There has also been a lot of interest in providing biophysical information on the state of the environment, and how it is affected by economic activity. Again, part of the motivation is the idea that if people were better informed there would be more chance of their behaving consistently with the requirements of sustainability, and voting for policies to protect the environment. There are two, related, problems about expecting such biophysical information to have much impact on public opinion. First, it is difficult for many people to take on board what scientific data mean. Second, there are lots of different biophysical indicators expressed in different units of measurement. One of the attractions of the economic indicators – GDP or ISEW – as sources of public information is that they come as single numbers, not as a whole series of different measurements.

One interesting attempt to get round these problems and to come up with a single easily understood indicator is the 'ecological footprint'. This is:

> The aggregate area of land and water in various categories that is claimed by participants in the economy to produce all the resources they consume, and to absorb all the wastes they generate on a continuing basis, using prevailing technology.
>
> (Wackernagel and Rees, 1996)

There are problems about actually measuring all of the land requirements, so footprint results should not be regarded as precise. However, they are striking and easily understandable, and are indicative of the basic situation. Wackernagel *et al.* (2002) estimated the footprint for the global economy for each year from 1961 to 1999. They found that the ratio of the footprint size to the land actually available increased from approximately 0.7 in 1961 to approximately 1.2 in 1999. In the latter year, as they put it, it would require 1.2 earths to regenerate what humanity used. There are wide differences in the size of the per capita footprint across different national economies. Whereas the global average is about 2 hectares per capita, for the USA the footprint is 9.7 hectares. The implication is that if all of the world's people were to consume at the US level, using current technology, it would require a few earths to support them.

11.3 COMMAND-AND-CONTROL INSTRUMENTS

Command-and-control instruments (or 'direct regulation' as they are also known) have been the dominant method of environmental regulation in the majority of countries. They are mostly used for pollution control and the management of common property resources (such as ocean fisheries within territorial waters). The benefits are measured in physical units, which avoids the problems of monetary valuation.

The command-and-control instruments which are currently in use operate at various stages of the production and pollution-generation process: inputs used, production technique, quantity of goods produced, emissions output, location of emissions and ambient pollution levels. Directing the controls at points closest to what is the target, namely ambient pollution levels, gives polluters most flexibility in how a pollution reduction is achieved. Examples of direct regulation in the US include ambient air quality standards (Clean Air Act), effluent emissions (Clean Water Act), standards for hazardous waste disposal (Resource Conservation and Recovery Act) and restrictions on the use of dangerous substances (Federal Insecticide, Fungicide and Rodenticide Act; Toxic Substance Control Act).

Direct regulation may ban substances or production techniques if they are deemed too dangerous for humans or the environment. Otherwise the quantity of a pollutant that can be produced or the share of a resource stock that can be used will be limited or the technology or location restricted. In the following we explore the three most commonly used types of command-and-control instruments.

11.3.1 Non-transferable emissions licences

Setting emission targets (or **environmental standards**) is normally a political process, which is based on scientific findings about safe emission levels and which take into account what policy makers and stakeholders consider as technically and economically feasible. Remember, as we saw in the previous chapter, that under conditions of uncertainty it is often difficult (or impossible) to establish what 'safe' emission levels are. However, the alternative of doing nothing is even less attractive. Therefore, targets are set and producers are asked to comply with them. Being aware of the problems which scientists and policy makers face when setting targets under conditions of uncertainty, we should not be surprised that we often observe targets which were considered safe in one year, but later have to be significantly revised as better or novel information arises and as we acquire a better understanding of the environmental impact of a production process.

In order to achieve a given overall emissions target for a particular kind of pollutant, the environmental regulatory authority creates licences (depending on the context, also called permits or quota), which limit the amount of emissions permissible for each production unit. As the name indicates, the licences cannot be transferred or traded – we look at licences that can be traded, which are a market instrument, later. The authority needs to have sufficient information for allocating the licences adequately. Examples of emissions licences are the 'eco-points' (Ökopunkte), which hauliers have to buy from the Austrian government when passing through Austria. Per EU Member State up to 8,000 trips through Austria are granted per year. Lorries with modern engines which produce less emissions require fewer points (they can make more trips) than lorries with older engines. How effective the system is environmentally depends on the total number of licences issued per year.

For the licence scheme to function well, levels of emissions need to be monitored regularly and penalties for non-compliance need to be in place (and enforced). If these conditions are fulfilled, emission licences will normally deliver the expected environmental improvement. If the cost structures for abatement are different for various emissions sources, the scheme will, however, cost society more than is strictly necessary. The reason for this is that each source has to fulfil its target independently, irrespective of how difficult (expensive) it is for it and it cannot ask (pay) somebody else to reduce emissions by a larger quantity on its behalf. The way in which this makes total abatement costs higher than they need be will be explained in detail when we look at how tradable licences, and uniform emissions taxes, work.

Box 11.2 Best Available Technology Regulation in Europe

While pollution from industrial installations in many European countries has been controlled to some extent for over 150 years, the definition of best available technology was only recently harmonised. In October 1999 a European Directive on Integrated Pollution Prevention and Control (EC/96/61) was passed, and by 2004 most Member States have integrated it into their respective national environmental laws. The Directive defines 'best available techniques' as: 'the most effective and advanced stage in the development of activities and their methods of operation which indicate the practical suitability of particular techniques for providing in principle the basis for emission limit values designed to prevent and, where that is not practicable, generally to reduce emissions and impact on the environment as a whole'. The Directive outlines a framework requiring Member States to issue operating permits for certain installations carrying on certain industrial activities. The Directive applies to new or substantially changed installations with effect from October 1999 and no later than October 2007 for existing installations. These permits must contain conditions based on BAT. The European Integrated Pollution Prevention and Control Bureau organises an exchange of technical information on best available techniques and creates reference documents (BREFs) which must be taken into account when the respective national authorities of Member States determine conditions for integrated pollution prevention and control permits. The BREFs inform the relevant decision makers about what may be technically and economically available to industry in order to improve their environmental performance.

11.3.2 Minimum technology requirements

Another command-and-control instrument regulates the technology which firms (and/or households) can use. The aim of required technology standards is to control pollution by banning or phasing out technologies which are known for causing severe or unnecessary environmental damage. In other words, it is a technology-forcing process which is intended to reduce future emissions to prevent unnecessary pollution. Required technology standards have been implemented as 'best practicable means' (BPM), 'best available technology' (BAT) and 'best available control technology' (BACT). More recently, policy makers have become more sensitive to causing high costs to firms in their country and therefore the 'best available technology not entailing excessive cost' (BATNEEC) was introduced.

For examples, such regulations have required polluters to use flue-gas desulphurisation equipment in power generation, designated minimum stack heights, required the installation of catalytic converters in vehicle exhaust systems and maximum permitted lead content in engine fuels. BACT regulations play an important role in the US clean air laws.

11.3.3 Regulation of location of polluting activities

For pollutants for which physical processes operate so that the pollutant quickly becomes dispersed to the point where the spatial distribution is uniform, the location of the pollution source is not important. Many pollutants are, however, not easily dispersed. For example, ozone accumulation in the lower atmosphere, oxides of nitrogen and sulphur in urban airsheds, particulate pollutants from diesel engines and trace metal emissions are cases where emissions source location matters. In such cases, in order to reduce human exposure it is common to regulate by zoning or planning procedures which control where sources or residences can be built. For example, incinerators are usually located on the outskirts of cities or industrial zones are separated from residential ones. Only on rare occasions are people relocated from existing residences in response to high pollution levels. For example, the radioactive pollution in Chernobyl was far too high for people to live there after the nuclear accident in 1986. Planning controls and other forms of command and control directed at location have a large role to play in the control of pollution with localised impacts and for immobile source pollution. They are used to prevent harmful spatial clustering of emission sources.

However, separating people from pollution sources does not reduce the impact on the non-human biophysical systems. In addition, people are not homogenous. Some groups are more able to avoid exposure to negative environmental impacts than others. As reported in Box 11.3, during the 1980s and 1990s evidence from many cases in the US showed that the poor and ethnic communities were more likely to be exposed to hazardous waste and pollution.

Box 11.3 Environmental racism and classism

There is evidence that poor people and residents in ethnic communities have experienced disproportionate exposure to hazardous waste and pollution, although there are conflicting views as to the cause. In the literature the proposition that environmental hazards disproportionately affect minorities and the poor is called the environmental racism–classism hypothesis. Quantitative research supports both propositions by reporting race/class correlates for a variety of environmental hazards. Studies, which were mostly conducted during the early to mid-1990s, documented particular exposure for air pollution and lead among urban African Americans, pesticide contamination for Chicano farmworkers, radiation exposure among the Navajo and waste management facilities in African American and Hispanic communities. Allen (2001) tests the environmental racism–classism hypothesis and focuses on environmental racism in relation to toxic releases in American counties. Allen uses data on 2,083 counties in 1995. The major findings in this study support the environmental racism and classism hypothesis. Minorities and the poor are disproportionately affected by environmental hazards. However, the results also demonstrate that the race–class–risk nexus, as regards toxic releases, is more complex than anticipated. As the percentage of the Black population in American counties in 1995 increased, so did the level of toxic releases; however, this relationship was stronger in the Sunbelt, indicating that larger proportions of Black Americans are exposed to higher levels of toxic releases in Sunbelt counties than is the case elsewhere in the nation. Class and race relationships are conditional: while high social class reduces the level of toxic releases, it does so by moderating the relationship between fiscal capacity, pollution potential and thus environmental harm.

Such outcomes do not need to be explained in terms of overt racism or classism. They could be the result of decision making based solely on efficiency or cost-effectiveness criteria. Locating waste facilities where they pose the least risk to the general population, have the lowest operation expenses and entail the smallest opportunity cost for alternate land uses, in combination with a low propensity for protest, seems to make a lot of sense in such terms. However, this is a clear reminder that public policy decisions should not be solely based on the efficiency goal. Efficiency and fairness need to be considered at the same time. In order to produce good recommendations for environmental policy, our economic framework needs to take into account equity as well as efficiency and cost effectiveness. This is what ecological economics aims to do.

