industrial process units.

So far in this chapter the discussion has been entirely in terms of a tax on emissions. However, as noted in Chapter 4, the ideas behind an emissions tax can be extended either to a tax on inputs or to a tax on products. With regard to inputs, Common (1977) showed that, so long as the 'pollution production function' relating inputs to emissions was known, a desired reduction in emissions could be achieved at least cost with a tax on inputs. Input taxes are conceptually very important in the analysis of the control of non-point pollutants where the monitoring of emissions is either difficult or impossible. But input taxes could also be utilised for point source emissions, an example being taxes on the sulphur content of coal as a means of reducing SO_2 emissions from power stations. Input taxes may involve problems where an input substitution occurs as a result of an input tax, and where the substitute input has adverse environmental effects. For example, taxing CFCs could lead firms to switch to HCFCs, which have been argued to be more damaging to global climate control, per molecule, than CFCs, as coolants, whilst taxing particular pesticides might cause farmers to substitute more harmful chemicals (see Box 5.2).

BOX 5.2

Pesticide taxes in Denmark

Worries over pesticide contamination of groundwater resources led the Danish government to take action over pesticide use by farmers. In 1995, a new tax on pesticides was introduced as a means of reducing these environmental damages. The tax is levied as a percentage of the wholesale price, at rates of 53% (insecticides), 33% (herbicides) and 3% (wood preservatives and rodenticides). Interestingly, these tax differentials do not reflect perceived differences in environmental risk – as economists might recommend – but rather the differences in treatment intensity (Schou and Streibig, 1999). The unfortunate implication of this design of tax policy is that products which exert higher environmental damages per kilogram used are effectively taxed at lower rates than those that are less harmful, since more 'environmentally friendly' pesticides tend to sell for higher prices.

Tax revenues, which in 2000 amounted to 375 million Danish Krone, are partly (60%) recycled to farmers as subsidies for organic farming and advice; the remaining 40% is spent on public research and monitoring programmes. However, it is not thought that the tax is set at a high enough rate to produce real incentive effects, especially given the nature of the apply/not apply decision that farmers need to take during the year (rather than how much to apply).

This example shows the importance of policy decisions over whether to tax pesticides using a sales tax (the Danish case), or a tax per kilogram of active ingredient, or a tax per unit of expected environmental damage. This in turn depends on what the exact objective of setting the tax is: to reduce pesticide use, to reduce environmental risks or merely to implement the polluter pays principle as a means of raising tax revenues. Note that the specific aim of the Danish tax was to raise revenues for pesticide research and extension advice. It is also the case that a given tax regime may actually encourage farmers to substitute away from currently used products into *more* environmentally risky pesticides.

Pesticide taxes now exist in other countries too, including Norway, Sweden, Finland and the Netherlands.

Finally, if a stable, predictable relationship between output of a product and emissions of a pollutant could be found, then a Baumol and Oates tax could be levied on products. For example, a tax on batteries might reduce cadmium and nickel pollution in drinking water. However, the relationship between emissions of a pollutant and product prices may be very difficult to estimate, and are likely to be depend greatly on how and when that product is both used *and* disposed of.

5.3 Efficiency properties of tradeable pollution permits

An alternative approach to pollution taxes as a way of achieving a target reduction in pollution is that of tradeable pollution permits (TPPs). This idea, which originated with Crocker (1966) and Dales (1968), has gained much popularity recently with environmental economists. However, as we will see, TPPs have their own set of problems. In this section, the basic theory of TPPs is first set out, both for uniformly and non-uniformly mixed pollutants. Later on in the chapter we review problem areas with TPPs, and compare the properties of TPPs and pollution taxes.

From Chapter 3, we know that the major economic explanation for pollution is the absence of a sufficiently defined and enforced set of private property rights in environmental resources. The main idea behind TPPs is to allocate such rights, and make them tradeable. This results in a market the right to pollute and consequently in the emergence of a market price for this right. Under certain conditions, this price provides the correct incentive for dischargers to arrange emission levels such that a cost-minimising solution is reached. For a uniformly mixed pollutant, we know from Section 5.2 that this involves an equality of MACs across polluters. Let us see how this works out for TPPs, considering first the simplest case, namely an assimilative, point-source, uniformly mixed pollutant – for example, carbon dioxide emissions from power stations. All that the control agency is concerned to achieve is a specified reduction in total emissions, irrespective of the locations of dischargers. Suppose current emissions from a region are 200,000 tonnes per year, and that the target reduction is 100,000 tonnes, leaving 100,000 tonnes of continuing emissions. The agency issues 100,000 permits, each one of which allows the holder to emit one tonne of CO_2 per year. Discharges are illegal without sufficient permits to cover them. These permits may be issued in two ways:

1. by giving them away, perhaps pro rata with existing emissions (this process is known as 'grandfathering')
2. by auctioning them. We discuss the role of auction design later on.

In either case, firms are then allowed to trade these permits. We expect firms with relatively high MACs to be buyers, and firms with low MACs to be sellers, assuming the initial allocation not to conform to the least-cost one. This is shown in Figure 5.3, where the horizontal axis measures both emissions and permits held by the firm.

Before any intervention by the EPA, the firm is at e_f, controlling no emissions. Suppose a TPP system of control is now introduced, and market price for permits of p^* is established. The firm will choose to hold e^* permits, since for any holding below this level, MACs lie above the permit price (it is cheaper to buy permits than to reduce emissions),

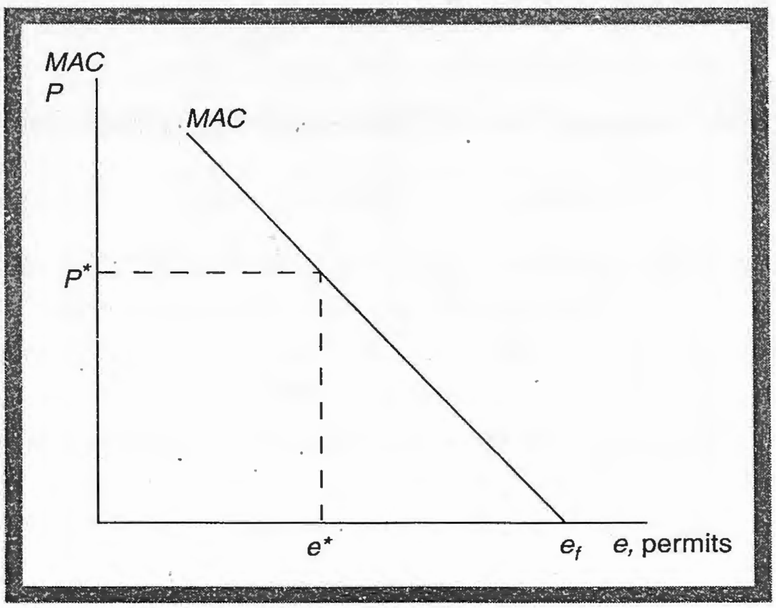

Figure 5.3 Firm's optimal response to a permit scheme

but if the firm initially holds more than e^* (and thus can emit to the right of e^*), it will choose to sell, since the price it can get (p^*) exceeds the marginal cost of making permits available for sale by reducing emissions. A firm with higher costs of controlling pollution will wish to hold more permits given a permit price of p^*.

Where does p^* come from? It is the equilibrium price in the permit market, as is shown in Figure 5.4. The agency issues a fixed number of permits, $\bar{E}$ (100,000 in this case). We know that each firm compares its MAC schedule with the permit price to decide how many permits to hold. If prices fall, the firm will hold more permits and control fewer emissions. The MAC curve for a firm is thus its demand curve for permits, and so the aggregation of MAC curves across $i = 1, \ldots, n$ firms in the control region $\sum_i \text{MAC}_i$ is the regional demand for permits. If the authority increases or decreases the supply of permits then, given a permit demand curve, the market-clearing permit price will fall or rise respectively.

The intuition behind the least-cost property of TPPs should now be clear. In Figure 5.3, the firm equates the permit price with its MAC schedule, so that for firm 1, say, we get $\text{MAC}_1 = p^*$. Another firm, firm 2, will make the same adjustment to its emission levels in the face of p^*, and if all k firms do the same, then we get

$$\text{MAC}_1 = \text{MAC}_2 = \cdots = \text{MAC}_k = p^*$$

for which a uniformly mixed pollutant is a necessary condition for cost-minimisation across the total of dischargers. These reactions by firms move them to their cost-minimising positions, and imply differing emission levels across firms.

