Economic incentives for optimal sulphur abatement in Europe

Halkos, George

University of Thessaly, Department of Economics

1993

Economic incentives for optimal sulphur abatement in Europe

by

George E. Halkos

Abstract
This paper reviews and develops a theoretical and empirical representation of economic incentives for the implementation of pollution control strategies. A number of alternative available economic instruments may be thought of which, if applied internationally, could encourage implementation of the desired abatement strategies by countries. The paper considers means of pushing the countries to minimize abatement cost with them. A comparison between the pollution targets achieved by the imposition of a uniform charge rate and by differentiated charge rates is discussed and empirical results are provided with associated conclusions. These results are then compared with a simple standards setting in the form of critical loads, in order to see in an empirical way if economic instruments work better than regulations.

Keywords: Economic instruments; optimal abatement; sulphur emissions; acid rain; mathematical programming, Europe.

JEL Classifications: C60, C71, Q50, Q52, Q53, N53.

An earlier version of this paper has been presented as:

Economic incentives for the implementation of optimal sulphur abatement strategies in Europe, University of York, Department of Environmental Economics and Environmental Management, Discussion Paper Series **93-04**.

INTRODUCTION

Over the past few years acid rain has become one of the most discussed environmental issues affecting Europe. It is a problem with technical, economic and political components. Technical problems results from atmospheric pollution which causes damage to forests, crops, lakes, human health and buildings. Economic issues arise in the form of negative externalities since emissions from one country may be deposited in the territory of its neighbours. Finally there is a political aspect of the problem because of the need of countries to formulate appropriate policies.

The recognition of acid rain as an externality is vital in economic policy. The OECD has studied the long range transport of air pollutants in Europe. The findings show that SO_2 is transported over long distances and that environmental quality in any one European country is significantly affected by the actions of others. It is thus a classic case of externality: the user-country may have little reason to concern itself with the sulphur content of the fuels it uses, except to the extent that legislation or ‘moral’ incentives force it to take account of emissions. The OECD (1979) results suggest that 4% of total U.K. emissions are deposited in Scandinavia and this constitutes 8% of total Scandinavia depositions. It has been estimated that more than 75% of the sulphur deposited in Sweden originates in other countries, mainly continental Europe (north of Alps) and U.K. On the other hand, a substantial part of the sulphur emitted in Sweden is deposited in Finland and in the northern Soviet Union. In this sense, the acidification problem is truly international, as there are external diseconomy problems between countries.

The presence of transnational externalities implies that gains can be realized in a cooperative behaviour. As there is no international or multinational ‘government’ that can enforce international environmental policy, these problems must be solved by voluntary agreements among the countries concerned. The problem is that of finding some institutional structure that will facilitate the appropriate agreements. Such a structure must be one that makes all parties (countries) better off. Otherwise, any agreement is unlikely. We thus seek structures that promise a Pareto-efficient outcome. This paper will review and develop the theoretical and empirical representation of economic incentives for the implementation of pollution control strategies. The analysis will be developed using the example of sulphur as the polluting substance to control.

BACKGROUND

Externalities can take either of two forms: a public (undepletable) form or a private (depletable or rival) form. An externality is undepletable (public) when an increase in the consumption of a good or a bad by one individual does not reduce its availability to others (e.g. noise). Freeman (1982) illustrates the phenomenon of depletable (private) externalities with the case of acid rain. In this instance, the sulphur emissions from a particular source are distributed in the form of acid-rain. The word 'depletable' means that 'each pound of sulphur emitted to the atmosphere must land somewhere and if the quantity falling on A's land increase, there is less to fall elsewhere'. The real issue is whether individual polluters can influence the distribution of pollution once it has fallen. For example, forests and plantation absorb more sulphur than do open areas. This filtering effect shifts the ultimate deposition location. Hence, self-protection through afforestation can filter the externality to another agent. Similarly, building a central waste disposal facility to handle point pollution in a particular location may filter and redistribute the waste. In this way, it is possible for polluters to influence the distribution and hence the value of acid deposition.

Acid rain is clearly a global problem in that pollution emissions may be deposited anywhere in the world, including the country of origin. For practical purposes acid rain may be viewed as a regional reciprocal externality where a group of countries is both the source and the victim of the problem. An important characteristic of this is that transfers of pollution between countries may be very unequal in quantity because of the existence of prevailing winds. There are two problems associated with regional reciprocal externalities which must be addressed in modelling the cost of acid rain abatement. The first is the need to achieve optimal economic efficiency across countries in pollution abatement. If this condition is not met scarce economic resources will be wasted. The second is the classic economic problem of the free rider: countries will benefit from pollution abatement in other countries at no cost.

The selection of the appropriate strategies to reach and implement pollution control objectives is of crucial importance to planners. Because of the existing differences between countries in energy–use patterns, emissions, source location and other economic factors, it is unlikely that a single, uniform program of abatement will be appropriate in all countries. Appropriateness is defined in terms of cost-effectiveness compatible with specific pollution control goal attainment, but also in terms of political and social acceptability as well as administrative feasibility. In other words, we must also take into account, for most developed countries, the increasing role of 'ecological' groups ('Greens') and various associations of environment–protection, the requirement for human population's protection, the necessity of purchase, installation and improvement of the appropriate abatement technologies, personnel training, maintenance costs, and so on. Therefore, required emissions reductions and the corresponding expenditures will vary between countries. A deposition target in a given country may be achievable at a lower cost by emissions' reduction in neighbouring countries. However, countries required to make large expenditures may be unwilling to pay, especially if the corresponding benefits are likely to be gained in other countries.

Economists have for many years proposed that decentralized-based policies are more efficient than centralized command–and–control approaches as the solution to the problem of how to regulate air pollution cost-effectively in these circumstances (Schultze, C., 1977)(1). Tietenberg (1990) has estimated the relative costs of regulations compared to market mechanisms in eleven cases. In four of these, regulations were 1 to 2 times as expensive as market mechanisms, in 5 cases regulations were 2 to 10 times more expensive, and in two cases more than 10 times more costly than market mechanisms.

Mäler (1989, 1990), assuming that marginal damage costs are constant and independent of the amount of depositions (i.e. a linear damage function), claims that achieving a second best solution by the imposition of a uniform charge rate per tonne of sulphur exported from one country to all other countries will be a satisfactory solution on the airborne export of sulphur. In the first–best optimum one should differentiate charge rates for different countries. Mäler believes that differentiation of charge rates would create not only practical problems but also obstacles to reaching an agreement and he suggests that the gains from going to the first-best optimum seem to be marginal, but he does not provide enough empirical foundation regarding the last of his conclusions. It is clear that a tax system such as the one proposed by Mäler will not achieve a first-best optimum, as it will not differentiate between exports to countries with high marginal damage cost and countries with low damage cost. Any attempt to apply the same charge to each unit of sulphur emitted, irrespective of where the emissions occur, is itself extremely inefficient.