In response to such academic studies and the strengthening environmental justice movement, on 11 February 1994 US President Clinton signed the Executive Order 12898 on Environmental Justice. In the spring of 1998 the US EPA issued Interim Guidance for investigating Title VI of the Civil Rights Act of 1964 to ensure that the issuance of pollution control permits does not negatively impact low-income and minority communities. The Office of Environmental Justice in the US EPA was founded to 'provide a central point for the Agency to address environmental and human health concerns in minority communities and/or low-income communities – a segment of the population which has been disproportionately exposed to environmental harms and risks'.

11.4 CREATION OF PROPERTY RIGHTS

The twentieth century has witnessed many efforts to improve the management of economy–environment interdependencies through reforms of **property rights**, the rights of an owner of property; these typically include the right to use the property as one sees fit (subject to certain restrictions, such as zoning) and the right to sell it. The reasoning behind this is that the absence of private property rights in natural resources or ecosystem services is responsible for many environmental problems. With the creation of private property rights in these assets, markets would change the relative prices of all assets. The prices of environmental assets being depleted would rise, so that their use would decline. This trend would also encourage the development and use of substitutes, where possible. As a result of the changed relative prices, production and consumption patterns would become less environmentally damaging, and the prices of goods and services would better reflect their environmental impacts.

This approach is problematic for three reasons. First, the range of applicability is limited since many environmental assets are inherently non-rival and/or non-exclusive in use. Second, where private property rights can be created, and other problems for the operation of markets are absent, the outcomes will be those that protect the environment as required by efficiency criteria. These outcomes are not necessarily consistent with sustainability requirements. Third, it overlooks the role of common property rights.

As discussed in Chapter 9 – see 9.2.4.2 on externalities – Ronald Coase showed that government intervention is not necessary if the people affected by the externality and the people creating it get together and bargain. However, the likelihood that bargaining will actually take place is low unless private property rights exist which can be enforced at low cost. The 'Coase Theorem' is of very limited applicability to important environmental problems, because they usually involve public goods/bads. This is the first point above. The second is that while bargaining would, in the circumstances assumed by Coase (rivalry and exclusion), lead to an efficient outcome, there is no reason to suppose that it would be equitable, or meet the requirements of sustainability.

The third problem with the focus on creating private property rights in environmental assets is that common property regimes may be an equally effective basis for managing environmental assets. The focus goes back to Hardin's 'tragedy of the commons', which was, as we discussed in Chapter 9, based on a confusion of open access and common property resources. Since then Elinor Ostrom and her co-workers have pointed out that common property resources are closer to a drama which may have a favourable or an unfavourable ending. A long list of historical and current examples show that common property regimes may be as effective a basis for managing environmental assets as private property rights. They also emphasise that it is not only formal institutions (laws), but also informal institutions (traditions, norms, etc.) that constitute an effective basis of property rights in many societies. References to this work were provided in Further Reading for Chapter 9.

This is not to deny a role for the creation of private property rights for using environmental assets more sustainably. There are, no doubt, situations where it is useful to create them where they do not currently exist and where the resulting outcomes would provide better protection for environmental assets than current arrangements. The point is that the usefulness of this type of policy instrument is to be judged on a case-by-case basis, in light of a full consideration of all particular circumstances. For some there is an ethical presumption that critical environmental assets should be collectively, rather than privately, owned. Collective or common ownership does not preclude leasing usage rights to individuals.

11.5 TAXATION

The idea of taxing environmentally damaging activities, such as pollution generation, in order to reduce their scale has a long history in economics. It was Arthur Cecil Pigou in his book *Economics of Welfare* (1920) who first introduced the idea that waste emissions were an externality and proposed imposing a tax equal to the marginal external cost so as to bring about the level of emissions that goes with allocative efficiency. Recall from Chapter 9 that externalities are the unintended effects of one agent's behaviour on some other agent or agents which the current institutional arrangement does not require the former to take responsibility for. While externalities can also be positive,[1] in environmental policy we are mostly concerned with negative externalities, such as, for example, air pollution from car exhausts causing respiratory problems for the human population living near a motorway.

Pigou argued that taxing emissions would reduce them to the level corresponding to allocative efficiency in the most efficient manner possible. An emissions tax is levied on each unit discharged. By raising the price for polluting to reflect social cost, environmental taxes ensure that polluters take responsibility for the full cost of their actions. In the absence of taxes or other control mechanisms, environmentally damaging activities are carried to excess by the operation of market forces. The reliance of emissions taxes on a price mechanism is a way to reduce overall compliance costs. This is so because taxes encourage the greatest pollution abatement by the firms able to adjust at the lowest cost, as we will demonstrate below when we look at the least cost theorem in section 11.7, and because they encourage deployment of new technologies.

We now want to consider at what level the regulatory authority should set the emissions tax in order to achieve: (1) the economically, *that is allocatively*, efficient level of emissions abatement; and/or (2) a sustainable level of production.

11.5.1 Taxation for allocative efficiency

Let us start with aiming for an economically efficient level of emissions abatement. First we need to identify the efficient level of pollution, which is the one that minimises the sum of total abatement cost plus total damage cost. Abatement cost is the cost to society of pollution reduction. Abatement may involve a reduction in the activity that produces the emissions, or the application of some emission-reducing measures, or some combination of the two. For example, emissions from driving a car could be reduced by driving less, by smoother driving, or a combination of the two. Damage cost is how affected individual firms value the damage. The sum of the total costs is minimised where the marginal damage and the marginal abatement cost are equal.

Back in Chapter 9, see section 9.1, we discussed the principle of equalising marginal costs and marginal benefits to maximise net benefits for allocative

1 For example, the effect child-rearing can have on society. Societies benefit greatly from an upbringing which makes young people responsible, well-educated and competent citizens. This is the reason why it makes sense for a society to support families, child-care centres, schools, training facilities, universities, etc. with public funds.

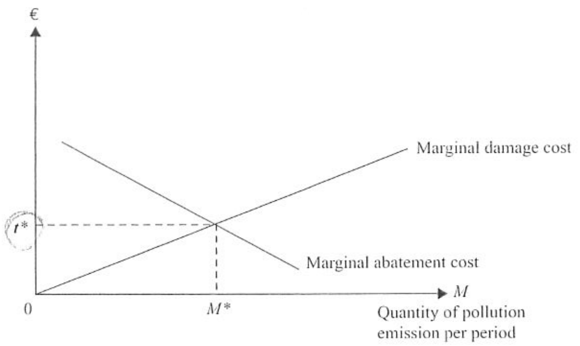


Figure 11.1 The economically efficient level of pollution minimises the sum of abatement and damage cost.

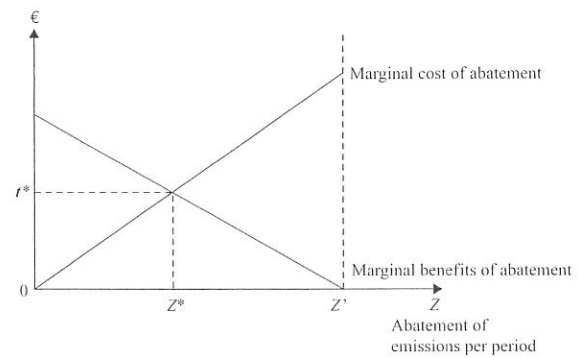


Figure 11.2 Economically efficient level of emissions abatement.

efficiency. Here we apply the same principle. By equalising the marginal damage cost and marginal abatement cost we find the economically efficient level of pollution emission per period. Figure 11.1 shows the intersection of the two curves. The marginal damage cost curve is upward sloping because the damage per additionally produced unit of pollution goes up because an additional ton, of say CO_2, at a high pollution level causes more environmental damage than the same quantity emitted in a situation with low pollution level. The marginal abatement cost curve is downward sloping because when pollution is high, incremental abatement, i.e. a small reduction, is relatively inexpensive, but the cleaner the environment, the more expensive it is to achieve further gains. If you think that the abatement cost curve slope should be the other way around, note that emissions are an economic 'bad', i.e. the less the better. In Figure 11.2 we will replace the label 'quantity of pollution emission per period' by 'abatement of emission per period', and then we are back to thinking in economic 'goods'.

At the intersection of the marginal abatement cost and the marginal damage cost, we find the economically efficient level of emissions M^*, which neoclassical economists call the 'optimal level of pollution', and the economically efficient tax

rate t^*. If it was the rate charged it would bring about the required amount of emissions. This is easier to explain if we look at things in a slightly different way as in Figure 11.2. Looking at the same situation from another angle, namely abatement of emissions instead of emissions level, we can, in Figure 11.2, identify the economically efficient level of emissions abatement Z^*. Note that on the horizontal axis Z' is zero emissions, and 0 is no abatement. The marginal benefit curve is the marginal damage from Figure 11.1, but from the abatement perspective instead of the pollution emission perspective. Z^* corresponds to the same level of emissions as M^*. This is the level at which the marginal costs and benefits of emissions abatement are equal. Z^* equals the pre-tax level of emissions minus the economically efficient quantity of emissions, i.e. the amount which needs to be abated in order to get from the original emissions level to the economically efficient emission level. Z^* is efficient, given that all social costs and benefits are included, because of the by now familiar argument about the equality of marginals. To the left of Z^* abatement cost is less than damage cost at the margin, so more abatement will reduce the sum of abatement and damage costs. To the right of Z^* abatement cost is greater than damage cost, so less abatement will reduce the sum of abatement and damage costs. You should see that the same argument gives M^* as the efficient level of emissions in Figure 11.1.

Taking Figure 11.2 as applying to a representative firm, we can see why profit maximising firms do what is needed in the same way – t^* is the (constant) marginal cost of emitting and the firm will maximise post tax profits when that is equal to marginal abatement cost (MAC). If MAC > t it pays to abate less, if MAC < t it pays to abate more.