Alternatively, we could view TPPs as a way of maximising the reduction in emissions subject to a given total expenditure on abatement. For example, Kling (1994) calculates that a system of tradeable permits for emissions from light-duty cars and trucks in California could cut emissions by 65% more than a uniform standard on exhaust emissions for the same level of costs. In this case, the TPP system would work by manufacturers

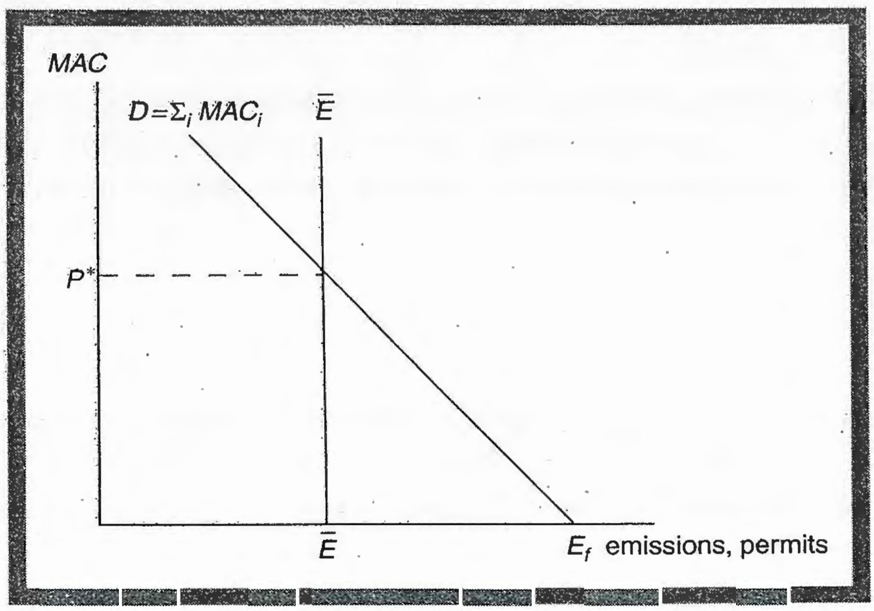

Figure 5.4 Supply and demand for permits

trading emission permits internally (higher design emission levels from some cars against lower design levels from others) or by trading with other car/van manufacturers.

As with the tax scheme, the total financial burden to any individual firm will be composed of resource costs (the sum under MAC) and transfer payments. In Figure 5.5, the financial burden for a particular firm is shown under three possible scenarios. In (a), the firm must pay for all the permits it wishes to hold (say in an auction, where the declared single price is p^*, and where the firm has no influence over this price). In (b), the firm is given some permits, but less than it requires for cost minimisation, so it buys additional permits from other dischargers. This is equivalent to the firm being only taxed on a fraction of its emissions. In (c), the firm initially receives more permits than it requires, and so sells some. It may be seen that the transfer payments for a given firm depend on the permit price and whether it is a net buyer or net seller (in all three cases, resource costs – i.e. control costs – are as shown in Figure 5.5a). For the industry, net transfers are zero under a 'grandfathering' scheme, since revenue from sales cancels out permit expenditures in aggregate (although transactions costs will impose an additional burden on firms – see p. 153). Under an auction, however, transfers leave the industry en bloc. Finally, in the case considered here (a uniformly mixed pollutant), it should be obvious that permits exchange at a rate of 1:1. If Bloggs sell 100 permits to Smith and Sons, then Bloggs must cut their emissions by 100 units, and Smith may increase theirs by 100. This is because control, as has already been said, is aimed at the total of emissions, not their spatial location. This will clearly not hold when we consider non-uniformly mixed pollutants.

Let us now establish our main results so far considered more formally. The original proof of the least-cost property of TPPs is due to Montgomery (1972), but our proof draws on Tietenberg (1984). Suppose that A represents the level of carbon dioxide (a uniformly mixed pollutant) emitted from the control region, and is given by:

$$A = \alpha + \sum_i (e_{fi} - x_i) \tag{5.15}$$

where α is emissions from other sources including natural sources, e_{fi} are 'uncontrolled' emissions from $i = 1, \ldots, n$ polluting firms (as point e_f in Figure 5.3) and x_i are reductions in emissions. Firms face control costs C_i which depend solely on its level of emission reduction:

$$C_i = C_i(x_i) \tag{5.16}$$

where $C_i(x_i)$ is a twice-differentiable function, with $C' > 0$ and $C'' > 0$ (with C' and C'' representing first- and second-order derivatives of C). The control agency wishes to hold total emissions at or below some level $\overline{A}$, which is assumed to be less than the current total of discharges. The agency's problem is thus to achieve

$$\underset{(x_i)}{\text{Min}} \sum_i C_i(x_i) \tag{5.17}$$

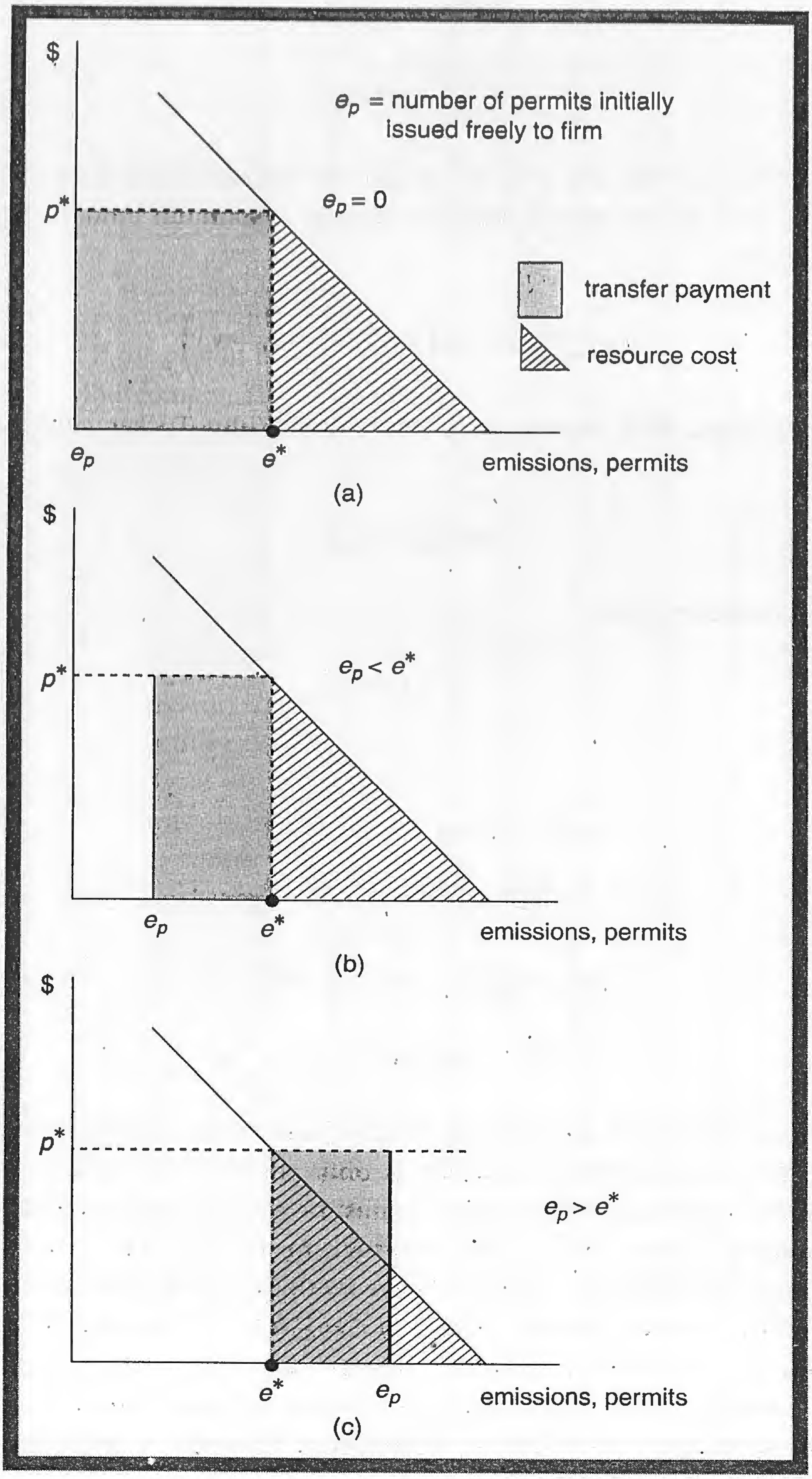

Figure 5.5 Permit revenues and expenditures

subject to

$$\alpha + \sum_i (e_{fi} - x_i) \leq \bar{A} \tag{5.18}$$

Constraint (5.18) says that the sum of background emissions plus firm emissions net of reductions must be no greater than the desired maximum amount. Forming the Lagrangian gives:

$$L = \sum_i C_i(x_i) + \lambda \left(\bar{A} - \alpha - \sum_i (e_{fi} - x_i) \right) \tag{5.19}$$

Then differentiating with respect to x_i yields the Kuhn–Tucker conditions for an optimum:

$$\delta C_i / \delta x_i - \lambda \geq 0$$

Or, using the notation above,

$$C_i' - \lambda \geq 0 \tag{5.20a}$$

and

$$x_i [C_i' - \lambda] = 0 \tag{5.20b}$$

$$\alpha + \sum_i (e_{fi} - x_i) \leq \bar{A} \tag{5.20c}$$

$$\lambda \left[\alpha + \sum_i (e_{fi} - x_i) - \bar{A} \right] = 0 \tag{5.20d}$$

$$x_i \geq 0; \qquad \lambda \geq 0; \quad i = 1, \ldots, n \tag{5.20e}$$

From the above we can see that λ is the shadow price of the pollution constraint, the same result we got in Section 5.2, which is only positive if the pollution constraint (Equation 5.18) is binding. All firms' MACs must be equal to this value, although some sources may have control costs that are too high for them to enter into the least-cost solution (so that, for this source, we would have $x = 0$). For a permit market to achieve this outcome, we need to issue a permit supply of $\bar{E} = \sum_i (e_{fi} - x_i)$, since this is the permitted level of emissions. Permits will then trade at a 1:1 rate between dischargers (i.e. if firm A reduces emissions by one unit and sells a one-unit permit to firm B, this allows firm B to increase its emission by one unit), giving what is known as an 'emissions permit system' (EPS) (Tietenberg, 1984).