This paper compares the costs of meeting pollution targets by the imposition of a uniform charge rate and by differentiated charge rates. These costs are then compared with a simple standards setting in the form of critical loads in order to see in an empirical way if ‘economic instruments’ work better than ‘regulations’. Section 1 offers a mathematical model for determining optimal abatement strategies. Section 2 explains the ways of implementing optimal abatement strategies for pollution control. A theoretical presentation of policy or economic instruments and the existing forms of these instruments in Europe follows. Section 3 reports the empirical results obtained. Finally, some general conclusions obtained from this comparison are presented.

1. A non-linear programming model for determining cost-effective control strategies

For determining the cost-effective abatement strategies in Europe, to be written in its simplest form as follows:

$$\text{Minimize} \qquad \sum_{i=1}^{27} C_i(SA_i) \qquad (1)$$

$$\text{Subject to} \qquad \sum_{i=1}^{27} d_{ji}(SE_i - SA_i) \leq \bar{D}_j - B_j \qquad j=1,2,...,27$$

$$SA_i \geq 0 \quad SE_i \geq 0 \qquad i=1,2,...,27$$

where the 27 European countries are considered simultaneously, and where SE_i is the quantity of sulphur emitted in country i, called unconstrained emissions and which are assumed to be exogenous in this mathematical problem; SA_i is the quantity of sulphur (in tones per year) to be abated in country i, i.e. the decision variable; C_i is the cost of abatement in country i ; and d_{ji} is the proportion of country i's emissions deposited ('exported') at receptor- country j, called the transfer coefficient ($0 \leq d_{ji} \leq 1$)$^{(2)}$; $\bar{D}_j$ is the maximum allowance of deposition in country j, called targeted or constrained depositions, and B_j is the level of depositions caused by natural sources in receptor- country j, called background pollution$^{(3)}$. The fact that pollution may remain in the atmosphere for long periods of time is addressed in the background deposition coefficient. SA_i, SE_i, $\bar{D}_j$ and B_j are expressed in tonnes (t).

Each of the 27 constraints (one for each European country) indicates the minimum annual abatement of sulphur depositions D_j to be secured in country j. These reductions are to be achieved by abating the sulphur emitted in each of the 27 European countries under consideration. Table 1 presents the unconstrained sulphur emissions and the associated deposition levels in the year 2000(4). The function C_i (SA_i) which is the objective function in (1), is a non-linear cost function (convex upward- sloping curve implying marginal cost increasing with removal level), giving the cost in country i of achieving any level of emissions reduction SA_i by means of the control technologies available for sulphur abatement(5).

The following abatement technologies, involving different levels of costs and applicability (depending on the physical and chemical characteristics of the fuel used), exist in most industrialized countries:

(a) gas oil desulphurization,

(b) heavy fuel oil desulphurization,

(c) hard coal washing,

(d) in furnace direct limestone injection,

(e) flue gas desulphurization and

(f) fluidized bed combustion.

The actual control costs of each abatement technology are defined by national circumstances and the abatement cost curves depend on the energy scenario adopted. Abatement costs differ considerably among countries even for the same technology, mainly due to country-specific factors such as sulphur content of fuels used, capacity utilization, size of installations and labour, electricity and construction cost factors. These cost input data were obtained country by country, sector by sector and fuel by fuel, where 27 countries (i.e. all Europe), 5 sectors and 10 fuels were considered, for the year 2000(6) (Halkos, 1992, 1993).

Table 1: Unconstrained emissions in the year 2000 (Emissions), actual depositions (Depositions) and targeted deposition according to critical loads (Targeted depositions) (all in thousand tonnes of sulphur)

Countries	Emissions	Depositions	Targeted depositions
Albania	200.00	154.00	84.40
Austria	187.00	376.00	226.70
Belgium	329.00	180.00	77.75
Bulgaria	961.00	702.00	217.53
Czechoslovakia	1216.00	1450.00	1128.80
Denmark	143.00	109.00	5.10
Finland	270.00	423.00	282.30
France	765.00	1183.00	194.95
GDR	2099.00	952.00	572.10
FRG	1556.00	1235.00	463.35
Greece	450.00	594.00	300.66
Hungary	467.00	436.00	22.94
Ireland	83.00	71.00	7.82
Italy	1715.00	1342.00	463.40
Luxembourg	15.00	7.00	1.00
Netherlands	242.00	199.00	48.00
Norway	69.00	247.00	82.00
Poland	2182.00	1859.00	685.70
Portugal	218.00	232.00	94.50
Romania	1374.00	1331.00	315.40
Spain	2573.00	2012.00	738.50
Sweden	248.00	534.00	319.30
Switzerland	50.00	134.00	58.30
Turkey	2203.00	1976.00	305.60
USSR	10890.00	13921.00	3531.80
UK	1844.00	1117.00	458.96
Yugoslavia	1891.00	1466.00	635.07
Total	34240.00	34242.00	11321.93

The important initial assumptions for the derivation of these national abatement cost curves are the following:

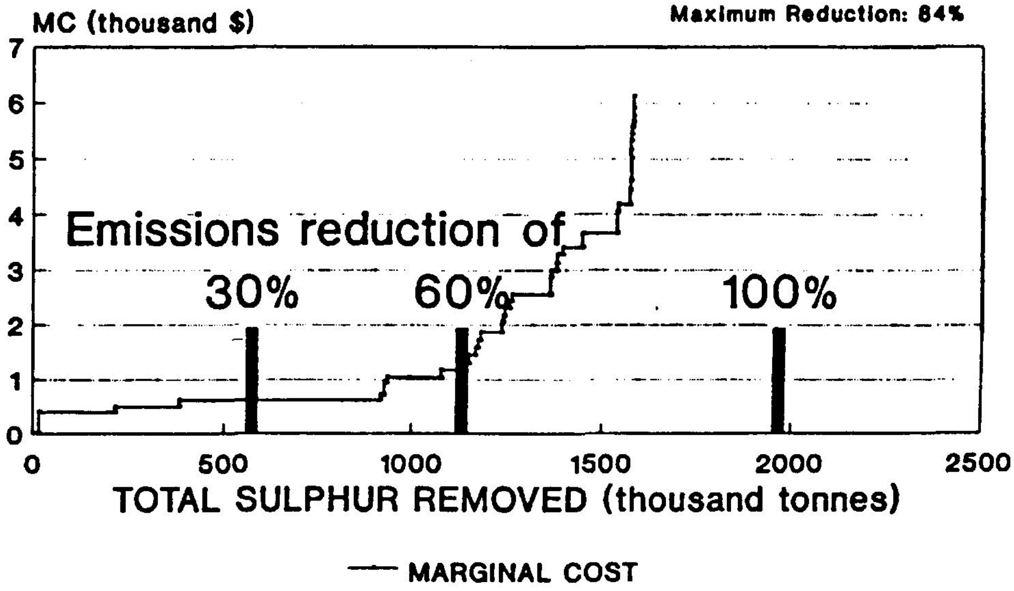
- First, control costs are independent of order of introduction.
- Second, each abatement technology has a fixed coefficient of abatement when operating at its defined capacity. For example, an FGD unit has an abatement efficiency of 90% (i.e. removes 90% of the sulphur content of the fuel in use) at the efficient plant size, while a sorbent limestone injection unit has an abatement efficiency of 50% at the efficient plant size.
- Third, it is assumed that the objective of private users is to minimize the costs of abating a given level of emissions.
- Further, fuel use and costs are assumed given independently of abatement policy. For the purposes of this exercise, then, abatement by means of reducing the output of electricity or other industrial output is ruled out.
- Finally, another basic assumption of the cost module is that there is a competitive market for sulphur abatement technologies accessible to all European countries.