11.5.2 Taxation for an arbitrary standard

In order to identify the allocatively efficient level of pollution as its target for emissions control the environmental regulatory authority would need to know the marginal costs and benefits of abatement. The intersection of the marginal cost and benefit functions determines the proper rate for (**Pigouvian** or externalities) taxation (t^*) and the desired emissions level. Generally the information that is needed to do this is not available to the authority. While neoclassical economists have devoted a lot of time and effort to devising means for estimating the marginal costs and, especially, benefits of pollution abatement, most recognise, with regret, that control to allocatively efficient levels is not feasible.

The policy-relevant questions are about methods of control to achieve what neoclassical economists call 'arbitrary standards'. This terminology merely means that the policy target does not derive from a precise balancing of the marginal costs and benefits of abatement. It does not, necessarily, mean that it is arbitrary in the sense of having been adopted impulsively or capriciously. Arbitrary pollution standards, in this sense, may be, and generally are, the result of a great deal of scientific research and political deliberation. For example, the standard could be a level which scientists and stakeholders consider to the best of their knowledge to be a sustainable level. For any target so determined we can identify the corresponding abatement level. Even though this level is not an allocatively efficient target, the argument used about the cost-efficiency of taxation as an instrument remains true. An arbitrary standard can be attained at least cost by the taxation of emissions, as we will show later. The regulatory authority does not need to know the aggregate marginal abatement benefit function, nor does it need to know the abatement cost function for each firm.

We need to note that choosing approach (1), aiming for allocative efficiency, or approach (2), aiming for an 'arbitrary standard', can have major implications for the biophysical environment. The economic efficiency approach is focused on economic costs and human perceptions of benefits, and knowledge that the result of emissions taxation is the economically efficient pollution level does not inform us that the outcome is sustainable. It may not be. In contrast, in approach (2) a standard is set that to the best of current knowledge and understanding is considered as a sustainable level of emissions, i.e. a level that can be absorbed by the sinks of the environment for an indefinite, or at least very long, time period without damaging it or humans. After this biophysical analysis in combination with democratic deliberation, the tax and the market mechanism are used to deliver what it is thought is the sustainable level at least cost. Approach (2) shows how ecological economists want to use the market mechanism. They have no problem with using it to attain standards set by sustainability criteria. They do not want standards set by consumer sovereignty criteria.

However, even with this more limited use of market mechanisms, we should not forget the problems resulting from uncertainty and irreversibility in the environment and society. We cannot be sure that the arbitrary standard adopted guarantees sustainability, and we cannot be sure that the tax actually imposed will deliver the standard aimed at. Joan Martinez-Alier, one of the founding fathers of ecological economics, reminds us that nobody knows for sure what the correct prices to bring about sustainability are. As he puts it: 'There are no 'ecologically correct' prices, although there might be 'ecologically corrected' prices' (Martinez-Alier, 2000:4018). He argues that when externalities are uncertain and irreversible, then it is impossible to set 'ecologically correct' prices. By 'ecologically corrected' he means prices modified by taxes and the like that move things in the direction of sustainability.

In some pollution problems, such as those arising with emissions from fossil fuel combustion for example, monitoring and enforcement costs will be lower if some input is used as the point of control, rather than emissions as such. It has been shown that if the relationships between the levels of some input and the level of emissions are known to the regulatory authority, then the least cost property of uniform emissions taxation carries over to input taxation – the authority can tax the input rather than the emissions and achieve an arbitrary standard at least cost. In the case of carbon dioxide releases in fossil fuel combustion when people talk of 'carbon taxation' in connection with mitigation of the enhanced greenhouse effect, discussed in Chapter 13, what they actually mean is taxation of the fossil fuels at rates which reflect their carbon contents. We come back to carbon taxation in the next subsection.

11.5.3 Taxation and the goods market

Taxing emissions, or the input which is the origin of the emissions, raises the costs of production. As we saw in Chapter 8, increasing the costs of production shifts the supply function upwards. This impact of emissions or input taxation is shown

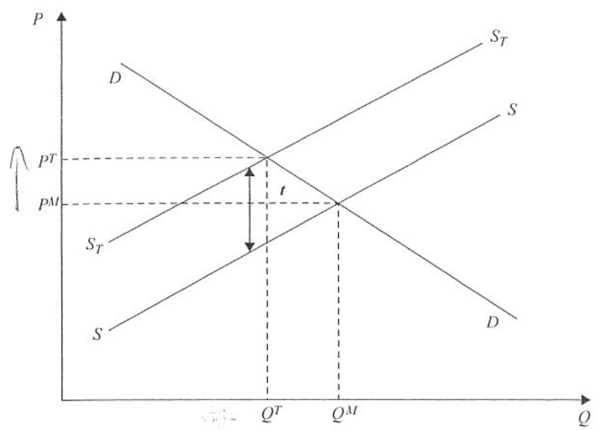


Figure 11.3 Impact of environmental tax on goods market.

in Figure 11.3, where *SS* is the supply function for some commodity before the imposition of the tax and $S_T S_T$ is the supply function with the tax in place. *DD* is the demand function for the commodity. As a result of the tax, the equilibrium quantity falls from Q^M to Q^T and the equilibrium price increases from P^M to P^T. As we saw in Chapter 8, the size of these changes due to the imposition of the tax will depend upon the elasticities of demand and supply, as will the incidence of the tax as between producers of the commodity and its buyers.

Let us now look briefly at carbon taxation. A carbon tax is aimed at reducing the carbon dioxide emissions that come from fossil fuels and which threaten to change the climate. More than 80 per cent of the world's CO_2 emissions come from fossil fuel combustion. In practice the tax would be applied not to emissions as such, but to fossil fuels. You can think of a carbon tax as a product charge placed on fossil fuels in proportion to their carbon content. Coal, which has a higher carbon content than oil and natural gas, is thus taxed relatively more. Once implemented, the rising prices of fossil fuels would induce people to use oil and gas in favour of coal; to use more renewable energy sources instead of fossil fuels; and to be more efficient in their use of energy generally. Applying such a tax enables an economy as a whole to reduce its level of carbon dioxide emissions for the lowest overall cost. Because of the scale of fossil fuel use in the economy, a carbon tax is attractive to governments as it can raise significant amounts of revenue, which could be used to finance environmental projects or to reduce other taxes, such as taxes on labour. In the latter case the relative prices of input factors would be changed. Labour would become cheaper and energy would become more expensive. This would increase the demand for labour, and reduce the use of energy, and hence carbon dioxide emissions. Increasing the demand for labour would work in the direction of reducing unemployment.

The amount of CO_2 released into the atmosphere when a barrel of oil is burned, for example, is known. Let us say it is *z* tonnes. Suppose that we want to tax CO_2

emissions at $w per tonne. Then taxing the extraction of oil at $$w \times z$ per barrel is equivalent to taxing CO_2 emissions from oil combustion at $w per tonne of CO_2. The extractor of oil would raise the price for which it sells oil to reflect the tax. Users of oil to produce other goods and services would have to pay a higher price for oil, and would raise the price for their outputs. The extent to which other commodity prices would rise would vary according to the amounts of oil used in their production. Final consumers would end up facing increases in the prices of all goods and services, with the sizes of the increases varying with the extent to which they used oil and, hence, were responsible for carbon dioxide emissions.

The effects on commodity prices can be figured out using input–output analysis, which we looked at in Chapter 5. The point about input–output analysis is that it picks up the ways in which commodity production uses inputs indirectly as well as directly. In section 5.1.3 we looked at input–output accounting and the environment and considered a numerical example economy in which manufacturing used 1,000,000 tonnes of oil, while agriculture used no oil directly. We saw that although agriculture does not directly use oil, the delivery of agriculture to final demand is responsible for some of the oil use in the economy. The production of the commodity agriculture uses the commodity manufacturing, and the production of the latter does directly use oil. We found that of this economy's total oil use, 25 per cent was indirectly attributable in this way to the production of agriculture for delivery to final demand.

Now suppose that each tonne of oil used in manufacturing releases 0.1 tonnes of CO_2 into the atmosphere, and that the government wants to tax CO_2 emissions at $1,000 per tonne – this is equivalent to a tax of $100 per tonne of oil. Clearly, the price of manufacturing will go up with the imposition of the tax. But so will the price of agriculture, because while agriculture does not directly use any oil in this example, it does buy manufacturing, and the selling price of agriculture will have to cover the increased costs on that account. In general, the carbon tax will affect the prices of all commodities in this way – their prices will increase by amounts reflecting their indirect as well as their direct responsibility for CO_2 emissions.

Using input–output analysis we can figure out from input–output accounts and information on the direct responsibility for CO_2 emissions in each industry what the price rises for all commodities would be for any given rate of carbon tax. In the numerical example here, for a $1,000 per tonne carbon tax, the price of agriculture would rise by 3.19 per cent and the price of manufacturing by 5.32 per cent – although agriculture does not use oil directly, as the transactions table shows, it does use a lot of manufacturing, and hence oil is an important indirect input to its production. The Appendix to this chapter shows how these results are obtained, and working through it will enhance your understanding of how carbon taxation would work.

In real economies all industries directly use energy inputs based on fossil fuels, and use inputs produced in several other industries using such energy inputs – all industries are directly and indirectly responsible for CO_2 emissions. Box 11.4 shows the estimates for the price rises that introducing carbon taxation would produce in a real economy. Table 11.2 lists the taxes levied on electricity consumption and production in various OECD countries. Electricity generation accounts for about 40 per cent of carbon dioxide emissions in modern industrial economies. Note, however, that much of the electricity produced is used in the production of other

Box 11.4 Input–output analysis of carbon taxation and commodity prices in Australia

Table 11.1 gives the results for carbon taxation-induced price increases, obtained from calculations of the same nature as those set out in the Appendix, using actual input–output data for Australia – the same data as was used to produce the results for CO_2 intensities given in Table 5.5. The main difference in the calculations is that those for Table 11.1 have to take account of the fact that Australia, like all industrial economies, uses not just oil, but also coal and gas. The three fossil fuels have different carbon contents per unit energy – the ranking from most to least carbon per PJ is coal, oil and gas.

