ECONOMICS OF NATURAL RESOURCES AND THE ENVIRONMENT

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7 · ENVIRONMENTAL STANDARDS, TAXES AND SUBSIDIES

7.1 THE INEFFICIENCY OF STANDARD-SETTING

The most common form of pollution regulation is through the setting of environmental standards. Chapter 6 indicated reasons as to why taxes are not widespread and are treated with some suspicion by polluters. Standard-setting tends to imply the establishment of particular levels of environmental concentration for the pollutant, for example X micrograms per cubic metre, or a percentage of dissolved oxygen in water or a level of decibels that are not to be exceeded. Standards are most likely to be set with reference to some health-related criterion, for example a level of contaminants that must not be exceeded in order that water is safe for drinking, concentrations of sulphur dioxide and particulate matter that are consistent with the avoidance of respiratory illness, and so on.

The problem with standard-setting is that it is virtually only by accident that it will produce an economically efficient solution, i.e. it is unlikely to secure the optimal level of externality. To see this consider Figure 7.1 which repeats the familiar pollution diagram. A standard S is set and this corresponds to pollution level W_s and economic activity level Q_s. Setting standards also entails having some monitoring agency which oversees polluters' activity and which has the power to impose some penalty. If it has no powers of punishment the only incentive the polluter has to stay within the standard is some form of social conscience. Typically, then, standards are associated with penalties – polluters can be prosecuted or at least threatened with prosecution. In many countries actual legal cases against polluters are rare because the pollution inspectorate uses its powers to alter the polluter's behaviour before the case comes to court.

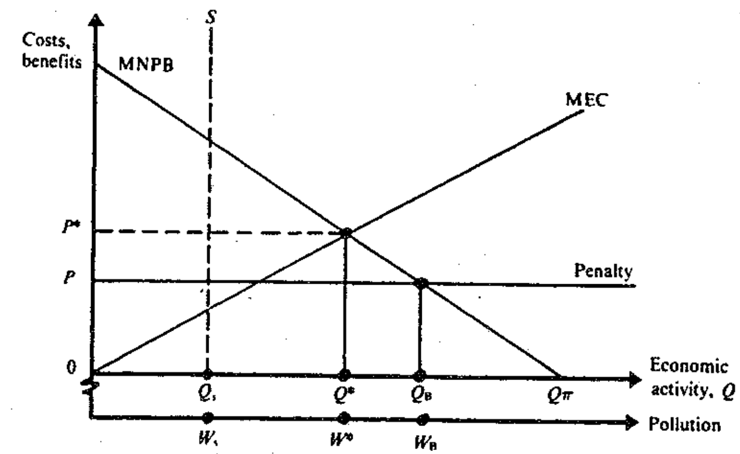


Figure 7.1 The inefficiency of standards.

Suppose the penalty in question is set at P in Figure 7.1. For the standard to work, then, the polluter must only pollute up to the maximum permitted level Q_s. It will be evident that Q_s is not optimal since it is less than Q^*. Indeed, unless the standard is set at Q^* it will not be optimal. The standard could coincide with the optimum provided the optimum was identifiable, a problem that is common to the Pigovian tax solution as well. So far, then, there is not much to choose between standards and taxes – both seem to require detailed information on the MNPB and MEC functons for an optimum to emerge.

But the penalty P also happens to be inefficient in this case. The polluter has an incentive to pollute up to Q_B. Why? He will do so because the total penalty up to Q_B is less than the net private benefits from polluting. He will not go beyond Q_B because further pollution attracts a penalty in excess of marginal net benefits. Strictly, we need to rephrase this finding in terms of the probability of the penalty being suffered. Remember, the polluter has to be caught by the pollution inspector and that is often difficult where, for example, there are many polluters in the area, each contributing a comparatively small amount to the total level of pollution. The calculation that the polluter does, therefore, is to compare the penalty *multiplied* by the probability of facing the penalty, with the net benefit of polluting. Even if the penalty is certain in Figure 7.1, it still pays to pollute up to Q_B.

This discussion should indicate quickly what the second broad requirement is for a standard to be optimal. It is that the penalty should be certain and that it should be equal to P^*. For the standard to be optimal we require that it be set in such a way that the output level corresponding to the standard is optimal, *and* the penalty level should be set equal to P^* and have 100 per cent certainty of being imposed for a transgression of Q^*.

The difficulties of securing these conditions explains why economists tend to be wary of standards.

7.2 TAXES VERSUS STANDARDS

The preceding section indicates a basic reason for preferring taxes to standards. Other considerations are also relevant and are discussed below.

Taxes as least-cost solutions
In Chapter 6 it has already been demonstrated that if a standard is to be adopted, a tax is the best way of achieving it. Clearly, this is not an issue of the superiority of taxes over standards, but a demonstration that a 'mix' of standards and taxes will, generally, be preferable to the adoption of standards alone.

Uncertainty and the benefit function
Figure 7.2 shows the basic pollution diagram but it is assumed that there is some uncertainty about the precise location of the benefit function. MNPB(true) shows the actual one and MNPB(false) the wrong one. The decision-maker assumes that MNPB(false) is the correct curve. Is the cost of his mistake bigger under a standard or a tax? So long as MEC and MNPB have the same (but opposite signed) slopes, the costs of being wrong are the same and there is no reason to prefer a tax to a standard. Thus, the tax t is set on the basis of trying to secure the optimal level of pollution assuming MNPB(false) is the correct curve. But MNPB(true) is the correct curve and hence the polluter, knowing this, goes to the point where MNPB(true) equals t. The effect is too much pollution (Q' instead of Q^*). The loss associated with the excess pollution is the area under MEC between Q^*Q' *minus* the area under MNPB(true) between Q^*Q'. This is shown as the triangle bde.

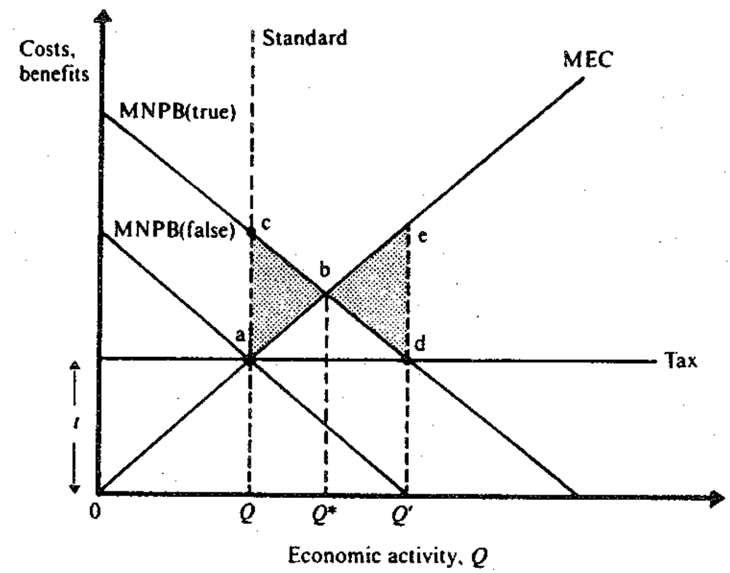


Figure 7.2 Equivalence of tax and standard.

Now assume the regulatory authority decides to set a standard, still believing in MNPB(false). The standard is set at Q. Provided the standards is rigidly enforced (but see Section 7.1), the level of activity is at Q, below the optimum Q^*, and with a loss of abc. It will be seen that the two shaded triangles are of equal size and hence there is nothing to choose between a tax and a rigidly enforced standard.

Figure 7.3 repeats the analysis but this time the two curves have different slopes. In case (a) the MEC curve is steeper than MNPB, and in case (b) it is less steep. Observation will show that in case (a) the tax solution produces a very much larger loss of welfare, i.e. the standard is to be preferred. In case (b) the standard produces the bigger loss – the tax is to be preferred. Notice that all these results hold just the same if it was the MEC function about which we are uncertain.

Clearly, the information requirements for making a rational choice between taxes and standards are quite formidable. Essentially, if the regulator does not know the *location* of MNPB but knows the *relationship between the slopes of MNPB and MEC* then he can make the right decision. But the regulator is very unlikely to know the relative slopes of the functions if he does not know even the scale of one of them.

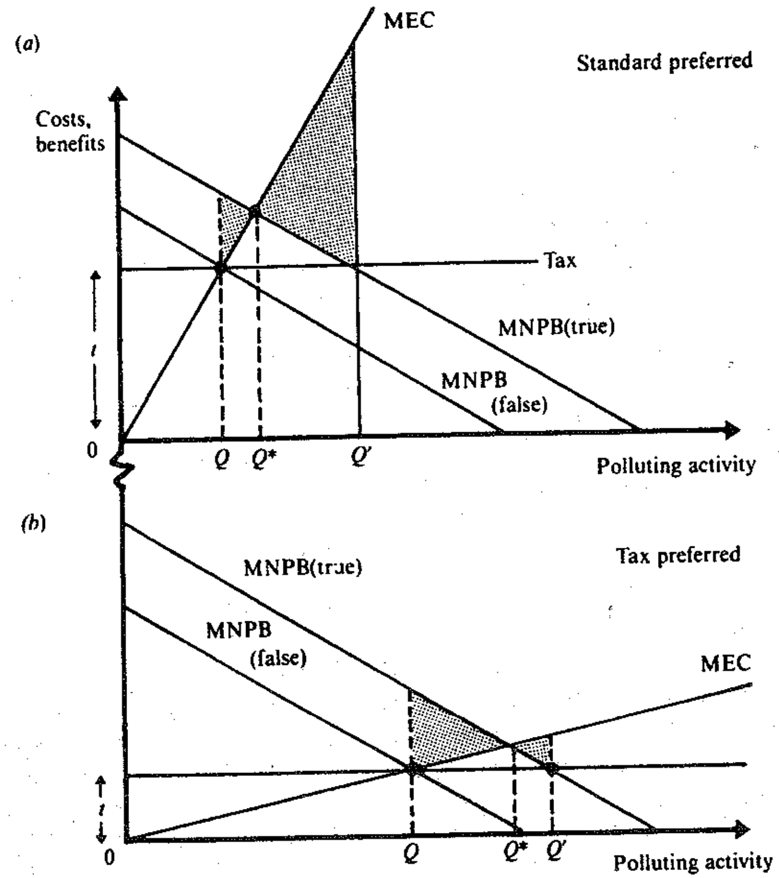


Figure 7.3 Standards versus taxes.

Dynamic efficiency
Taxes are superior to standards in one other respect. Inspection of Figure 7.1 shows that up to Q_s the polluter has no incentive to abate pollution. He faces no penalty for wastes emitted up to that point. But it may be socially desirable to encourage polluters to search continually for lower cost technologies for reducing pollution. Under the standard-setting approach this incentive does not exist. With a tax, however, the polluter still pays the tax on the optimal amount of pollution – recall the discussion in Chapter 6 – and hence has a continuing incentive to reduce pollution.

Administrative costs
The tax solution is certainly costly to implement. It is also open to legal wrangling if the tax is based on a measure of the economic value of damage which is disputed by the polluter. Since industry typically spends significant sums on challenging standards and regulation in general, it is not clear that this is a real criticism of the tax solution. The administrative costs of imposing the tax may also differ little from those involved in ensuring that standards are kept. In both cases monitoring is required. Standard-setting implies that a penalty system be in place and implementable. Taxes require that fees be collected. Some economists have argued that technology-specific controls are cheapest to administer, i.e. regulations of the form that a given technology must be used. Again, however, there must be monitoring and a penalty system for disobeying the requirement. Overall, it is far from clear that standards are cheaper to administer than taxes – only individual case studies will decide the issue.

Outright prohibition
There is one circumstance in which a tax is self-evidently inferior to a standard. This is where the pollutant is so damaging that an outright ban on its use is called for. In such circumstances we are effectively saying that the MEC curve is vertical – there are infinite marginal damage costs associated with the use of the pollutant. Alternatively, there is such uncertainty that we decide it is too risky to use the pollutant. This situation fits a number of ecotoxins and food additives. Clearly, there is no point in having a tax in these circumstances since the revenues would never be collectable.