Resulting annualized costs of the abatement technologies used for sulphur reduction are expressed in United States $ at their 1985 levels (Halkos, 1992, 1993).

The economic efficiency of alternative abatement options (expressed as $ per tonne pollutant removed) depends on site specific conditions, and a least cost emission control function for each source can be estimated by ranking alternative options in order of increasing marginal cost of control. To do so, technologies are sorted so that marginal abatement costs increase with the level of abatement sought. Marginal costs increases are due to the effect of switching between technologies as the scale or level of abatement rises. The marginal cost curve has a staircase shape (i.e. it is a discontinuous step function) with each step representing the incremental effect of a particular discrete abatement technology. The level of each step indicates the incremental cost of a technology, and the range of each step the maximum incremental amount of sulphur removed by introducing that technology. The sequence of efficient technologies gives us the long run marginal cost of abatement. If an abatement technology is introduced which has a lower marginal cost at some level of abatement than the technology applied before, then this technology should have been applied first. The control methods applied before are not taken into consideration. The most cost–effective techniques are the proper abatement techniques for the national decision maker.

It is assumed that the regulatory authority seeks to maximize abatement subject to a budget constraint: a cheaper option will always be preferred over a more costly one. It would be economically inefficient to introduce relatively costly control options unless opportunities for using cheaper alternatives had already been exhausted. The relative economic efficiency of alternative options is compared by reference to "cost-effectiveness" which, for a given option at a given site, is the total annualized cost divided by the annual tonnes of pollutant removed. This type of cost function is potentially useful to policy-makers because it indicates the maximum level of emission abatement that can be achieved with a given budget constraint. That is, we look for an efficient frontier or a minimal cost envelope, which will give us the optimal total cost function; i.e. the corresponding point on the marginal cost curve specifies the set of country control options which minimize total abatement costs (Rubin et al., 1986; Baumol & Oates, 1979; Kneese & Schultze, 1975; Mäler, 1990). Figure 1 presents an example of the marginal abatement cost for the FRG (Halkos, 1992)(7). Let us now turn to the presentation of the available economic incentives for the implementation of the optimal abatement strategies.

FIGURE 1: FRG YEAR 2000
MARGINAL ABATEMENT COST CURVE



2. Policy or economic instruments

Environmental control approaches include actions to protect the environment using policy instruments, such as regulations or economic instruments like charges, taxes and marketable emission permits (or licences). Economic instruments tend to operate more directly than regulations. For example, by taxing polluters, the government raises their costs ideally bringing their cost curves up to the true social curve (including the cost of pollution) thereby internalizing the externality. On the other hand, regulations will generally affect internalization indirectly, by requiring countries to change behaviour in a given way to reduce pollution. Regulations can vary from the extreme case of prohibition and/or the imposition of production quotas on producers to the setting of ‘standards’ for protecting selected target populations. In the air pollution field, the use of standards is based on motions governing the relationship between emissions at the source level and concentrations in specific air sheds. Direct regulatory instruments, also known as ‘command and control’ mechanisms, are enforcement mechanisms and regulations on activities affecting the environment, such as environmental (air, water, soil) quantity regulations (in the form of emission standards, fuel quality regulations and fuel use regulations).

Enforcement policies rely on a variety of legal instruments, ranging from licence withdrawal to criminal prosecution. Environmental quality standards protect human health or ecosystems. Indicators of ‘quality’ are precisely defined as allowable average concentrations over a specific time period for a given pollutant in a particular region. The standards are usually based on scientific dose-response relationship. That is, the expected health response results from a given dose of pollutant. Critical loads are used in some countries as a basis for the definition of environmental quality standards. From a relative sensitivity map of ecosystems applied to the indirect effects of acidic depositions in Europe constructed by Chadwick and Kuylenstierna (1990) it is possible for each country to calculate the area of land in each of five "sensitivity" categories.

To each class (category) corresponds a maximum deposition allowance, or ecosystem sensitivity threshold, which represents the maximum acceptable level of sulphur deposition on these points, denoted by β_k (k=1, 2, 3, 4, 5), and expressed in tonnes of sulphur per square kilometre (t/km^2) per year. These are:

$$\begin{array}{ll}
\text{Class 1} & \beta_1 \geq 5.12 \ \text{t/km}^2 \text{ per year} \\
\text{Class 2} & \beta_2 = 2.56 \ \text{t/km}^2 \text{ per year} \\
\text{Class 3} & \beta_3 = 1.28 \ \text{t/km}^2 \text{ per year} \\
\text{Class 4} & \beta_4 = 0.64 \ \text{t/km}^2 \text{ per year} \\
\text{Class 5} & \beta_5 = 0.32 \ \text{t/km}^2 \text{ per year}
\end{array} \tag{2}$$

These critical loads are based on ecological criteria for which data for the entire continent are available, like geology, soil type, vegetation and amount of rainfall (for more details regarding the way that these factors are weighted to give these classes of relative sensitivity to acidic depositions, see Chadwick and Kuylenstierna, 1990). The authors of this categorization estimate that the most sensitive type of terrain (class 5) is able to tolerate at most 0.32 tonnes of sulphur depositions per square kilometre per annum without suffering ecological damage. At the other extreme, class 1 terrain is assumed to be capable of tolerating 5.12 tonnes per square kilometre per annum. Therefore, the classes are ranked in order of increasing sensitivity: class 1 refers to the least sensitive regions and class 5 to the most sensitive regions of each country. Let us call these regions "sensitivity areas".

