Acid rain

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1. Introduction

Air pollution is not a new phenomenon - Londoners in the twelfth century complained about the noxious fumes from burning sea coal, and the corrosive effects of sulphur dioxide (SO_2) dissolved in rain has been well understood for at least a century. But the focus of concern constantly shifts. In Britain the process started with the Smoke Abatement Acts of 1853-56, and via various other measures to the landmark Clean Air Act of 1956. The cause for concern were the very obvious health hazards associated with the unregulated burning of coal, and in particular the large number of people, estimated at 4,000, who died in the great London smog of December 1952. America has also been concerned with reducing coal pollution, but was also active in reducing automobile pollution from quite early on. Here the impetus was the deteriorating air quality in urban areas like Los Angeles and Washington DC, where photochemical smog led to high levels of ozone. This was traced to exhaust emissions of hydrocarbons and nitrogen oxides (NO_x), and California lead the way in introducing successively tighter emissions controls. In Europe, the impetus for environmental policy developed because acid rain - from the sulphur dioxide and NO_x emissions from the burning of coal and oil - is no respecter of national boundaries. Each locality and country discovered that some of the immediately harmful effects of burning coal could be avoided by building tall chimneys, but this merely dispersed the pollutants elsewhere, often great distances to other countries. These countries could not directly control the deposition of acid rain, and could instead only complain and negotiate for some coordinated solution to the perceived problem. Different countries responded to different facets of the pollution problem. The Scandinavian countries were troubled by the death and disappearance of fish from lakes and rivers. Germans worried about forest die-back. Glasnost revealed the full extent of the environmental disasters in East Europe, and provided the focus for local hostility to the environmental insensitivity of central planning.

Environmental awareness has grown rapidly over the past decade, and with it the growing realisation that we live on a rather small and fragile planet. The green movement has had to work hard to capture the attention and imagination of the public and politicians, and has had to

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resort to emotional arguments to get over its message. There is no doubt that the hidden environmental costs of current technology may be high, even life-threatening, but it is also clear from recent experience that the costs of carelessly designed environmental regulation can also be high. As economists, we have a duty to argue for cost-effective environmental policies. Inefficient policies not only achieve less than they should, but they also run the risk of alienating taxpayers and consumers who ultimately pay for the regulation and may undermine the aims of the environmental movement. Economists, with honourable exceptions, have tended to ignore environmental economics because it seems to raise few new ideas. Most of the useful techniques have been the stuff of undergraduate welfare economics since Pigou's day, and though each generation adds to the stock of knowledge and technique, the subject has not been at the theoretical cutting edge for some time. This might not have mattered if economists had been supplied with accessible facts with which to clothe the theory and to bring the policy issues into sharp perspective. But these facts are largely produced by scientists unfamiliar with the economic style of argument, and often unconcerned with economic costs and benefits. There has been too little communication between the disciplines.

It is interesting to compare the situation in the US. The style of regulation exemplified by the Environmental Protection Agency, and the separation of legislative and executive power means that environmental legislation has to be argued in a quasi-judicial way before being enacted, and economists have been centrally involved in the ensuing debates—not necessarily successfully. As a result of having to make a quantified case in public, economists have investigated the scientific evidence, have conducted empirical enquiries, and have identified the gaps in our knowledge. Environmental economics has received a considerable impetus and a solid body of theory and evidence on which to build. We in Europe lag behind, though there are signs that the times are changing.

This contribution deals with a small part of the environmental debate, that concerned with acid rain. It is an important topic—not as important as the greenhouse effect, which is global in scale, probably not as important as traffic congestion, which is a domestic matter for each country. Nevertheless, substantial sums of money have been spent and are now being committed in an attempt to alleviate the problem of acid rain. The thrust of this paper is that this programme as currently interpreted is flawed, unnecessarily expensive, and if it succeeds, it runs the risk of high political cost. Relatively simple economic principles applied to the appropriate facts ought to be able to achieve the same environmental benefits at substantially lower cost, and in a more decentralised and less politically problematic way.

I make no apologies for the high ratio of facts to theory in what follows. The environmental debate has been long on emotional argument and short on substance for too long. I am not an expert in this field, and have had to rely on secondary sources for the data. On the crucial issue of quantifying the benefits of reducing acid rain I have not been able to find adequate evidence and so cannot finally quantify the efficient policy. But I have found enough evidence to cast considerable doubt about the priorities for abatement, and to suggest where research effort should be concentrated. Several findings surprise me. Fish death from acid rain is sad, but economically unimportant. Tree death may be far more important, though there are worrying uncertainties about the cause and cure of this problem. Health problems associated with coal emissions, particularly the combination of SO_2 and particulates (smoke particles) are potentially of the first importance, those associated with NO_x and ozone seem trivial.

2. Acid rain and its effects

In order to understand the acid rain problem it is necessary first to describe the causes and consequences of acid rain. Considerable scientific research over the past decade has illuminated the phenomenon, though uncertainties remain. The next step is to identify the sources and measure the amounts of pollutant released, and their destination. What is it that causes the damage, where does the main damage occur, and what are economically the most expensive consequences of acid rain? Finally, one needs to determine the techniques available for reducing emissions, and the costs of abatement, in order to identify cost-effective abatement policies. This last step is usually ignored by ecologists and politicians, who are content once they have found ways of reducing acid rain to press for the maximum (politically?) feasible degree of abatement. This section addresses each issue in turn.

2.1 Defining acid rain

Acid rain is normally understood to include the deposition of the acidic combustion products sulphur dioxide, SO_2, various nitrogen oxides, NO_x, and chloride, Cl^-, either as dry gases or particles, or as wet deposits in rain, snow, sleet, hail, mist or fog. These pollutants usually undergo a series of chemical transformations into sulphuric acid, H_2SO_4, nitric acid, HNO_3, and hydrochloric acid, HCl. These acids in turn have a variety of effects on the environment, both directly and indirectly in causing the release of further harmful chemicals such as aluminium. Acid rain can be measured in a variety of ways—in terms of tonnes of the original gases released, or tonnes of elemental sulphur, or in terms of the acidity of the rainfall, run-off, streams or lakes. Acidity is measured in pH units on a logarthmic scale.[1] As the scale is logarithmic, rainfall with a pH of 5 is 10 times as acid as that with a pH of 6. Unpolluted rain is slightly acidic from dissolved carbon dioxide and has a pH of about 5.6. Sea water is naturally alkaline, having a pH of 8.3.