Table 11.1 Price increases for a carbon tax of $20 per tonne in Australia

Sector	*Percentage price increase*	*Ranking*
Agriculture, forestry and fishing	1.77	9
Mining	1.69	12
Meat and milk products	1.77	9
Food products	1.46	16
Beverages and tobacco	0.84	24
Textiles clothing and footwear	0.95	21
Wood, wood products, furniture	1.31	15
Paper, products, printing, publishing	1.12	20
Chemicals	1.56	16
Petroleum and coal products	9.97	4
Non-metallic mineral products	1.89	8
Basic metals products	9.00	5
Fabricated metal products	2.76	6
Transport equipment	0.82	23
Machinery and equipment	0.71	26
Miscellaneous manufacturing	0.89	23
Electricity	31.33	1
Gas	21.41	2
Water	1.34	18
Construction	1.60	13
Wholesale and retail	10.14	3
Transport and communication	2.28	7
Finance and business services	1.21	19
Residential property	0.42	27
Public administration and defence	1.73	11
Community services	0.93	21
Recreational and personal services	1.62	13

Source: adapted from Common and Salma (1992).

The figures in the rightmost column of Table 11.1 are the rankings by proportionate price increase – electricity goes up by the largest percentage, residential property by the smallest. If you look back at Table 5.5 you will see that the rankings there and in Table 11.1 are very similar. The commodities whose production involves more CO_2 emissions – when indirect as well as direct pathways are taken into account – are those with larger price increases following the imposition of the carbon tax. All commodity prices go up because all production involves energy, and hence fossil fuel, use when indirect use is accounted for. The reason that the rankings in the two tables are not exactly the same is because of the impact of existing commodity taxes in the data, and because of the way these input–output tables handle payments for distribution services.

While these kinds of calculations bring out the implications of the indirect use of fossil fuels in commodity production and show how the effects of carbon taxation would hence cascade throughout the whole economy, they would not accurately predict the long-run pattern of commodity price changes. This is because the input–output analysis assumes that the imposition of the carbon taxation has no effect on production technologies – the input–output coefficients that it uses are constant. This is a reasonable assumption in the short run, but not in the long run – on the difference between the long and short runs see Chapter 8. In the long run, there would be some substitution of other inputs for fossil fuel inputs, and the input–output coefficients would change. Also, as you can see in the Appendix, these calculations assume that the producers are able to pass all of the tax increase price effects forward to the buyers of their outputs. As we saw in Chapter 8 when looking at tax incidence, they are not generally able to do this to an extent that depends on the elasticities of supply and demand.

goods and services, rather than delivered to final users of electricity. Hence, the proportion of CO_2 emissions accounted for by deliveries to final demand of the commodity electricity is typically well below 40 per cent. If you look back at Table 5.5, you will see that for Australia that proportion is just 15 per cent – this is a representative figure for a modern industrial economy. Table 11.2 shows that electricity is widely taxed, and that the rates of taxation differ greatly between countries.

Carbon taxation as we have described it is not currently in use in any economy. For the United Kingdom, Table 11.2 shows, as a tax on electricity consumption, something called the Climate Change Levy. This was originally intended to be a tax on carbon emissions that would reduce them so as to enable the United Kingdom to meet its international obligations and domestic policy objectives. In the event what emerged from the political process was a tax on some forms of energy consumption, which will not be very effective in terms of reducing CO_2 emissions. The story of how the original intention became the actual Climate Change Levy throws lots of light on the actual process by which policy is determined. Unfortunately, there is no space to tell it properly here – you will find out about it on the book's companion website. Basically, it looks as if the UK government retreated from a proper carbon tax because it feared the electoral consequences of the price rises that would have been attributed to such a tax.

11.5.4 Environmental taxes raise revenue

For policy makers emissions taxes, or energy taxes, are welcome for another reason. They generate income for the government. As discussed in Chapter 8, the revenue from the tax depends on the elasticity of demand, which is the percentage change of quantity demanded of the taxed good in response to a one per cent increase in the price. In our context of energy taxes the elasticity of demand measures how firms and households react to the increase in the prices of energy. From empirical estimates we know that the price elasticity of demand for energy is low, at least in the short run. This is due to energy demand arising from earlier decisions, which are difficult to revise in the sort run. For example, energy used for home heating is the result of a housing decision, which can only be corrected in the longer term (e.g. by installing better insulation or moving to a different house). Other decisions relating to energy depend on collective decisions. For example, one may prefer to use a train or bus instead of a car as sole occupant, but if there is no, or an inadequate, public transport system in place, the possibility for substitution does not exist in the short run. In the longer run, policies may change or the individual can move to a place which offers public transport.

Table 11.2 Taxes in OECD member countries levied on electricity

Country	*Tax*	*Specific tax-base*	*Tax rate national currency*	*Units*	*Tax rate – euro*	*Units*
Austria	Energy tax	Electricity consumption	0.0150	€ per kWh	0.015	€ per kWh
Belgium	Cotisation sur l'energie	Électricité basse tension	55.0000	BEF per MWh	1.3641	€ per MWh
Denmark	Duty on CO_2	Electricity	0.1000	DKK per kWh	0.0134	€ per kWh
Denmark	Duty on electricity	Electricity consumption for heating of dwellings and other purposes	0.5010	DKK per kWh	0.0673	€ per kWh
Denmark	Duty on electricity	Electricity consumption for other purposes	0.5660	DKK per kWh	0.076	€ per kWh
Finland	Excise on fuels	Electricity used in the manufacturing sector, etc.	25.0000	FIM per MWh	4.2073	€ per MWh
Finland	Excise on fuels	Electricity used in the rest of the economy	41.0000	FIM per MWh	6.9	€ per MWh
Finland	Strategic stockpile fee	Electricity consumption	0.7500	FIM per MWh	0.1262	€ per MWh
Germany	Duty on electricity	Electricity consumption	0.0250	DEM per kWh	0.0128	€ per kWh
Italy	Additional tax on electricity – towns / provinces	Electricity consumption for private dwellings				
Italy	Additional tax on electricity – towns / provinces	Electricity consumption for industrial purposes				
Italy	Tax on electrical energy – State	Electricity consumption for industrial purposes	6.0000	LIT per kWh	0.003	€ per kWh
Italy	Tax on electrical energy – State	Electricity consumption for private dwellings	4.1000	LIT per kWh	0.0021	€ per kWh
Japan	Promotion of power resources development tax	Electricity consumption	0.4450	JPY per kWh	0.0041	€ per kWh
Netherlands	Regulatory Energy Tax	Electricity consumption – up to 10,000 kWh per year.	0.0601	€ per kWh	0.0601	€ per kWh
Netherlands	Regulatory Energy Tax	Electricity consumption – between 10,000 kWh and 50,000 kWh per year.	0.0200	€ per kWh	0.02	€ per kWh

(cont.)

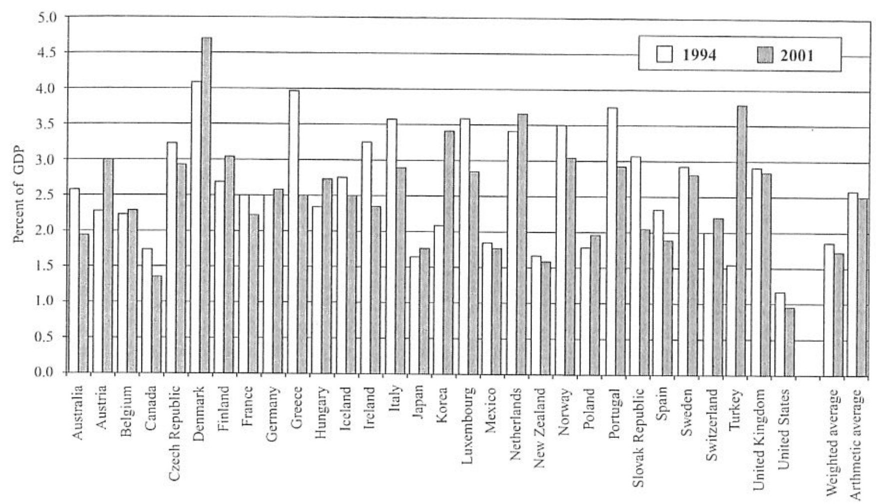
Table 11.2 *(cont.)*

Country	*Tax*	*Specific tax-base*	*Tax rate national currency*	*Units*	*Tax rate – euro*	*Units*
Netherlands	Regulatory Energy Tax	Electricity consumption – between 50,000 kWh and 10 million kWh per year.	0.0061	€ per kWh	0.0061	€ per kWh
Norway	Tax on consumption of electricity	Electricity consumption	0.1030	NOK per kWh	0.0128	€ per kWh
Spain	Tax on electricity	The production or importation of electricity		4.864%		4.864%
Sweden	Energy tax on electricity	Electricity consumption – households	0.1980	SEK per kWh	0.0214	€ per kWh
Sweden	Energy tax on electricity	Electricity consumption – manufacturing and commercial greenhouses	0.0000		0	
Sweden	Energy tax on electricity	Electricity consumption – other sectors	0.1740	SEK per kWh	0.0151	€ per kWh
Sweden	Energy tax on electricity	Material permitted for abstraction > 200 000 tons	0.1400	SEK per kWh	0.0015	€ per kWh
United Kingdom	Climate change levy	Electricity consumption – ordinary rate	0.0043	GBP per kWh	0.0069	€ per kWh
United Kingdom	Climate change levy	Electricity consumption – reduced rate	0.0008	GBP per kWh	0.0014	€ per kWh
United States	Delaware – public utilities tax	Electricity consumption		4.25% of gross receipts		4.25% of gross receipts

Taxes on Electricity production

Country	*Tax*	*Specific tax-base*	*Units*	*Units*
Lithuania	Excise tax on energy products	Electricity	1% of the value of kWh	1% of the value of kWh
United Kingdom	Non-fossil fuel obligation levy	Electricity production	0.7% of the price.	0.7% of the price.
United States	Connecticut – electricity company tax	Income from electricity distribution to residential service	6.8% of income.	6.8% of income.
United States	Connecticut – electricity company tax	Income from electricity distribution to other than residential service	8.5% of income.	8.5% of income.