7.3 POLLUTION REDUCTION SUBSIDIES

We have concentrated on regulatory mechanisms that use the 'stick' – a tax or a penalty for exceeding a standard. But why not approach the issue differently and encourage polluters to install abatement equipment by having a subsidy on the amount of pollution reduced? Like standards, subsidies are not popular with economists. It is important to understand the nature of a subsidy in this context. The idea is to give payments to firms who pollute below a certain prescribed level. Let the subsidy be S per unit of pollution, the

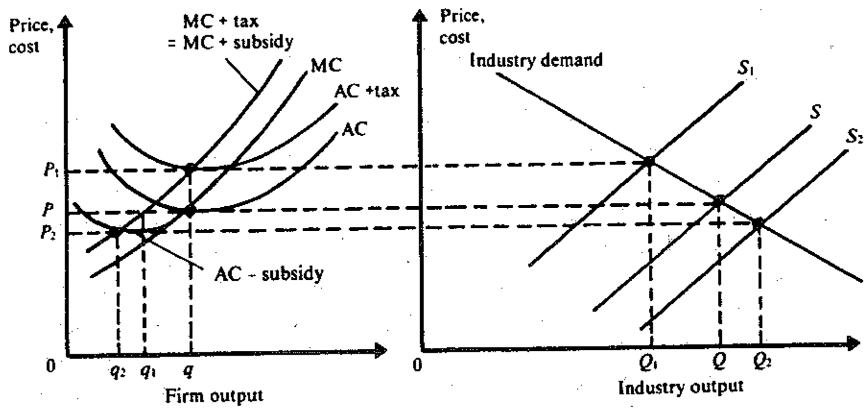


Figure 7.4 Taxes versus subsidies for individual firms (*left*) and the industry (*right*)

prescribed level be W and the actual level achieved by a polluter be M; M is below W. The subsidy payment is then

Subsidy $= S(W - M)$

Figure 7.4 illustrates what happens. The diagram shows the position of each individual *firm* on the left and the *industry* on the right. The distinction turns out to be important. The initial points are P,q for the firm, with price being equal to the lowest point on the average cost curve AC, and P,Q for the industry with aggregate supply curve S. Note that the P = AC condition means that we are considering an industry in which there is free exit and entry. First consider the effect of a tax. This will shift AC and MC upwards for the firm, bringing about a new *short-run* equilibrium where the ruling price, P, equals the new marginal cost at q_1 for the firm. But the ruling price is now below the new average cost so firms will exit the industry, shifting the industry supply curve to the left. A new long-run equilibrium is therefore P_1,Q_1 for the industry and P_1,q for the firm. This is fairly straightforward and as we would expect.

The effect of the subsidy is a little more difficult to analyse. This *raises* the firm's MC curve. If the subsidy is the same amount as the tax, the curve will shift to (MC + subsidy) which is the same as (MC + tax). This seems odd – surely subsidies will *lower* the MC curve? In this case this is not so and the formulation of the subsidy explains why. As the firm expands output, it *foregoes* a subsidy which it could

get by pollution reduction. Foregoing a subsidy is the same as paying a tax – there is a financial loss in each case. So MC shifts upward. But *average* cost falls for the firm since it gets a payment for lowering output. So, the MC curve for the firm becomes (MC + subsidy) which is the same as (MC + tax), but the AC curve for the firm falls to (AC – subsidy).

The short-run equilibrium is where price equals the new marginal cost, i.e. q_1, the same as with the tax. The short-run responses to the subsidy are therefore the same as those for the tax – there is no difference between them. The long-run response is very different however. In the short run, price now exceeds the new *average cost* (AC – subsidy) and hence new firms will enter the industry, shifting the supply curve to the right. A new long-run equilibrium occurs at P_2,Q_2, and P_2,q_2 for the individual firm.

What happens to pollution? The relevant comparison is what happens in the long run. Under the tax, industry output falls and hence pollution falls. Under the subsidy, however, industry output expands the pollution expands. Even though pollution per firm has fallen in Figure 7.4, the number of firms has increased. A subsidy, then, runs the risk of altering the exit and entry conditions into the polluting industry in such a way that, instead of reducing pollution, it may actually increase it.

8 · MARKETABLE POLLUTION PERMITS

8.1 THEORY OF MARKETABLE PERMITS

The idea of pollution permits was introduced by J.H. Dales (1968). As with standard-setting, the regulating authority allows only a certain level of pollutant emissions, and issues permits (also known as pollution 'consents' or certificates) for this amount. However, whereas standard-setting ends there, the pollution permits are tradeable – they can be bought and sold on a permit market.

Figure 8.1 illustrates the basic elements of marketable permits. MAC is the marginal abatement cost curve which, as Chapter 6 showed, can also be construed as the MNPB function if the only way of abating pollution is to reduce output. The horizontal axis shows the level of emissions and the number of permits: the easiest assumption to make is that one permit is needed for each unit of emission of pollution. The optimal number of permits is OQ^* and their optimal price is OP^*. That is, the authorities, if they seek a Pareto optimum, should issue OQ^* permits. S^* shows the supply curve of the permits: their issue is regulated and is assumed not to be responsive to price.

The MAC curve is in fact the demand curve for permits. At permit price P_1, for example, the polluter will buy OQ_1 permits. He does this because, in terms of control strategies, it is cheaper to abate pollution from Q_2 back to Q_1 than to buy permits. To the left of Q_1, however, it is cheaper to buy permits than to abate pollution. MAC is thus the demand curve for permits.

8.2 THE ADVANTAGES OF MARKETABLE PERMITS

Why do the permits have to be marketable? There are six main attractions of marketability.

1. Cost minimisation

Figure 8.2 repeats Figure 8.1, but omits the MEC curve. It also shows the overall MAC curve as being the sum of the individual polluter's MAC curves. We assume just two polluters for simplicity. This aggregation is legitimate because it was shown above that the MAC curve is the demand curve for permits: adding the curves up is therefore the same as aggregating any set of demand curves. By reference to the individual MAC curves of the two polluters we can see how many permits are purchased. Polluter 1 buys OQ_1 permits, and polluter 2 buys OQ_2 permits at price P^*. Note that the higher

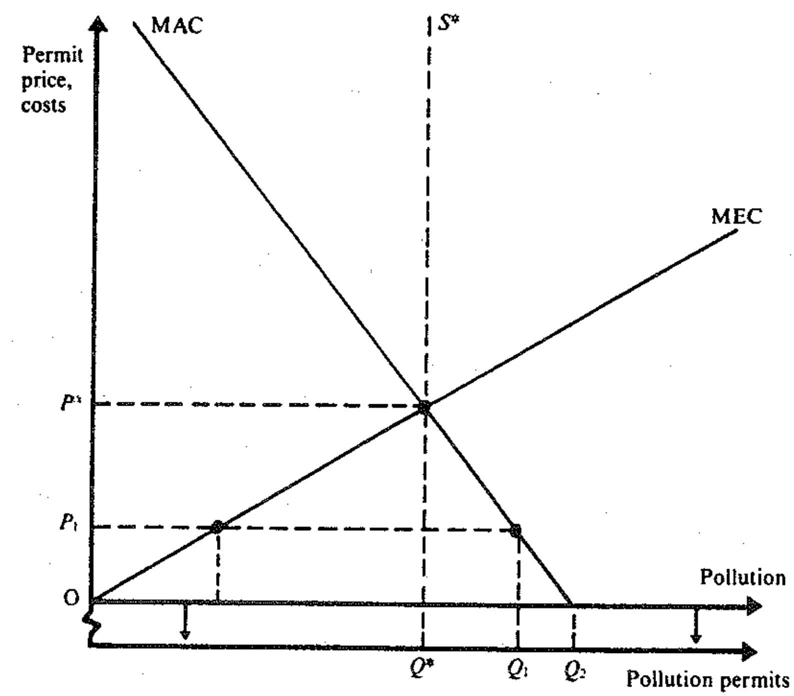


Figure 8.1 The basic analytics of marketable permits.

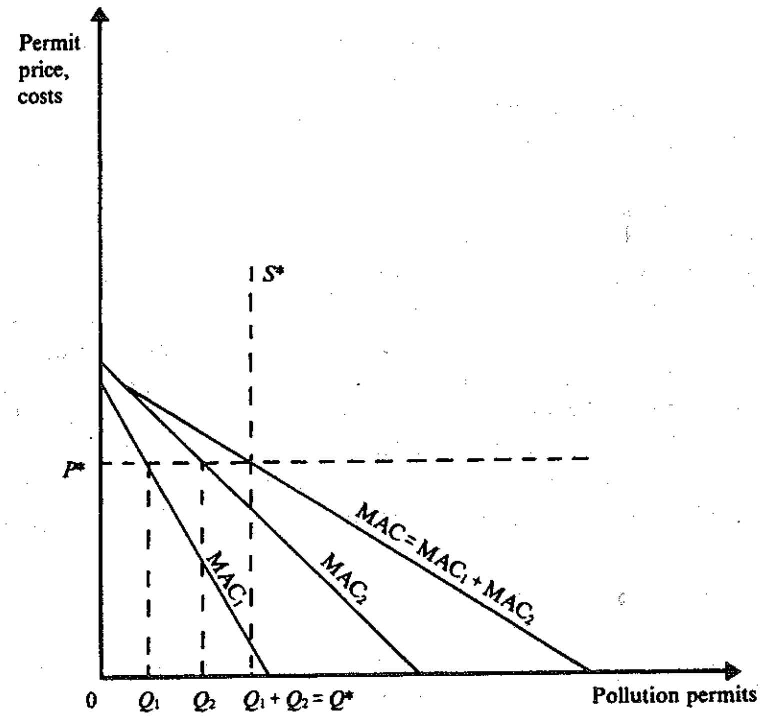


Figure 8.2 Cost minimisation with marketable permits.

cost polluter (2) buys more permits. This gives us a clue to the cost-effectiveness of permits. Polluters with low costs of abatement will find it relatively easier to abate pollution rather than buy permits. Polluters with higher costs of abatement will have a greater preference for buying permits than for abating pollution. Since polluters have different costs of abatement there is an automatic market - low-cost polluters selling permits and high-cost polluters buying them. By giving the polluters a chance to trade, the total cost of pollution abatement is minimised compared to the more direct regulatory approach of setting standards. Indeed, what we have is an analogue of the Baumol–Oates theorem about taxes being a minimum-cost way of achieving a standard (see Section 6.7).

2. New entrants

Suppose new polluters enter the industry. The effect will be to shift

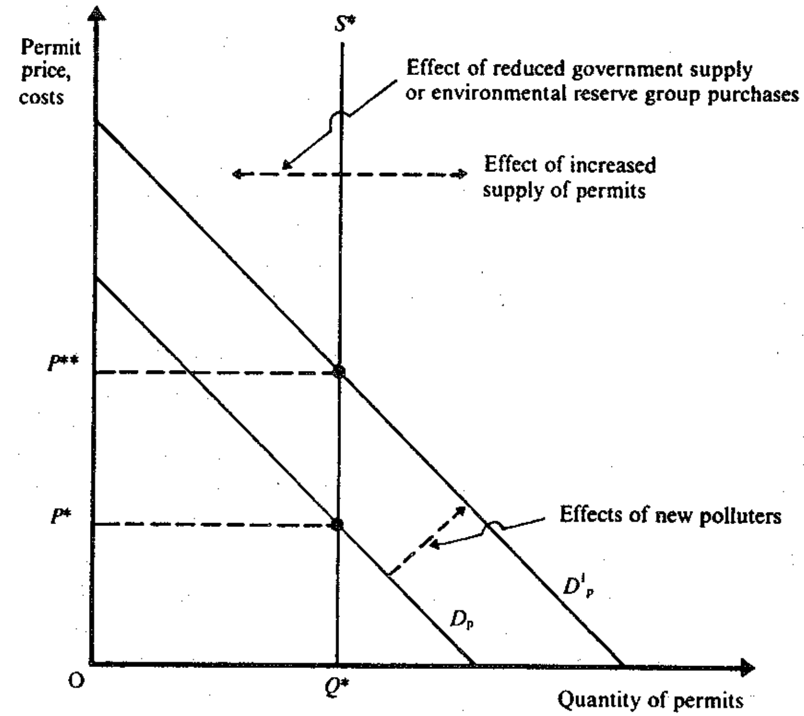


Figure 8.3 Changing the supply and demand for permits.

the aggregate pollution permit demand curve to the right, as in Figure 8.3. As long as the authorities wish to maintain the same level of pollution overall, they will keep supply at S^* and the permit price will rise to P^{**}. The new entrants will buy permits if they are high abatement cost industries, otherwise they will tend to invest in pollution control equipment. Once again, the overall cost minimisation properties of the permit system are maintained. But suppose the authorities felt that the increased demand for permits should result in some relaxation in the level of pollution control. Then they could simply issue some new permits, pushing the supply curve S^* to the right. Alternatively, if they felt that the old standard needed tightening they could enter the market themselves and buy some of the permits up, holding them out of the market. The supply curve would shift to the left. In short, the permit system opens up the possibility of varying standards with comparative ease to reflect the conditions of the day. The authority would simply engage in market operations, rather like a central bank buys and sells securities to influence their price.

3. *Opportunities for non-polluters*
Although it is not regarded as an intended feature of the permit system, there is another intriguing feature of them. If the market in permits is truly free, it will be open to anyone to buy them. An environmental pressure group, concerned to lower the overall level of pollution, could enter the market and buy the permits, holding them out of the market, or even destroying them. Such a solution would be efficient because it would reflect the intensity of preference for pollution control, as revealed by market willingness to pay. The danger with this idea is, of course, that a government might react adversely to a situation in which the level of pollution it had decided was optimal or acceptable was being altered by people who disagreed with it. They might simply issue new permits each time the environmental group bought the permits. In practice, the environmental group would lobby the government to issue only a small number of permits, so that environmental quality would not be undermined.

4. *Inflation and adjustment costs*
Permits are attractive because they avoid some of the problems of pollution taxes. As we saw in Chapter 6, even where a standard is set and taxes are used to achieve it, there are risks that the tax will be mis-estimated. With permits it is not necessary to find both the desirable standard and the relevant tax rate; it is necessary only to define the standard and find a mechanism for issuing permits. Moreover, if there is inflation in the economy, the real value of pollution taxes will change, possibly eroding their effectiveness. Because permits respond to supply and demand, inflation is already taken care of. Taxes also require adjustment because of entry to, and exit from, the industry. Permits, as we have seen, adjust readily to such changes, whereas taxes would require adjustment.