Most SO_2 comes from large combustion plants—thus in 1987, 85 percent came from large combustion plants, and 73 percent from power stations. Of UK emissions from fossil fuel combustion, 79 percent came from coal combustion and 12 percent from fuel oil.[2] Sulphur dioxide pollutes the environment through two different routes. Much of the SO_2 gas falls to earth within 300 km of the source in its dry form, and this process is described as dry deposition. Long range transport occurs because SO_2 is oxidised to sulphate particles, which are not readily deposited in dry form. Their main removal is by scavenging in rain-making processes as wet deposition, which may occur 1000-2000 km from the source. Long term monitoring of rain chemistry has been carried out since 1978 only at between 6-12 sites throughout the UK. During the period 1986-89 a rather dense network of up to 60 sites was operated to establish the spatial pattern of rain chemistry across the UK. The basis spatial pattern has changed little from year

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[1] pH is defined as the negative logarithm of the hydrogen ion (H^+) concentration, having the perverse effect that lower numbers correspond to higher acidity. Thus 0 is the most acid, 7 is neutral and 14 is most alkaline. Lemon juice has a pH of 2, milk of magnesia 10.5.

[2] UK data are taken from the *Digest of Environmental Protection and Water Statistics, 1988*, (Department of the Environment, 1989). Figures after 1970 are based on revised emission factors described therein.

to year during this period and so 30 sites have been retained for longer term monitoring.[3] Wet deposition can be reported in two ways—by its intensity and cumulative deposition. Intensity is shown by the maps of the average acidity of precipitation (in pH), and cumulative deposition by wet deposited acidity in grams of hydrogen ions per square metre per year. Deposited acidity is the product of the acidity of the rainfall and the amount of rain—wetter areas in the west may have more acid deposited even though the precipitation is less acidic.

2.2 Measuring acid rain

The European Monitoring and Evaluation Programme (EMEP) was set up in 1978 to monitor the movement of pollutants, and to determine where the deposition of pollutants released from each source occur. Until recently the only pollutant tracked was SO_2, though now NO_x is also monitored. The surface of Europe is divided into squares with grid lines 150 km apart. There are about 720 grid line intersections on land and about 100 monitoring sites which are used in the EMEP model and these are termed arrival points. Using detailed meteorological information, the track of air which arrived at each of the 820 or so points is followed backwards in time for 96 hours. An air parcel is then studied forwards in time as it follows each back-track precisely, picking up pollution and depositing pollution until it arrives back at its arrival point. This whole procedure for each of the 820 points is repeated at six-hourly intervals 365 days of the year. The model also keeps a record of the pollution produced by each country. Not all the deposition can be traced back to an identified source, as meteorological data is only accurate enough to track back for 96 hours. Table 1 gives a subset of the basic data from this exercise for 1987, and is to be read as follows.[4] Britain (GB) appears as a receiver in the left hand column and as an emitter in the top row. Looking along the row against GB the table shows that Britain received 14,000 tonnes of sulphur[5] (ie about 27,000 tons SO_2) from France, 11,000 tonnes from West Germany (DE), and 571,000 tonnes from domestic sources. Looking down the column headed GB the Table shows that Britain emitted 1,271,000 tonnes sulphur whose final destination could be established, and of this 43,000 tonnes fell on France, 45,000 tonnes on Germany, and 437,000 tonnes on North Africa within the monitoring area (demonstrating just how far the plume can travel).

The large numbers on the diagonal of Table 1 shows how important domestic sources of pollution are. The large off-diagonal numbers indicate where the major impacts of one country on another occur, and it is striking that they primarily occur in East Europe, confirming the view that central planning has been an environmental disaster for its participants.

Table 2 presents the information from the same programme in a different way. The first two columns give total emissions (not just those whose final destination can be identified) for

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[3] Personal communication from R Derwent.

[4] The fuller table is given in the Appendix, and is the source of the calculations reported below. Using the abbreviated table leads to considerable biases in estimating total damages and the complete table should be used for all calculations. An earlier table for 1980 is given in Environmental Resources, Limited, (1983), p40.

[5] To convert from sulphur to sulphur dioxide, multiply by 1.9, or roughly double the numbers.

the base year 1980 and the most recent year available, in order of magnitude. This allows an estimate of the extent to which countries have succeeded in

Table 1 Origins of Sulphur Deposition in Europe

Thousand tonnes S/year

Receivers		*Emitters* CS	FR	DD	DE	BL	HU	IT	PL	ES	SC	SU	GB	OE	UI	Sum
Czechoslovakia	CS	385	11	128	28	5	45	10	95	1	0	2	7	13	28	765
France	FR	19	332	41	40	28	5	21	15	65	0	0	43	8	140	760
GDR	DD	84	14	725	61	11	2	2	32	1	0	1	15	3	24	979
West Germany	DE	47	69	163	330	44	3	13	23	6	0	1	45	6	64	821
Benelux	BL	4	32	15	51	102	0	0	4	2	0	0	31	0	19	267
Hungary	HU	31	3	16	6	1	190	12	25	0	0	1	1	28	18	337
Italy	IT	13	21	15	8	2	11	353	14	10	0	1	4	19	86	562
Poland	PL	145	15	310	47	10	40	10	790	1	1	18	15	21	64	1492
Spain	ES	2	11	5	3	1	2	2	3	523	0	0	6	16	98	674
Scandinavia	SC	17	5	48	18	6	4	2	44	0	59	33	32	30	194	501
Soviet Union*	SU	107	10	167	36	8	84	13	337	1	8	2204	16	97	491	3584
Britain	GB	5	14	15	11	8	0	1	3	2	0	1	571	6	60	702
Other Europe	OE	95	40	97	49	8	141	136	101	29	3	95	70	825	435	2163
N Africa	NA	105	136	253	131	71	64	182	194	210	28	196	437	254	821	3087
Sum	Σ	1064	721	2005	823	322	594	759	1685	856	107	2558	1271	1377	2553	16695
Error		5	8	7	4	17	3	2	5	5	8	5	-22	51	11	1

Source: *Acid Magazine* Sep 1989, from EMEP data.