Suppose each firm is given an initial allocation of e_i^0 permits, where $\sum_i e_i^0 = \bar{E}$, and that a price of p is initially set for permits. The representative firm's problem is now to:

$$\underset{(xi)}{\text{Min}} \; C_i(x_i) + p(e_{fi} - x_i - e_i^0) \tag{5.21}$$

that is, to minimise the sum of abatement costs and net spending on permits. The solution to this problem for the firm implies

$$C_i' - p \geq 0 \tag{5.22a}$$

$$x_i\,[C_i' - p] = 0 \tag{5.22b}$$

$$x_i \geq 0 \tag{5.22c}$$

Comparing these equations with (5.20a,b and e) we can see that the least cost solution will be replicated if the permit price p is equal to λ, which it will be if the permit market is competitive (Montgomery, 1972), such that all possible gains from trade are realised.

If non-uniformly mixing pollutants are to be managed using a TPP system, then trading in a permit market such as that above (i.e. under an EPS system) may result in violations in local ambient environmental quality standards, for instance if a high-impact source buys permits from a low-impact source. In this instance, even though the total of emissions is unchanging, environmental damages can increase. The possible solutions to this problem within TPPs are many, and designs such as the offset and ambient permit systems have been proposed. Indeed, many pollutants are non-uniformly mixed; for example, organic wastes discharged to a watercourse and sulphur dioxide discharged to the air. In this case, the control agency is interested in both the amount of discharges and their spatial distribution, since these two factors combine to determine the effect of the pollutant on ambient air or water quality at monitoring points. As will be recalled from above, *transfer coefficients* can be estimated which relate discharges at any point i to ambient air/water quality at some other point j. Admitting non-uniformly mixed pollutants changes the nature of the cost-minimisation problem, by changing the pollution constraint. Ambient pollution concentration at any point j is given by

$$A_j = \alpha_j + \sum_{i,j} d_{ij}\,(e_{fi} - x_i) \tag{5.23}$$

where α_j is pollution from other sources arriving at point j; and the d_{ij} terms are the transfer coefficients. The problem now is to obtain

$$\underset{(x_i)}{\text{Min}} \sum_i C_i\,(x_i)$$

subject to

$$\alpha_j + \sum_{i,j} d_{ij}\,(e_{fi} - x_i) \leq \bar{A}_j \qquad \text{for all } j$$

where $\bar{A}_j$ are the maximum allowable pollutant concentrations at each point j. Assuming for simplicity that all sources undertake some reduction in emissions (i.e. $x_i > 0$), then the Kuhn–Tucker condition of interest is

$$C_i'\,(x_i) - \sum_i d_{ij}\lambda_j = 0 \tag{5.24}$$

so that each source's MAC is equal to the weighted average of the shadow cost of emissions reductions needed to hit the targets. Put another way, there is now a shadow price (λ_j) at each monitoring point, so that we have got away from the simple 'equalise MACs' rule that was relevant in the uniform mixing case. This system of permits is known as an ambient permit system.

One design issue with permit markets, for either uniformly or non-uniformly mixed pollutants, is whether firms should be allowed to 'bank' emission reduction credits. For instance, a firm could decide to abate more than was required in the present period, earn credits and then bank these for use in a future period, when perhaps it thought abatement costs would be higher or permit prices higher. Allowing the banking of permits has been argued to be desirable since it can even out spikes in permit markets due to, say, sudden increases in the demand for electricity – as happened in California in 2000, which produced big increases in NO_x permit prices (Ellerman *et al.* 2003), can act as a hedge against uncertainty; and can encourage early reductions in emissions. However, regulators may worry that banking will result in violations in environmental standards in some time periods.

A second design issue relates to the possibility of allowing trade between point and non-point sources of pollution. For instance, both point sources (such as industrial plants and sewage treatment works) and non-point agricultural run-off are responsible for severe oxygen depletion in the northern Gulf of Mexico (Ribaudo *et al.* 2005). Allowing for trade between these two source types allows for cost-savings in pollution control, since marginal abatement costs for point sources were found to be typically greater than marginal abatement costs for non-point sources. Simulations showed that a net welfare gain of $46 billion was possible with such trading; although, interestingly, the modellers made the assumption of a 1:1 rate of trading between the (expected) reduction in agricultural-source pollution and each unit of reduction avoided for point sources within each of 21 districts included in the model.

5.4 Problems with pollution taxes

In Section 5.2, problems facing a regulator when a non-uniformly mixed pollutant is the environmental concern were noted. We now list some further problems with tax policies for the achievement of pollution reduction targets.

First, the pollution control agency must set the tax rate (or vector of rates for a non-uniformly mixed pollutant) at the appropriate level(s) to achieve the desired improvement in environmental quality. To get this exactly correct requires full information on abatement costs and transfer coefficients. As Baumol and Oates originally argued, agencies could iterate onto the correct tax rate (for a uniformly mixed pollutant) by setting a best-guess rate and then observing the consequent reduction in emissions. If this was too great, the tax rate should be reduced; if too little, then the tax rate should be increased. However, this neglects three problems: (1) setting an initially incorrect tax rate can lock firms into incorrect investments in pollution control equipment, preventing them from minimising costs (Walker and Storey, 1977); (2) setting an initial rate too low may result in irreversible, or reversible but serious, damage to the environment and (3) the aggregate MAC function is not stable through time. It will be changing in real terms owing to fluctuations in energy costs, input costs and product prices, and also in nominal terms as the result of inflation. Getting the tax rate correct may thus be a very tricky task.

A second problem concerns the issue of new entrants to a region. Suppose the major pollution problem in an estuary is emissions from oil refining. If new refineries are established in the area, then the aggregate MAC function will shift to the right, implying that, unless the tax rate is increased, aggregate emissions will increase. This is really just another aspect of the information problem discussed in the preceding paragraph.

A third issue relates to the case of pollution problems where the undesirable environmental effect is brought about by a number of pollutants. The problem for the environmental regulator is then to set the correct taxes across this 'basket' of pollutants to achieve the environmental target. Perhaps the best example is global warming, which is caused by a number of gases, carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O) and chlorofluorocarbons (CFC_{11} and CFC_{12}). The increased accumulation of these gases is, according to many global models, producing an increase in global mean temperature. A comprehensive account of economic analysis of the greenhouse effect is provided in Owen and Hanley (2004). Michaelis (1992) considers this problem from the point of view of how to design a comprehensive GHG tax system. The important question here is the level of efficient relative tax rates for the four main GHGs. Michaelis also considers the dynamics of this problem, in that there is not only a finite assimilative capacity in each time period for GHGs, but also a constraint on the total stock if undesirable warming is to be avoided. Each pollutant has a different contribution to global warming potential. The solution to this control problem implies that higher tax rates will be imposed on GHGs with higher α and lower η values, where α represents the relative warming potential of each gas, and where η represents the natural degradation rate of each gas in the atmosphere. Furthermore, for a given GHG the optimal tax rate evolves over time at a rate of $(1 + r/1 - \eta)$, where r is the rate of discount. Michaelis shows that absolute tax rates depend on the initial stock of GHGs, the time period over which the problem is considered, the absolute level of abatement costs and the initial period level of emissions.

Pollution taxes can be objected to on equity grounds. Pollution taxes might have undesirable re-distributive effects on households: for instance, a tax on energy consumption by households aimed at cutting CO_2 emissions might well hit poorer households harder than richer households, since the former tend to spend a higher proportion of their income on energy than the latter. For instance, the Danish CO_2 tax has been found to impose proportionately higher costs on poorer households in Denmark than on richer households (Wier *et al.* 2005). Box 5.3 gives more evidence on this issue. Firms could also raise objections to the equitability of taxes. Pezzey (1988) has argued that pollution taxes can over-penalise firms in terms of what is conventionally understood about the polluter pays principle (PPP). In Figure 5.6, a single polluter on a river is shown, in terms of the MAC schedule, and a marginal damage cost (MDC) schedule, which relates the amount of emissions to the monetary value of environmental damages caused by these emissions.

BOX 5.3

Distributional effects of economic instruments

Environmental taxes send signals to consumers by making consumption of environmental resources more expensive. However, there are concerns that their effect could be 'regressive', by hitting lower income households disproportionately. Research by Dresner and Ekins (2004a,b) investigated the possible impact on low-income households in four areas: domestic use of energy, water and transport, and domestic generation of waste. It also considered whether any negative impacts could be reduced if the tax or charge were designed appropriately, or if a compensation scheme were introduced. The study found the following:

- Low-income households' use of energy, water and waste disposal services and their use of cars where they own them, is disproportionate in relation to their income. This confirms that a flat-rate tax or charge applied to such usage would be regressive.
- For the average low-income household, the disproportionate impact could be removed through an appropriate (i.e. non-flat rate) design of the tax or charge scheme and/or by introducing a compensation scheme along with the tax or charge – although this would clearly have transactions costs associated with it, and could produce knock-on incentive effects.
- However, use of environmental resources tends to vary widely within a given income group. This means that, in practice, some low-income households would end up as net losers from any charging-plus-compensation scheme, even when the scheme leaves low-income households better off on average.
- It may be possible (e.g. with water use) to relieve this re-distributional burden through further special arrangements. Alternatively, it may be necessary to tackle the underlying cause of the hardship (such as energy-inefficient buildings) if pricing is to be used as an instrument of policy.