5. *The spatial dimension*
We have tended to assume that there are just a few polluters and that the points at which the pollution is received (the 'receptor points') are also few in number. In practice we are likely to have many emission

sources and many receptor points. If we are to set taxes with at least a broad relationship to damage done, it will be necessary to vary the taxes by source since different receptor points will have different assimilative capacities for pollution. Additionally, there are likely to be *synergistic* effects. That is, several pollutants may combine to produce aggregate damages larger than the sum of the damages from single pollutants. This raises the spectre of a highly complex and administratively burdensome system. To a considerable extent permits avoid this spatial problem. To investigate this further we need to look briefly at different types of permit systems.

6. *Technological 'lock-in'*
Permits are also argued to have an advantage over charges systems with respect to 'technological lock-in'. Abatement expenditures tend to be 'lumpy'; to increase the level of effluent removal, for example, it is frequently necessary to invest in an additional type of abatement process. Adjustments to changes in charges are therefore unlikely to be efficient unless the changes in the charge can be announced well in advance and can be backed by some assurance that a given charge level will be fairly stable over the short and medium term. The charge approach also risks underestimating abatement costs. For example, if the aim is to achieve a given standard, then, together with the regulating authority's assessment of abatement costs, this will determine the relevant charge. If the authority is wrong about the abatement costs, however, the charge could be set too low in the sense that polluters will prefer to pay it than to invest in abatement equipment, thus sacrificing the desired standard. This reluctance of polluters to invest in equipment will be strengthened by the previously discussed 'lumpiness' factor. A permit system generally avoids this problem of lumpy investment, the authority's uncertainty about abatement costs, and polluters' distrust of charges. This is so because the permits themselves are issued in *quantities* equal to the required standard, and it is prices that adjust. The consequences of an underestimate of abatement costs in the presence of permits is simply that the price of permits is forced up (since the demand for them is determined by abatement costs, as we saw), whereas the environmental standard is maintained (Rose-Ackerman, 1977).

8.3 TYPES OF PERMIT SYSTEMS

The literature has tended to classify three types of permit system. The *ambient permit system* (APS) works on the basis of permits defined according to exposure at the receptor point. Quality standards might vary according to the receptor point: there is no need for each receptor point to have the same ambient quality standard. Under an APS, then, permits have to be obtained from the market in permits at the receptor point. This means that the trade in permits will not be on a one-for-one basis; it will be necessary to trade on the basis of the number of permits required to allow a given amount of pollution concentration at the receptor point. Each polluter, then, may face quite complex markets - different permit markets according to different receptor points, and hence different prices.

The *emissions permit system* (EPS) is much simpler. It simply issues permits on the basis of source emissions and ignores what effects those emissions have on the receptor points. Within a given region or zone, then, the polluter would have only one market to deal with and one price, the price of a permit to emit pollutants in that area. Trade in permits is on a one-for-one basis.

The APS has obvious complications for the polluters and may well be an administrative nightmare for the regulators as well. The EPS is simpler but has other problems. By not discriminating according to receptor points it is unlikely to discriminate between sources on the basis of the damage done. It will therefore be inefficient. Put more formally, the price of permits will not approximate the marginal external cost. Second, any one area is likely to experience some concentration of pollution in specific small areas - so-called 'hot spots' - where actual concentrations exceed the standard. Because the EPS is emission-based across a wider area, it will not take account of this failure to observe the standard at all points. The simple technique of re-defining the area so that the hot spot is contained within a narrower zone to which the standard applies really amounts to turning the EPS into an APS, and we are back to the complexities of many markets and prices. The EPS also works on the basis of a one-for-one trade within the defined zone - there is no trade outside the zone. With the APS, however, all receptor points are taken into account. EPS could thus result in damage outside the zone being ignored.

To overcome these difficulties a third system has been proposed.

This is the *pollution offset* (PO) system. Under the PO system, the permits are defined in terms of emissions, trade takes place within a defined zone, but trade is not on a one-for-one basis. Moreover, the standard has to be met at all receptor points. The exchange value of the permits is then determined by the effects of the pollutants at the receptor points. The PO system thus combines characteristics of the EPS (permits are defined in terms of emissions, and there is no trade outside the defined area) and the APS (the rate of exchange between permits is defined by the ambient effects).

Which is the best system? Tietenberg (1985) has reviewed much of the evidence. His review suggests that EPS is more expensive than APS in terms of the total abatement costs likely to be involved. But the APS is also judged to be a largely unworkable system because of its complexity. How then does EPS fare in comparison to the more traditional standard-setting, or 'command-and-control' systems? The evidence is varied and is not easy to compare as the two systems might have different amounts of emission control because of difficulties in the spatial configuration of the requirements to meet the standard. The PO system was not evaluated.

8.4 PERMIT TRADING IN PRACTICE

There is some experience of pollution permit trading in the United States. The Clean Air Act (1970) established National Ambient Air Quality Standards (NAAQSs) which were to be implemented by the individual states under State Implementation Plans (SIPs). The Act marked the introduction of federal control, through the Environmental Protection Agency (EPA), over what had previously been a state responsibility alone. The SIP for each state had to indicate to EPA how the state would implement the ambient standards for all pollutants other than 'new sources' which were controlled directly by standard-setting by EPA.

In 1977 the Clean Air Act was amended to allow for the fact that many states were not meeting the ambient standards. Areas not meeting the standards were declared to be *non-attainment* regions. Stringent regulations were applied to these regions. All 'reasonably available control technologies' (RACTs) had to be applied to existing plant, and there had to be 'reasonable further progress' in achieving annual reductions so that the standard could be achieved. New sources were subject to construction permits which were conditional on the use of the 'lowest achievable emission rate' (LAER), the lowest emission rate demonstrated to have been achieved elsewhere. In the area where standards had been met, the focus switched to prevention of significant deterioration (PSD), i.e. to ensuring that the areas did not deteriorate.

The other main change in 1977 was the introduction of an emissions trading programme. Basically this operates through an *emission reduction credit*. Suppose a source controls emissions more than it is required to do under the standard set. Then it can secure a credit for the 'excess' reduction. The credit could then be traded in several ways. The first way is through a policy of *offsets*. These can be used in non-attainment areas, allowing new sources to be established, and which thus add to emissions, provided there is a credit somewhere else in the region. The new source effectively buys the credits from existing sources, the overall pollution level is not increased, and new industry is not unduly deterred from setting up in non-attainment regions that would otherwise suffer a loss of income and employment.

The second way is through a *bubble policy*. A 'bubble' is best thought of as an imaginary glass dome covering several different sources of pollution, either several points within one plant, or several different plants. The aim is not to let the overall emissions from the imaginary bubble exceed the level required by the standard-setting procedure. If any one point exceeds the RACT standard, for example, it can be compensated for by securing emission reduction credits elsewhere within the bubble.

The third procedure utilises *netting*. This is similar to the bubble, but relates to sources undergoing modification and which wish to avoid the rigours of being classified as a new source and subjected to the stricter standard (LAERs). Again, so long as plant-wide emissions do not increase, the modified source can increase emissions if there are emission reduction credits to offset the increase.

Lastly there is *banking* whereby sources can store up emission reduction credits for use later in a netting, bubble or offset context.

These components have a clear affinity with the permit trading systems discussed previously. The actual progress of these legislative features of the US policy is complex and varied. An overall evaluation of the policy is difficult, but several general observations stand out. First, trading has tended to result in better air quality,

although there are exceptions. Second, there appear to have been significant cost savings. Third, the offset policy probably has assisted regions which would otherwise have suffered economically because of firms being unable to set up in non-attainment regions. Fourth, administrative costs have been high. Fifth, it is probable that abatement technology introduction has been stimulated by the policy. By 1986 the total number of bubbles in existence was thought to be about 250; 3,000 offset transactions were reported. The amount of netting appears not to be known and banking has had a very limited impact.

9 · MEASURING ENVIRONMENTAL DAMAGE I: TOTAL ECONOMIC VALUE

9.1 THE MEANING OF ENVIRONMENTAL VALUATION

The preceding chapters have discussed alternative ways of correcting excessive pollution levels – letting a market in externality develop, taxes, standards and marketable permits. It was shown that some form of regulatory approach will generally be required – it is very unlikely that markets in externality will develop. The remaining instruments of regulation can be used in two sets of circumstances:

1. Situations where no attempt is made to identify the economically optimal level of pollution.
2. Situations where efforts are made to determine the optimum and then achieve it.

In the first case there is no requirement to measure the external cost curve (MEC). We determine a standard, perhaps on health-related criteria, and find the best way of achieving that standard. We saw that taxes and marketable permits had attractive characteristics in this respect. In the second case we have first to identify the optimum, or approximate it, and then set the standard or tax accordingly. As we saw, to do this we need also to know the private benefit function of the polluter (MNPB).

This provides the first justification for trying to *measure* environmental damage, i.e. to identify the MEC curve. It is important to recognise that the measurement in question is in *money* terms. If it was in any other units we could not identify the optimum because the MNPB curve (or the abatement cost curve, MAC) is measured in these units. For the purposes of this chapter, therefore, 'valuation' means money valuation.

The idea of putting a money value on damage done to the environment strikes many as illicit, even immoral. The justification for monetary valuation lies in the way in which money is used as a *measuring rod* to indicate gains and losses in utility or welfare. That is, money is the means of measurement. It must not be confused with more popular concepts about making money as an *objective* – crude greed, profit at the expense of others, the pusuit of Mammon. The reason money is used as the measuring rod is that all of us express our preferences every day in terms of these units – when buying goods we indicate our 'willingness to pay' (WTP) by exchanging money for the goods, and, in turn, our WTP must reflect our preferences. We might use any other units provided they can be applied meaningfully to both the benefit and cost sides of the pollution picture, and provided both reflect the preferences of individuals. Some attempts have been made to find other units – notably energy units – but, even if they can be applied to both sides of the picture, they have no meaning in terms of *preference* revelation. Accordingly, money units remain the best indicator we have. Environmental economists simply have to bear the burden of trying to explain what the use of money measures means, and what it does not mean. Misunderstanding is something we can reduce, but probably not eliminate.

Because money valuation relates back to individual preferences, it does however follow that any rejection of preference as the proper basis for decisions about the environment will entail rejection of the use of money values, or economic values as we shall call them. This is important. Many commentators on environmental economics observe that there is a multiplicity of values – we cannot subsume duty, obligation, keeping promises, love, and natural justice under economic values. What is more, each type of value has a different *moral standing* according to the viewpoint of the individual. Some see duty as the dominant moral rule; others see consistency (doing unto others only that which you would wish to see done to you, for example); still others see natural justice as the important rule. Chapter 15 discusses these profound issues in more detail. In this chapter we begin with the assumption that it is economic value that counts, although, as we shall see, the detection and measurement of those values seems to raise many of the concerns that the critics express about economic values.

Table 9.1 Pollution damage (in billions) in the Netherlands.

Pollution	Cumulative damage to 1985		Annual damage 1986	
	Dfl	US$	Dfl	US$
Air pollution	4.0–11.4	1.2–3.0	1.7–2.8	0.5–0.8
Water pollution	n.a.	n.a.	0.3–0.9	0.1–0.3
Noise nuisance	1.7	0.5	0.1	0.0
Total	5.7–13.0	1.7–3.5	2.1–3.8	0.6–1.1

Sources: (1) Netherlands Ministry of Public Housing, Physical Planning and Environmental Management, *Environmental Program of the Netherlands 1986–1990*, The Hague, 1985. (2) J. B. Opschoor, 'A Review of Monetary Estimates of Benefits of Environmental Improvements in the Netherlands', OECD Workshop on the Benefits of Environmental Policy and Decision-Making, Avignon, France, October 1986.

9.2 THE USES OF ECONOMIC VALUE

We have already identified a major use to which economic value measurements can be put: they should enable us to identify, or at least approximate, the optimum. We may wish to do this *ex ante*, i.e. before deciding on a type of environmental regulation. We may wish to do it *ex post*, i.e. after a regulation has been imposed, to see if the regulation has got us nearer to the optimum.

A separate use for economic value measurements is to demonstrate the importance of environmental policy. Many of the gains from environmental policy do not show up in the form of immediate monetary gain: the benefits are to be found more in the quality of life than in any increment to a nation's economic output. But it is essentially a historical accident that some gains in human welfare are recorded in monetary terms in the national accounts and others are not. By and large, this is explained by the fact that the accounts measure gains to economic sectors in which property rights – whether private or public – have been well defined. The third party effects of economic activity – noise, air pollution, water pollution, etc. – do not show up in the accounts either because the ill-defined or absent rights to clean air, peace and quiet and pure water mean that no monetary transfer takes place between polluter and polluted, or because such transfers as do take place (e.g. through court action) are not part of the national accounting conventions. Thus environmental

benefits tend to be less 'concrete', more 'soft' than market-place benefits. The temptation is to downgrade them by comparison.

We can view the widespread support for environmental policy as a reflection of the inappropriateness of this downgrading process. In reality, the environment is valued highly and one task in environmental policy is to record and measure these environmental values in whatever ways possible.

It is possible to illustrate the way in which benefit estimation techniques have been used to measure the importance of damage to the environment and, conversely, the benefits of environmental policy.