Source: OECD/EEA Environmentally Related Taxes Database, http://www1.oecd.org/scripts/env/ecoInst/index.htm



Source: OECD Environmentally Related Taxes database
http://www.oecd.org/document/29/0,2340,en_2649-34295 1894685_1_1_1_1,00.html

Figure 11.4 Revenue from environment-related taxes as percentage of GDP.

Revenues from environmental taxes may or may not be earmarked for funding environmental projects. Environmental taxes face the fundamental problem that the tax base shrinks if the tax is effective. An example here is the recently introduced congestion charge in London, which reduced traffic by 18 per cent within the first year, and consequently did not produce enough revenue to increase capacity of public transport as had been planned. For more information see http://www.londontransport.co.uk/tfl/downloads/pdf/congestion-charging/cc-12monthson.pdf.

Figure 11.4 shows that the degree to which environmental taxes are charged and raise revenue in various countries differs widely. This is an issue, which we will revisit in the next chapter, where we will discuss the international dimension of environmental problems and environmental policy instruments. An often-used argument against the introduction of environmental taxes in a country has been the fear that doing so would make it less competitive in international trade.

11.6 TRADABLE PERMITS

Tradable permits are another type of market-based policy instrument, under which rights to discharge pollution or exploit resources can be exchanged through either a free or controlled permit-market. Like taxes they generate economic incentives to economic agents to move towards less environmentally harmful behaviour. As used to date, they have mainly impacted directly on firms.

Tradable permits differ from taxes in a fundamental way. Consider the control of pollution. With taxes the monetary incentive facing the agent is the fixed price to be paid for each unit of emissions. With permits the agent faces a quantitative emissions target which is fixed by the amount of permits held, and the agent can vary her holding by buying or selling permits at a variable price. From the point of view of the regulatory authority, taxes fix the price at which agents can emit but not the amount that they emit individually or collectively, whereas permits fix the amount that agents can collectively emit but not the price at which they do it or the amount they emit individually.

So, how do tradable permits work? A tradable permit is an environmental policy instrument which organises the exchange of rights to discharge a particular kind of waste into some environment, or to use a particular natural resource. In the former case, instead of being charged for releases, one needs to hold a permit to discharge the amount that one actually does discharge. The regulatory authority sets a limit (expressed in biophysical units) on the total amount of emissions permits in existence. By controlling the total number of permits, the regulating authority can control the aggregate pollution quantity. While the regulating authority sets the total amount permissible, they do not attempt to determine how that total allowed quantity is allocated among individual sources of emissions. Tradable permits are based on the principle that any increase in emissions from one source must be offset by an equivalent decrease elsewhere. Sources can vary the amount of permits that they hold, and hence their emissions levels, by buying and selling from and to other sources. The main advantage of the scheme is that it is cost-effective and generates dynamic incentives for cost reduction. The regulatory authority could issue a total amount of permits equivalent to the allocative efficiency target, if it knew what it was, or an amount equivalent to an 'arbitrary standard'.

The basic principles are the same in the case of permits to use a natural resource – a regulatory authority determines what the level of use is to be and issues a corresponding total amount of permits, denominated in terms of tonnes or number of individuals for a fish stock, for example. Once issued by the authority, permits can be bought and sold among the resource-harvesting firms and individuals.

Examples include individual transferable quotas in fisheries, tradable depletion rights to mineral concessions and marketable discharge permits for water-borne effluents. The United States began emissions trading after passage of the 1990 Clean Air Act, which authorised the US EPA to put a cap on how much sulphur dioxide (which causes acid rain) the operator of a fossil-fuelled plant was allowed to emit. The tradable permit scheme as applied under the US Clean Air Act led to reductions of smog and acid rain emissions at lower economic cost than would have been the case with a command-and-control instrument. Other initiatives for more sector-specific use of the tradable permit concept have been developed, e.g. for municipal waste management (in the UK) and the use of renewable energy sources (e.g. in Denmark and Italy). Currently researchers are working on options for more systematic use of tradable permits in the transport sector. For an overview of past and current tradable permit schemes see Table 11.3. More details can be found in Further Reading.

Box 11.5 Emissions trading in the European Union

On 1 January 2005 the European Union launched its Europe-wide Emissions Trading Scheme. This aims to reduce the impacts of climate change through lowering carbon emissions by providing clear incentives for investment in energy efficiency and cleaner technologies. Under the terms of the Scheme, industrial emitters of carbon dioxide (CO_2) will be allocated tradable allowances, on an installation-specific basis, specifying the amount of CO_2 they can emit each year. The key principle of the scheme is the ability of companies to trade their allowances in order that emissions reductions are achieved at least cost to Europe as a whole.

The EU Emissions Trading Scheme (EU ETS) will establish the world's largest-ever market in emissions. Participation in the scheme will be mandatory for companies in sectors covered by the first phase of the scheme. These are electricity generators, oil refineries, iron and steel production, cement clinker and lime production, glass manufacturing, brick and tile manufacturing and pulp and paper. In addition, installations in any sectors that have combustion plants of a thermal input of over 20MW, including aggregated plants on a single site, are also covered (hospitals, universities and large retailers may find themselves included under this provision). For further details see http://www.europa.eu.int/comm/environment/climat/emission.htm.

There are two broad types of tradable permit system – the 'cap-and-trade' scheme and the 'emission reduction credit' scheme. For the latter, a baseline is agreed for every participant source before the start of the operation of the system. A participant is credited with any over-achievement, and is allowed to sell the credits arising. The 'cap-and-trade' scheme involves a decision by the regulatory authority about the total quantity of emissions (or natural-resource use) that is to be allowed – the 'cap' – and shares the total among the participating agents. This scheme establishes a quantified ceiling assigned to each participant for a given period. No one is allowed to emit (or use) more than the amount for which it possesses emission (or natural-resource use) permits. As with the emissions reduction scheme, the participants can buy and sell permits from and to one another.

As with all environmental policy instruments monitoring emissions (or natural-resource use) and implementing a system of penalties for non-permitted emissions (use) is crucial. The control authority devises the total quantity of permits issued and the initial allocation. The initial allocation to firms and/or individuals can be either by auction, which generates revenue for the government, or it can be without charge. Adequate safeguards need to be put in place to ensure that permits can be freely traded between participating firms and/or individuals at whichever price they have agreed for that trade. In Table 11.3 you will notice a 'Greenhouse gas trading scheme' for Sweden and UK. We will be looking at greenhouse gas emissions and the problems that they cause in Chapter 13, where we will come back to the use of tradable permits for their control.

11.7 THE LEAST COST THEOREM

What economists (and in fact many others) find attractive about market instruments is the prospect of their delivering the desired outcome at least cost. The **least cost theorem** says that total abatement costs are minimised where the regulatory authority taxes emissions from all firms at a uniform rate per unit. However, the theorem does not, as is widely believed, establish a presumption in favour of emissions taxation over command-and-control instruments of the form that specify

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Table 11.3 Past and current tradable permit schemes in OECD countries

Country	*Name of permit system*	*Type of permit system*
Australia	Individual transferable fishing quotas	Quota
	New South Wales – Hunter River Salinity Trading Scheme	Quota
	Pilot Interstate Water Trading Project	Transferable usage rights
Canada	Alberta – Tradable hunting rights	Transferable usage rights
	Allowance system for HCFCs	Quota
	Allowance system for methyl bromide	Quota
	Maple grove permits	
	NO_x and VOC emissions	Quota
	Transferable fishing quota	Quota
Denmark	Emissions trading in the electricity sector	Quota
France	Tradable development rights for land preservation	
Iceland	Individual transferable fish quota	Quota
Mexico	Tradable hunting permit	Transferable usage rights
Netherlands	Tradable fishery quota	Quota
New Zealand	Revised district scheme (tradable development rights)	
	Transferable fishing quota	Quota
Norway	Quota system for greenhouse gases	Quota
Poland	Chorzow – VOC control (demonstration project 1991–92)	Quota
Sweden	Greenhouse gas emissions trading scheme	Quota
Switzerland	Basel Canton – Control of VOC and NO_x emissions	Quota
UK	Greenhouse gas emissions trading scheme	Quota
US	Acid rain allowance trading	Quota
	Colorado – Tradable phosphorous discharge rights – Dillion Reservoir	Quota
	Mobile sources averaging, banking and trading	Averaging
	Montgomery County – Land management	
	Northeast USA – Ozone transport commission NO_x Programme	
	Ozone-depleting substances (ODS)	Quota
	RECLAIM (Regional Clean Air Incentives Market)	Quota
	Tradable development rights for Lake Tahoe watershed management	
	Tradable development rights for pinelands management	
	Tradable permits for lead in gasoline (1983–1987)	Quota
	Transferable fishing quota	Quota
	Transferable rights for wetlands conservation	
	Wisconsin – Lower Rox River Trading Scheme	Quota

Notes: VOC control: control of Volatile Organic Compounds, which comprise any carbon compound that evaporates under standard test conditions. All paint and caulk solvents except water are classified as VOCs. Government regulations limiting the amount of volatile organic compounds permitted in paint are in place in several countries.

NO_x emissions: Nitrogen Oxide emissions. Nitrogen Oxides is a term used to refer to nitric oxide (NO) and nitrogen dioxide (NO_2). The major sources of man-made NOx emissions are high-temperature combustion processes, such as those occurring in automobiles and power plants.

Source: OECD/EEA database on economic instruments and voluntary approaches used in environmental policy and natural resources management.

allowable emissions levels. The regulatory authority could achieve the same target at the same minimum cost in other ways:

- by setting the emissions level for each firm;
- by creating permits tradable as between firms with the total amount of emissions permitted equal to the arbitrary standard for total emissions.

We will look first at uniform emissions taxation, and then at these alternatives.