Most of the data required by the model are readily available. The country–specific cost functions were explained in section 1. EMEP (1989) gives estimates of the flows of sulphur depositions between countries and these are used to compute the transfer coefficients $d_{ji.}$ Accordingly, we construct the following sensitivity index:

$$\overline{D}_j = D_j - \sum_{k=1}^{5} (D_{jk} - a_{jk}\beta_{jk}) \qquad (3)$$

Where a_{ik} is the area in nation i lying in sensitivity category k; D_{jk} are the actual depositions in each sensitivity area of a given country j; D_j are the actual depositions in country j; and β_{jk} the critical load for each sensitivity class k (k= 1, 2, 3, 4, 5) given by (2) in country j. Table 1 presents the targeted depositions according to critical loads. The abatement requirement of achieving $\overline{D_i}$ therefore reflects current annual depositions in excess of the environmental sensitivity limits proposed by Chadwick and Kuylestierna. It is an index of severity of current pollution in the country in excess of some uniform benchmark.

Of course the abatement of sulphur pollution cannot be directed to specific geographical areas. Thus, although the scheme described above gives a realistic measure of levels of depositions harmful to the environment, it does not necessarily reflect a realistic level of abatement required to eliminate ecological damage. It implicitly assumes that abatement can be directed in the precise quantities to those areas suffering environmental harm from sulphur depositions. In practice, of course, acid rain is indiscriminate in the areas it pollutes. Therefore, an important assumption in expression (3) is that air pollutants fall uniformly on the country's territory, i.e. that climatic factors which could determine a higher concentration of depositions in some regions rather than others such as rain, prevailing winds, variability in the level of precipitations and other atmospheric phenomena do not have any effect here. This assumption is made for simplicity, because of the obvious difficulty of measuring such effects and because the analysis needs to be made practicable.

In practice, environmental quality standards take many forms like emissions restrictions, restrictions on pollution per unit of an input and restrictions on the use of a polluting input. The control of sulphur emissions by emission standards is widely used in air and water pollution (Vernon, 1990). They set a maximum allowable rate of pollution output for each generic type of source (electricity, industry, petroleum refineries and transport) by type of pollutant(8). Furthermore, fuel quality regulations are structured around the types of fuels in use (e.g. coal, oil etc) and are limited by the technical possibilities and the costs of cleaning process for the different fuels. At present, varying types of fuel-quality standards are in use in nearly all OECD countries. The range of standards varies from 0.2% on light and medium fuel oil to 0.3% on gas diesel oil. Control on fuel use has been applied as a strategy for air pollution reduction on a permanent or temporary basis to satisfy general environmental and health concerns. In some heavily polluted areas such as Ankara, coal is restricted in winter

(Vernon, 1990)

The second way of implementing optimal abatement strategies for pollution control is by the use of emissions' charges or taxes to encourage abatement. Pollution taxes or charges are based on Pigou's concept of increasing the costs of pollution activities so that they reflect the true social cost to society of those activities through environmental damage(9). The essence of charges approach is for a tax to be imposed on each unit of sulphur emitted. This implies revenues that can go to a Central Authority for the creation of a European fund for air pollution control (Bergman, 1986; Hettelingh and Hordijk, 1986; and Mäler, 1990). In this case this International Authority would impose a charge on uncontrolled emissions. The Authority could then distribute any tax revenue to subsidize further emissions' abatements.

However, the Authority would need to know the shape of the corresponding abatement cost functions to be able to set the appropriate emissions' charge. The emissions' charge yielding the cost-effective **uniform emissions** abatement strategy can be determined as follows:

$$\underset{SA_i}{Minimize} \quad \sum_{i=1}^{27}\left[C_i(SA_i) + p(SE_i - SA_i)\right] \tag{4}$$

p being the emissions' charge, and where all the other variables have the meaning explained in section 1. But acidic depositions vary greatly by location and with time as well. Uniform targets are set with no reference to marginal damage done by the transportation of pollutants over long distances and their acidic depositions. Additionally, if the relationship between source location and receptor location are not taken into account then the fundamental aspect of externality (represented by the transfer coefficients) is not taken under consideration.

By contrast with the uniform emissions abatement strategy, the emissions' charge yielding the most cost-effective **differentiated emissions** abatement strategy would be determined as follows:

$$\underset{SA_i}{Minimize} \quad \sum_{i=1}^{27} C_i(SA_i) + \sum_{j=1}^{27} p_j[d_{ji}(SE_i - SA_i)] \tag{5}$$

where $\mathbf{p_j}$ is the charge for depositions at receptor-country (or source) j. In the case of our mathematical model, the emissions' charge paid by polluters in the cases corresponding to expressions (4) and (5) should then be set respectively as follows:

$$\sum_{i=1}^{27} p(SA_i - SA_i^*) \tag{6}$$

and:

$$\sum_{i=1}^{27}\sum_{j=1}^{27} p_j d_{ji}(SA_i - SA_i^*) \tag{7}$$

where SA_i^* denotes the optimal required emissions' reduction of country i according to the mathematical model and under the critical loads scenario (see expressions (1) –(3), while SA_i denotes the emissions' reduction strategy effectively chosen by country i (10).

Economic theory indicates that the optimum rate of pollution charges is at the level where the marginal abatement cost is equal to the marginal damage cost of the pollution it is intended to abate. In a 'first-best policy' one should differentiate the tax rate between different exporters according to the size of their damage costs. In the case of limited available information a 'second-best' but still cost-effective solution is to set a level of uniform charge high enough to ensure that polluters will abate pollution to a target level of pollution. Such a level of charges is often too high to be acceptable or enforceable for political or other reasons. A feasible form of tax in the case of most sulphur emissions is one related to the sulphur content of fuels burnt. Any given tax on the sulphur content of fossil fuels will lead to desulphurization up to the point where the marginal desulphurization cost per unit of sulphur abated equals the tax.

Market incentives such as taxes, charges and permits can lead to solutions superior to that of a regulation instrument, provided, however, that the prices or quantities designed to achieve a given air quality objective reflect accurately the social costs of pollution, i.e. some estimate of the damage suffered by the community if the pollution targets (under the critical loads scenario) are not respected. In any case, even if this assumption is relaxed (as in this model, which does not consider any estimate of the social costs of air pollution), such market approaches can still be considered superior to regulatory ones, since they afford polluters the opportunity to avoid paying penalties by striving for a greater abatement cost-effectiveness (whereas regulated standards would just be 'imposed' on them). It would be efficient for each country to arrange for implementation of emissions' abatement up to the point where the marginal cost of abatement is less than or equal to the emissions' charge; and to pay the charge emissions which are relative more expensive to abate.