Note: Sulphur dioxide figures will be about twice as large. Total wet plus dry deposition of sulphur for period 11.12.87 to 6.11.88, sums may not add due to rounding errors (shown).
* Norway plus Sweden
** European part of USSR within EMEP area of calculation.
*** Deposition in North Africa within areas of calculation
UI = unattributable to any country, plus small amount from NA

moving towards the target 30% reduction now widely accepted. The next two columns give depositions within the country, and an estimate of the fraction of depositions which can be traced back to domestic sources (using Table 1 data). This confirms the importance of the diagonal element in Table 1, and the importance of domestic sources of pollution. The final column gives the ratio of exports of SO_2 to imports. As total depositions in 1987 are only 78% of total emissions, this ratio can be expected to be significantly above unity on average. The smaller it is, the more the country is sinned against, than sinner. Britain stands out as the greatest sinner on this criterion, and the Scandinavian countries as those most sinned against.

Table 2 Sulphur Emissions

Thousand tonnes or per cent

	Emissions 1980	1987	Change %	Depositions 1980	1987	Own contrib tot. depos	Exp/Imp
Soviet Union*	6400	5100	-20	5101	3584	0.61	2.1
GDR	2500	2500	0	963	979	0.74	7.0
Poland	2050	2270	+10	1443	1492	0.53	2.1
United Kingdom	2335	1840	-21	803	702	0.81	9.7
Spain	1625	1581	-3	670	674	0.74	7.0
Czechoslovakia	1550	1450	-6	818	765	0.50	2.8
Italy	1900	1252	-34	916	562	0.63	4.3
German Fed Rep	1600	1022	-36	1083	821	0.40	1.4
France	1779	923	-48	1160	760	0.44	1.4
Hungary	817	710	-13	416	337	0.56	3.5
Yugoslavia	588	588	0	662	497	0.39	5.0
Bulgaria	517	570	+9	293	235	0.65	5.0
Belgium	400	244	-39	162	121	0.41	2.7
Greece	200	180	-10	150	119	0.38	1.8
Turkey	138	177	+22	209	210	0.29	2.7
Finland	292	162	-44	273	210	0.23	0.7
Denmark	219	155	-13	110	83	0.37	2.4
Netherlands	244	141	-42	175	139	0.23	1.0
Portugal	133	116	-13	83	83	0.42	1.7
Sweden	232	116	-50	333	307	0.12	0.3
Romania	100	100	0	405	330	0.10	0.2
Irish Republic	110	84	-24	66	68	0.31	1.3
Austria	177	75	-58	282	207	0.09	0.3
Norway	70	50	-29	199	194	0.07	0.2
Switzerland	63	31	-51	121	70	0.11	0.4
Total	26078	21471	-18	20484	16695	0.52	

Source: *Acid Magazine* Sep 1989, from EMEP data.

Note: Sulphur dioxide figures will be twice as large. Countries ordered by 1987 emissions.
* European part of USSR within EMEP area of calculation.
Figures for emissions in 1987 based on interpolation except for USSR, UK, Czecholslovakia, Finland and Netherlands.

Only part of total SO_2 emissions come from man-made sources; other important sources include volcanoes, biological decay and forest fires. These natural sources might account for 80 - 290 million tonnes per year world-wide, compared to total man-made emissions of 75 - 100 million tonnes. The levels and mechanisms responsible for natural emissions are imperfectly understood, but they may play an important role in the European acid rain problem.

The information generated by EMEP is remarkably useful, not only in quantifying the level of pollution, but also identifying efficient and feasible abatement policies. The information on deposition can be used to draw maps showing the average acididy of precipitation over Europe using contour lines of increasing levels of acidity. Such maps show that in 1987 most of Yorkshire and the East Midlands had precipitation of average acidity below pH 4.3, (ie 5 times as acid as 'normal' rain with a pH of 5.0) whereas Wales, South-west England and the west coast of Scotland was above 4.6, and so less acid. Substantial areas north of a line joining the Wash and Liverpool received more than 0.05 grams H^+ per square metre by wet deposition (ie 0.5 kg per hectare or 50 kg per square km), with Wessex, East Wales, Northern Ireland, and North-east Scotland receiving less than 0.02 grams. Most of the sulphur deposition in the UK occurs through dry deposition as SO_2, particularly in the south and east of the country. In the north and west where rainfall is more frequent and intense, wet deposition of SO_2 and sulphate particles becomes more significant.

In a European context, the lines of equal rainfall acidity show the highest concentrations in Germany and Poland, with pH below 4.1, with most of Southern France, almost all Italy, Spain, Portugal, West Yugoslavia and West Greece having a pH greater than 4.9 (ie less acid than any part of the UK and Eire).

The EMEP tables can also be used to throw light on the political economy of pollution control. Consider first the column in Table 2 which gives the fraction of total deposition which can be attributed to domestic sources. The unweighted average of these figures is 41% (with a standard deviation of 21%). The weighted average (heavily influenced by the larger countries, especially the USSR, which, by their size, have higher domestic absorption) is 52%. The weighted average domestic absorption as a fraction of total production is 33%, and the unweighed average is 29% (with a standard deviation of only 5%). What this means is that the average unilateral cost of reducing a tonne of domestic deposition is equal to the cost of reducing domestic emissions by about 3 tonnes. The ratio of total European depositions to total European emissions is 66%, so that if all European countries acted in concert, the cost of reducing depositions by one tonne would be only half as great. Put another way, many countries could reduce depositions within their borders by about 50%, but at twice the cost per tonne reduced as if all countries acted together. There are thus considerable benefits to coordinated action, but these should not be dramatised—SO_2 pollution is far from a pure public good at the country level, and self interest ought to go a considerable way towards alleviating the problem.

The next question one can ask of the EMEP data is whether there are significant opportunities for bilateral bargaining between pairs of countries over pollution levels. One way to identify such is to look for instances where the volume of bilateral pollution trade is large relative to total depositions, and where trade is bilateral rather than unilateral. The volume can be measured by one-half exports plus imports, and bilateralism can be measured by the difference between exports and imports. Table A1 of the Appendix gives the net exports of each country and can be used to identify the extent of bilateralism. The following country pairs have a difference between these two measures of 5% of depositions or less for the smaller of the partners: Czechoslovakia-GDR; Czechoslovakia-Hungary; Czechoslovakia-Poland; GDR-Poland; Poland-Hungary; Soviet Union-Czechoslovakia; Soviet Union-GDR; Soviet Union-Hungary; Soviet Union-Poland. It is notable that significant balanced trade in pollution is confined to Eastern Europe, and does not affect any of the other countries identified in Table 2. (It may be substantial for smaller countries as a proportion of their deposition.)