Table 9.1 shows estimates for the costs of environmental damage in the Netherlands. Note that these are damage estimates arising from pollution. A good many types of damage did not prove capable of 'monetisation', so that, if the monetised figures are accepted, actual damage exceeds the estimates shown. Various techniques were

Table 9.2 Pollution damage in the Federal Republic of Germany (1983–85)

Pollution	DM billion	US$ billion
Air pollution		
Health (respiratory disease)	2.3–5.8	0.8–1.9
Materials damage	2.3	0.8
Agriculture	0.2	0.1
Forestry losses	2.3–2.9	0.8–1.0
Forestry recreation	2.9–5.4	1.0–1.8
Forestry (other)	0.3–0.5	0.1–0.2
Disamenity	48.0	15.7
Water pollution		
Freshwater fishing	0.3	0.1
Ground water damage	9.0	2.9
Recreation	n.a.	n.a.
Noise		
Workplace noise	3.4	1.1
House price depreciation	30.0	9.8
Other	2.0	0.7
Total	103.0	33.9

Source: Adapted from data given in W. Schulz, 'A Survey on the Status of Research Concerning the Evaluation of Benefits of Environmental Policy in the Federal Republic of Germany', OECD Workshop on the Benefits of Environmental Policy and Decision Making, avignon, France, 1986.

used to derive the figures and considerable caution should be exercised in quoting or using them. They are, at best, 'ball park' numbers. Nonetheless, they show that even measured damage is a significant cost to the economy – the totals shown are 0.5–0.9 per cent of the Netherlands' GNP.

Table 9.2 presents similar estimates for the Federal Republic of Germany. Again, many items have not been valued and differing techniques are used to derive the estimates. The figures shown total to over 100 billion Deutschmarks annual damage (about US $34 billion), the major part of which is accounted for by the disamenity effects of air pollution (which is likely to include some of the separately listed air pollution costs), and the effects of noise nuisance on house values. The important point is that, if the estimates can be accepted as being broadly in the area of the true costs, pollution damage was costing an amount equal to 6 per cent of the Federal Republic of Germany's GNP in 1985.

Table 9.3 shows estimates for the USA for the year 1978. However, in this case the figures are for *damage avoided* by environmental policy. That is, taking the total of $26.5 billion, the argument is that, in the absence of environmental policy, pollution damage would

Table 9.3 The benefits of pollution control in the USA (1978).

Pollution	US$ billion
Air pollution	
Health	17.0
Soiling and cleaning	3.0
Vegetation	0.3
Materials	0.7
Property Values[a]	0.7
Water pollution[b]	
Recreational fishing	1.0
Boating	0.8
Swimming	0.5
Waterfowl hunting	0.1
Non-user benefits	0.6
Commercial fishing	0.4
Diversionary uses	1.4
Total	26.5

Source: A.M. Freeman, *Air and Water Pollution Control: A Benefit-Cost Assessment*, Wiley, New York, 1982.
[a] Net of property value changes thought to be included in other items.
[b] At one half the values estimated for 1985.

have been $26.5 billion higher in 1978 than it actually was. The total shown in Table 9.3 would be 1.25 per cent of GNP in 1978. The marked divergence between this figure and the percentage suggested for Germany is partly explained by the absence of estimates for noise nuisance, and by the very low figure for property value changes.

9.3 COSTS, BENEFITS, WILLINGNESS TO PAY AND WILLINGNESS TO ACCEPT

We have seen that an underlying purpose in attempting a monetary measure of the environment is to provide a check on the economic rationality of investing in environmental improvement. The cost of such improvements is measured in money terms and the monetary sum involved should approximate the value to society of the resource used up. Since resources are scarce it is important to establish that the gain from the policy exceeds the resource cost, and this can only be done by measuring the benefit in the same units as the costs. In fact expenditures should be undertaken until the extra *benefits* are just equal to the extra *costs*. In formal terms, *marginal benefit* should equal the *marginal cost* of providing that benefit. In turn, this equivalence meets the requirement that the scarce resources in the economy be used in their most efficient way, i.e. given a certain level of resources, the 'marginal benefit equals marginal cost' rule maximises the total net benefit that can be achieved with these resources.

As noted previously, it is important to understand that the concept of benefit is interpreted in a particular way. The basic idea is that 'what people want' – individuals' preferences – should be the basis of benefit measurement. The easiest way to identify these preferences is to see how people behave when presented with choices between goods and services. We can reasonably assume that a positive preference for something will show up in the form of a *willingness to pay* for it. In turn, each individual's willingness to pay will differ. Since we are interested in what is socially desirable, we can aggregate the individual willingness to pay to secure a total willingness to pay. The willingness-to-pay (WTP) concept thus gives an automatic monetary indicator of preferences. While we can safely assume that people will not be willing to pay for something they do not want, we cannot be sure that WTP as measured by market prices accurately

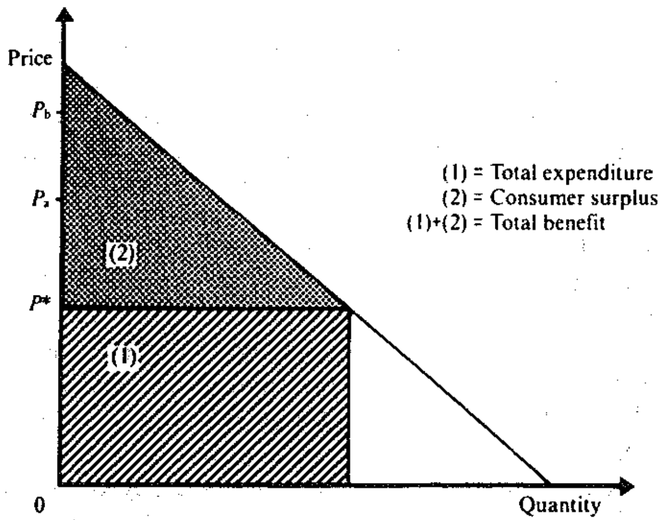


Figure 9.1 A demand curve for environmental goods.

measures the whole benefit to either individuals or society. The reason for this is that there may be individuals who are willing to pay *more* than the market price. If so, their benefit received is larger than market price indicates. The 'excess' that they obtain is known as *consumer surplus*.

Accordingly we can write the following fundamental rule:

Gross WTP = Market price + Consumer surplus

The idea can be illustrated with the aid of a diagram showing a demand curve. Figure 9.1 shows that the market price, determined by forces of supply and demand in this case, is $P*$. Since it is not possible to charge a different price to each and every individual buying the good, $P*$ becomes the market price for everyone. But individual A can be seen to be willing to pay a higher price: P_a. Similarly, individual B is willing to pay a price P_b. The total amount of benefit obtained is in fact the entire area under the demand curve shown by the two shaded areas. The shaded rectangle is the total expenditure by individuals on this particular good, and the shaded triangle is the consumer surplus. The two areas together then measure total benefit.

The intuitive basis to monetary benefit measurement is thus rather simple. People reveal their preferences for things they desire by

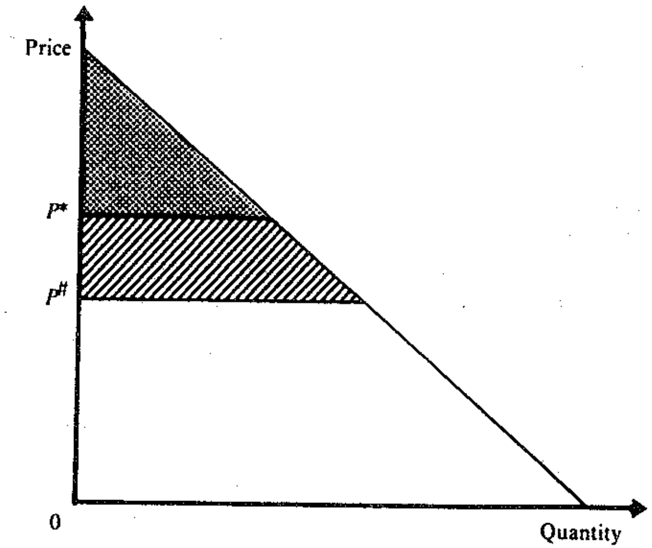


Figure 9.2 Changes in price and welfare gains.

showing their willingness to pay for them. Market price is our initial guide to what people are willing to pay and hence total expenditure on the good is our first approximation of benefit received. But since there will be people willing to pay more than the market price, and hence who secure a surplus of benefit over expenditure, *gross* WTP will exceed total expenditure. What we seek in benefit measurement, then, is a measure of areas under demand curves.

As it happens, the strict requirements for areas under demand curves to measure benefits is more complicated than this. Demand curves of the kind shown in Figure 9.1 have the same income level as we move up or down the demand curve. Along such demand curves, known as *Marshallian demand curves*, income is held constant. We require that individuals' welfare, well-being or 'utility' be held constant, which somehow means correcting the demand curve for the fact that utility varies as we move up and down the demand curve. Such adjustments have been worked out in the economics literature. Figure 9.2 shows the same demand curve as Figure 9.1 but this time $P*$ falls to $P^{\#}$ because of some change in the market. It will be evident that the price falls make the consumer 'better off' because the total shaded area (consumer surplus) has actually increased. The gain from the price fall is shown by the heavy shaded area.

Hypothetically, we can ask the consumer what he is willing to pay to secure the price fall so as to leave him as well off at $P\#$ as he was at P^*. This measure, based on the income and relative price pertaining to P^*, is known as the *compensating variation* measure of benefit. If instead we ask the consumer how much he would be willing to accept in order to forego the price fall, the relevant base point will be $P\#$. That is, the consumer will want a sum of money that will make him as well off as he would have been if the price fall has occurred, i.e. as well off as he would be at $P\#$. This sum, pertaining to the income and price levels at the subsequent position, is known as the *equivalent variation*.

Either the compensating variation or the equivalent variation is the technically correct measure of benefit. The compensating variation measure will be less than the area under the demand curve shown in Figure 9.2, which in turn will be less than the equivalent variation measure.

This digression into the technical basis of benefit measurement is important because it reveals that we have two basic concepts of benefit: one based on willingness to pay (WTP) and another based on willingness to accept (WTA). The theory of economics tells us that these ought not to differ very much but, as we shall see, some empirical studies suggest that there may be marked differences between the two. To obtain some idea of why this appears to happen consider the intuitive basis of the two measures. WTP has already been explained: individuals reveal their preferences for an environmental gain by their willingness to pay for it in the market place (we consider in a moment the fact that most environmental goods and services have no markets). But we are often faced with the problem of how we value an environmental *loss*. In that case we can ask how much people are willing to pay to prevent the loss or how much they are willing to accept in the way of compensation to put up with the loss. In short, there will be two measures of benefit gained from an environmental improvement and two measures of loss, or 'damage', from an environmental deterioration. The measures are:

1. WTP to secure a benefit.
2. WTA to forego a benefit.
3. WTP to prevent a loss.
4. WTA to tolerate a loss.

Why should these measures differ? Individuals appear to view losses differently to gains, a phenomenon that psychologists refer to as 'cognitive dissonance'. Given an initial position, they see an extra benefit as being worth so much, but a removal of some part of what they already have is seen differently, perhaps as containing some infringement against what they regard as being theirs 'by right'. Certainly, the phenomenon of asymmetry in the valuation of gains and losses in relation to some initial position is known to psychologists. They differentiate the benefit case from the loss case, referring to the former as having a 'purchase structure' and the latter as having a 'compensation structure'. How the values differ in the two contexts depends very much on what is considered by the individual as being the 'normal' state.

If WTA and WTP do differ significantly, then we have a problem for the measurement of environmental benefits, for many cases will involve the prevention of a loss rather than securing a benefit. It is likely then that the 'compensation structure' will be more important in these cases than the 'purchase structure'. A policy of preventing the loss may not be justifiable if the measure of benefit is based on WTP to prevent the loss, but justifiable if the benefit is measured as WTA compensation to tolerate the loss. It seems fair to say that this issue is not resolved in the environmental economics literature. Psychologists express little surprise that WTP and WTA are not the same; some economists find that they differ in many studies; others find that they may converge if the study is formulated in a particular way, and economic theorists tend to dispute that WTP and WTA can differ so much simply because the theory says that they ought not to differ (and hence there must be something wrong with the empirical studies).

9.4 TOTAL ECONOMIC VALUE

We are now in a position to explore the nature of the economic values embodied in the demand curve of Figure 9.1. While the terminology is still not agreed, environmental economists have gone some considerable way towards a taxonomy of economic values as they relate to natural environments. Interestingly, this taxonomy embraces some of the concerns of the environmentalist. It begins by distinguishing user values from 'intrinsic' values. User values, or user benefits, derive from the actual use of the environment. An angler, wildfowl hunter, fell walker, ornithologist, all use the natural environment and derive benefit from it. Those who like to view the countryside, directly or through other media such as photograph and film also 'use' the environment and secure benefit. The values so expressed are economic values in the sense we have defined. Slightly more complex are values expressed through *options* to use the environment, that is, the value of the environment as a potential benefit as opposed to actual present use value. Economists refer to this as *option value*. It is essentially an expression of preference, a willingness to pay, for the preservation of an environment against some probability that the individual will make use of it at a later date. Provided the uncertainty concerning future use is an uncertainty relating to the availability, or 'supply', of the environment, the theory tells us that this option value is *likely* to be positive (see below). In this way we obtain the first part of an overall equation for total economic value. This equation says:

Total user value = Actual use value + Option value

Intrinsic values present more problems. They suggest values which are in the real nature of the thing and unassociated with actual use, or even the option to use the thing. Chapter 1 drew attention to one meaning of 'intrinsic' value, namely a value that resides 'in' something *and that is unrelated to human beings altogether*. Put another way, if there were no humans, some people would argue that animals, habitats, etc. would still have 'intrinsic' value. We drew attention in Section 1.10 to a separate, but not wholly independent concept of intrinsic value, namely value that resides 'in' something but which is captured by people through their preferences in the form of non-use value. For the rest of this chapter it is this second definition of intrinsic value that we use. That is, values are taken to be entities that reflect people's preferences, but those values *include* concern for, sympathy with, respect for the rights or welfare of non-human beings and the values of which are unrelated to human use. The briefest introspection will confirm that there are such values. A great many people value the remaining stocks of blue, humpback and fin whales. Very few of those people value them in order to maintain the option of seeing them for themselves. What they value is the *existence* of the whales, a value unrelated to use although, to be sure, the vehicle by which they secure the knowledge for that value to exist may well be film or photograph or the recounted story. The example of the whales can be repeated many thousands of times for

other species, threatened or otherwise, and for whole ecosystems such as rainforests, wetlands, lakes, rivers, mountains, and so on.