Under a system of emission charges, regulators set prices for emission levels that are designed to achieve a given air quality objective. With the third class of instruments, i.e. marketable emission licences, regulators establish the quantity of emissions that would achieve a given air objective, issue permits to pollute this amount and leave it to the market to decide the value of these permits. In theory, both approaches are equivalent in a perfect world of zero cost information, administration and legal enforcement in so far as they both minimize the cost of achieving a given level of air quality but in practice the two approaches are different. With marketable emission permits, the problem of the Authority having to know the shape of abatement cost functions is avoided, because in this case countries would be allocated a specific number of licences defining the amount of pollutants they would be allowed to emit in any given period. The initial number of licences issued in each country would depend on their contribution in sensitive receptor areas and on their required depositions' reduction. In countries upwind of sensitive areas a greater number of licences would be required to emit one unit of pollutant than in countries downwind.

Grubb (1989), Hoel (1990) and Pearce (1990) have suggested a number of approaches to the initial allocation of permits once the total limit on emissions has been agreed. Such an approach should rely on the current pollutant's emissions (in this case sulphur) or the current gross national product as far as energy use is linked to economic activity. However, both approaches reward polluting countries and restrict developing countries. Grubb (1989) suggests that pollution should be the most equitable basis for allocation, but under the condition that only adult population counts, to prevent giving an incentive to increase population and to reduce the relative benefit to developing countries, which have much higher proportion of children in their populations. Finally, another approach is by using the land area. This approach has the advantage that can be measured easily and would discourage high population densities, but the lack of a link to human activity makes it impracticable.

In order to determine whether the market will be sufficiently competitive to produce an efficient result (enough participants and transactions), one needs to be able to forecast the final distribution of permits. Whether the market for permits has monopolistic features that undermine its efficiency depends on whether one, two or more countries account for a large share of either sales or purchases (Hahn and Noll, 1983). To predict the concentration in permits transactions also requires solving the cost-minimizing problem for participants in the market. From this, one can predict a final distribution of permits. This can be compared to the initial distribution to generate an estimate of net sales (or purchases) by each country (through the central authority), which then can be used to calculate expected market shares. The main feature of emissions' licences is that they are tradable, so that countries in which abatement is relatively expensive would be able to buy extra permits rather than pollution control equipment.

Formally, the trading system which would yield the cost effective deposition reduction strategy involves a separate market in emission licenses for all receptor countries. Each country (or source) would have to purchase sufficient licences (defined in terms of deposition units at the receptor) in each market to cover its emissions' rate. The problem faced by the country (or source) is then to minimize the sum of expenditures on emissions' controls and licences. This can be expressed as follows:

$$\underset{SA_i}{Minimize} \quad \sum_{i=1}^{27} C_i(SA_i) + \sum_{j=1}^{27} p_j[d_{ji}(SE_i - SA_i) - q_{ji}^0] \tag{8}$$

Where **p_j** is the prevailing price in receptor-market j and q_{ji}^o is the initial allocation of licences at receptor-market i allowed to country (or source). However, in such a situation the Authority would not have to set **P_j** : the market itself would find this price, given the initial allocation of licences. The role of the Authority would essentially be to determine the initial allocation of licences and to supervise the buying and selling of licences between the countries. Separate bodies could be set up to supervise swapping of licences within Western and Eastern Europe, with an overall coordinating body to arrange such swaps. In the case of our mathematical model, one possible initial allocation of licences for a given country could be determined as follows:

$$q_{ji}^0 = f(d_{ji}, D_j - \bar{D}_j) \tag{9}$$

where q_{ji}^o depends on each country's contribution to deposition in sensitive receptor areas and on its required depositions' reduction, being the constrained depositions $\overline{D_j}$ determined according to the critical loads scenario (see table 1).

It would be then convenient for countries to buy permits rather than further reduce emissions, whenever $q_{ji}^o < MAC_i\,(SA_i^*)$ where $MAC_i\,(SA_i^*)$ is the marginal abatement cost of country i and $\mathbf{SA}_i^*$ the corresponding optimal emissions' reduction determined by the non-linear programming model. It is worth mentioning that a permanent allocation of permits and trading on a permanent basis, creates the problem of powerful parties hoarding rights for future use rather than trading. Instead the periodic 're-issuing' of permits according to the initial allocation would amount to a system in which permits were leased rather than sold and hoarding would not be possible. Finally, the trading requires some sort of enforcement procedure, with penalties for countries exceeding the permitted emissions. Hahn and Hester (1989) showed that excessively bureaucratic monitoring systems in the USA impede trading.

To sum up, the obvious advantage of permits over charges systems is that the former avoids the problem of the authority's uncertainty about abatement costs. The consequence of an underestimate of abatement costs in the presence of permits is simply that the price of permits is forced-up, whereas the environmental standard is maintained (Rose-Ackerman, 1977). The charge approach also risks underestimating abatement costs. If the authority is wrong about the abatement costs, the charge could be set too low in the sense that polluters will prefer to pay in than to invest in abatement equipment, thus sacrificing the desired standard. Our next step is to consider the empirical results obtained.

3. Empirical results

Every European country has its own specific national environmental legislation and regulations. Some regulations are general, but many are specific on, for example, ambient air quality standards, fuel quality standards, emission standards, licences etc. Environmental standards and the relevant legislation in the eastern European countries are more lax than the EU countries and monitoring is extremely poor. Economic instruments generally have proved to have several advantages over regulations. Some of the main aspects of the available instruments are now considered to see where the differences are, when they are used for the implementation of optimal abatement strategies.

For optimal economic efficiency, a regulatory system that considers each specific source of emissions assumes that regulators know enough about the production process they are inspecting, and the abatement opportunities applicable to it, to be able to determine the optimal emissions reduction for it. Countries are likely to be reluctant to provide accurate information to regulators, because some abatement strategies may involve changes in the production process which, if revealed as a result of regulation would give away commercial secrets. Consequently, standards are unlikely to provide the most cost–effective method of air pollution abatement. If countries are cost-minimizers, emissions taxes can lead to the cost-minimizing solution (Burrows, 1980; Baumol and Oates, 1989; Kneeze and Shultze, 1975; Pearce and Turner, 1990).

On the contrary, a policy of regulation could achieve this least-cost allocation only if the individual polluters' abatement costs were revealed to the policy maker(11). Moreover, marketable permits and emissions charges allow polluters with low abatement costs to benefit from reducing pollution to a low level. Polluters with high control costs can pay rather than spending on controls. In this way market mechanism allow greater reduction in pollution for the same cost, or the same reduction for a lower cost, compared with regulations and standards (Vernon, 1990). By giving the polluters a chance to trade, the total cost of pollution abatement is minimized compared to the more direct regulatory approach of setting standards (Pearce and Turner, 1990).