Another possible question to ask is which pairs of countries have large net trade balances in pollution which might lead to financial negotiations over pollution levels. The following countries have net imports from another country which are greater than 5% of total depositions:

Poland from GDR (19%); Denmark from GDR (12%); Scandinavia from GDR (10%); Scandinavia from Poland (9%); Soviet Union from Poland (9%); Czechoslovakia from GDR (6%); Scandinavia from Soviet Union (5%). Scandinavia is thus the only West European country which receives large net imports from single countries—otherwise it is the Eastern bloc countries that stand out as large net importers from each other.

Nitrogen oxide emissions Nitrogen oxide, or NO_x, emissions are measured in terms of tonnes nitrogen dioxide equivalent, NO_2. Table 3 gives per capita emission levels for both SO_2 and NO_x for the major member countries of the UN Economic Commission for Europe Convention on Long-range Transboundary Air Pollution. It shows that the UK is not high in comparison with Europe and North America taken together, but is rather higher than the West European countries in the sample. The table shows that whereas SO_2 has decreased between 1980 and 1985, NO_x has if anything increased.

Table 4 gives further information about NO_x for 1985. The first column shows that mobile sources contribute about one-half of all NO_x emissions in Western Europe (actually OECD Europe), though the range is from 28 per cent in Ireland to 84 per cent in Norway. Much of the rest comes from large combustion plants—thus in Britain 35 per cent of the total came from power stations. The next column shows total emissions of NO_x from mobile and stationary sources in relation to total energy use. The coefficient of variation (CV) is 34 per cent, showing that emissions correlate quite closely with energy consumption, but there are important variations in the degree to which energy use causes NO_x pollution. Britain does poorly by this score, almost as badly as Portugal and Greece. (Figures for Luxembourg seem rather low and may be explained by some energy sales, especially of transport fuel, being consumed abroad.) Column (3) shows mobile emissions of NO_x per unit of GDP (which has a lower CV than total emissions per unit of GDP). Column (4) gives mobile emissions in gm NO_2 per km driven by cars. White (1982, Table 2) shows that if there were no emissions regulations, then for the US 64 percent of total NO_x emissions would come from cars, and the balance of 36 per cent from trucks. Uncontrolled emissions are 5.44 gm/km for cars, and 38.6 gm/km from heavy diesel trucks. The expected uncontrolled emissions per km driven by cars alone might therefore be 8.5 gm/km (ie 5.44/0.64) if the proportion of car km in total vehicle km were the same as the US. The European average is 3.15 or only 37 per cent of that predicted for uncontrolled emissions. Perhaps more impressive, if the total emissions from all mobile sources are attributed entirely to cars, then the average achieved is about as good as those achieved in the US by cars of later than 1976 model year (White, 1982, Table 10). It is interesting to note that Britain does quite well by comparison with other countries—at least as far as emissions from mobile sources are concerned. The last two columns relate mobile emissions to two transport fuels. Column (5) gives mobile emissions in kg per tonne of gasoline, again showing Britain in a favourable light. The final column gives total mobile emissions divided by total fuel consumed in the transport sector, and thus accounts for diesel emissions, which are potentially quite serious.

Table 3 Emissions per head

Kilograms/head/year

	SO_2		NO_x	
	1980	1985	1980	1985
United Kingdom	85	65	35	33
Austria	47	18	29	28
Czechoslovakia	202	203	78	73
France	65	31	34	29
Germany Fed Rep	52	43	49	49
Greece	83	73	13	15
Irish Republic	63	39	21	19
Netherlands	28	25	35	34
Poland	115	116	5	18
Spain	87	75	21	24
Sweden	64	33	38	37
Switzerland	19	15	31	33
Canada	193	150	72	72
United States	102	90	90	83
Averages: All	86	70	39	39
Europe	76	61	32	32
West Europe	59	42	31	30
Ratios:				
UK/sample	0.99	0.93	0.89	0.84
UK/Europe	1.12	1.06	1.08	1.01
UK/W Europe	1.43	1.56	1.14	1.10

Source: UNECE (1987) *National strategies and policies for air pollution abatement*

Note: Averages are unweighted.

Nitrogen oxides also come from natural as well as man-made sources. Again estimates are very imprecise, but natural sources may account for 20 - 90 million tonnes compared to estimated total man-made emissions of about 90 million tonnes. One might therefore argue that perhaps only half the total acid rain emissions are man-made, but whereas natural emissions are world wide, man-made sources are concentrated in the northern hemisphere, and specifically in Europe and North America.

Table 4 NO_x emissions from mobile sources

Country	Mobile total/ %	Total $kgNO_x$/ toe	NO_x/ gdp	gm NO_x/ km car	Mobile NO_x/ gas	NO_x/ fuel
	(1)	(2)	(3)	(4)	(5)	(6)
Austria	68	10.1	1.27	3.76	61	37
Belgium	55	9.0	0.84	2.73	44	22
Denmark	34	15.9	1.04	2.48	61	35
Finland	58	14.8	1.80	7.62	86	47
France	66	13.1	1.29	2.81	62	38
Germany	59	14.6	1.54	3.73	70	46
Greece	63	18.6	3.35	4.36	75	44
Ireland	28	10.7		1.08	22	14
Italy	51	14.8	1.06	2.03	70	34
Luxembourg	64	6.9	2.32	5.61	45	28
Netherlands	60	10.5	1.53	3.27	95	56
Norway	84	12.3	2.27	9.65	113	81
Portugal	38	19.6	3.38	4.13	123	54
Spain	46	18.5	1.49	4.50	67	36
Sweden	68	9.2	1.48	3.26	52	37
Switzerland	74	11.1	0.92	3.04	49	41
UK	45	16.4	1.65	2.96	44	32
Europe	49	14.1	1.42	3.15	63	39
SD unweighted	14	4.76	0.83	2.01	25	14
CV	0.29	0.34	0.58	0.63	0.40	0.37

Sources: UNECE (1987) and OECD; emissions for 1985 or latest available year; other figures for 1985 or 1986.

Notes: Col (1): NO_x Emissions from mobile sources divided by total emissions NO_x; Col (2) total emissions divided by total energy consumption in tonnes oil equivalent (toe); Cols (3-6): Emissions from mobile sources divided by: GDP; km driven by cars; kg gasoline; kg total road transport fuel.
SD is standard deviation; CV is coefficient of variation.