These *existence values* are certainly fuzzy values. It is not very clear how they are best defined. They are not related to vicarious benefit, i.e. securing pleasure because others derive a use value. Vicarious benefit belongs in the class of option values, in this case a willingness to pay to preserve the environment for the benefit of others. Nor are existence values what the literature calls *bequest values*, a willingness to pay to preserve the environment for the benefit of our children and grandchildren. That motive also belongs with option value. Note that if the bequest is for our immediate descendants we shall be fairly confident at guessing the nature of their preferences. If we extend the bequest motive to future generations in general, as many environmentalists would urge us to, we face the difficulty of not knowing their preferences. This kind of uncertainty is different to the uncertainty about availability of the environment in the future which made option value positive. Assuming it is legitimate to include the preferences of as yet unborn individuals, uncertainty about future preferences could make option value negative. Provisionally we state that:

Intrinsic value = Existence value

where, for now, existence values relate to values expressed by individuals such that those values are unrelated to use of the environment, or future use by the valuer or the valuer on behalf of some future person.

In this way we can write our formula for total economic value as:

Total economic value = Actual use value + Option value + Existence value

Within this equation we might also state that:

Option value = Value in use (by the individual) + Value in use by future individuals (decendant and future generations) + value in use by others (vicarious value to the individual)

The context in which we tend to look for total economic values should also not be forgotten. In many of those contexts three important features are present. The first is *irreversibility*. If the asset in question is not preserved it is likely to be eliminated with little or no chance of regeneration. The second is *uncertainty*: the future is not known, and hence there are potential costs if the asset is eliminated and a future choice is foregone. A dominant form of such uncertainty is our ignorance about how ecosystems work: in sacrificing one asset we do not know what else we are likely to lose. The third feature is *uniqueness*. Some empirical attempts to measure existence values tend to relate to endangered species and unique scenic views. Economic theory tells us that this combination of attributes will dictate preferences which err on the cautious side of exploitation. That is, preservation will be relatively more favoured in comparison to development.

There is no particular agreement on the nature of the equation for total economic value. Some writers regard intrinsic value as part of existence value rather than as its equivalent. Others regard intrinsic value as being inclusive of option value. To a considerable extent the variations in definition appear to relate to what is meant by 'use'. Thus if it means actual current use by the individual expressing the preference, bequest values are not use values. The view taken here, however, is that the issue of when use occurs and by whom cannot be regarded as differentiating characteristics: all uses, whenever they occur and whoever they are by, give rise to use values. Equally, all use values are conceptually distinct from the intrinsic value of the environment which we currently equate with existence value. It is clear that the concepts of option and existence value need further investigation.

9.5 OPTION VALUE

The willingness to pay for an environmental good, e.g. wildlife preservation, a national park, improved water or air quality, is related to the consumer surplus that the individual expects to receive from that good. We saw that gross WTP was made up of the intended expenditure on the good plus the consumer surplus (CS). The benefit to the individual will therefore be the excess WTP over what is actually paid out, since the latter is the cost to the individual. This excess is CS. Since decisions are made on the basis of what is *expected*, we can say that the relevant CS is *expected* CS, which we write as E(CS).

If we are sure of our capability of buying the good, and of our future preferences, and of the availability of the good when we want

it, E(CS) is a proper measure of the benefit of the good. It is this that we would wish to put into our cost–benefit assessment. If it costs an amount C to preserve a wildlife habitat, for example, we can say that it is worth preserving it if $C < E(CS)$. However, the idea that we are certain of both the factors influencing our *demand* for the wildlife habitat, and the factors influencing its *supply* is not realistic. On the demand side we might be unsure of our income and unsure of our preferences in the future. On the supply side, we may be unsure that the habitat will be there for us to enjoy. It is this presence of *uncertainty* that requires us to modify the use of E(CS) as our measure of benefit.

We can illustrate the required modification by considering supply uncertainty. This is very relevant in the real world because natural environments are everywhere being reduced in size and number. We cannot be sure that a given environment will be available to us in the future. The basic idea is that, given this supply uncertainty, and given the fact that most people do not like risk and uncertainty (they are said to be *risk averse*), an individual will be willing to pay *more* than the expected CS in order to ensure that he or she can make use of the environment later on. The total WTP is called *option price* (OP) and it comprises the expected consumer surplus *plus* 'option value' where option value (OV) is the extra payment to ensure future availability of the wildlife habitat; that is:

Option price = Expected consumer surplus + Option value

or

$$OP = E(CS) + OV$$

On this basis, simply estimating future use of the wildlife habitat will give us only E(CS) and will ignore OV. We will have underestimated the true value of the habitat.

Once different attitudes to risk are introduced and the uncertainty is extended to the 'demand side', we cannot be sure OV is positive. Indeed, even with supply side uncertainty, there is ambiguity over the sign of OV. The analytical basis for these judgements is complex (see the notes on further reading for this chapter) but the general outcome is as shown in Table 9.4, although the reader is warned that the signs shown for supply uncertainty require certain technical assumptions to be fulfilled.

A further source of value is *quasi option value*. Imagine a development that threatens to destroy the wildlife habitat we have

Table 9.4

	Sign of OV		
	Risk loving	Risk neutral	Risk averse
Demand uncertainty			
Income	+ve	0	–ve
Preferences	?	?	?
Supply uncertainty	–ve	0	+ve

been hypothesising. The development has a certain value in terms of people's willingness to pay for its outcome. An illustration might be a tropical forest which contains a rich range of diverse species which may have future value for scientific and commercial purposes. Many experts argue this, for example, with respect to plant species for pharmaceuticals and for crop breeding. There are uncertain benefits from the preservation of the habitat, but these benefits could become more certain through time as information grows about the uses to which the forest species can be put. But if the development takes place, this source of genetic information is lost for ever. Quasi option value (QOV) is the value of preserving options for future use given some expectation of the growth of knowledge. If QOV is positive it would tend to support the view that the development should be postponed in order to make a better decision later.

The literature suggests that if the expected growth of information is independent of the developments, i.e. we do not need the development to generate the information, then QOV will always be *positive*. If, on the other hand, the information depends on the development, QOV could be positive or negative: positive when the uncertainty is about the benefits of preservation, and negative when the uncertainty is about the benefits of the development. It seems fair to say that the types of information growth in question in the real world are *not* related to development. Hence the presumption must be that QOV is always positive.

9.6 EXISTENCE VALUE

Existence value is a value placed on an environmental good and which is *unrelated to any actual or potential use of the good*. At first sight this may seem an odd category of economic value for, surely, value derives from use? To see how existence values can be positive consider the many environmental funds and organisations in existence to protect endangered species. The subject of these campaigns could be a readily identifiable and used habitat near to the person supporting the campaign. It is very often a remote environment, however, so much so that it is not realistic to expect the campaigner to use it now, or even in the future. Nonetheless, many people support campaigns to protect tropical forests, to ban the hunting of whales, to protect giant pandas, rhinoceros, and so on. All are consumable vicariously through film and television, but vicarious demand cannot explain the substantial support for such campaigns and activities. This type of value, unrelated to use, is existence value.

Existence value provides one of the building bridges between economists and environmentalists, for it is not readily explained by the conventional motives. Economists have suggested a number of motives, all of which reduce to some form of *altruism* – caring for other people or other beings:

1. *Bequest* motives relate to the idea of willing a supply of natural environments to one's heirs or to future generations in general. It is no different to passing on accumulated personal assets. As noted above, however, we prefer to see bequest motives as part of a *use* value, the user being the heir or future generation. It is possible, of course, to think of a bequest as relating to the satisfaction that we believe will be given to future generations from the mere existence of the asset, but the very notion of bequest tends to imply that the inheritor makes some use of the asset.
2. *Gift* motives are very similar but the object of the gift tends to be a current person – a friend, say, or a relative. Once again, gift motives are more likely to be for use by the recipient. We do not therefore count the gift motive as explaining existence value – it is one more use value based on altruism.
3. *Sympathy* for people or animals. This motive is more relevant to existence value. Sympathy for animals tends to vary by culture and nation, but in a great many nations it is the norm, not the exception. It is consistent with this motive that we are willing to pay to preserve habitats out of sympathy for the sentient beings, including humans, that occupy them.

Much of the literature on existence value stops here. The reason for this is that altruistic motives are familiar to economists. They make economic analysis more complex but, by and large, altruism can be conveniently subsumed in the traditional model of rational economic behaviour. In terms of the idea that individuals maximise utility, or welfare, what we can say is that altruism gives utility to the giver, and the giver's utility depends on the utility of other people, or other beings. This interpretation fits neatly into the rational economic man concept, and avoids facing up to still other motives that may be relevant to explaining existence value.

What might these other motives be? One suggestion is that non-human beings have rights, and that when people express an existence value unrelated to their own or anyone else's use of the environment, they are, as it were, voicing those rights because the beings in question cannot do so. But if this is a motive for existence value, then it appears to cause problems for the model of rational economic man, or so some economists fear. It means, for example, that actions may be motivated by factors other than maximising utility. In turn, this means that we will not be able to explain the world (completely anyway) in terms of utility maximisation. Nor, if the rights of others have superior moral standing over utility maximisation, can we prescribe policy on the basis of maximising utility (or benefits). Given the powerful superstructure that economists have built up on the basis of utility maximisation, it is very understandable that they should be unwilling to sacrifice its generality. But the idea that behaviour often *is* motivated by the respect for the rights of others is hardly surprising. It is a fact of life. Why, then, is it any more odd to think of valuations reflecting the rights of other beings? We are, after all, used to the idea that we can pursue our own pleasure only within limits set by society, limits that attempt to embody rights.

The issue here may then be one of deciding when it is, and is not, proper to take account of existence values. If the aim of society is to allocate resources so as to maximise, as far as possible, the utility of individuals in society, then it will be correct to take account of existence value if it is altruistically based. If, on the other hand, existence value relates to a rights motive, and we do not wish such motives to be relevant to the design of policy, it will be improper to take account of it. The reader must decide for himself or herself. For the record, we see no inconsistency in taking account of existence value – whatever its basis – because the values in question are of

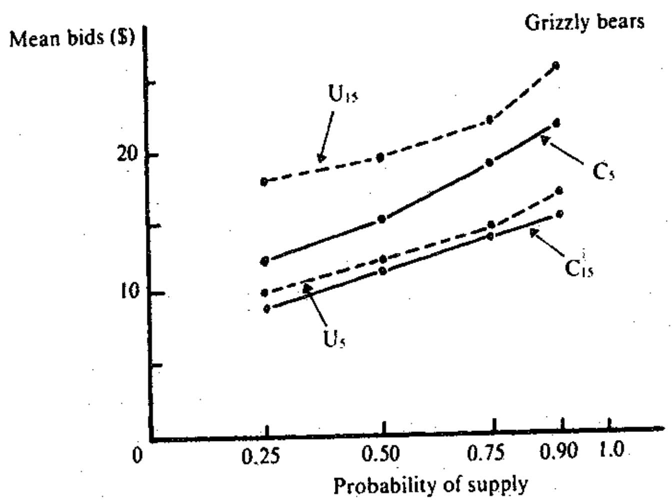
people and because social policy typically does reflect both wants and rights.

A second motive for existence value unrelated to altruism is *stewardship*. We might also refer to this motive as *Gaian*, after the Greek goddess of the earth, Gaia. A Gaian motive might be based on the idea that the Earth is something far greater and more important than the multitudes of people it supports, and that its population has a responsibility to see that it survives. The implication, of course, is that individual wants may have to be sacrificed to some greater good but, again, we should not be surprised at this idea. Families engage in such activities frequently. There exists also a modern Gaian movement based on a scientific hypothesis that the Earth is a living organism which adjusts in a self-regulating manner to external shocks. An ironic twist to the Gaian motive for existence value, however, is that, in this view, humans are rather unimportant in the self-regulation.

9.7 EMPIRICAL MEASURES OF OPTION AND EXISTENCE VALUE

It is possible to secure empirical estimates of option and existence value by the use of procedures which adopt a questionnaire approach to the WTP for benefits. This approach, the *contingent valuation* approach, is described in Chapter 10. In this section we report several studies which have attempted to obtain actual measures.