When economists refer to pollution standards, they mean either a uniform reduction on pollution emissions or a uniform level of emissions (Baumol and Oates, 1989; Besanko, 1987). Emission standards and regulations which require uniform reduction in pollution are inefficient, because the costs of reduction are not uniform for all polluters. Under uniform emission standards, some polluters will be reducing emissions less than it would be cost-effective to do so, while others will be reducing emissions by more than is cost-effective. The regulatory approach of differentiated individual standards, selected on the basis of environmental impact without any consideration of abatement costs, is an expensive means of achieving an emission target. To reach the same emission target by implementing a uniform charge means that this target will generate more damage than under regulated individual standards, because the market allocation of pollution shares ignores the difference in environmental impact between different locations. On the other hand, implementing differentiated rates of charge which take into account the differences in environmental impact of various polluters' effluent have an advantage over differential individual standards when abatement costs differ between polluters.

Mäler (1990) proposes a uniform tax rate of 4 DM per kg of sulphur exported (or \$1.36 in 1985 US \$) on the airborne export of sulphur from one country to all other countries. It is clear that a tax system such as the one proposed by Mäler will not achieve a first-best optimum, as it will not differentiate between exports to countries with high marginal damage cost and countries with low damage cost. Any attempt to apply the same charge to each unit of sulphur emitted, irrespective of where the emissions occur, is itself extremely inefficient. In a first- best policy one should differentiate the tax rate between different exporters, Mäler claims that a differentiation would create not only practical problems but also obstacles to reaching an agreement and he suggests that the gains from going to the first-best optimum seem to be marginal. In order to check this conclusion, a uniform charge of \$1.36 per tonne of sulphur exported (obviously for reasons of comparisons) was used in the first case and a set of differentiated charge rates implied according to what the marginal cost of abatement is at the level of achieving the optimal emission level (SA_i^*), i.e. $p^* = MAC(SA_i^*)$. Tables 2 and 3 present the results obtained.

These tables present the emission and deposition reductions and the associated costs of achieving these levels under the scheme of a uniform tax of $1.36 per tonne of sulphur exported (table 2) and of differentiated charge rates (table 3). Comparing these tables with table 4 of the mathematical model's results for the critical loads scenario, it can be seen that the imposition of charges implies a higher emissions reduction than when we apply simple standards in the form of critical loads (i.e. 20908 thousand tonnes sulphur removal for the uniform charge case; 19,445 thousand tonnes for the differentiated charge rates; and 17072 thousand tonnes in the case of simple standards). At the same time, the cost of achieving these targets is almost the same in the standards' case and in the differentiated charge rates (even if the latter achieves a much higher emissions reduction) and more expensive in the case of the uniform charge rate. This happens, because, as can be seen from table 2 (from the column that gives the different charge rates for each country), the uniform charge rate of $1.36 per tonne of sulphur exported is, for some major polluters like USSR, Spain, Romania, Turkey, Bulgaria, and Italy, much higher than these countries' marginal cost of abatement of achieving the optimal SA_i^* (i.e. in many cases $p^* > MAC(SA_i^*)$).

This implies that these countries abate more than they ought to. For instance, USSR would abate 1076 thousand tonnes more under the imposition of the uniform charge rate of $1.36 than under the case of differentiated charge rates. Its marginal abatement cost at the optimal emissions reduction given by the non–linear programming model is only $MAC(SA_i^*)$ = $0.296 (see table 3). It follows that such countries would prefer to pay the differentiated charge (abating less) rather than the uniform charge. On the other hand, the rest of the European countries would pay less for achieving the pollution control targets under the

Table 2: A uniform charge rate of \$1.36 per tonne sulphur exported
(Emissions and depositions are in 1000 tonnes S; costs and charges in m 1985 \$)

Countries	Emissions reduction	Depositions reduction	Total Costs	Charges
Albania	118.86	92.31	132.25	1.79
Austria	112.99	240.57	28.84	39.07
Belgium	228.94	125.69	191.50	54.28
Bulgaria	635.01	458.28	207.92	77.42
Czechoslovakia	749.08	925.88	347.05	34.73
Denmark	84.55	68.94	47.97	53.25
Finland	154.99	245.32	140.06	55.11
France	512.48	845.07	501.13	158.66
GDR	1329.32	616.19	1099.50	46.46
FRG	1212.84	879.67	1639.05	84.25
Greece	352.12	414.59	230.79	58.10
Hungary	263.69	272.06	92.21	30.48
Ireland	42.59	39.15	11.84	24.68
Italy	1035.99	840.57	729.84	335.51
Luxembourg	5.50	4.00	0.69	4.96
Netherlands	166.18	139.32	168.85	35.41
Norway	31.56	150.00	6.39	28.31
Poland	1352.95	1168.45	723.67	108.21
Portugal	106.18	142.46	75.22	84.57
Romania	978.59	877.11	304.11	98.40
Spain	2134.68	1632.86	850.76	343.64
Sweden	138.60	322.20	47.42	62.02
Switzerland	12.62	31.93	2.70	35.55
Turkey	1059.47	1008.42	449.04	277.29
USSR	5610.20	7618.51	1619.90	1272.27
UK	1209.20	735.90	1104.84	325.29
Yugoslavia	1259.85	962.03	476.25	93.96
Total	20908		11235	3824

Table 3: Differentiated charges rates imposed tonne of sulphur exported (emissions and depositions in 1000 tons S; costs and charges in m and differentiated charge rates in 1000 1985 $)

Countries	Emissions reduction	Depositions reduction	Total costs	Differentiated Charge rates	Charges
Albania	118.96	92.31	133.96	18.02	36.02
Austria	117.13	241.44	36.75	2.429	82.11
Belgium	247.94	133.21	261.87	3.470	43.37
Bulgaria	629.28	446.50	201.42	1.197	64.60
Czechoslovakia	751.69	926.73	356.96	3.261	211.24
Denmark	114.05	80.20	159.08	4.348	27.33
Finland	185.20	254.05	330.07	23.53	883.72
France	527.82	843.93	545.89	2.520	359.88
GDR	1316.91	616.19	922.27	19.60	567.41
FRG	1221.17	891.50	1679.36	4.910	340.28
Greece	360.94	410.23	247.39	3.200	159.16
Hungary	264.78	270.82	94.07	1.096	27.10
Ireland	30.06	32.00	4.02	1.600	0.34
Italy	992.30	811.98	673.40	1.100	222.18
Luxembourg	6.61	4.27	02.74	0.500	0.42
Netherlands	178.51	146.03	215.64	3.676	44.82
Norway	41.48	159.56	32.27	3.184	70.59
Poland	1380.54	1174.91	814.61	7.215	541.42
Portugal	134.35	156.83	121.33	2.900	103.59
Romania	968.84	851.12	293.98	0.322	32.61
Spain	2000.31	1546.97	690.04	0.248	73.77
Sweden	154.17	336.32	90.32	3.468	164.00
Switzerland	13.54	30.73	03.89	1.860	0.43
Turkey	616.48	667.78	186.79	0.320	165.43
USSR	4534.00	6543.34	922.58	0.296	650.20
UK	1278.33	770.76	1285.80	1.787	190.85
Yugoslavia	1259.66	955.22	475.95	1.470	130.96
Total	19445		10783		5194