2.3 Assessing the damage caused by acid rain

Acid rain has ecological consequences in that it affects the soil, vegetation, especially forests, lakes (and hence fish). It causes economic damage to man-made structures (buildings, fabrics, metals), and it can affect human health. The ecological consequences are complex and still subject to scientific uncertainty and hence dispute. Soils vary widely in their ability to buffer (ie neutralise) acid rain, and natural processes add to the man-made sources of acid rain. Recent work undertaken for the UNECE Convention on Long-range Transboundary Air Pollution[6] attempts to establish critical loads for various kinds of soils, which, if exceeded, would mean that the soil could no longer neutralize additional acid rain depositions.[7] Many sensitive areas

[6] and reported in *Acid News* No 3, October 1988

[7] Two measures are given for these critical loads - acid input in keq H^+ ions/km^2/year and sulphur deposition in kg S/ha/year. (It appears that 1 keq H^+ is 1 kg.) For very slow weathering rocks (Class 1, which have poor buffering ability and hence low critical loads, the figures are < 20 keq H^+ or < 3 kg S/ha; for moderate weathering

especially in Scandinavia experience both high rates of deposition and soils for which critical loads are low.

The effects of acid runoff on lakes have been intensively studied in Scandinavia and the UK (and doubtless elsewhere). One of the main mechanisms leading to the decline in fish stocks is the release of aluminium caused by acidification, rather than the direct effects of acid (see eg Environmental Resources, Limited, 1983). Palaeoecological studies of core samples can trace back acidity levels into the distant past and show substantial falls in pH in many lakes after the industrial revolution. Thus Battarbee *et al* (1988) analyzed lake acidification in sensitive areas in the UK and found that before 1850 most lakes studied had pH levels of about 6.0, but that since then pH values had declined by 0.5 - 1.5, (ie acidity had increased by between 3 and 30 times) depending on deposition rates and buffering capacities. Other studies show similar trends, and also show that lake acidification may be reversed (even if temporarily) by the addition of lime either to the lake or the rivers in the cachement area. This is expensive, as about $5g/m^3$ of bicarbonate (usually in the form of limestone) are required to raise the pH from 4.5 to 6.5, and it does not by itself restore the lake to its original condition—restocking may also be required. (See eg Dudley, *et al* 1985; Britt, 1986.)

One implication which does not seem to have been adequately emphasised is that acid rain is a stock pollutant as well as a flow pollutant. That is, part of the final damage caused will depend on stocks of acid in the environment, not just the rate at which acid rain is deposited. Even if the environment is capable of neutralising or disposing of some of the acid each year, if inflows exceed this rate of disposal, then the stock of acid will increase. In the UK it is believed that current levels of soil acidification are a legacy of the Industrial Revolution, and that water quality will not be restored until the soil recovers. This recovery is a slow process and relatively insensitive to near-term rates of emission reduction, requiring liming for rapid recovery.[8] This may go some way to explaining the paradoxical relationship between decreasing levels of SO_2 emissions on the one hand, and apparently deteriorating ecological conditions on the other. On the other hand, Battarbee *et al* (1988) note that acid deposition has been declining in Scotland over the past 15 years, and that the uppermost sediments are already recording an improvement, which suggests a possibly swift improvement if deposition levels could be further reduced. The speed of response will presumably depend on the ecological circumstances, but may imply high rates of 'depreciation' of acid stock levels.

Lake and fish damage appears quite well understood compared to the damage suffered by trees. The problem was highlighted in Germany in the early 1980s, and shown to occur elsewhere. Forest damage has been attributed to acid rain, weather changes and droughts, the age of trees, fragility of soils at high altitudes, and inappropriate forest management. Ozone attack appears to be important, and may have synergistic interactions with acid rain (Environmental Resources, Limited, 1983). Even if the exact proportion of damage attributable

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capacity soil (Class 3) the figures are 50-100 keq and 8 - 16 kg respectively, and for very rapid weathering soils in Class 5 > 200 keq and > 32 kg respectively. Currently deposition rates in parts of West Germany are in the range 50 - 80 kg S/ha/yr and in the worst affected parts of Europe the figures exceed 100 kg. 1 kg of sulphur is about 2 kg of SO_2.

[8] Personal communication from David Pearce

to acid rain is not known, there seems widespread agreement that reductions in acid rain would be beneficial to forests. Similar uncertainty pervades the study of crop damage, though again ozone appears to be more directly harmful than acid rain. To the extent that ozone plays a major role in crop and forest damage, NO_x, which is a major contributor to ozone production, is more damaging than SO_2.

Damage to buildings and materials occurs primarily in urban areas as a consequence of relatively high concentrations of SO_2, with little effect detected from exposure to NO_x. The effects have been observed and correctly attributed for centuries, and the estimated damage costs are thought to be high.[9]

Health effects of intense pollution can be dramatic—it is estimated that 4,000 people died in the great London smog of December 1952. Similar levels of SO_2 concentration were attained in a subsequent episode in London from 3 - 7 December 1962, after the Clean Air Act of 1956 had lead to a dramatic fall in smoke concentrations. This time an estimated 340 died, suggesting that the earlier episode was so deadly because of synergistic interactions between smoke particles and SO_2 (Park, 1987, p127). It appears that it is the gas SO_2 that is harmful, rather than the wet form of acid rain. Acid rain in its wet form can have indirect effects by releasing toxic heavy metals into water supplies. Individuals vary considerably in their tolerance to these gases, but there is some evidence from epidemiological studies that long-term exposure at lower levels than these dramatic episodes can be harmful to health (Park, 1987, p127. See also Pearce and Markandya, 1989 for a summary of the extensive economics literature on the health impacts of SO_2.)

If SO_2 and particulates are lethal, the health case against NO_x is at best unproven, for it appears that NO_x is much less active biologically. There is a certain irony in the fact that the impetus to reducing automobile emissions initially came from California, where it was suspected, and later established that photochemical smog was caused by vehicle exhaust. The case mounted by the US Environmental Protection Agency for reducing emissions was based on the supposed adverse health effects of high concentrations of ozone, though subsequent studies (Lave, 1982; White, 1981; 1982) cast considerable doubt on the evidence. To quote White (1981, p59-60): '. . the ozone-related health effects under discussion were short term and reversible. .. Thus far, ozone exposure has not been demonstrated to have long-term debilitating consequences in humans. ... the contrast with other studies of pollutants, such as particulates and sulphates, was striking. .. Particulates and sulphates probably killed; ozone appeared to do little more than cause coughing!'