David Brookshire, Larry Eubanks and Alan Randall (1983) measured the *option price* (option value plus expected consumer surplus) and *existence value* of grizzly bears and bighorn sheep in Wyoming, both species being subject to threats to their existence. By asking hunters for their WTP in a context where the probability of there being adequate supplies of these species was variable, the authors were able to uncover different types of economic value. A hunter who was certain of his own intentions nonetheless faced uncertain supply. The pattern of bids is shown in Figure 9.3. The U refers to respondents who were uncertain if they would hunt, the C to respondents who were certain they would. This captures an element of demand uncertainty. The subscripts 5 and 15 refer to the number of years before a programme of protection would permit the hunting to take place, the programme being hypothetically paid for by the licences for which the respondents were bidding.



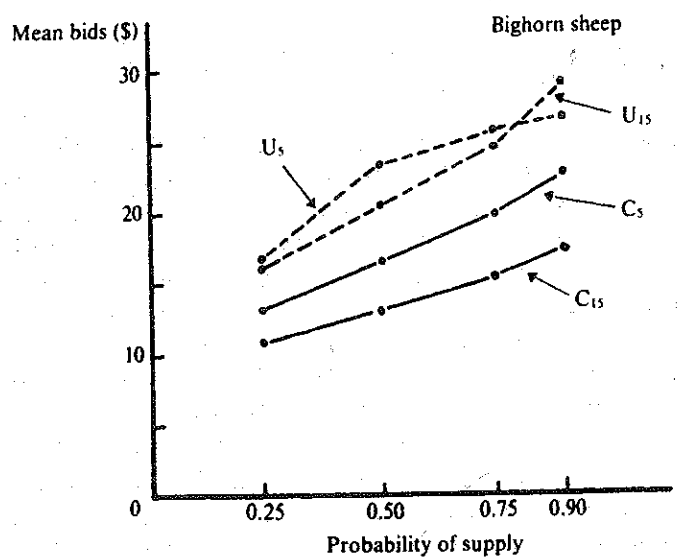


Figure 9.3 Mean grizzly and bighorn bids for certain (C) and uncertain (U) hunting demands over alternative time horizons (5 and 15 years). (*Source*: D. Brookshire, L. Eubanks and A. Randall, 'Estimating option prices and existence values for wildlife resources', *Land Economics*, 59(1), February 1983.)

The overall option price should increase as the probability of supply increases. This was the result predicted by the theory and it is seen to hold in this case. One might also expect the bids based on certain demand to exceed those based on uncertain demand, but the diagrams show that there is no systematic relationship. Respondents who indicated they would never hunt the bears or sheep were asked what they would nonetheless pay to preserve the species. They were further divided into *observers* (a form of use value) and *non-observers* ('pure' existence value). The results provided estimates of 'observer option price', i.e. the option price associated with keeping the species for recreational observation, and existence value. The results are shown in Table 9.5. Clearly, these are significant sums. To see this compare them to the average option prices for hunting under, say, 90 per cent probability of future supply. For grizzly bears and the five-year time horizon the sequence would be $21.50 option hunting price compared to $21.80 option observer price and $24.00 for existence value. Average existence value is on a par with the bids to maintain the population for hunting and observation.

In a later paper, Brookshire *et al.* (1985) detail findings relating to the Grand Canyon. By looking at the bids made by respondents to experience improved visibility (regardless of whether visits take place or not), the authors find that the total 'preservation bid' for the Grand Canyon's visibility was $4.43 per month, compared to a 'user bid' of $0.07 per month. Interpreting existence value as the difference between total preservation value and use value, the finding is thus that existence value dominates preservation in this case. Existence value stands in the ratio of 66:1 to user value (note that what is being preserved is visibility, not the site itself). The explanation for such a

Table 9.5 Existence values and option prices for grizzly bears and bighorn sheep.

	Bears		Sheep	
	5 years	15 years	5 years	15 years
Average observer option price ($)	21.8	21.0	23.0	18.0
Average existence value	24.0	15.2	7.4	6.9

Source: Brookshire *et al.* (1983).

large ratio is that the resource in question is unique – it has no substitutes. Where substitutes exist one would expect existence values to be lower, and this tends to be the picture in other studies on existence value.

Jon Strand (1981) reports a CVM-type study of acid rain for Norway. After indicating the nature of the environmental problem – damage to freshwater fish from acid rain – respondents were given a starting point figure for the global cost of stopping acid pollution which was translated into a special income tax. They were then asked if they were willing to pay this sum. The approach was thus of the 'take-it-or-leave-it' kind rather than one involving iterative bids in which respondents could vary their bid according to different levels of clean-up. But the hypothetical tax rates were varied across the four samples of respondents interviewed, i.e. the tax rate was the same for each sample but varied between samples. The 'yes' responses were found for the lower taxes. Strand then estimates 'bid curves' using this information in a conditional probability framework, i.e. estimating the probability that a respondent would pay a particular tax given a certain income. Strand estimates that the average bid was 800 Norwegian krone per capita. Given a population of 3.1 million, this translates to a 'national' annual benefit of 2.5 billion krone. Earlier work by Strand suggests that user values are about 1 billion krone, so that subtracting this from the implied total preservation value of 2.5 billion krone gives an existence value of 1.5 billion krone. In 1982 terms this translates to some $270 million or about 1 per cent of the Norwegian GNP. Note that, by asking for WTP, the Strand study probably underestimates the true value of benefits of reduced aquatic acidification. The reason for this is that a good deal of the acidity arises from 'imported' pollution and respondents will generally have been aware of this. Accordingly, they may well have had the attitude that others besides themselves should pay for the clean-up.

10 · MEASURING ENVIRONMENTAL DAMAGE II: VALUATION METHODOLOGIES

10.1 TOTAL ECONOMIC VALUE AND DECISION-MAKING

Chapter 9 showed that the relevant concept when measuring the benefit of an environmental improvement is total economic value (TEV). In the same way, if we wished to measure the damage done to the environment, say by a development project, we would want to calculate the TEV that is lost by the development. Damage and benefit are obverse sides of the same concept.

The relevant comparison when looking at a decision on a development project is between the cost of the project, the benefit of the project, and the TEV that is lost by the development. More formally, we can write the basic rules as:

(i) proceed with the development if

$$(B_D - C_D - B_P) > 0$$

and

(ii) do not develop if

$$(B_D - C_D - B_P) < 0$$

where B_D refers to the benefits of development, C_D refers to the costs of the development and B_P refers to the benefits of preserving the environment by not developing the area.

TEV is in fact a measure of B_P, the total value of the asset left as a natural environment. The benefits and costs of the development will be relatively simple to measure, primarily because they are likely to be in the form of marketed inputs and outputs which have observable prices. This is clearly not going to be the case with TEV, so we need now to investigate ways in which we can measure the component parts of TEV.

10.2 DIRECT AND INDIRECT VALUATION

The approaches to the economic measurement of environmental benefits have been broadly classified as *direct* and *indirect* techniques. The former considers environmental gains – an improved scenic view, better levels of air quality or water quality, etc. – and seeks directly to measure the money value of those gains. This may be done by looking for a *surrogate market* or by *experimental* techniques. The surrogate market approach looks for a market in which goods or factors of production (especially labour services) are bought and sold, and observes that environmental benefits or costs are frequently attributes of those goods or factors. Thus, a fine view or the level of the air quality is an attribute or feature of a house, risky environments may be features of certain jobs, and so on. The experimental approach simulates a market by placing respondents in a position in which they can express their hypothetical valuations of real improvements in specific environments. In this second case, the aim is to make the hypothetical valuation as real as possible.

Indirect procedures for benefit estimation do not seek to measure direct revealed preferences for the environmental good in question. Instead, they calculate a 'dose-response' relationship between pollution and some effect, and only then is some measure of preference for that effect applied. Examples of dose-response relationships include the effect of pollution on health, the effect of pollution on the physical depreciation of material assets such as metals and buildings, the effect of pollution on aquatic ecosystems and the effect of pollution on vegetation.

However, indirect procedures do not constitute a method of finding the willingness to pay, WTP, for the environmental benefit (or the willingness to accept, WTA, compensation for environmental damage suffered). What they do is to estimate the relationship between the 'dose' (pollution) and the non-monetary effect (health impairment, for example). Only then do they apply WTP measures taken from direct valuation approaches. Accordingly, we do not discuss indirect procedures further in this chapter.

10.3 THE HEDONIC PRICE APPROACH

The value of a piece of land is related to the stream of benefits to be derived from the land. Agricultural output and shelter are the most obvious of such benefits, but access to the workplace, to commercial amenities and to environmental facilities such as parks, and the environmental quality of the neighbourhood in which the land is located are also important benefits which accrue to the person who has the right to use a particular piece of land. The property value approach to the measurement of benefit estimation is based on this simple underlying assumption. Given that different locations have varied environmental attributes, such variations will result in differences in property values. With the use of appropriate statistical techniques the hedonic approach attempts to (a) identify how much of a property differential is due to a particular environmental difference between properties and (b) infer how much people are willing to pay for an improvement in the environmental quality that they face and what the social value of improvement is. Both the identification and the inference activities involve a number of issues which are discussed in some detail below.

The identification of a property price effect due to a difference in pollution levels is usually done by means of a *multiple regression* technique in which data are taken either on a small number of similar residential properties over a period of years (time series), or on a large number of diverse properties at a point in time (cross section), or on both (pooled data). In practice, almost all property value studies have been cross-section data, as controlling for other influences over time is much more difficult.

It is well known of course that differences in residential property values can arise from any sources, such as the amount and quality of accommodation available, the accessibility of the central business district, the level and quality of local public facilities, the level of taxes that have to be paid on the property, and the environmental characteristics of the neighbourhood, as measured by the levels of air pollution, traffic and aircraft noise, and access to parks and water facilities. In order to pick up the effects of any of these variables on the value of a property, they *all* have to be included in the analysis. Hence such studies usually involve a number of *property* variables, a number of *neighbourhood* variables, a number of *accessibility* variables and finally the *environmental* variables of interest. If any variable that is relevant is excluded from the analysis then the estimated effects on property value of the included variables could be biased. Whether the bias is upward or downward will depend on how the included and excluded variables relate to each other and to the value of the property.

On the other hand if a variable that is irrelevant is included in the analysis then no such systematic bias results, although the estimates of the effects of the included variables are rendered somewhat less reliable. This would suggest then that we include as many variables as possible. However, doing so creates another difficulty. Typically many of the variables of interest are themselves very closely correlated. So, for example, accessibility to the town centre is often closely related to some measures of air pollution, and one measure of air pollution, such as total suspended particulate matter, is very closely correlated to other measures such as sulphur dioxide. To overcome this, many studies use only one 'representative' measure of pollution.

The first stage in the hedonic price approach, then, is to estimate an equation of the form:

property price = f(property variables, neighbourhood variables, accessibility variables, environmental variables)

or, symbolically,

$$\text{PP} = \text{f}(\text{PROP, NHOOD, ACCESS, ENV})$$

where f(.....) simply means 'is a function of' (depends upon). The actual specification of this equation is a matter of professional choice. A familiar one is:

$$\ln \text{PP} = a \ln \text{PROP} + b \ln \text{NHOOD} + c \ln \text{ACCESS} + d \ln \text{ENV}$$

where 'ln' simply refers to logarithm. By feeding in the observed values for property prices, the property variables, the neighbourhood, accessibility and environmental variables, a simple computer program will generate the values of *a, b, c* and *d*. In this case, the value of *d* will tell us by how much property prices vary if we alter the value of the environmental variable. Provided we can relate the property price to the willingness to pay, we have nearly solved the problem of valuing environmental damage (or improvement).

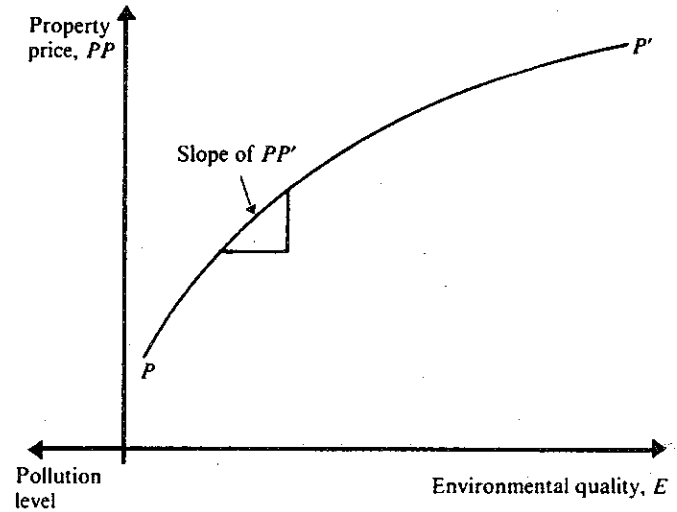


Figure 10.1 Property prices and environmental quality.

Figure 10.1 shows a typical relationship between pollution and property values that might be uncovered by the hedonic price techniques. It shows that as the pollution level decreases, so property values rise, but at a declining rate. Figure 10.2 plots the *slope* of the relationship in Figure 10.1 against the level of pollution. This is shown as AB. Hence it gives, for each level of pollution, the amount by which property values would fall if pollution levels were to be increased by a small amount.