Table 4: Standards imposition in the form of critical loads (emissions and depositions are in 1000 tonnes S; costs and benefits in m 1985 $)

Countries	Emissions reduction	Depositions reduction	Total costs
Albania	118.97	92.31	134.06
Austria	116.82	234.19	36.04
Belgium	236.55	125.69	213.22
Bulgaria	634.60	429.47	207.42
Czechoslovakia	749.38	903.00	348.11
Denmark	109.50	77.02	141.44
Finland	189.05	245..32	400.60
France	473.77	669.65	434.80
GDR	1336.4	616.19	1228.3
FRG	1209.6	855.78	1625.3
Greece	365.80	401.12	260.69
Hungary	262.02	261.72	89.66
Ireland	8.92	17.10	0.17
Italy	979.24	765.56	657.66
Luxembourg	3.50	3.49	0.06
Netherlands	167.44	137.61	173.17
Norway	35.47	148.62	14.11
Poland	1384.2	1156.22	831.94
Portugal	140.77	128.72	136.05
Romania	713.79	719.99	192.56
Spain	1030.5	845.41	241.15
Sweden	154.88	324.02	94.11
Switzerland	9.34	73.40	0.46
Turkey	390.09	483.64	121.12
USSR	3881.4	5762.22	671.95
UK	1115.8	668.99	906.43
Yugoslavia	1254.4	925.71	468.53
Total	17072	17072	9634.1

uniform charge rate than under the differentiated charge rates. The reason is the same. The differentiated charge rate corresponding to the optimal abatement level is much higher than the uniform charge rate of $1.36/tonne S exported and so these countries would prefer to pay the uniform charge.

Obviously a high uniform charge rate achieves a high emissions reduction but it is not the most cost-effective way of achieving a pollution target. Mäler's conclusion that the difference between the first and the second best solutions is marginal, may be valid for certain countries and regions (for example for Europe's aggregate emissions reduction) but is not valid for all countries and regions. **If** $p^* > MAC\ (SA_i^*)$, which indicates that the uniform charge rate is higher than what corresponds to the first-best optimum, then countries will prefer to pay a differentiated charge than to abate, and vice versa. Finally, from tables 2 and 3, it can be seen, that the uniform charge rate raises revenue of $3824 million, while the differentiated charge case raises revenue of $5194 million(13). Therefore, moving from the first-best optimum to a uniform charge rate does make a difference.

CONCLUDING REMARKS

Various economic incentives exist to implement 'optimal' abatement strategies. Using a simple mathematical model it was shown

- First, that the imposition of charges implies a higher emissions reduction than when we apply simple standards in the form of critical loads.
- Second, that the cost of achieving these targets is almost the same for the simple standards' case and the differentiated charge rates (even if the latter achieves a much higher emissions reduction) and more expensive for the case of the uniform charge rate.
- Third, it was shown that a high uniform charge rate achieves a high emissions reduction

but this result may be due to the fact that for some major polluters it is cheaper to abate than to pay the charge.

Therefore, although there may not be a significant difference moving from the first to the second best solution across countries it may be quite different within countries. That is, it was shown that moving from the first-best optimum to a uniform charge rate does make a difference. Finally, it was shown that differentiated charge rates raise more revenues than a uniform charge rate.

Charges have two purposes other than raising general revenues: namely, to act as an incentive to encourage polluters to reduce harmful emissions because it is cheaper than paying the charge and to provide a fund for financing pollution control. The implementation of differentiated charge rates has an advantage over differentiated standards in so far as abatement costs differ between polluters. Polluters with high abatement costs can pay rather than spending on control. However, regulations will work better than charges for countries where the pollutant's abatement cost curve is flat, and charges will achieve better results for countries where the control cost is steep. In the richer countries, efficiency would be enhanced if the PPP were more adhered to, using a combination of economic instruments and regulatory approaches. The former generally have proved to have several advantages over regulatory laws: they are relatively easy to administer, cost-effective, flexible, they provide a source of finance, and they are less rigid and static and encourage innovation. They are also likely to be used with an increasing frequency in the richer European countries.

It is worth mentioning that market mechanisms do not invalidate regulatory approaches; they are, and must be, an adjunct to them. But in doing so, one must be careful that the charges imposed, as well as the levels at which they are imposed, are not introduced for financial reasons but for economic efficiency.

ACKNOWLEDGEMENTS

Some of the work reported here was undertaken whilst the author was with the Stockholm Environment Institute at York (SEIY). The support of Michael Chadwick and the SEIY is acknowledged. Thanks are due to Charles Perrings and Paul Burrows at the University of York for helpful comments on an earlier draft. The use made of data sources and their interpretations, as well as any remaining errors, are solely the author's responsibility.

NOTES

(1) There is a wide body of literature addressing issues of the economics of regulation and public choice reviewed in, for example Stigler (1971) and Mueller (1989).

(2) The estimates of the flows of sulphur emissions and the subsequent depositions between countries are based on the EMEP model (European Monitoring and Evaluation Program, Norwegian Meteorological Institute) from which the cross-country "transfer coefficients" d_{ji} and d_{ii} which measure the trade-off of sulphur (total dry and wet deposition of sulphur) between all single European countries can be obtained. The proportional transfer coefficients of the EMEP's transfer matrices have been used with early and provisional unconstrained sulphur emission estimates from SEIY, since modified, for the year 2000 (see note 4) to derive a transfer matrix of a closed system of 27 countries. There is obviously an uncertainty, regarding the elements of this matrix, because of weather variability and different meteorological conditions year by year and so on. This means that actual transport matrix coefficients will vary from year to year due to these meteorological variations.

(3) The background depositions have been excluded in our model, as far as they are attributable to natural sources (such as volcanoes, forest fires, etc) but also to emissions whose origin cannot be determined and therefore it is impossible to be tracked by our model. Of course, if the background depositions are included, then actual depositions in each European country will become even larger.

(4) The estimates of the unconstrained sulphur emissions used in this paper are based on early work undertaken by the Stockholm Environment Institute (SEI) at York. They should, however, be regarded as indicative only. Obviously, subsequent revision to estimate of energy balances and fuel sulphur content for the year 2000 will lead to revisions of the cost estimates.