The least-cost property of uniform emissions taxation derives from the fact that it loads total abatement across firms such that those for which it costs less do more. Each firm would abate up to the level where its marginal abatement cost

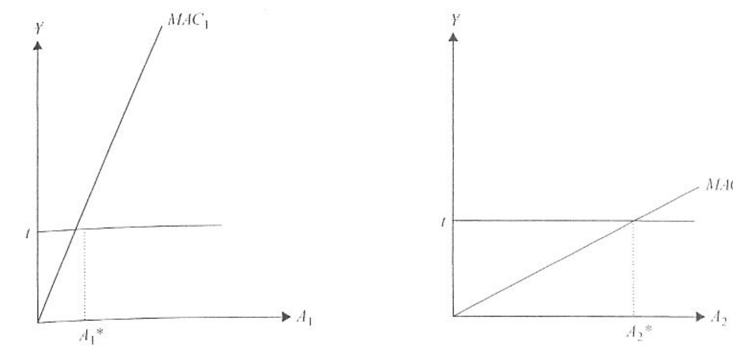

Figure 11.5 Least cost theorem.

was equal to the tax rate, so as to minimise its costs inclusive of tax. As a result, all firms would be operating with the same marginal abatement cost. Given this, there would be no reallocation of the total abatement cost as between firms that could reduce the total cost of abatement. The regulatory authority does not need to know each firm's abatement cost function to identify the differential abatement targets for each firm that minimise total cost. The reaction of the firms to the uniform tax identifies the least-cost solution.

Figure 11.5 shows how the least-cost property works for the case of two firms. The regulatory authority levies a tax at the rate t per unit of emission. Both firms will move toward the point t = MAC, because if t > MAC it would pay to abate more and if t < MAC it would pay to abate less. They will abate to levels A_1^* and A_2^*, which adds up to A^* total abatement. To see that this is least-cost combination of abatement by firm 1 and firm 2 that gives total abatement A^*, suppose that firm 1 abated less than A_1^* and firm 2 an equal amount more in order to stay at total abatement level A^*. The increase in cost for firm 2 is more than the decrease for firm 1. Try to think through for yourself the same idea, but with firm 2 abating less than A_2^* and firm 1 making up for it. What is the outcome in terms of total abatement costs? If the regulating authority knew MAC_1 and MAC_2, desiring total abatement A^*, it could work out what the cost minimising A_1^* and A_2^* were, and use command-and-control to require firms to abate these specified amounts of emissions. But the regulator does not know the necessary details of abatement costs for individual firms. Hence, it cannot work out these abatement levels and will normally go for an overall target of A^* via equal proportional cutbacks in each firm. This procedure is generally considered as 'fair'.

If command-and-control is to involve the same abatement cost total as uniform taxation, then the levels of abatement enforced by the environment regulatory authority will have to differ across firms such that more is done by those firms where abatement costs are lower. For the environment regulatory authority to be able to issue the individually tailored regulations, it would have to know the marginal abatement-cost function for each firm. This is generally infeasible, and the operational version of command-and-control is seen as involving each firm cutting back emissions by the same x per cent so as to achieve an overall x per cent reduction. Because firms' abatement-cost functions differ, uniform percentage cutbacks would not be efficient.

Not requiring complete information on the costs of abatement in each firm is what gives uniform taxation the cost advantage over command-and-control. However, a problem for the argument for uniform taxation as we have just presented it is that without complete information, the authority cannot calculate the tax rate that goes with its target level of abatement. In the absence of complete information in the authority about the firms' marginal abatement-cost functions, uniform taxation is not guaranteed to achieve the target level of total abatement. Hence, uniform taxation is not dependable.

In the absence of such information, the authority can set a tax rate, which will achieve some overall abatement level at the lowest cost that is possible. You can see that whatever is done is done at the least cost from Figure 11.5 – the least-cost property follows from the fact that both firms face the same tax rate. But, the achieved overall abatement level will not, except by chance, be the reduction in total emissions desired. The regulatory authority could proceed by trial and error, setting a tax rate, observing the outcome, and adjusting the rate up or down as necessary. This would involve additional adjustment costs, and would be very unpopular with the affected firms.

Tradable permits are both least cost and dependable without requiring that the regulatory authority has complete information about the costs of abatement in each firm. They are dependable because the total quantity of permits created by the environment regulatory authority is equal to the emissions total that corresponds to the desired overall cutback. They are least cost because the market in permits allocates them to firms where it is cheaper to buy a permit than to abate. Given that the permits are tradable, a single price for a unit permit will be established. The argument as to why they are least cost is exactly as for the uniform tax. All firms face the same price. The price will depend on the amount of permits issued, and given ideal conditions will turn out to be what the tax rate would have needed to be to reach the target desired. Firms where abatement is relatively cheap will abate rather than buy permits. The loading of total abatement across firms so as to minimise total cost is automatically generated by the cost-minimising behaviour of the firms.

This outcome, however, is crucially dependent on two other assumptions, namely that all of the firms are price-takers and that the environment regulatory authority can both monitor emissions and enforce compliance without cost. Where there are few firms, one may be able to exercise power in the permit market and distort its operation. Monitoring firms' compliance, ensuring that their actual emissions correspond to their permit holding, may involve the environmental regulatory authority in substantial costs. Note that such potential problems aside, the dependability and least-cost properties of permits hold irrespective of how the environmental regulatory authority initially allocates the permits. It could give polluters permits in proportion to past pollution (**grandfathering**) or it could sell them (e.g. by auction).

To conclude, how do tradable permits compare to environmental taxes? If we were ready to make strong simplifying assumptions, cost-minimising solutions could be calculated by the regulatory authority and either type of market-based policy instrument could be used with the same result. In reality, uncertainty about abatement costs (present and future), interdependencies of impacts, and responses of economic actors make the design of such policy instruments a less straightforward task. We know now that uncertainty about abatement costs may make quantity-based regulation (e.g. standards or tradable permits) more attractive. Because of the risk of missing the target when using taxes, in cases where it is considered important not to miss the target the regulatory authority will be attracted to tradable permits by virtue of their dependability.

11.8 ENVIRONMENTAL PERFORMANCE BONDS

Most environmental problems involve uncertainty and hence it has always been a difficulty in formulating and implementing environmental policy. Recall from the previous chapter that uncertainty differs from risk. While policy makers may know about the possible states following a decision, if they cannot attach probabilities to those states they are facing uncertainty rather than dealing with risk. In the previous chapter we focused on the severe problems which uncertainty causes for setting policy goals. In regard to implementation and policy instruments, we have just seen the difficulties arising from uncertainty.

A good example of the problems which uncertainty arising from major novel technologies creates for policy makers is provided by Genetically Modified Organisms (GMOs). They are particularly difficult to assess for policy makers, as there is no past experience according to which probabilities can be assigned to some of the possible adverse environmental outcomes. Some of these, according to some scientists, could have potentially huge irreversible impacts. How can governments deal with such situations? On what basis can they decide whether to embrace such innovations and reap the benefits for society or whether to stop them because they are too dangerous? What if it is impossible to make a well-informed decision at the beginning of a technological development? Are there any policy instruments that can help to address irreversibility and uncertainty in a satisfactory way?

Until a few decades ago, it was common practice in policy circles to ignore or deny the existence of uncertainty, or to apply arbitrary numerical fudge factors, and then to proceed as if everything was known with certainty, or at least that probabilities were known. More recently uncertainty has become more widely recognised and the need for the adoption of safe minimum standards and the precautionary principle, discussed in the previous chapter, is gaining widespread acceptance in policy-making and advice circles.

As we saw in the previous chapter, a major problem for the implementation of the precautionary principle remains that its definition and goals are vague, leaving its application dependent on interpretation by the regulators in regard to any particular problem. Another problem is a lack of experience with policy instruments aiming to operationalise and implement the principle. A rather recent idea for a policy instrument is that of environmental performance bonds. In general terms, a performance bond is a promise to pay compensation in the event of non-fulfilment of a particular contract. Performance bonds are common in the construction industry. Before a job begins, a construction company puts up a bond, i.e. an amount of money that is held by a third party. If the construction is completed satisfactorily and on time, the bond monies are returned to the construction company. However, if the work is unsatisfactory or late, part of, or the entire bond will be forfeited. An environmental performance bond is a deposit that possible polluters and violators of environmental standards must pay into an environmental fund to be held there until a specified period of time has elapsed.

It is aimed at providing a financial incentive to a firm undertaking a project or running some process to adhere to environmental requirements and to deal with uncertainty. Environmental performance bonds have a different aim to the command-and-control, taxation or tradable permit schemes, which we looked at earlier in this chapter. They do not directly focus on the reduction of emissions, but are designed to make companies responsible for the unknown environmental impact of their future activities. In keeping with the precautionary principle, they require a commitment of resources up front to offset potentially catastrophic future effects.

The basic idea is that before a firm introduces, for example, a new (chemical) substance or a new technology, a bond is fixed, which is equal in size to the current best estimate of the money value of the largest potential future environmental damage. The bond plus part of the interest is returned if the polluter proves that the suspected damage has not occurred or certainly will not occur. If damage does occur, the bond will be forfeited to a corresponding amount to the value placed on the damage. While the bond is being held it earns interest, and the part of the interest that is not returned to the polluter is used to finance the administration necessary for the environmental bonding system and research into environmental pollution-control technology and management.

Environmental performance bonds also create an incentive for the proponent of a project to conduct research to reduce the uncertainties about its environmental impacts. The incentive effect could be enhanced by having the size of the bond posted periodically adjustable. If the firm could show that the worst case was very unlikely to happen, part of their bond would be refunded to them. This would give proponents an incentive to fund independent research or, alternatively, to change to less damaging technologies. The implementation of the scheme requires an environmental regulatory agency with assistance from a scientific advisory board consisting of independent environmental experts.