If MDC were known, then an (optimal) tax of t^* could be set, realising emissions of e^* if the firm is a cost minimiser which is fully informed as to its MAC schedule. However, the total financial burden to the firm (the shaded area, being the sum of abatement costs and tax payments) exceeds what Pezzey calls the conventional PPP and the 'extended PPP'. The conventional PPP is interpreted as meaning that firms should pay their own control costs up to the socially desired level of control e^*. The extended PPP adds to this burden the value of damages done by this socially desired level of emissions, the area under MDC up to e^*. That the financial burden to the firm under the tax of t^* exceeds both these amounts might be judged to be unfair. The size of transfer payments implied by a pollution tax policy has been argued to have been a major barrier to the acceptance of pollution taxes in the OECD. However, in principle this obstacle is surmountable at the aggregate level, since transfers could be returned to industry as lump sum payments (e.g. as capital grants for investment in pollution control). A tax system could also be

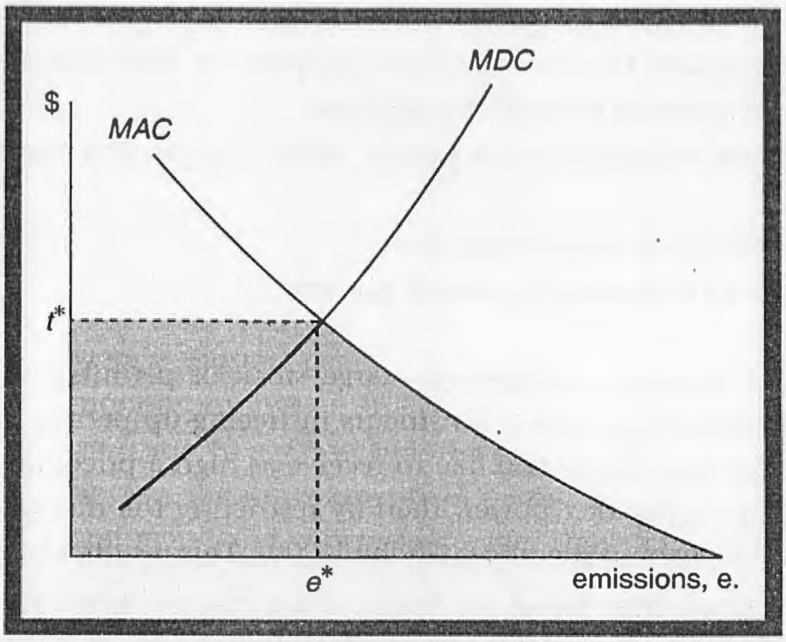

Figure 5.6 Financial burden of a pollution tax
Note: The shaded area equates to the financial burden.

introduced where firms were granted a tax-free baseline emissions level, and then only taxed on emissions above this baseline: this would clearly impose a lower financial burden on firms than taxing all of their current emissions (Pezzey, 2003).

Pollution taxes can also be criticised from the perspective of uncertainty over outcomes. A pollution tax will only achieve the environmental target if (1) all polluters are cost minimisers; (2) all are well informed about their MAC schedules and (3) no untaxed emissions are possible. Point (1) is important since, unless dischargers wish to minimise costs, they will not behave in the manner suggested by the models presented earlier in this chapter. Firms might emit at levels where MAC > t. Whilst the assumption of cost minimisation seems reasonable for single owner, partnership and equity-financed companies irrespective of market structure, it may not describe nationalised companies and municipal sewage treatment works. Point (2) is important since firms cannot make optimal cost-minimising adjustments to emission levels if they do not know their MAC functions. Finally, if firms can cheat, and escape paying taxes on emissions, then again the target reduction in pollution will not be achieved.

5.5 Problems with tradeable pollution permits

5.5.1 Market power

In the formal models in Section 5.3 we made the assumption that the permit markets being studied were perfectly competitive, in that each individual firm had no control over the market price. This seems a reasonable assumption where a large number of similarly sized traders operate in a market. However, if only a few firms are present, or if one of these firms is large enough to influence the permit price through its own buying and selling behaviour, then the least-cost property of TPPs may not hold, and the aggregate costs of the permit scheme can increase.

Why should a firm seek to influence the permit price? Two possible motivations are

1. to minimise its costs of compliance; and
2. to disadvantage its rivals in the product market.

Consider a firm which holds a relatively large stock of permits. This firm can earn revenues by selling permits; the costs it incurs in freeing-up permits for sale are given by its MAC. Clearly, the firm would like to receive as high a price for each permit as it can; if it has monopoly market power, then by restricting the number of permits sold on the market it can push up the price (Godby, 2000). This results in a welfare loss, with pollution by the dominant firm being too high, and pollution from the competitive fringe being too low. The extent to which a firm will choose to engage in such behaviour clearly depends on the price elasticity of demand for permits and the slope of the firm's MAC schedule, since the latter determines the price of freeing up permits for sale, whilst the former (which in turn depends on other firms' MAC schedules) dictates the degree to which the permit price will rise as the number of permits offered for sale decreases (Hahn, 1984; Misiolek and Elder, 1989). Alternatively, with monopsonistic power in the market, by buying fewer permits the firm can reduce the price it must pay for those permits it does purchase. Again, the cost of this price-setting behaviour is given by the firm's MAC schedule, the slope of which will influence the degree to which the firm engages in such behaviour. Summarising the above, we may say that in the monopoly case the market power firm spends too little on abatement, as it sells fewer permits than it would do in the competitive outcome, to keep permit prices high. Other firms spend too much on abatement. In the monopsony case, the market power firm spends too much on abatement and buys too few permits relative to the competitive case, to keep permit prices low. Hahn (1984) found that the initial allocation of permits affected both the post-trading outcome and the permit price, unlike the 'neutrality of initial allocation' result in a competitive market. However, Cason *et al.* (2003) argue that the extent to which market power affects the least-cost outcome depends also on *how* trading occurs: in their study, a continuous double auction (where buyers and sellers post sell/buy prices publicly, and can accept each others' offer at any time) resulted in only minor losses in market efficiency due to monopoly or duopoly, relative to the competitive equilibrium.

There is little empirical evidence from actual permit markets of the effects of price-setting behaviour. Godby (2000) and Stavins (1998) report no evidence of market power affecting outcomes in the US sulphur market, whilst the UK National Audit Office note that this was not a problem in the UK carbon trading scheme, since institutional rules prevented any one firm from holding more that 20% of all permits. Many studies have however pointed out the *potential* for such uncompetitive outcomes to emerge. OECD (2000) estimate that Russia and other former Soviet Union countries will command a large share of permits for sale under international carbon trading due to their 2005 emissions being well below their 1990 levels: this market power is estimated to reduce potential gains from trade under a European carbon trading scheme by 20%. At the national scale, Crampton and Kerr's (1999) simulation of a US CO_2 market predicted that no one firm would control more than 6% of the market. At a more local scale, Eheart *et al.* (1980) found that only two sources would control 80 per cent of all permits sold for phosphorus discharges into Lake Michigan.

Maloney and Yandle (1984) modelled price-setting behaviour through the establishment of cartels. For monopoly power, the increases in total control costs over the competitive base line were at most 41 per cent (with 90 per cent of the sources owned by the monopolist); for monopsony power, the greatest increase in total abatement costs was only 8 per cent, again at a 90 per cent holding for the monopsonist. However, even in the worst monopoly case, the (uncompetitive) permit market still achieved a 66 per cent saving over the command-and-control outcome. Hahn (1984) considers a permit market for particulate sulphates in the Los Angeles region, a market in which earlier work by the author had shown that one source (a power station) could be responsible for over 50 per cent of controllable emissions. In this case, the market clearing permit price varies from $3200/ton with monopsony, to the competitive price of $3900, to a price of $21,000/ton with full monopoly power. Cason *et al.* (2003) use experimental economics methods to simulate the effects on prices, trading and efficiency of both monopoly and duopoly in an emissions trading scheme for nitrogen pollution permits in Port Phillip Bay, Victoria, Australia. As noted above, a double-auction design is used, along with actual marginal abatement cost functions estimated for polluters in the Bay. The authors find rather small effects on cost-efficiency from monopoly or duopoly, and little impacts from changing the initial allocation of permits across sources – a rather encouraging finding for the development of TPP approaches to real-life pollution control problems where large firms dominate waterbodies, and/or where few traders are involved.

Moving onto the second of the two motivations set out above, Misiolek and Elder (1989) analyse the case where firms seek to raise the permit price so as to make entry to a product market less attractive for potential rivals. This may occur when actual or potential rivals must purchase permits in the same market as that of the firm wishing to take exclusionary action. Misiolek and Elder argue that such exclusionary action is most likely to be taken by large firms with relatively low MACs, to exclude smaller potential or actual entrants with higher MACs. They show that exclusionary action can increase both short-run and long-run profits for a firm. In a sense, exclusionary behaviour counteracts cost-minimising manipulation: we have seen above that the latter can lead to a firm with monopsony power buying too few permits; yet exclusionary behaviour will cause it to wish to buy too many permits. For a monopolist, however, whose cost-minimising manipulation involves selling fewer permits than in the competitive case, the effect of exclusionary manipulation is to worsen the distortion. As an interesting twist on this story, Carlen (2003) speculates that, in international permit trading of carbon dioxide, national governments could put pressures on firms not to trade permits with foreign competitors in the same industry, in order to protect the domestic industry from international competition.