If we are to obtain an estimate of the demand for environmental quality we would like to know how much households are willing to pay for given levels of environmental quality. In Figure 10.2 consider an individual or household who is living in an environment with an ambient pollution level P^0. It is assumed in the hedonic methodology that this choice has been arrived at in a rational manner. That is to say, the household concerned has weighed the benefits of living in alternative locations against the costs and on balance has chosen location P^0. To arrive at this decision it must have concluded that the extra payment required in higher property prices for an improvement in the environment from a pollution level slightly higher than P^0 to P^0 is just equal to the benefits of that improvement. Hence we can define the amount W^0 as that household's willingness to pay for the last unit of environmental quality. But such a willingness to pay is a

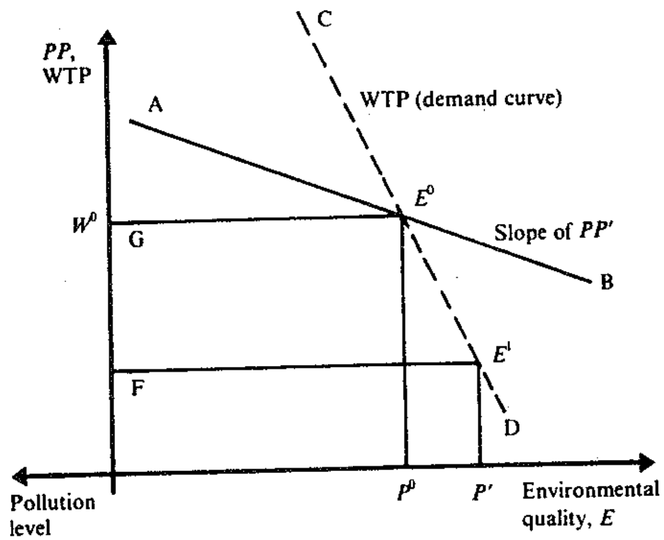


Figure 10.2 Willingness to pay and hedonic property prices.

point on the household's demand curve, and other such points are indicated by the broken line CD through E^0.

What this shows us is that the estimated hedonic price relationship can be used to obtain a point on each household's demand curve, and that the slope of the estimated relationship is a locus of points on the demand curves of many different households. If all such households were identical in every respect then the derived curve AB in Figure 10.2 would also be the demand curve for environmental quality. Each household's willingness to pay for a small improvement at every level of pollution P must also be every other household's willingness to pay if they are all identical, and the locus of willingness-to-pay points defines the demand curve. In general, however, households will differ in income and preference for environmental quality. When that is the case the hedonic approach as outlined so far only gives us partial information on the demand structure. What is now required is to see how this marginal willingness to pay varies with household income and other household characteristics. This involves a further statistical exercise which would then estimate the demand function for environmental quality.

In order to value any environmental improvement we would now use the estimated inverse demand functions CD. Suppose that

pollution falls from P^0 to P'. Then the gain in consumer surplus to *each* household at P^0 is the area E^0E^1FG. By adding up all such consumer surpluses we obtain the overall value of the environmental improvement. In fact, most empirical studies work with schedules such as AB, i.e. this second stage is not carried out.

Table 10.1 reports the results of hedonic price air pollution studies where significant effects of air pollution on property values have been found and where these effects can be expressed irrespective of the units of measurement of pollution or property values (i.e. in percentage terms). As stated earlier, many such studies find it difficult to distinguish between different forms of air pollution because of their strong inter-correlation. In these cases the one pollution measure included inevitably picks up the effects of all forms of air pollution with which it is strongly correlated. The results in Table 10.1 suggest that a 1 per cent increase in sulphation levels will result in falls in property values between 0.06 and 0.12 per cent. A similar increase in particulates lowers property values by between 0.05 and 0.14 per cent. Where the pollution variable is picking up more that one measure of air pollution, property value falls of

Table 10.1 Impact of air pollution on property values.

City	Year of: (a) property data (b) pollution measure	Pollution	Percentage fall in property value for a percentage increase in pollution
St Louis	(a) 1960	Sulphation	0.06–0.10
	(b) 1963	Particulates	0.12–0.14
Chicago	(a) 1964–67	Particulates	
	(b) 1964–67	and sulphation	0.20–0.50
Washington	(a) 1970	Particulates	0.05–0.12
	(b) 1967–68	Oxidants	0.01–0.02
Toronto-Hamilton	(a) 1961	Sulphation	0.06–0.12
	(b) 1961–67		
Philadelphia	(a) 1960	Sulphation	0.10
	(b) 1969	Particulates	0.12
Pittsburg	(a) 1970	Dustfall and	0.09–0.15
	(b) 1969	sulphation	
Los Angeles	(a) 1977–78	Particulates	
	(b) 1977–78	and oxidants	0.22

Source: D. W. Pearce and A. Markandya, *The Benefits of Environmental Policy*, OECD, Paris, 1989.

between 0.09 and 0.5 per cent are recorded. Again we should note that the fall in property values per unit increase in pollution could vary with the level of pollution.

10.4 CONTINGENT VALUATION

The contingent valuation method (CVM) uses a direct approach – it basically asks people what they are willing to pay for a benefit, and/or what they are willing to receive by way of compensation to tolerate a cost. This process of 'asking' may be either through a direct questionnaire/survey, or by experimental techniques in which subjects respond to various stimuli in 'laboratory' conditions. What is sought are the personal valuations of the respondent for increases or decreases in the quantity of some good, contingent upon a hypothetical market. Respondents say that they would be willing to pay or willing to accept if a market existed for the good in question. A contingent market is taken to include not just the good itself (an improved view, better water quality, etc.), but also the institutional context in which it would be provided, and the way in which it would be financed.

One major attraction of CVM is that it should, technically, be applicable to all circumstances and thus has two important features:

- it will frequently be the *only* technique of benefit estimation
- it should be applicable to most contexts of environmental policy.

The aim of the CVM is to elicit valuations – or 'bids' – which are close to those that would be revealed if an actual market existed. The hypothetical market – the questioner, questionnaire and respondent – must therefore be as close as possible to a real market. The respondent must, for example, be familiar with the good in question. If the good is improved scenic visibility, this might be achieved by showing the respondent photographs of the view with and without particular levels of pollution. The respondent must also be familiar with the hypothetical means of payment – say, a local tax or direct entry charge – known as the payment *vehicle*.

The questioner suggests the first bid – the 'starting point bid (price)' – and the respondent agrees or denies that he/she would be willing to pay it. An iterative procedure follows: the starting point price is increased to see if the respondent would still be willing to pay

it, and so on until the respondent declares he/she is not willing to pay the extra increment in the bid. The last accepted bid, then, is the maximum willingness to pay (MWTP). The process works in reverse if the aim is to elicit *willingness to accept* (WTA): bids are systematically lowered until the respondent's minimum WTA is reached.

A very large part of the literature on CVM is taken up with discussion about the 'accuracy' of CVM. Accuracy is not easy to define. But since the basic aim of CVM is to elicit 'real' values, a bid will be accurate if it coincides (within reason) with one that would result if an actual market existed. But since actual markets do not exist *ex hypothesi* (otherwise there would be no reason to use the technique), accuracy must be tested by seeing that:

- the resulting bid is similar to that achieved by other techniques based on surrogate markets (house price approach, wage studies, etc.)
- the resulting bid is similar to one achieved by introducing the kinds of incentives that exist in real markets to reveal preference.

There are various ways of classifying the nature of the biases that may be present in the CVM. A classification is shown in Table 10.2.

Table 10.2 Sources of bias in CVM.

Strategic	Incentive to 'free ride'?
Design	(a) starting point bias (b) vehicle bias (c) informational bias
Hypothetical	Are bids in hypothetical markets different to actual market bids? Why should they be?
Operational	How are hypothetical markets consistent with markets in which actual choices are made?

The concern with *strategic bias* is long-standing in economics and emanates from the supposed problem of getting individuals to reveal their true preferences in contexts where, by not telling the truth, they will still secure a benefit in excess of the costs they have to pay. This is the *free rider problem*. For example, if individuals are told that a service will be provided if (a) the total aggregated sum they are willing to pay exceeds the cost of provision, and (b) that each will be charged a price according to their maximum WTP, then the presumption is that each individual will understate his or her true demand. The context is one in which the good in question is a 'public good', or has features of a public good. Such goods are difficult to provide in a way that excludes anyone from enjoying them, and the consumption of the good by each individual tends not to be at the cost of consumption to other individuals. Environmental quality has these features. Hence the relevance of the 'free rider' problem. Typically, however, CVM studies have not found strategic bias to be significant.

The potential for *design bias* arises from various sources. The first of these is *starting point bias*. It will be recalled that the interviewer suggests the first bid, the starting point. It is possible that this will influence the respondent in some way, perhaps by suggesting the range over which the 'bidding game' would be played by the interviewer, perhaps by causing the respondent to agree too readily with bids in the vicinity of the initial bid in order to keep the game as short as possible.

CVM studies have attempted to test for this source of bias, usually by offering different starting bids, and sometimes by letting the respondent make the first bid. Statistically, then, it is possible to see if the mean (average) bid is affected by the choice of starting bid. The results are not conclusive, some studies finding no correlation between starting bids and mean bids, others finding that mean bids were very much affected by starting bids.

Vehicle bias

This arises from the choice of the 'vehicle', or instrument of payment, used in the approach. Such vehicles include changes in local taxes, entrance fees, surcharges on bills (e.g. electricity bills), higher prices for goods, and so on. Respondents may be 'sensitive' to the vehicle, perhaps regarding $1 paid through taxes as being more costly to him than $1 paid through an entrance fee.

The tests for vehicle bias are conceptually very simple. The average bid should not differ significantly between type of vehicle, e.g. the value of an improvement to the environment should be roughly the same whether the hypothetical payment is a tax increase or an entrance fee to the area, etc. If mean bids do vary by type of vehicle, vehicle bias may be said to exist. There are exceptions to this basic rule, but tests of the rule – by seeing how mean bids do vary with

choice of instrument – seem to suggest some source of bias. The research issue that arises is then how to choose a 'neutral' vehicle.

Information bias

This may arise from various aspects of the CVM. Starting point bias, for example, could be regarded as a form of information bias since it is the interviewer who 'informs' the respondent of the first bid. The sequence in which information is supplied may also influence respondents, e.g. indicating the 'importance' of a feature before explaining the nature of the choice. The general amount and quality of information is also of significance, particularly if the total cost of the environmental improvement is included in the information. The tests for such bias are difficult and usually involve either withholding information from one group and supplying it to another, or measuring the degree of information thought to be held by respondents. Various studies suggest no effect, while others derive measured differences in WTP according to information differences.

Hypothetical bias

The basic idea of CVM is to elicit hypothetical bids that conform to actual bids if only actual markets exist. The basic difference between actual and hypothetical markets is that in actual markets purchasers will suffer a cost if they get it wrong – regret at having paid too much, for example. One obvious test is to carry out the CVM using hypothetical *and* actual payments. What work there has been suggests that hypothetical bias is still a problem in the CVM approach. Both information and hypothetical bias problems seem to produce random variation in study results and are therefore more properly to be regarded as reliability rather than bias problems.

Operational bias

This may be described in terms of the extent to which the actual 'operation conditions' in the CVM approximate actual market conditions. This had led researchers to suggest various 'reference operating conditions' (ROCs) which should be met. The lists vary but all would include the requirement that respondents be familiar with the good they are being asked to value, and that they have either prior experience of varying the quantities of the good, or can 'learn' how to do this through repeated bids. One might add to the list the requirement for the general absence of uncertainty, but it is worth

Table 10.3 Comparisons of CVM with other techniques.

	CVM results		Indirect market study	
Study	Commodity	Value[a]	Method	Value[a]
Knetsch and Davis (1966)	Recreation days	$1.71 per household/day	TCM	$1.66 per household/day
Bishop and Heberlein (1979)	Hunting permits	$21 per permit	TCM value of time = 0 value of time = ¼ median inc. value of time = ½ median inc.	 $11.00 $28.00 $45.00
Desvousges *et al.* (1983)	Water quality improvements: (a) loss of use (b) boatable to fishable (c) boatable to swimmable	User values:[b] average (across question format) $21.41 $12.26 $29.64	TCM	User values $82.65 $ 7.01 $14.71
Seller *et al.* (1984)	Boat permit to: Lake Conroe Lake Livingston Lake Houston	Close-ended consumer surplus: $39.38 $35.21 $13.01	TCM	Consumer surplus $32.06 $102.09 $13.81
Thayer (1981)	Recreation site	Population value per household per day: $2.54	Site substitution	Population value per household per day: $2.04
Brookshire *et al.* (1982)	Air-quality improvements: (a) poor to fair (b) fair to good	Monthly value[c] $14.54 $20.31	HPM (property values)	Monthly value: $45.92 $59.09
Cummings *et al.* (1983)	Municipal infrastructure in: (a) Grants, NM (b) Farmington, NM (c) Sheridan, WY	Elasticity of substitution of wages for infrastructure –0.037 –0.040 –0.042	HPM (wages)	Elasticity of substitution of wages for infrastructure; 29 municipalities: –0.035
Brookshire *et al.* (1984)	Natural hazards (earthquakes) information	$47 per month	HPM (property values)	$37 per month

Source: Cummings, Brookshire and Schulze (1986), p. 125.
[a] Mean values amongst respondents.
[b] Values apply to post-iteration bids for users of the recreation sites.
[c] Value for sample population.

noting that this automatically raises problems for the use of CVM in eliciting option values which arise precisely because of uncertainty.' It was noted above that the concept of 'accuracy' is a little elusive when considering benefit measurement techniques. Validity is a multidimensional concept and no one test will prove definitive. But some reassurance is likely to derive from any discovery that differing techniques secure similar valuations. Table 10.3 summarises several studies that have attempted comparisons of CVM and other valuation approaches. The studies compared CVM with one or other of the travel cost methods (TCM) (see below), hedonic property price approach (HPM = house price method), and site substitution approach (not discussed here). The *ranges* of values all overlap if accuracy is expressed as ± 60 per cent of the estimates shown, and overlap in thirteen of the fifteen comparisons if the range is ± 50 per cent. These are familiar ranges of error in estimates of demand functions in economics. This does not mean that the CVM is 'correct' since, as noted above, we have in turn to make some judgement as to how correct the comparator techniques are. But it does tend to be reassuring.