2.4 Measuring the costs of acid rain damage

Estimates for the costs of different types of damage are scattered in the literature, and vary greatly in their reliability. Pearce and Markandya (1989) provide a useful methodological discussion of cost-benefit analysis applied to environmental pollution, summarise a variety of these estimates, and note the criticisms to which they are vulnerable. One approach familiar to economists is to ask what people would be willing to pay for property located in less rather than

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[9] Exactly how high the damage costs might be is currently the subject of active research, especially in Germany, but at the time of writing the buildings component of damage cost estimation has not reported.

more polluted air, as reflected in the response of property values to pollution levels. Most of the estimates here come from the 1960s and 1970s, and suggest that for each 1% increase in SO_2 concentration, property values fall by between 0.06 - 0.15 of 1% of their value for a house of average value.

An alternative approach is to look at the direct economic costs caused by the acid rain, and this has been done by for the Netherlands for 1986. It was estimated that current costs were about \$53 - 175 million per year,[10] but if the costs of dealing with future damage were taken into account (loss of timber etc) this might rise to \$120 - 380 million. Looking at the current costs, the large proportion of the total comes from agricultural damage, thus extra liming of the soil to counteract acidification might cost \$18-60 m, and falls in crop yield might be \$36 - 360 m. One should of course be most wary about estimating the value of lost agricultural output given the distortions of the CAP. Indeed, as an aside, agriculture is responsible for considerable ground water pollution (notably nitrates, and possibly algal blooms). Much of this is in turn the consequence of intensive agricultural practices induced by the high agricultural prices enjoyed under the CAP, notably high fertiliser levels. If for various reasons it is difficult to reform agricultural output price levels, then there is a strong case for raising agricultural input price levels to the same ratio to world prices as output prices enjoy. This would improve the efficiency of resource use and reduce the deadweight losses associated with the CAP. Thus if output prices are twice import parity levels, then fertiliser prices should be taxed to raise their price to twice world market levels. This would go some way to reducing another form of environmental pollution. (See Newbery, 1989, for the details of the arguments on efficient input taxation.)

Other ecological damage estimates are given in Environmental Resources, Limited, (1983). German forest damage was put at \$0.25 billion p.a., and rough estimates of potential EEC wide damage can be deduced from the annual value of spruce and fir forestry production of \$6.6 billion p.a. Thus if 20% of forests are adversely affected so that their production drops by 10%, the loss is \$0.13 billion p.a.[11] OECD (1981) estimated that the value of fish loss in Scandinavia was \$38 million p.a., and in Scotland might be \$0.7 m p.a.

Acid rain causes material damage, whose costs, excluding the costs of restoring historic buildings, have been estimated by UNECE (1982) at between \$4 - 17 per head (ie \$1.0 - \$4.7 billion for the 1983 European Community as a whole). The figures in the Netherlands are somewhat higher (\$10-19/head) and in Germany are estimated at \$19/head. Environmental Resources Limited, (1983) gives figures from OECD (1981) for the estimated total corrosion damage for 12 OECD European countries to galvanised steel and its paint coatings in 1974. For the UK the figures were \$5.9 billion p.a., for W Germany \$9.5 billion p.a., and for Belgium, Luxembourg, Denmark, France and the Netherlands together \$3.3 billion, or in total \$18.7 billion. How much of this can be attributed to acid rain is still under study.

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[10] Unless specifically stated to the contrary, all cost estimates have been expressed in mid-1989 purchasing power, updating from the US CPI.

[11] But note that there is still, considerable disagreement as to how much forest damage has occurred, and how much is caused by acid rain.

Reducing car emissions would also reduce photochemical smog in some areas—-particularly those which experience temperature inversions combined with strong sun. Los Angeles is the leading example, but clearly Athens suffers similarly. There is no doubt that those living in such areas would be willing to pay for reductions in smog levels, and Schechter *et al.* (1989) estimate that households in metropolitan Haifa, Israel, would be willing to pay £12 (1987 £) per household per year to reduce pollution levels by 50 per cent. It is difficult to imagine that this would amount to a large total sum for Europe as a whole compared with the other damage costs, given the relative infrequency of photochemical smog in more Northerly climes.

2.5 The costs of abatement

The Department of Environment estimated the costs to the UK of retrofitting 6GW of coal fired plant with Flue Gas Desulphurisation (FGD) and all 12 major coal fired power stations (23GW) with low NO_x burners at over £1 billion. It now seems doubtful that more than a small part of this programme will go ahead, as the liability to install FGD would make the privatisation sale of the CEGB unattractive. Instead it appears that the successor companies to the CEGB, Powergen and National Power, will meet the emissions standards by a combination of installing high efficiency gas turbines and importing low sulphur coal. The impact of this on British Coal will be substantial, and it is in interesting example of how the (private) cost-minimising solution to the emissions standards may differ from centrally imposed solutions. The Government also intends to apply the EC large car emission standards ‘as soon as practical, probably in the early 1990s,’ at an estimated annual cost of £550 million. The second stage, applying to small cars, was estimated to add an additional £250 million per year, a total of about 4 per cent of UK motoring costs. (Department of Environment, 1988, 7.14-15.) This section examines various estimates of the costs of abatement in somewhat more detail to see if mandating particular solutions is likely to be cost effective, and to check on the consistency and plausibility of various estimates. The results are summarised in Table 5 and then briefly explained.