5.5.2 Transactions costs

Transactions costs are the costs of trading: of finding someone to trade with, of negotiating and concluding this deal and then of clearing the deal (if needed) with the regulator. High transactions costs can form a barrier to permit trading, and thus prevent all possible cost savings from being realised (Stavins, 1995). Transactions cost can determine whether the initial allocation of permits across sources matters. Also of importance is whether transactions costs are increasing or decreasing with the volume of trade: for example, they could fall as the volume of trade rises due to learning-by-doing effects (Cason and Gangadharan, 2003). Uncertainty as to whether trades will be approved by regulators, for instance where trading rules are in place to handle non-uniform mixing problems, is also important in deciding the effects of transactions costs on the efficiency of a TPP system (Montero, 1997).

How serious is the problem of transactions costs in reality? Tietenberg (1990) has argued that the relatively small cost-savings achieved by a variety of permit trading schemes operating for air pollution in the USA prior to 1990, such as the 1977 offsetting scheme and the 1979 'bubble' scheme, was due to high transactions costs. High transactions costs have also been put forward as the main reason why the Fox River scheme in Wisconsin failed to realise any trades at all after it was introduced in the 1980s, despite prior simulation work which showed that very significant potential cost savings existed (O'Neil *et al.*, 1983). One practical solution to reducing transactions costs as a barrier to cost-minimisation is for the regulator to run an electronic 'bulletin board' service, where offers to trade are posted in terms of prices and quantities of permits: this is likely to be a feature of the new EU Emissions Trading System, which is described in Box 5.4.

BOX 5.4

The European Union's Emission Trading Scheme

As part of its programme of measures to achieve its Kyoto targets on reducing GHG emissions (see Chapter 6), the European Union introduced an Emissions Trading Scheme (ETS) for carbon dioxide with effect from January 2005. Those covered by the scheme will include the largest point source emitters of carbon dioxide, including electricity generators, oil refineries and the iron and steel industry. Some 12,000 'installations' are covered by the scheme. Individual member states published National Allocation Plans in 2004, which set out the initial allocations of permits to affected firms/sectors: these National Plans were then vetted by the European Commission. Interestingly, the ETS scheme came about partly because of severe political pressure from member states against a putative carbon tax being proposed by the European Commission.

The ETS scheme is a cap-and-trade scheme, in that, under the National Allocation Plan, each sector/firm is given a number of emission permits equal to its allowance or cap, for each period over which the scheme operates. In the first phase (2005–2007), these allowances are mainly being issued free, with 5% being retained for auction. Firms that cut emissions below their individual cap can then either sell the freed-up permits, or bank them for their own future use (although not all member states may allow banking). Similarly, firms who fail to reduce emissions to their cap can cover the difference by buying permits, either from emission reductions within the EU, or from emission reductions outside the EU which are sanctioned under the Flexible Mechanisms of the Kyoto protocol. A second phase of the scheme runs from 2008–2012.

Much criticism has been levelled at the ETS. For example, Vertedal and Svendson (2004) have criticised the implications for competitiveness and rent-seeking behaviour in the EU given the grandfathering method of allocation, whilst Boemare and Quirion (2002) note that problems may exist due to national non-compliance on the part of member states. For an excellent overview of the scheme, including the challenges it faces and comparisons with US schemes, see Kruger and Pizer (2004).

5.5.3 Trading rules and non-uniform mixing

In Section 5.3, two designs of permit system were mentioned: an emissions permit system (EPS) and the ambient permits system (APS). Under the former, permits are denominated in units of pollutant emitted (one permit permits one tonne of BOD, for example). Trades of permits between firms take place at a one-for-one rate. In other words, if source A sells one permit, it must reduce its emissions by the amount of emission covered by the permit. When source B buys this permit, it can increase its emissions by the same amount. Total emissions therefore do not increase. The EPS is a simple system, and for a uniformly mixed pollutant it may work well. For non-uniformly mixed pollutants, however, trades under an EPS could result in violations of ambient quality targets, since if source B is located in a more sensitive part of, say, a river, then its increase of x tonnes of emissions will do more damage than is avoided by A reducing its emissions by x tonnes.

To get around this problem, the APS was proposed. However, this has the problem that it is a very complicated market. Permits are denominated in units of damage at receptors. There is a separate market in permits at each receptor, and firms must trade in as many markets as their emissions affect receptors. For a pollutant such as sulphur dioxide, this could be a very large number of markets. Transaction costs would therefore be relatively high, whilst the number of traders in each market would be relatively low, giving rise to potential problems of imperfect competition (see above). What is more, total emissions can rise as a result of trading, which may cause knock-on environmental problems. If firm A sells permits which permit a reduction of 1 mg/l in dissolved oxygen at receptor point z, and if B's emissions have a relatively small impact on dissolved oxygen at point z, then B can increase its emissions by more than A reduces its own. Cost savings under the APS are to an extent realised by allowing a degradation of air or water quality down to the target level at receptors where, pre-trade, air/water quality is better than the target. An APS may also result in an increase in the long-range transport of pollutants (Atkinson and Tietenberg, 1987).

A variety of other trading rules have been proposed. All basically work on the principle of permits being denominated in units of emissions (one permit per tonne of BOD), but with rules governing trades in permits to stop the violation of ambient quality targets. The three best known of these trading rules systems are the pollution offset, the non-degradation offset and the modified pollution offset. The pollution offset system (Krupnick *et al.*, 1983) works by imposing a rule on trades that they may not violate the ambient quality target at any receptor point. However, this is consistent with worsening ambient quality up to the target level and an increase in total emissions. The non-degradation offset imposes the additional constraint that total emissions may not increase as a result of trades (Atkinson and Tietenberg, 1982). Finally, the modified offset (McGartland and Oates, 1985) allows trades so long as neither the pre-trade quality level nor the target level, whichever is the stricter (cleanest), is not violated. As Atkinson and Tietenberg (1987) point out, there is no general conclusion which can be drawn as to the relative cost-effectiveness of the modified and non-degradation offset systems (they rule out the simple offset system as being incompatible with environmental quality objectives). Comparisons must instead be made on a case-by-case basis. In their empirical analysis, they find the following for models of two US cities (St Louis and Cleveland) for the control of sulphur oxides (Cleveland) and particulate emissions (St Louis). In each case, the two offset systems are compared with the theoretically obtainable least-cost solution (which in this case would result from a perfect implementation of an APS) and with the command-and-control alternative of uniform design standards (denoted SIP, for state implementation plan). (SIPs were prepared by all US states in response to the Clean Air Act). Simulation results are set out in Table 5.1.

These results show the sensitivity of both the economic and the environmental outcomes of TPP systems to the trading used put in place to cope with spatial variability

Table 5.1 Abatement costs and emission reductions under different trading rules in Cleveland and St Louis, USA

	Cleveland – SO_2		St Louis – particulates	
Policy	*Total abatement costs ($m/yr)*	*Emission reductions, grammes/second*	*Total abatement costs ($/day)*	*Emission reductions, tons/day*
Least Cost	7.19	1328	82	13.49
SIP	11.18	1391	2314	23.50
Non-degradation offset	7.41	1391	116	23.50
Modified pollution offset	9.71	1440	190	22.24

in environmental impacts. Note, however, that both TPP systems are cheaper than regulation, represented by the SIP outcome.

It is worth mentioning that even where the damages done by emissions are dependent on their spatial location, TPP schemes may choose to ignore this fact, and opt for a simpler EPS design. For instance, this is what has happened under the Acid Rain trading programme in the US (see Box 5.5).

BOX 5.5

Sulphur trading in the USA

In 1990, the US Congress passed Clean Air Act Amendments which introduced a TPP system for the control of sulphur dioxide emissions from large point sources (primarily power stations). Stavins (1998) gives an excellent analysis of the political economy factors underlying the introduction of the scheme. The impetus for this measure came from Bush's campaign promises to take action on acid rain, and from support for the idea of TPPs to achieve this from the Environmental Defence Fund and members of the President's Council of Economic Advisors. Rising pollution control costs in the US gave a further impetus to the use of an economic instrument, rather than more regulation. What is more, great variations were known to exist in MACs across sources, implying large potential cost savings to be made from trading. Finally, there was no existing system for controlling acid rain causing emissions, so no status quo bias was present in the minds of regulators.

The system was intended to bring about a 10 million ton (50%) reduction in emissions of SO_2 from large stationary sources relative to 1980 levels. Most permits were 'grandfathered' (a great deal of time was spent in arguments over this allocation, both across regions and industry groups), although a small proportion were retained by the EPA for allocation to new sources, and for auction at the Chicago Board of Trade. As Joskow and Schmalensee (1998) point out, the decision to go with grandfathering rather than auctions is partly explainable by the greater control this gave politicians and administrators over the geographic distribution of financial burdens to firms. Compliance with the scheme is encouraged by a penalty of $2000/ton for unauthorised emissions.