One significant feature of the CVM literature has been its use to elicit the different kinds of valuation that people place on environmental goods. In particular, CVM has suggested that existence values may be very important. Schulze *et al.* (1983), for example, have suggested that the benefits of preserving visibility in the Grand Canyon are of the order of $3.5 billion per year, and some $6.2 billion per year if the visibility is extended to the southwestern parklands of the USA. Making allowance for future population trends, annualised benefits rise to $7.4 billion. These compare with the control costs of some $3 billion per year. Existence value estimates are as yet few in number and should be treated with caution. CVM results are best interpreted as indicating respondent total economic valuation i.e. a global assessment encompassing use, option and existence values.

10.5 TRAVEL COST APPROACHES

Travel cost models are based on an extension of the theory of consumer demand in which special attention is paid to the value of time. That time is valuable is self-evident. What precisely its value is, remains a question on which there is some disagreement, as will become clear later. However, as a starting point let us imagine a household consisting of a single person who works as a driver. He can work

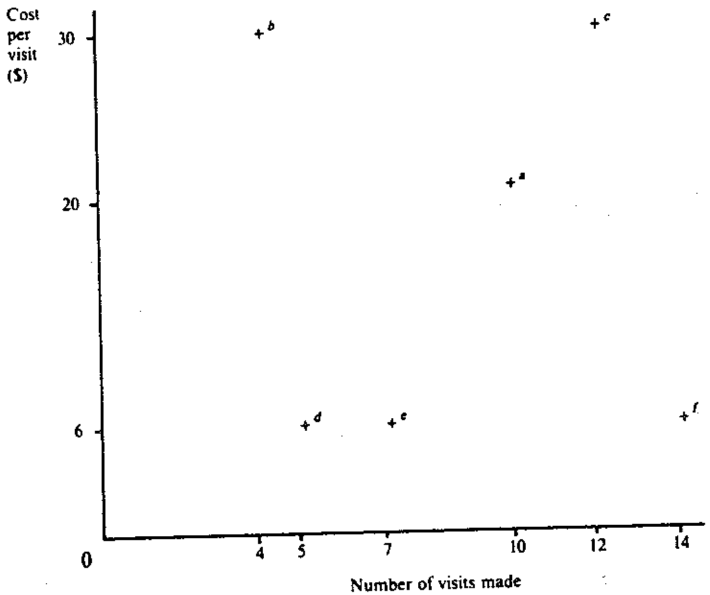


Figure 10.3 Observations of recreational visits and visiting costs.

as many or as few hours as he wishes and he earns $5 an hour. He is fortunate enough not to pay taxes, and enjoys (or dislikes) driving for work or for recreation equally much. On a particular day he can either drive to a park that takes an hour to get to, and spend some time there, or he can go to work. In these circumstances he is faced with possibly two decisions. The first is whether to go to the park or to go to work. The second is, if he goes to the park, how much time to spend there. Suppose that the cost of the journey in terms of petrol and wear and tear is $3 and there is an entry fee of $1. If he goes to the park and spends a couple of hours there, then it will have cost him $4 in cash *plus* the loss of income of $20. The true cost of the visit consists of the entry fee, plus the monetary costs of getting there, plus the foregone earnings. If we had information on all these variables and we could obtain it for a large number of individuals, along with the information on the number of visits that each had made (and would make) during the season, then we could attempt to estimate the household's willingness to pay for a given number of visits. However, at first glance the data would not look very orderly. Figure 10.3 shows the kind of data that we might find.

Our single earner household, for example, could be represented by the point *a*: he makes ten visits at a cost per visit of $20. Points *b* and *c* represent two households, each of whom face very high costs ($30). Of these *b* makes very few visits because it is a poor household living far from the recreational site, and *c* is a high earning household located near the park that makes a lot of short visits (being a high earner it has a high foregone-earnings component to its costs). Points *d*, *e* and *f* also represent households with the same costs per visit. Whereas both *d* and *e* make few visits, *d* does so because it has no attraction to the facilities offered, but *e* does so because it has access to another park close to its residential location. Household *f*, on the other hand, makes a lot of visits. Although it is identical to *e* in every other respect, it is not located close to another recreational area.

It is clear from the above that if we are to trace out how a particular household, such as *a*, would react to changes in the cost per visit, then we need to group together households that are similar to *a*. The locus of points linking such households would then constitute their demand curve for the recreational facilities that that site has to offer. Similarity here means grouping our observations according to income, preference for recreation and access to other recreational facilities. Given the demand curves we can calculate the benefits of the site by taking the area under these curves to obtain the consumer surplus as indicated in Chapter 2. Adding up the consumer surpluses for different categories or households gives us the overall benefit of the site.

If the model developed here is to be used to evaluate the benefits of environmental improvements, then further work has to be done. It is no longer enough to separate out the groups according to what other recreational facilities they may have access to. We now need to know how much of the willingness to pay of a category of households will increase if the facility at a particular site is improved to allow, for example, the possibility of fishing in a lake where none was possible before. This in turn requires knowledge of how much of the willingness to pay for each site is due to each of its specific facilities. Then by looking across sites we will be able to trace out changes in this wilingness to pay as facilities change. The data required for such an exercise would include the facilities of each site and the location of each household relative to all the sites. This is clearly a very large amount of information and so some simplifying assumptions will be necessary in many cases. What these are and

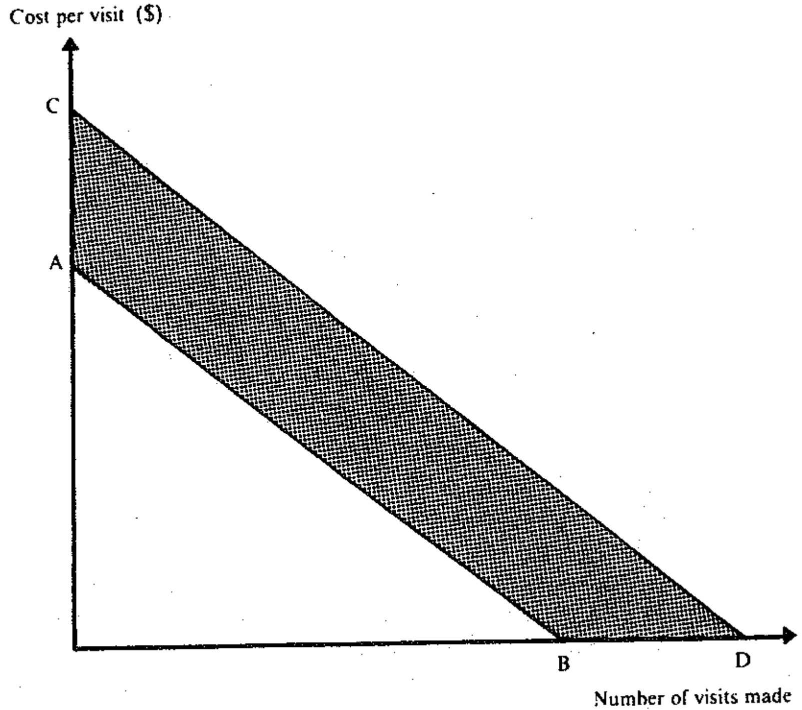


Figure 10.4 Benefits of improving a recreational site.

how they affect the results is discussed later in this section.

If we can derive the demand curve for recreation for a particular category of households defined by household characteristics such as income, education and the liking for recreational facilities, and we can show how this demand curve would shift if facilities improved, then the benefit of the improvement can be derived as shown in Figure 10.4. AB is the curve prior to the change and CD is the curve after the change. The benefits of this group of consumers are given by the area ABCD. Adding across groups gives the total benefit.

10.6 WILLINGNESS TO PAY VERSUS WILLINGNESS TO ACCEPT

The CVM has been particularly instrumental in a debate over the relationship between WTP and WTA measures of environmental change. It will be recalled that WTP is generally elicited when

Table 10.4 Disparities between WTP and WTA (in year-of-study dollars).

Study	WTP	WTA
Hammack and Brown (1974)	$247.00	£1,044.00
Banford *et al.* (1977)	43.00	120.00
	22.00	93.00
Sinclair (1976)	35.00	100.00
Bishop and Heberlein (1979)	21.00	101.00
Brookshire *et al.* (1980)	43.64	68.52
	54.07	142.60
	32.00	207.07
Rowe *et al.* (1980)	4.75	24.47
	6.54	71.44
	3.53	46.63
	6.85	113.68
Hovis *et al.* (1983)	2.50	9.50
	2.75	4.50
Knetsch and Sinden (1983)	1.28	5.18

Source: Cummings *et al.* (1984).

considering the valuation of a potential environmental *benefit*, whereas WTA seems more appropriate if we are asking someone to 'accept' a *cost*. If we take a given state of the environment, then, we could ask for the WTP to improve the environment still further and the WTA to reduce environmental quality from the initial position. Economic theory tells us that these two values should not differ significantly. Yet the CVM studies tend to suggest quite major disparities. Table 10.4 shows some examples of the kinds of differences that have been found. How are the differences to be explained? There are various options. The main ones are as follows:

1. Economic theory is wrong and people value gains and losses 'asymmetrically', attaching a lot more weight to a loss compared to the existing position than to a gain.
2. The relevant CVM studies are flawed and no reliance can be placed in the disparate estimates.
3. CVM studies tend to deal with large, discrete changes and 'instant' valuations. These cannot be compared to the context in which economic theory concludes that WTA and WTP must be very similar.

The literature is divided on which explanation is correct, proposition

1, in particular, exciting considerable controversy. Psychologists suggest that *prospect theory* explains a good deal of the difference. In prospect theory individuals' values relate to gains and losses in comparison to some 'reference point'. This contrasts with the economic assumption that individuals maximise 'utility'. What matters is the point from which the gains and losses are measured. This may, for example, be the status quo. Second, prospect theory suggests that values for negative deviations from the reference point will be greater than values placed on positive deviations. Gains will be valued less than losses, just as the CVM studies suggest. Third, the manner in which the gains and losses are to be secured matters a great deal. An 'imposed' loss, for example, will tend to attract a much higher value than a voluntarily secured gain of equal quantity. Thus, it is suggested that a loss of something that is already owned is regarded as more important than the gain of something not yet possessed; the regret attached to going without something one never had is less than the cost of losing what one already has.

Other writers suggest that the disparity between WTP and WTA disappears when proper incentives are established for people to tell the truth in response to questions about their valuation of environmental quality. These 'proper' responses, it is argued, are present in the market-place. Moreover, markets contain in-built mechanisms whereby buyers and sellers learn about the commodity they are trading, whereas hypothetical market situations do not. Brookshire and Coursey 61987), for example, report experiments in valuation of tree-growing in a city area and conclude that

> when the market-like elicitation process is repeated even a small number of times, values for the public good are more consistent with the traditional economic notions of diminishing marginal utility. Although individuals may initially exaggerate their preferences for the public good, they modify their stated values as functions of the incentives, feedback, interactions and other experiences associated with the repetitive auction environment (Brookshire and Coursey, 1987, p. 565).

A great deal more research is needed to investigate these issues. The issue is important, for if WTA and WTP really do differ by multiples of three or more (as some of the empirical literature suggests) then the kind of values placed on the environment are similarly affected.

11 · POLLUTION CONTROL POLICY IN MIXED ECONOMIES

11.1 POLLUTION CONTROL: THEORY VERSUS PRACTICE

In Chapters 4 – 8 we analysed the economic approach to pollution and pollution control policy. We noted that, from the point of view of economic theory, market-incentive policy instruments (e.g. effluent charges and rights) can be shown to be the least-cost solution to the problem of attaining ambient environmental standards. In the context of pollution abatement it appears that economic-incentive instruments possess the inherent advantages of cost-effectiveness and also provide inducements to technological innovation. Economists therefore have often been critical of the direct regulation and standards approach to pollution control, despite the fact that this 'command and control' strategy has been universally favoured by governments and their regulatory agencies. The use of charges and rights (permits) is not entirely absent in operational control strategies, but their adoption and implementation has been quite restricted and they have been limited to a supplementary role.

A number of practical or political reasons for the lack of widespread acceptance of market-incentive instruments in pollution control were briefly examined in Chapter 6. It may also be the case that some of the advantages claimed for effluent charges and some rights schemes exist only under unrealistic or restrictive assumptions. We noted in Chapter 8 that a number of real pollution situations involve a range of pollutants and a complex 'receiving' environmental system or set of systems. Quite complicated charging schemes may be required in such circumstances with significant informational requirements. The bargaining (over charge levels, timing of