In general, the applicability of environmental performance bonds is impeded by several factors. One difficulty is to measure the value of environmental damage in monetary terms. If it is impossible to express the damage in monetary terms, the part of the bond which is forfeited and the damage cannot be equivalent. The application of bonds might also be restricted by the necessity to prove causation. An additional problem arises if the actual damage is higher than the originally estimated maximum possible loss, as then the size of the bond is not sufficient to pay for the damage. Hence, the scheme only works successfully if the regulatory body takes a cautious view of the available evidence, implying a high amount for the performance bond, so that society would not find itself under-compensated. In order to avoid favouring big corporations over small and medium-sized enterprises (SMEs), organisational arrangements for cooperation of SMEs need to be devised.

The problems relating to SMEs are not unique to environmental performance bonds. While in most countries the share of firms classified as SMEs is about 98 per cent, it is by no means certain that environmental policy instruments that are effective with large companies are equally effective when applied to small and medium-sized ones. It is likely that they will need to be adapted to the particular circumstances of SMEs. In Europe there are few specific allowances made for SMEs in terms of environmental legislation; most Member States apply the same requirements to all companies. The UK and the Netherlands have given most consideration to sectoral and even company-specific economic issues. Other Member States worry that lighter administrative and legislative burdens for SMEs could lead to a lowering of environmental standards in such companies.

Given the problems encountered with uncertainty and irreversibility, environmental performance bonds are clearly an interesting idea for a potential addition to the existing menu of environmental policy instruments. To our knowledge they are not in use yet anywhere in the form set out above. Box 11.6 describes a proposed application of the idea. As with all environmental policy instruments, the success of an environmental bond scheme would depend heavily on the specific institutions created for its implementation. Especially, the procedure for setting the amount of the performance bond and the approach to addressing the problems related to SMEs will be crucial.

Box 11.6 Suggestion for an application of environmental performance bonds to wetland restoration

Wetlands are increasingly recognized as valuable natural systems providing useful services to society such as flood abatement, water purification, groundwater recharge, erosion control and biological diversity. International recognition of the value of wetlands is apparent through collective action in the Convention on Wetlands of International Importance (Ramsar Convention). Historical degradation of wetlands in the United States has led to a federal 'no net loss policy' for wetlands authorised by the Clean Water Act. The Clean Water Act mandates the restoration and maintenance of the chemical, physical and biological integrity of the nation's surface waters including certain wetlands. Gutrich and Hitzhusen (2004) review the situation in the state of Ohio and find that 90 per cent of the original wetlands have been converted over the last two centuries. Under the rules guiding the protection of water quality in Ohio, wetland losses are prohibited without compensatory 'mitigation': restoring or creating a wetland to make up for the one destroyed in the process of development.

Wetland mitigation is viewed as a means to balance the need for economic development with environmental protection. The extent and rate to which mitigation wetlands can replace the functions of natural ones remains uncertain and the value of the temporary loss of social wetland benefits have yet to be adequately addressed.

In an attempt to identify the ecological substitutability of mitigation inland freshwater marshes for natural ones, to estimate economic restoration lag costs to society and to address least-cost approaches to successful mitigation, Gutrich and Hitzhusen (2004) assessed sixteen mitigation wetlands, comprising of eight low-elevation inland freshwater emergent marshes in Ohio and eight high-elevation (>2,285 m) freshwater emergent marshes in a wetland complex in Colorado, USA. Years required for achieving full functional equivalency for both flora and soils for the Ohio sites under logistic growth ranged from 8 to 50 years with a median of 33 years. Years required for achieving floristic functional equivalency for the Colorado sites ranged from 10 to 16 years with a median of 13 years. Restoration lag costs are a function of ecological rates of wetland substitution and decrease with the increased ability of a mitigation wetland to restore all the functions of a natural site quickly. Per acre (0.4 ha) in Ohio restoration lag costs ranged from $3,460 to $49,811 per acre with an average of $16,640 per acre (US$ in 2000). This suggests that society is currently incurring significant wetland restoration costs due to time lags of mitigation sites. Therefore, Gutrich and Hitzhusen (2004) suggest the posting of an interest-accruing performance bond, which can serve to internalise the time lag costs to the permittee and provide an incentive for more cost-effective wetland restoration efforts.

11.9 INTERDEPENDENCE OF POLICY GOALS

While the main aim of environmental policy is clearly to achieve environmental targets, it cannot be pursued independently of other policy goals. In fundamental terms, we find that in complex evolving systems, such as the social–ecological systems which we are dealing with here, the effects of different policies interact and therefore policies in different areas cannot be pursued independently of one another.

For example, as we will see in Chapter 13, consideration of climate policy really needs to take into account the link between the extremely skewed international distribution of income, and human history over the last few centuries. The relevance of history has two dimensions. First, the risks of climate change are the result of an accumulation and long residence time of green house gases (GHGs) in the atmosphere. Second, economic history is characterised by unfair trade, colonialism and other historical contingencies. Western countries have a historical responsibility because they have enjoyed high economic growth since the Industrial Revolution associated with the intensive use of fossil fuels, the fundamental cause of the human contribution to GHGs in the atmosphere. The neglect of historical responsibility, and of equity issues, in some of the current analyses may serve to reinforce the pursuit of opportunistic strategies in international climate negotiations.

Interdependencies among policy goals will also influence whether a proposed environmental policy gets the necessary political support to become a reality. The differing distributional consequences of the various possible pollution control policy instruments will be very important in determining which instrument is selected in practice. For example, as we have seen, an emissions tax imposed upon fossil fuels will affect final consumers who purchase goods that have energy inputs, directly or indirectly, to their production as well as affecting purchases of the fuels for heating and the like. That means that all final consumers are affected to an extent that depends on the energy intensity of their consumption behaviour. Individuals, for example, for whom heating comprises a large proportion of their budget – as is typically the case with the poor in northern countries – may well experience quite large falls in real income following the imposition of fossil fuel taxation. Indeed, many kinds of 'green taxes' (environmental taxes) are likely to have regressive effects upon income distribution, that is to reduce the real incomes of the poor more than the better-off. Many European governments initially rejected energy taxes because they feared the negative distributional effects.

On the other hand, environmental policies may generate so-called double dividends. The first benefit, or dividend, is an improvement in the environment. The second dividend is a reduction in the overall economic costs associated with the tax system by using the revenue generated to displace other more distortionary taxes that slow economic growth at the same time. For more details on the second dividend see Further Reading.

SUMMARY

In this chapter we have studied the main characteristics of the available environmental policy instruments, and looked at examples in various countries. The main lesson is that there is no single instrument type that is best in all situations. Tradable permits can be depended on to meet the target, whereas taxation cannot. On the other hand, taxation raises revenue. Both taxation and tradable permits have abatement cost advantages over command-and-control-type instruments. Changing the information and/or preferences that people have can affect their behaviour and its environmental impact. The question of instrument choice, like the question of policy targets, is made much more difficult to deal with by the fact that there is usually a lot of uncertainty involved, and by the fact that an instrument adopted in pursuit of one target is likely to affect other sustainable development policy objectives.

KEYWORDS

Command-and-control instruments (p. 410): policy instruments, used for pollution control and the management of common property resources, which require polluters to meet specific emission-reduction targets and often require the installation and use of specific types of equipment to reduce emissions.

Environmental standard (p. 411): a quantifiable characteristic of the environment against which environmental quality can be assessed. It is a surrogate for the environmental values that are to be protected.

Grandfathering (p. 430): An initial allocation of emission permits which rejects the relative amounts emitted by the various sources prior to the introduction of the permit system.

Green taxes (p. 434): taxes with a potentially positive environmental impact, hence comprising energy taxes, transport taxes and taxes on pollution and resources; also called environmental taxes.

Least cost theorem (p. 427): total abatement costs are minimised where the regulatory authority taxes emissions from all firms at a uniform rate per unit.

Market-based instruments (p. 405): policy instruments which seek to address environmental problems via a price mechanism.

Moral suasion (p. 406): A type of approach used by an authority to get members to adhere to a policy, goal or initiative. It involves applying pressure on members, rather than using legislation or force, to achieve a desired result.

Pigouvian tax (p. 417): A tax on an externality, such as pollution, designed to use market forces to achieve an efficient allocation of resources. Named after A. C. Pigou, one of the first economists to study market failure due to externalities.

Property rights (p. 413): social institution carrying the right of ownership.

Regulation (p. 405): control by means of rules and principle; includes formal rules introduced by national and international governing bodies as well as traditions and group norms.

Tradable permit (transferable pollution permits) (p. 425): An environmental policy instrument under which rights to discharge pollution or exploit resources can be exchanged through either a free or a controlled permit-market. Examples include individual transferable quotas in fisheries, tradable depletion rights to mineral concessions and marketable discharge permits for water-borne effluents.

FURTHER READING

Common* (1996), Tietenberg* (2003), Harris* (2002) and Hanley *et al.** (2001) are introductory environmental economics textbooks which discuss in greater detail how policy instruments work, especially market-based instruments. Perman *et al.* (2003) is a good intermediate/advanced text which offers a thorough discussion of the least cost theorem. Common* (1995) explores policy options for sustainable development, not only environmental policy but also population and welfare policy. Articles in van den Bergh (1999) give an overview of different environmental policy instruments. A classic article which explores the advantages and disadvantages of price incentives compared to setting quantitative standards is Weitzman (1974). The role of the decision process for the outcome is explored in Frey and Stutzer (2002c). For further details about the double dividend of environmental taxation see Goulder (1994).

For the reader who wishes to explore tradable permits in greater depth, Tietenberg (2001) is a collection of leading articles important in the development of the use of emissions trading to control air pollution, from its earliest implementation in the USA in 1976, to its application to global warming in the Kyoto Protocol. A recent innovative suggestion for using tradable permit schemes for households is presented in Anderson and Starkey (2004). The Domestic Tradable Quotas (DTQ) scheme is premised on the assumption that stabilising greenhouse gas concentrations in the atmosphere at a level that will prevent dangerous anthropogenic interference with the climate system will require very large reductions in global greenhouse gas emissions. Furthermore it is assumed that these reductions will be achieved through some form of international agreement establishing binding national emissions reduction targets. The DTQ scheme is a new instrument designed to enable nations to meet the component of their emissions reduction targets that is related to energy use. A nation implementing a DTQ scheme establishes the maximum quantity of greenhouse gases that it can emit from energy use during any given year. This carbon budget is reduced year on year up to and including the year by which a nation must have achieved its emission reduction target. Each carbon budget is divided into carbon units, with, for example, 1 carbon unit representing 1 kg of carbon dioxide. A proportion of these units is allocated by government, free and on an equal per capita basis, to all adult citizens. This free allocation is known as the Entitlement. The remaining carbon units are allocated to firms and other organizations through a government-regulated auction. A computer database holds the carbon unit account for all citizens and organizations. The database records all carbon unit transactions – issuing, surrendering, buying or selling. See also OECD (2002).