Sulphur permits are denominated in annual tons of emissions, and can be banked. Permit prices fell from $131 to $95/ton during the first five years of trading (Ellerman *et al.*, 1999). While many trades have occurred, most have been internal rather than external, and high monitoring costs may have eroded the cost savings of the scheme. But the volume of trading has risen over the life of the scheme (e.g. from 130,000 sales in 1993 to 5.1 million in 1997), and large cost savings have still resulted, estimated by the General Accounting Office at $2 billion per year, and by Ellerman *et al.* as between 1/3 and 1/2 of the cost without trading. This cost saving is partly due to the phenomenon whereby the existence of trading possibilities has reduced prices of scrubbers, while fuel switching is also allowed. Estimates suggest that the benefits of the scheme have been considerably in excess of the costs (Burtraw, 1999).

The US also runs a 21-state NO_x trading scheme, whilst a regional air pollution trading scheme (RECLAIM) is in operation around Los Angeles.

5.5.4 Grandfathering or auctions?

As noted above, an EPA has two options for how it allocates permits: it can auction them or give them away freely (grandfathering). What does economics have to say about the relative merits of these two approaches? An auction will imply an additional financial cost to firms, namely the payments they initially make for their permits. These transfer payments leave the industry en bloc, unlike with grandfathering, which decreases the political attractiveness of auctions. For example, Lyon (1982) calculated that, for point source dischargers of phosphates to Lake Michigan, the total financial burden on firms (abatement costs plus permit purchases) was approximately three times the sum of abatement costs alone. Grandfathering has therefore been favoured in existing permit schemes, such as the US SO_2 trading scheme (Schmalensee *et al.*, 1998). The up-side of auctions is that the EPA will collect revenues which it could use for restoring environmental damage, or for subsidising improvements in pollution treatment capital. Revenues could also be used to allow reductions in non-environmental taxes, resulting in a possible 'double dividend' (Chapter 4): Goulder *et al.* (1997) estimate that the costs of SO_2 trading would have been 25% lower in the US if an auction system had been used rather than grandfathering, due to this double dividend effect. Revenues under auctions could indeed be substantial: Jensen and Rasmussen (2000) estimate Denmark could earn $200 million from auctioning carbon permits, whilst Crampton and Kerr (1999) suggest a similar scheme in the US could raise $126 billion annually. Grandfathering systems have been criticised in terms of discrimination against new entrants to an industry/area, who must pay for permits which existing firms were given for free. (Verterdal and Svendson, 2004). Grandfathering creates rents for those firms who receive permits: we could therefore expect resources to be wasted in rent-seeking behaviour by potential permit holders. Firms could also increase emissions in the run-up to a permit system in order to qualify for a higher number of permits; whilst in permit market where permits are re-issued in future time periods, a similar incentive exists to increase emissions over the cost-minimising level in order to be awarded more permits.

If an EPA does decide to auction permits, how should this be done? Lyon (1982) considers two alternative designs for an auction system. The first is the simplest design, a single price auction, whereby firms submit sealed bids for a specified number of permits. Permits are then sold to the highest bidder for 'a price that could represent either the lowest accepted bid or highest rejected bid' (p. 18). This mechanism is known to encourage strategic behaviour in that, if bidders believe that their bid could be the marginal bid, they benefit from understating their true WTP. The second alternative is an incentive-compatible Groves mechanism, whereby the highest bidders win the permits, but where a discharger's own bid never affects the price it pays. The Groves mechanism used by Lyon is an adaptation of the Vickrey second price auction, whereby the highest bidder pays the price bid by the second-highest bidder. Cason (1993) considered the rules used by the US EPA in their SO_2 permit auction (under this scheme, some permits are kept back for an annual auction, even though most are grandfathered – see Box 5.5).

Cason finds that EPA auction rules will encourage firms to understate their maximum WTP for permits, thus resulting in an inefficient level of permit purchases.

Finally, we note that a variety of 'mixed' allocation mechanisms could be used. For instance, the firm could be given an initial allocation equal to less than 100% of its current emissions, and then be invited to bid in an auction for any further permits it wishes to hold at the start of the scheme. Transfer payments by the firm would thus be lower than under a pure auction, but higher (on average) than under pure grandfathering (Pezzey, 1992, 2003).

5.5.5 Sequential trading

One possible explanation for why actual permit trading schemes have realised fewer cost savings than those predicted by economists lies with a comparison between the way the trading process has been modelled in simulation analysis and how trading actually occurs (Netusil and Braden, 2001). Many simulation studies assume that trading happens in a multilateral, simultaneous, fully informed manner, since this is the implication of representing the least-cost outcome from a mathematical programming model, which is achieved without violating environmental standards, as the trading outcome (see Box 5.6). However, actual trades are bilateral, sequential and often take place without traders being fully informed as to the minimum compensation demanded (supply price) and maximum WTP (willingness to pay) (demand price) of potential trading partners. Atkinson and Tietenberg (1991) consider the implications for the number of trades and the level of cost saving of this difference. In sequential trading under early US EPA rules, each trade was restricted from (a) violating ambient standards and (b) allowing an increase in emissions. This is much more restrictive than requiring the total of trades to meet these conditions.

BOX 5.6

Simulating a tradeable permit market for water pollution control in an estuary

There is relatively little empirical evidence concerning the ability of TPP systems to deliver actual cost savings with respect to water pollution control, since relatively little actual use has been made of them in practice. The infamous Fox River trading system was the subject of initial studies which suggested large cost savings from permit trading (O'Neil *et al.*, 1983). However, in practice, the trading scheme was so hamstrung by regulations that only one trade ever occurred. Most 'evidence' for cost savings from TPPs for water pollution control comes from simulation studies.

The Forth Estuary, in Central Scotland, is a tidal water body which is subject to many demands, including providing water for industrial cooling, for recreation, a habitat for birds and a sink for waste disposal. Most wastes come from industry, notably from a large petro-chemical complex and from a yeast factory. A seasonal 'sag' in Dissolved Oxygen (DO) in the upper estuary due to too much pollution has been noted in many summers: this has a bad effect on salmon migrating upstream. Control is currently exercised by the Scottish Environmental Protection Agency, who use performance standards ('consents') to regulate discharges of pollution from firms and sewage works. Hanley *et al.* (1998) report results from a simulation exercise to study the potential cost-savings from introducing a TPP system to improve DO levels. They found that such a system could generate very large cost savings over regulation, although these were reduced once uncertainty over water quality impacts was allowed for. For example, a TPP system could achieve a 20% improvement in DO in the most polluted part of the estuary at one-ninth of the cost of uniform regulation. This very large saving occurs because marginal abatement costs vary greatly over firms at the current level of control. Under uncertainty, the TPP system generates higher costs, but still achieves the target (in probabilistic terms) at a much lower cost than standards. These results were obtained by combining an economic model of polluters, based on abatement costs, with a water quality model which allows for firms located in different parts of the estuary to have different impacts on DO per unit of emission.

Atkinson and Tietenberg modelled a number of trading scenarios, using data from the St Louis area in the USA. Four scenarios were modelled which impose gradually more restrictive outcomes on the trading process. They were the following:

1. Simultaneous, full information: no increase in total emissions allowed.
2. Sequential, full information: first, a matrix M of possible cost savings from each pair-wise trade was identified. The biggest cost-saving trade was allowed, and the emission vector updated. These two traders were then eliminated before M was recalculated, and the next pair chosen. This process continued until all cost–saving trades were exhausted.
3. Partial information (a). The matrix M is not known. The firm with the lowest cost is chosen as first seller, then the best trading partner identified. These two firms are then eliminated and the process is repeated.
4. Partial information (b). The matrix M is not known. A firm is selected randomly as the first seller and its best trading partner identified. The elimination process then continues as in (3).

In all cases, the percentage of cost savings associated with the least cost solution were calculated. Results were as shown in Table 5.2, for two air quality standards, a primary standard and a stricter secondary standard.

As may be seen, fully informed but sequential trading incurs a large cost penalty over the hypothetical least-cost solution; this penalty is greater the stricter is the target

Table 5.2 Effects of sequential trading and information deficiencies on the realisation of all potential cost savings from trading pollution permits, Cleveland, USA

Scenario	*Percentage of least-costs savings realised, primary standard*	*Percentage of least-cost savings realised, secondary standard*
1. Simultaneous, full information	91	66
2. Sequential, full information	88	50
3. Partial information (a)	13	39
4. Partial information (b)	48	25

environmental improvement. Under full information, the best trades (those which save most resources) proceed first; but under partial information '... early sub-optimal trades reduce future opportunities (for cost-saving) considerably' (Atkinson and Tietenberg, 1991, p. 27). While the partial information outcomes are probably too pessimistic, since they ignore firms' abilities to find the best bargains going, they do point out the desirability of increasing information flows in the permit market: it is possible that the EPA could help here, by increasing the amount of information available to potential traders. The sequential scenario shows, more importantly, that not all of the cost savings available in the (hypothetical) least-cost outcome will be realised, given the way trading actually occurs.