The sources of these estimates are as follows. Environmental Resources Limited (1983) gives estimates of the capital cost of FGD at $175-200/kW or about 15-20% of the capital cost of the plant. Retrofitting, where practical, may increase this cost by a further 30-50%. FGD reduces thermal efficiency by about 2% (eg from 36% to 34.1%) and so can increase the operating costs by 10-20%. The first station which the CEGB plans to retrofit is Drax A+B, which has a total capacity of 4000 MW, and which burns 11 million tonnes of coal per year with a sulphur content of 1.7%. If 95% of this were previously released as SO_2, the annual emissions would be 178,000 tonnes S, or 338,000 tonnes SO_2. After fitting FGD, 90% of the SO_2 will be removed, and the reduction in SO_2 would be 287,000 tonnes. The Layfield report gives the costs of FGD as £17/kw/year, broken down into £5 capital, £2.5 operating and £9.5 for loss of thermal efficiency. This last figure depends on the cost of coal which has since fallen and Jeffrey (1988) estimates the efficiency cost as £6.5. Using these figures the annualized cost of the Drax programme would be £56m, or about £200/tonne SO_2 reduction. Longhurst *et al* (1987) gives the CEGB’s estimates of the cost of retrofitting FGD for a 2000MW plant as £160 million plus £35 million is lost output, or £440-£740/tonne of sulphur removed from the gas stream (ie £230-390/tonne SO_2 removed). Dudley *et al* (1985, p121) try to cost a programme to achieve a 60% reduction in SO_2 in UK emissions from its 40 GW coal-fired capacity using CEGB data and data presented at the Layfield enquiry. The levelized lifetime cost of sulphur abated ranges from £442/tonne to £738 with an average value of £550, all in £1983. (The average figure is thus £745 at £1989, or \$1200/tonne S. The costs per tonne SO_2 removed would be about half this.) Brackley (1987) estimates the cost of SO_2 removal using FGD as \$1,150 /tonne S removed with 90 per cent removal, rising to \$3,000 /tonne S for the next 5 per cent removed in going from 90 to 95 percent removal, in both cases using coal with 1% sulphur content. This figure is very close to Dudley's estimate.

The effect on the cost of electricity generation would be about 10 - 15 per cent of the cost of generation from coal-fired power stations, or possibly 6 percent of the price paid by customers.[12] This can usefully be compared with the predicted size of the 'nuclear levy' of 11 per cent of the sales price, which will be paid by consumers of fossil-fuel generated electricity after the privatisation of the CEGB to cover the cost of supplying 20 per cent of total electricity by non-conventional (mainly nuclear) means.[13]

Dowlatabadi and Harrington (1989), in a rather critical account of US estimates of the costs of large programmes to reduce total emissions by 8 million tonnes per year from the current levels of 25 million tonnes, cites various estimates of the average cost per ton SO_2 reduction. Thus the Congressional Budget Office (1986) gives the least cost method of making this reduction (by allowing the utilities to choose how best to meet the standards) as \$360/ton, and \$400/ton if they must continue to use the same coal as originally instead of substituting to lower sulphur content coal. These are average costs, and there is considerable agreement that the marginal costs of high levels of sulphur removal are substantially higher than small reductions. If the marginal cost were twice the average cost at this programme level then the US figures would be close to the CEGB figures. Estimates based on German data applied to East Germany suggest that the capital cost per kw capacity are DM600 using the Wellman-Lord method.[14] At present the capacity of the 15 larger plants is 13.3 GW which is responsible for 1.98 million tons of SO_2 per annum. The capital cost would be DM 8 billion, and the estimated annual cost of operating less the value of the sulphur sold on world markets would be DM 1.6b, or DM 600 per ton SO_2 removed, or \$300 per ton SO_2, which appears rather low.

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[12] Based on estimated capital costs of £100-£200/Kwe capacity, 5 - 7% discount rate, 30-40 years lifetime, 70% load factor, and 1.7% sulphur content coal (ie for Drax). The calculated figure is reassuringly close to that given in 1984 by the CEGB in HMSO (1984, para 5.93).

[13] This suggests that the cost of non-conventional alternative is 44 per cent of the final price (ie 11% borne by 80% of the total allocated to the 20% non-conventional), which seems unreasonably high. But detailed estimates of the cost of nuclear power are not yet available.

[14] See *Acid News*, No 3, July 1989, p9.

Table 5 Estimates of costs of reducing SO_2 by various means

Source	*Action*		*Cost 1989$/tonne SO_2 removed*
A	moving to low sulphur gas oil		$2,560
A	moving to low sulphur fuel oil		640
A	Fluidised bed combustors (FBC):		
		new boilers	96
		existing	2,240
A	Flue gas desulphurisation (FGD):		
		new plant	256
		existing plant	640
B	Drax FGD new plant		350
C	FGD retrofit 2000 MW plant		400-750
D	60% reduction from 40 GW CEGB coal capacity		600
E	FGD 90% removal		600
E	FGD marginal cost of next 5% removal		1,600
F	US coal generators, coal switching, Av cost		400
F	US coal generators, no coal switching, Av cost		460
G	East Germany, Wellman-Lord FGD, net of S sales		300
H	move from 2.15% to 1% sulphur heavy fuel oil		380
H	move from 1% to 0.7% sulphur heavy fuel oil		825

Sources:
A Environmental Resources Limited (1983, p137, uprated by 1.28 to $1989).
B Based on Layfield (1987) and Jeffrey (1988).
C Longhurst *et al* (1987).
D Dudley *et al* (1985, p121).
E Brackley (1987).
F Congressional Budget Office (1986) in Dowlatabadi and Harrington (1989). These are average costs. Marginal cost might be twice average cost.
G *Acid News*, No 3, July 1989, p9.
H Alfsen *et al* (1986).

An alternative option for reducing SO_2 emissions is to switch to lower sulphur content fuels. Thus Alfsen *et al* (1986) calculate the cost of switching from high to low sulphur heavy fuel oil as 2,300 NOK/ton SO_2 removed (ie $380) when moving from 2.15% to 1% HFO, and 5000 NOK/ton removed (= $823) when moving from 1% to 0.7% HFO. Table 5 shows similar calculations of switching from high to low sulphur fuels for the EEC given in Environmental Resources Limited (1983, p137).

What stands out from Table 5 is the wide variation in the costs of the most common proposed method of dealing with large power stations—flue gas desulphurisation, or FGD. In part this variation may be explained by differing degrees of sulphur removal—estimate E shows that the marginal cost is sharply increasing. The East German estimates may be based on lower construction costs or a more optimistic view of the value of recovered sulphur. The figures for FGD from source A seem rather low when compared to other estimates. Unfortunately there are no European estimates for the important option of shifting to low sulphur coal. In part this is because, unlike oil, there is no clearly defined world market price for the two grades of coal that would allow a robust estimate to be made of the differential cost of shifting from high to low sulphur coal.