In section 11.2.3 we discussed novel ways for measuring and reporting economic and environmental performance. The ISEW proposal first appeared in Daly and Cobb (1989), where it is calculated for the USA. Another example, showing a similar path over time, is in Stockhammer *et al.* (1997). Further references, and discussion, can be found in of Perman *et al.* (2003) ch. 19. A good straightforward account of the motivation for, and methods used in, ecological footprinting is Wackernagel and Rees (1996). The March 2000 issue (vol. 32, no. 3) of *Ecological Economics* has twelve articles on ecological footprinting, including several setting out its limitations.

Further discussions on environmental racism/classism can be found in Faber (1998), Cole and Foster (2001), Gottlieb (2001), Lester *et al.* (2001), Roberts and Toffolon-Weiss (2001) and Fletcher (2003).

WEBSITES

http://www.epa.gov/ – Environment Protection Agency (EPA), which develops and implements national environmental policy in the US.

http://www.eea.eu.int/ – European Environmental Agency (EEA), which delivers background information for environmental policy in Europe; their webpage has a helpful glossary: http://glossary.eea.eu.int/EEAGlossary

http://europa.eu.int/comm/environment/index_en.htm – European Commission (EC) – General Directorate Environment, which implements environmental policy in Europe.

http://www.oecd.org/department/0,2688,en_2649_33713_1_1_1_1_1,00.html – OECD – Environment Directorate, which advises OECD countries on environmental policy.

http://www.oecd.org/document/29/0,2340,en_2649_34295_1894685_1_1_1_1,00.html – OECD Environmentally Related Taxes database.

http://www2.oecd.org/ecoinst/queries/index/htm/ – OECD and EEA have also prepared a complementary database with information concerning other economic instruments (such as tradable permits schemes, deposit-refund systems and environmentally motivated subsidies) and voluntary approaches used in environmental policy.

http://www.colby.edu/personal/t/thtieten/trade.html – Bibliography on tradable permits compiled by Tom Tietenberg.

http://www.ieta.org/ – international tradable permit schemes.

http://www.europa.eu.int/comm/environment/climat/emission.htm – EU tradable permit scheme

http://www.defra.gov.uk/environment/climatechange/trading/uk/index.htm – UK tradable permit scheme

http://www.futureforests.com/acabalog/index_shop_calculators.asp/ – Carbon calculator for air travel

http://www.fairtrade.net/ – Fairtrade labelling organisation.

http://www.rprogress.org/ – the website for the organisation Redefining Progress, which has lots of material on ecological footprinting and the Genuine Progress Indicator.

DISCUSSION TOPICS

1. Proponents of eco-labels consider them as an alternative to traditional governmental regulation. Is this a viable option? What are the drawbacks of eco-labels?
2. Some environmental activists argue that tradable pollution permits are immoral because they confer the right to damage the natural environment. Do you agree?

EXERCISES

1 (a) Energy prices rose sharply in the early 1970s, stimulating interest in energy conservation. Companies installed more energy-efficient equipment, individuals insulated their homes, and the average fuel efficiency of the vehicle fleet was improved. Despite all this, UK energy demand was 210.1 million tons of oil equivalent in 1970 while by 2000 it was 231.9 million tons of oil equivalent. How can you explain this result?

(b) Part of the explanation may be the 'rebound effect', which describes the increase in demand for a resource, although more efficient technology is used. How does this relate to the claim of ecological economists that the focus on efficiency is insufficient, and that the total level of energy and material throughput needs to be targeted as well?

(c) Many ecological economists regard the current level, and pattern in terms of the fuels and resources used, of energy consumption as unsustainable. What scenarios and policy measures for this can you think of that would address their concerns?

APPENDIX INPUT – OUTPUT ANALYSIS OF CARBON TAXATION

This appendix builds on the Appendix in Chapter 5, and you may find it useful to go back and look again at it before working through this one.

We first extend the symbolic transactions table to include primary inputs:

	Sales to			
Purchases from	*Agriculture*	*Manufacturing*	*Final demand*	*Total Output*
Agriculture	0	Q_{AM}	F_A	Q_A
Manufacturing	Q_{MA}	0	F_M	Q_M
Primary input	V_A	V_M		

Note that we have now consolidated W and S and OFP into one primary input so as to simplify the algebra, and recall that OFP includes the payments for oil.

The corresponding coefficient table is:

	Agriculture	*Manufacturing*
Agriculture	0	a_{AM}
Manufacturing	a_{MA}	0
Primary inputs	v_A	v_M

where

$$a_{AM} \equiv \frac{Q_{AM}}{Q_M}$$
$$a_{MA} \equiv \frac{Q_{MA}}{Q_A}$$
$$v_A \equiv \frac{V_A}{Q_A}$$
$$v_M \equiv \frac{V_M}{Q_M}$$

With P for price and Q for quantity, we have

$$\text{Agriculture sales receipts} = P_A \times Q_A$$
$$\text{Agriculture expenditures} = (P_M \times Q_{MA}) + V_A$$

and

$$\text{Manufacturing sales receipts} = P_M \times Q_M$$
$$\text{Manufacturing expenditures} = (P_A \times Q_{AM}) + V_M$$

Since receipts are always equal to expenditures in input–output accounting, this is

$$P_A \times Q_A = (P_M \times Q_{MA}) + V_A$$

and

$$P_M \times Q_M = (P_A \times Q_{AM}) + V_M$$

The definitions of the coefficients mean that

$$Q_{AM} = a_{AM} \times Q_M$$
$$Q_{MA} = a_{MA} \times Q_A$$
$$V_A = v_A \times Q_A$$
$$V_M = v_M \times Q_M$$

and making these substitutions in the two sales equals receipts statements gives

$$P_A \times Q_A = (P_M \times a_{MA} \times Q_A) + (v_A \times Q_A)$$

and

$$P_M \times Q_M = (P_A \times a_{AM} \times Q_M) + (v_M \times Q_M)$$

where dividing both sides of the first by Q_A and of the second by Q_M gives

$$P_A = (a_{MA} \times P_M) + v_A \tag{1}$$

and

$$P_M = (a_{AM} \times P_A) + v_M \tag{2}$$

as a pair of simultaneous equations which can be solved for P_A and P_M in terms of the coefficients a_{MA}, v_A, a_{AM} and v_M.

Substituting from (2) into (1) and rearranging gives

$$P_A = \left(\frac{1}{1 - a_{AM} \times a_{MA}}\right) \times v_A + \left(\frac{a_{MA}}{1 - a_{AM} \times a_{MA}}\right) \times v_M \tag{3}$$

and using this to substitute in (2) leads to

$$P_M = \left(\frac{a_{AM}}{1 - a_{AM} \times a_{MA}}\right) \times v_A + \left(\frac{1}{1 - a_{AM} \times a_{MA}}\right) \times v_M \tag{4}$$

The prices of both commodities depend on all of the coefficients describing the structure of the economy – the input–output coefficients and the primary input coefficients.

For the numerical illustration from the chapter, repeated from Chapter 5, we have

	Agriculture	*Manufacturing*
Agriculture	0	$a_{AM} = 0.1$
Manufacturing	$a_{AM} = 0.6$	0
Primary inputs	$v_A = 0.4$	$v_M = 0.9$

and substituting these values in (3) and (4) leads to $P_A = 1$ and $P_M = 1$. This seemingly strange result arises from the fact that the entries in the transactions table are expenditures, that is they are price times quantity. When we treat them as quantities, as we have done here, we are measuring quantities in units that are (millions of) dollars' worth. This is standard practice in input–output accounting; as the industry sectors distinguished each produce many different actual commodities, this is the only way to proceed. The price of such a unit is just 1.

So, solving the equations (3) and (4) for the numerical coefficient values derived from the transactions table data gives the prices implicit in that data, which is as it should be. We can now use these equations to derive the result stated in the body of the chapter for the effect on prices of a carbon tax. With oil as base, the tax is \$100 per tonne, so that given the use of 1,000,000 tonnes of oil it costs Manufacturing \$100,000,000 or \$100 × 10^6. The tax is treated as an expenditure on primary inputs, which increase from \$1,800 million to \$1,900 million for Manufacturing. Hence, with the tax in place

$$v_M = 1900 \div 2000 = 0.95$$

and substituting in (3) and (4) for the original a_{AM}, a_{MA} and v_A and this value for v_M gives $P_A = 1.0319$ and $P_M = 1.0532$.

Using these prices with the original quantities for commodities

	Sales to			
Purchases from	*Agriculture*	*Manufacturing*	*Final demand*	*Total output*
Agriculture	0	200	800	1000
Manufacturing	600	0	1400	2000

gives expenditure flows, in \$ million as:

	Sales to			
Purchases from	*Agriculture*	*Manufacturing*	*Final demand*	*Total output*
Agriculture	0	206.38	825.52	1031.9
Manufacturing	632.92	0	1474.48	2106.4

where subtracting intermediate purchases from total sales revenue gives 399.98 for Agriculture and 1,900.02 for Manufacturing. Allowing for rounding errors,

these are the payments for primary inputs in each sector with the carbon tax in place.

With many sectors, the algebraic method used here will not work to solve many simultaneous equations. However, multiple simultaneous equations can readily be solved using matrix algebra, where the arithmetic can be done quickly using a spreadsheet such as Excel™. The interested reader can find out more about matrix algebra from references in the Further Reading section of Chapter 5.