However, economists are now beginning to cast doubt on the extent of cost penalties due to sequential trading if one allows for a dynamic market process (the Atkinson and Tietenberg results essentially comes from a static, one-shot formulation of the permit trading process). Ermoliev *et al.* (2000) showed that if the market structure allows for the price formation process between buyers and sellers to be separated in time from the process of finalising contracts, then a bilateral, sequential process could replicate the least-cost solution. An experimental study, simulating trading of global carbon permits, by Klaasen *et al.* (2005) confirmed this theoretical finding in a sense, since almost 96% of the potential cost savings were realised under a bilateral, sequential trading system over a two-and-a-half-hour trading period with repeated interactions between buyers and sellers and common knowledge on average trade volumes and prices. However, gains from trade were very differently distributed than in the least-cost, perfect competition outcome. The role of strategic behaviour, cheating and market power seem future issues to be explored in the context of sequential trading.

5.6 Taxes versus permits

Tradable Pollution Permits and pollution taxes are both capable, in theory, of achieving the least-cost solution to pollution control problems. How can we choose between these policy alternatives?

5.6.1 Innovation and cost-savings over time

One important advantage of economic instruments over design or performance standards concerns dynamic cost-savings (Requate, 2005). Suppose a firm could adopt a production process which had lower marginal abatement costs (MAC_{new}, in Figure 5.7) associated with it, relative to the firm's existing technology (MAC_{old}). Installing this cleaner technology incurs a cost, but benefits accrue in terms of abatement cost savings. These benefits can be shown to be greater under a pollution tax than under a uniform standard.

Under the uniform standard $\bar{e}$, the firm saves total abatement costs of area (xze^f) by switching. Under a tax set at t^1, the firm would find it cost-effective to reduce its emissions from e to e_t if it switched to the new technology. The firm, under the old technology, incurred control costs of ($\bar{e}xe^f$) and tax charges of ($ot^1x\bar{e}$). With MAC_{new}, control costs are ($e_t^1ye^f$) and tax payments total (ot^1ye_t) This produces net savings under the new technology of area (yxe^f) which exceeds the savings under the uniform standard e *and* results in lower emissions. A tax system, relative to a standard, would thus, over time, result in a progressive reduction in both abatement costs and emission levels.

But how do taxes compare on this score with tradable permits? Milliman and Prince (1989) argued that emission taxes provide higher incentives for firms to innovate into cleaner technologies, for the diffusion of these technologies and for pressure on regulators to then adjust environmental targets, than emission subsidies, or certain forms of tradeable permits or uniform standards. These three stages in dynamic adjustment are shown in Figure 5.8 (note that the horizontal axis shows emission reductions, rather than emissions, and refers to industry-wide emissions, rather than emissions from one firm).

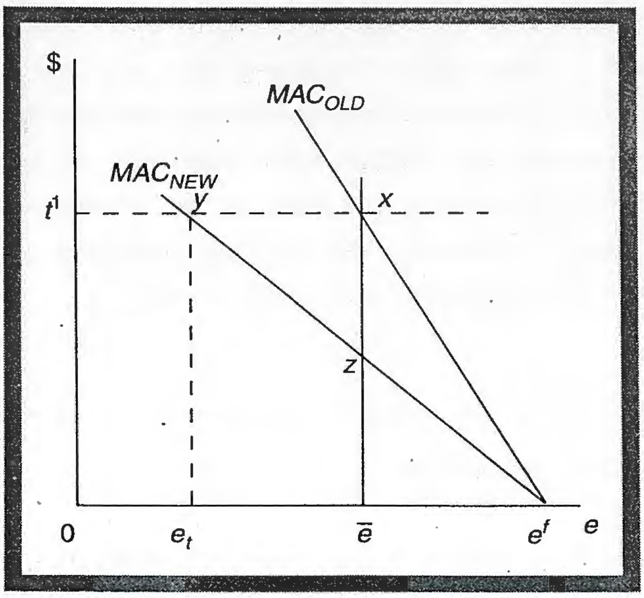

Figure 5.7 Savings under innovation with a pollution tax (Milliman and Prince, 1989)

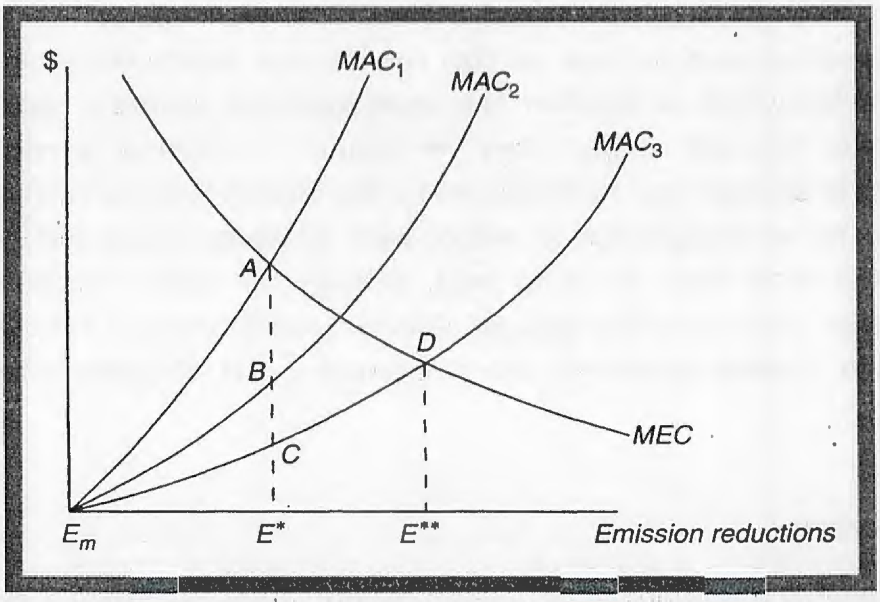

Figure 5.8 Savings under innovation, diffusion and regulatory response (Milliman and Prince, 1989)

Innovation, the process considered in the preceding paragraph, shifts the industry marginal abatement cost curve to MAC_2 from MAC_1. This produces savings equal to area (E_mAB). Diffusion of this cleaner technology produces a further fall in the industry MAC curve to MAC_3, and a further cost saving of (E_mBC). This changes the optimal level of emissions control to E^{**}, which, if the agency recognises this, further increases benefit by area (CAD). While tradeable permits, emission taxes and emission subsidies offer identical advantages over uniform standards in terms of incentives to promote innovation (the case we discussed in Figure 5.7), once diffusion is considered, emission taxes and auctioned permits emerge as the policy instruments most likely to maximise incentives, whilst allowing for agency response produces a preference for taxes. Milliman and Prince show that cost savings from innovation and diffusion of a cleaner technology will be greater under an auctioned TPP system than under grandfathering: they state that '... only emission taxes and auctioned permits clearly reward an industry innovator from the entire process of technological change' (p. 257), whilst '... the incentive mix from auctioned permits may be superior to that generated by taxes' (p. 261), once the response of the regulator is taken into account. A similar line of argument is presented by Jung *et al.*, 1996, allowing for heterogeneity across firms.

More recently, Requate and Unold (2003) show that these earlier analyses fail to distinguish between potential cost savings to the industry as a whole and the incentives facing individual firms in equilibrium. They note that individual firms can free-ride on the effects of others adopting cleaner technology. Taxes now provide a bigger incentive to innovate than permits, since if some firms innovate, this reduces permit prices (since the demand for permits is now lower following the shift down in MAC), which reduces incentives for other firms to innovate since the opportunity cost of not doing so is now lower. Moreover, no difference exists between auctioned and grandfathered permits in terms of incentives to innovate. Requate and Unold flag up the importance of another issue in this debate over which instrument encourages innovation best: that of whether the environmental regulator anticipates the new technology or not, and whether they pre-commit to a change in environmental policy (in terms of its type and its level) before the firms invest in cost-saving technology or not. The encouragement to reduce costs provided by market mechanisms has been referred to as 'over the long haul, perhaps the most important criterion on which to judge environmental policies' (Kneese and Schultze, 1978). Taxes seem to do better than tradable permits in this important aspect of environmental policy design.

5.6.2 Uncertainty

In Chapter 4, analysis was provided of how uncertainty over damage and control costs functions can complicate both the implementation of economic instruments and the choice between them. What can be said about this issue in the specific context of the relative merits of pollution taxes and tradable permits?

Pizer (1999, 2002) looks at this issue in the specific context of uncertainty over the costs of controlling GHG emissions. If there was no uncertainty over future control costs, both tradeable quantity controls and carbon taxes would produce similar outcomes. But considerable uncertainty does in fact exist, since (i) we have little experience with such large cuts in emissions, (ii) we do not know what future technological options will be and (iii) we do not know what the 'do nothing' level of emission will be, relative to which achievements are measured and targets set.

Pizer constructs a scenario in which there is uncertainty over control costs, which then turn out to be higher than thought. With a permit system, emissions are unchanged over the perfect information scenario, but costs of control rise. This stability of emission in the face of uncertainty is one reason why TPPs are indeed favoured over taxes by some environmental lobby groups (Stavins, 1998). Under a tax system, emissions are reduced by less, but the price per ton of carbon stays constant. Permit systems thus result in more uncertainty about costs than emission reductions, whilst this situation is reversed with taxes. Pizer then subjects these alternatives to a run of 1000 different predictions based around current IPCC calculations. He finds that emissions are below the full-information 8.5 GtC (Giga tonnes of Carbon) level in 75% of cases with the tax, but exceed it in the remaining cases. The permit market ensures that emissions never go above 8.5 Gt. However, the same simulation of possible future scenarios shows the costs of the permit scheme to be in the range of 0–2.2% of global GDP, a much larger range than that for the tax at 0.2–0.6% of global GDP. The variation of control costs is thus much greater under the permit system than with a tax.