(5) To solve this non–linear programming model, a computer – routine was developed in FORTRAN. The routine provides, for each scenario, the optimal values of: (a) the decision variable (SA_i^*), i.e. the emissions reduction achieved, (b) the deposition reduction achieved and (c) the corresponding total abatement costs($C_i(SA_i)$). Of course, these results are given country by country and for Europe as a whole.

6) Other types of abatement options that are omitted in this approach are abatement through energy conservation in its broadest sense (energy demand suppression, fuel switching, efficiency measures, fuel substitution and use of low sulphur fuels).

(7) The data on which these estimates are based are projections made prior to the unification of Germany. For these reason the report refers to the Federal Republic of Germany, not Germany.

(8) A further distinction between types of point sources is made on the availability of control and new 'cleaner' process technologies and their cost-effectiveness. These standards are closely linked to technology and so are often referred to as technology standards.

(9) Some people refer to effluent charges as the 'Polluter Pay Principle, (PPP)'. The PPP relies on the idea that one who causes an environmental problem has the responsibility to take the necessary measures to eliminate the problem and bear the full cost of the measures. It was adopted by the OECD countries in 1972. For further details see OECD (1975).

(10) To solve problems (4) – (7), a computer routine, similar to that of the non linear program mentioned in note 5, was developed in FORTRAN in order to use the proper NAG library routines.

(11) In general, when pollutant's abatement cost curve is flat, regulations work better than price regulations,. In concern, when a pollutant's abatement cost is steep, price regulations probably work better (UK, CEED, 1986; Weitzman, 1974).

(12) Mäler does not provide any explanation for the choice of the level. The Swedish Environment and Energy Ministry in 1989 suggested a charge on sulphur dioxide emissions from the combustion of coal and peat equal to $5 per tone of sulphur emitted (for details see Vernon, 1990).

(13) Regulation does not have the same revenue-generating capability as a pollution charge, at least if there are no violations of the limits set under regulations. This is due to the absence of a charge for the pollution units below the limits regulated. Once violations are allowed for, even regulation generates revenue from fines. On the contrary, a charge has the extra source revenue from intramarginal units of pollution which is lacking under regulation (Burrows, 1980).

REFERENCES

Baumol W.J. & Oates W.E. (1979). *Economics, environmental policy and the quality of life*, Prentice-Hall, Inc., Englewood Cliffs, N.J.07632.

Baumol W.J. & Oates W.E. 1989. *The theory of environmental policy*, 2nd edition. Cambridge University Press, Cambridge.

Bergman L. (1986). *Some observations of the economics of the acidification in Europe*. Unpublished manuscript. Stockholm School of Economics. Stockholm.

Besanko D. (1987); Performance versus design standards in the regulation of pollution; *Journal of Public Economics,* **34:** 19-44

Burrows P. (1980). *The economic theory of pollution control.* MIT Press, Massachusetts.

Chadwick M.J. & Kuylenstierna J.C.I. (1990). *The relative sensitivity of ecosystems in Europe to acidic depositions*. Stockholm Environment Institute at York, United Kingdom.

Coase R. (1960). The problem of social cost. *Journal of Law and Economics*, **3:** 1-44.

EMEP (1989). *Airborne transboundary transport of sulphur and nitrogen over Europe: model description and calculations*, The Norwegian Meteorological Institute, EMEP/MSC-W Report 2/89.

Freeman A.M. (1982). *Air and water pollution control: a benefit cost assessment.* Willey, NY.

Grubb M. (1989). The greenhouse effect: negotiating targetsα; London, U.K., Royal Institute of International Affairs.

Hahn R.W. & Hester G.L. (1989). Where did the markets go? An analysis of EPA's emission trading program. *Yale Journal of Regulation*, **6**: 109-153.

Hahn R.W. & Hester G.L. (1983). Barriers to implementing trade able air pollution permits: problems of regulatory interactions. *Yale Journal of Regulation***, 1**: 63-91.

Halkos G. (1992). *Economic perspectives of the acid rain problem in Europe*, D.Phil Thesis, Department of Economics and Related Studies, University of York.

Halkos G. (1993). An evaluation of the direct costs of abatement under the main desulphurisation technologies, MPRA Paper 32588, University Library of Munich, Germany.

Hettelingh J.P. & Hordijk L.(1986). Environmental conflicts: the case of acid rain in Europe, *The Annals of Regional Science*, **20(3)**: 38-52.

Hoel M. (1990). Efficient international agreements for reducing emissions CO_2. Paper prepared for the Conference on actions for a common future; Norway, May 8-16.

Kneese A.V. & Schultze C.L. (1975). *Pollution, prices and public policy*, The Brookings Institution Washington D.C., Chapters 2 and 6.

Mäler K.G. (1989). *The acid rain game: valuation, methods and policy making in environmental economics.* Edited by H. Folmer and E. van Ierland, Chapter 12.

Mäler K.G. (1990). International environmental problems. *Oxford Review of Economic Policy*, **6(1)**: 96-98.

Mueller D.C. (1989). *Public Choice II: a revised of Public Choice*. Cambridge U.P.

OECD (1975). *The Polluter Pays Principle*. OECD Environment Committee, Paris.

OECD (1979). *An assessment of the implementation of the Polluter Pays Principle.* OECD Environment Committee, Paris.

Pearce D. (1990). *Economics and the global environmental challenge.* Henry Sidgwick Memorial Lecture, Newnham College, Cambridge, U.K. (23 February 1990).

Pearce D.W. & Turner R.K. (1990). *Economics of natural resources and the environment.* Simon and Schuster International Group ISBN -0-7450-0202-1.

Pigou A.C. (1920). *The economics of welfare.* Mac Milan.

Rose-Ackerman S. (1977). Market models for pollution control: their strengths and weaknesses. *Public Policy*, **25:** 385-392.

Rubin E.S., Cushey M., Marnicio R.J., Bloyd C.N. & Skea J.F. (1986). Controlling acid deposition: the role of FGD. *Environment Science and Technology*, **20(10):** 960-968.

Schultze C.L. (1977). *The public use of private interest.* Washington: The Brookings Institution.

Stigler G. (1971). Theory of Economic Regulation. *Bell Journal of Economics and Management Science*, **2**: 3-21.

Tietenberg T.H. (1990). Economic instruments for environmental regulation. *Oxford Review of Economic Policy*, **6(1):** 17-32.

UK CEED (1986). *The use of market mechanisms in the regulation of air pollution.* UK Center for Economic and Environmental Development, February 1986.

Vernon J.L. (1990). *Market mechanisms for pollution control: impacts on the coal industry.* IEA Coal Research, IEACR/27 IEA Coal Research.

Weitzman M.L. (1974). Prices vs quantities. *Review of Economic Studies*, **41:** 477-491.