Which policy option should we choose? Pizer argues that it depends on what we believe about damages. If there is some threshold beyond which further CO_2 emissions will impose very high (and maybe irreversible) costs, the greater certainty over emission levels that comes with permit markets is preferable. If instead damages rise smoothly with increasing emissions, the threats are not so bad, and we prefer the greater certainty over control costs which comes from taxes. This preference for taxes over permits is re-enforced when one remembers that it is not current emissions that most worry us about climate change, but the overall stock of GHGs in the atmosphere, which changes very slowly.

Pizer's more recent simulations reveal, by assuming the marginal benefits from GHG abatement schedule is relatively flat (which is what the literature tends to assume), that a carbon tax increases welfare by five times the value achieved under tradable permits. What is more, he finds that the original Wietzman result, noted in Chapter 4, must be changed for stock pollution problem like climate change, where costs depend on the current flow of pollution but damages (and thus the benefits of abatement) depend on the stock. Now the taxes-versus-TPPs argument is no longer just about the relative slopes of the marginal cost and benefits of control functions – with relatively flat marginal benefits favouring taxes – but must also take into account growth, depreciation, discounting and the correlation of shocks to the cost function over time.

Montero (1997) considers a rather different form of uncertainty, namely that over whether the regulator will actually approve a proposed permit trade. In his model, increasing uncertainty moves firms away from the overall least-cost solution. Firms may indeed be uncertain about many aspects of future permit trading, including for instance the effect of the number of permits they hold in the current round on the number they are grandfathered in subsequent rounds. The ability to be able to bank permits for future use is a key factor in determining the extent to which such uncertainty drives a wedge between the least-cost outcome and the actual trading outcome (Godby *et al.*, 1997).

5.7 Why don't governments make more use of economic instruments for pollution control?

A conclusion which might be drawn from both the theoretical and the empirical studies reported in this chapter is that economic instruments, such as taxes and TPPs, offer the possibility of considerable efficiency gains over 'command-and-control' approaches such as design and performance standards. However, as several recent surveys of actual pollution control policy in the OECD have shown, governments have made relatively little use of economic instruments as yet (OECD, 1997). Use of economic instruments has certainly increased over time: this is evident if one compares earlier OECD surveys (e.g. Opschoor and Vos, 1989) with more recent surveys, such as NCEE (2004; see Box 5.7). Major recent pieces of environmental legislation, such as the EU's Water Framework Directive, actively encourage the wider use of economic instruments; whilst in the UK the 1990s saw the introduction of several new economic instruments, such as a landfill tax and a tax on aggregates extraction. We have also provided examples of current uses of pollution taxes and tradable permits in both the main text of this and the previous chapter, and in the box sections.

BOX 5.7

Recent experience in the use of economic instruments to control pollution

A report by the National Centre for Environmental Economics reviews experiences outside the US with economic instruments for managing the environment, including air and water quality, water quantity, solid and hazardous wastes. A comparison was drawn with a previous EPA-funded survey in 1997. The main findings of the report were:

- *Direct fees and taxes* are the most used economic instruments internationally. Noteworthy trends include more applications and higher rates, as well as some acceptance in parts of the world where charges heretofore have been difficult to implement.
- *Pollution permit trading regimes* have gained greater acceptance worldwide, particularly for the control of GHG emissions. New applications of marketable permits for conventional pollutants in nations such as Chile, China and Slovakia are noted.
- *Greenhouse gas emission control* is an important and rapidly growing application of economic instruments. In 1997 just a handful of nations imposed carbon taxes. Now many more nations rely on carbon taxes and GHG trading regimes are in place.
- *Reductions in environmentally harmful subsidies* has been encouraged by international lending institutions. Leading lenders often make the elimination of environmentally harmful subsidies a condition for lending.
- *Liability* for harms caused to the environment is increasingly being used as a tool to limit polluting and environmentally damaging activities.
- *Information* is used in many new applications, including product labelling, categorising firms according to their environmental performance and disclosure of pollution releases.
- *Voluntary programs* now exist in a host of programmes to encourage firms to improve their environmental performance. Much greater attention is also being paid to rewards that can be offered in such programmes.

Overall, the report found a clear *increase* in the use of market mechanisms for environmental protection worldwide, which is good news for environmental economists!

Yet the impression remains that economic instruments are still used rather less than their theoretical advantages would seem to warrant. And even where economic instruments are brought in, this is sometimes with the major purpose of raising revenue, rather than changing behaviour (Barthold, 1994).

Why? This question has been investigated by many authors (Hahn, 1989; Cumberland, 1990; Hanley *et al.*, 1990; Barthhold, 1994; Keohane *et al.*, 1997; Stavins, 1998). An early explanation was that of ignorance on the part of policy-makers. Beckerman (1975) suggested that the main reason economic instruments were not used was that policy-makers were unaware of their potential. However, this is no longer true in the OECD. The UK government, for example, has recently published several documents supporting the concept of economic instruments (Department of the Environment, 1993), while economic instruments have been debated in the USA since the early 1970s (Nelson, 1987).

A second reason is practical problems. These include the use of either spatially differentiated taxes or complex trading rules for non-uniformly mixed pollutants (for instance, an ambient permit system has never been implemented, with the partial exception of the two-zone system used in the RECLAIM trading scheme in Los Angeles); interactions between regulated pollutants, stochastic influences on pollution emissions and resultant concentrations; and the way the regulator could reduce the supply of permits (or the level of pollution that a given supply permits) before the end of the permits' expiry dates. For many water pollutants in many physical settings, the potential number of traders in a TPP scheme is very small; whilst the financial transfers implicit in pure pollution tax schemes mitigate against their political acceptability.

A third possible reason is institutional problems. This is perhaps the most important category. The logic behind a preference for economic instruments is that regulators prefer more cost-effective policies to less cost-effective ones. Yet cost-effectiveness may be ranked low by regulators in a list of policy objectives – although the Water Framework Directive mentioned above places a specific duty on EU governments to design and implement cost-effective catchment management plans for water quality. A second institutional problem is connected with the ethical implications of economic instruments. Kelman (1981) has argued that pollution taxes, by putting a price on the right to pollute, somehow debase the notion of environmental quality: his survey of the US environmental lobby found that 68 per cent of those questioned took this view of pollution taxes (interestingly, the same survey found that 85 per cent of industrialists were opposed to pollution taxes, on the grounds that these increased the financial burden on firms relative to those imposed by the regulatory system). Tradeable permits could also be thought of as 'rights to pollute' and thus also subject to ethical opposition from environmental groups.

Fourth, some economic instruments could actually increase the financial burdens on firms, relative to command-and-control options, even though they minimise aggregate abatement costs from society's viewpoint. This is certainly very important in any analysis of why pollution taxes at incentive rates have been so little used (see Section 5.4), and is also an impediment to the introduction of auction-based tradeable permit schemes. Allowing firms some level of emissions over which they are not taxed, or using partial grandfathering schemes, are both ways around this problem. It is interesting to note in this regard that all sizable tradeable permit systems implemented to date are grandfathered rather than auctioned.

Finally, both pollution taxes and TPPs are significant changes in pollution control policy. Consequently, resistance can be expected from those with a vested interest in the preservation of the existing system, while bureaucracies in general may resist wholesale changes in policy. It is interesting in this regard that some of the best examples of economic instruments being introduced in the OECD are where changes to the legislation are brought in gradually, rather than as a dramatic change. In Germany, for example, charge levels for water pollution were gradually increased over the first four years of the scheme. A related tactic to increase firm approval is to introduce a voluntary scheme initially – this was the approach taken in the UK to introducing the concept of trading in CO_2 emissions across large point sources. It can also be the case that industry will lobby in favour of economic instruments where it sees these as a way of saving costs over regulatory alternatives.

As environmental standards become stricter, it is likely that the cost savings offered by economic instruments will become more attractive to policy-makers. Indeed, industry may make use of TPPs under its own initiatives, in order to reduce the costs of meeting expected future legislation. A good example of this is the in-house trading system implemented by BP in 1998, which covered GHG emissions from 150 BP units operating in 100 countries. In 2001, some 4.5 million tonnes of emission reductions were traded under this scheme worldwide (Akhurst *et al.*, 2003). However, the lesson from the last 20 years is that most changes come gradually, that cost-savings can be over-estimated by eager economists and that lobby groups will seek to retain regulation which is relatively favourable to their own interests. As Stavins (1998) notes, the main reason why command-and-control policies have dominated historically is '. . . because all of the main parties involved had reasons to favour them: firms, environmental advocacy groups, organised labour legislators and bureaucrats' (p. 72). Standards can improve firms' competitive positions by protecting them from new entrants; can be more popular with environmental lobby groups who find them easier environments in which to bring their influence to bear than taxes or tradable permits, and who worry about localised pollution problems emerging; are a more conducive working environment for regulators trained in law rather than economics; and serve to partly hide the cost of environmental standards from the public, unlike pollution taxes. Finally, Stavins notes that regulators care more about the geographic distribution of environmental costs and benefits than their total cost-benefit effectiveness, which is the main selling point of economic instruments.

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