The study by Environmental Resources Limited (1983) concludes that the estimated total damage caused by acid rain might be in the range $0.6-4.5 billion per year, of which the larger part is damage to buildings, then to forests, then to crops, with fisheries negligible. The costs of reducing SO_2 emissions in the year 2000 by 10-27% by adopting FGD and FBC in new boilers might be $1.9-6.5 billion per year, depending on the amount of new capacity installed. The cost of retrofitting might be $9 billion to achieve a further 38-42% reduction. The cost of new FGD might be to raise the cost of generating electricity from coal by 10-15 per cent, and the effect of retrofitting existing stations might be to raise generating costs by nearly twice that. Since the price of electricity would be determined by the marginal cost of new plant, the effect of forcing older plant to adopt FGD would be to decrease their profitability and equivalently, to write down their capital value sharply. Once installed, though, their operational life should be the same as new plant. New gas-fired generating capacity will become relatively more attractive than coal-fired capacity, as low sulphur gas is more readily available. Recent developments which have greatly raised the thermal efficiency of gas turbines makes these cost-effective for base load at current European gas and coal prices, and may offer a lower cost alternative way of reducing acid rain emissions. The combined impact of more stringent emissions standards, movements towards a single market in energy with the consequential reductions in Government subsidies and protectionist policies, and the new gas turbine technology may sound the death knell for the European coal industry.

2.5.1. Costs of reducing NO_x emissions Estimates of the cost of reducing NO_x emissions are somewhat harder to find, partly because most studies have concentrated on emissions from cars, where other pollutants were also being reduced, and partly because the technology for removing NO_x from stationary sources is not yet commercially proven. There are extensive studies of the cost of reducing NO_x emissions from vehicles by the use of catalytic converters and engine modifications, mostly for the US. It is possible to use the most recent to calculate costs per tonne NO_x removed. Using the information supplied by Crandall *et al* (1986) it appears that the costs of introducing the 1975 standards compared to no regulation might be $5,500/tonne removed, and the additional costs of introducing the 1981 standards starting from the 1975 standard would be $9,000/tonne removed—illustrating the rising marginal cost of abatement as emissions standards are tightened.[15] Crandall *et al* (1986, p114-5) estimate that the programme costs for the US of the more stringent 1984 standards might be about $20 billion per year with a replacement rate of 10.5 million cars, which is several times rather optimistic estimates of the

[15] These are based on a vehicle lifetime of 12 years, annual distance driven of 8,000 miles, and a real discount rate of 3%. Higher discount rates increase the cost.

potential benefits of reducing pollution. (Safety regulations in contrast, though expensive, seem to have been justified on cost-benefit criteria.) Another way to put the costs of emissions into perspective is to note that the capital cost of emissions equipment in 1981 was 6.7 per cent of the new car price, and the reduced fuel efficiency increases fuel consumption by 5 - 15 percent. The British estimate that emissions controls might raise driving costs by about 4 per cent is consistent with these figures.

Recent figures from Europe given in IME (1987) suggest that the cost of meeting proposed emissions standards are \$5,800 - \$10,600/tonne NO_2 removed, with lean-burn engines somewhat cheaper than three-way catalytic converters. This compares with costs of between \$200 - \$4,000/tonne removed from stationary sources (using modifications to burner design) and between \$4,000 - \$10,000/tonne removed using (relatively untried) catalytic converters. The best method of dealing with emissions from large combustion plants is disputed. Selective catalytic reduction is costly (perhaps \$150-200 per kw capacity) and possibly unreliable. Low-nox burners are currently under development and are increasingly adopted in new plants. According to *The Economist* (Dec 23, 1989) there is a new technology, tested on a 150 MW power plant in West Germany, and in about 20 industrial applications, called selective non-catalytic reduction. The technique involves injecting a urea-based liquid ('Noxout') into the furnace to reduce the NO_x to nitrogen. The capital cost of retro-fitting is low (\$15-20/Kw capacity) and the cost of NO_2 removal is about \$1,500/tonne, or less than one quarter that of selective catalytic conversion.

3. Acid rain and environmental policy

Acid rain is no respecter of national boundaries and has thus raised international concerns. Debates about international rights and obligations appear to be based on legalistic rather than economic principles—of equity, uniformity and the appeal to simple principles embodying such notions, rather than to the more finely adjusted and individually variable notions of costs and benefits. We shall see first the outcome of this international debate, and then contrast its proposed solution with an economically rational solution.

3.1 Policy to date

The debate on acid rain and on appropriate responses has been conducted in two different forums. The initial pressures came from the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP) which has 34 members from Europe and North America. Much of the pressure here was exerted by the Scandinavian countries and Canada, who are both large net importers of acid rain because of their unfortunate downwind location relative to their polluting neighbours in Europe and the United States. In 1982, Norway and Sweden pressed for the signatories to reduce SO_2 emissions to 30% below 1980 levels by 1993. This led to an informal '30% Club' founded in Ottawa in March 1984, and, in July 1985 21 countries, but not including the US and the UK, signed a protocol at the third meeting of the UNECE LRTAP Convention in Helsinki.

Whereas the Scandinavians were initially primarily concerned with the acidification of lakes and streams and consequent loss of fish, West Germans were worried about the impact their own industry was having on the environment, concerns which were reflected in the growing political power of Green parties in the early 1980s. An emotive campaign in 1982 drew attention to the problem of *Waldsterben* or forest death, in which official estimates showed that over half the forest area had suffered damage, attributed to acid rain. For a variety of political reasons described in more detail in Berkhout *et al* (1989), a Large Combustion Plant Ordinance (Grossfeuerungsanlagen-Verordnung or GFAVo) was enacted in June 1983, under which flue gas desulphurization equipment would be fitted to 37 GW of coal fired power stations and to the early closure of 12 GW. Not surprisingly, industry protested that the costs of this programme, which were to be borne by electricity consumers, would harm West Germany's competitive position in international markets, and this led the government to press for similar standards being adopted for the whole of Europe. The European Commission proposed a Large Combustion Plant (LCP) Directive based on the GAFVo in December 1983, calling for a cut in SO_2 emissions by 1995 by 60% to 40% of their 1980 level. After much debate, described in Skea (1988), the UK finally agreed to reduce SO_2 emissions from existing large plant to 20% below its 1980 level by 1993, by 40% below by 1998 and to 60% below by 2003, and nitrogen oxides (on the same basis) by 15% to 1993 and 30% by 1998. The Directive also provides for stringent emissions standards for new large combustion plants which the UK accepted.[16] In November 1988, the UK Environment Minister, Lord Caithness, signed a UN protocol in Sofia committing the UK and most leading industrial countries to freeze the level of nitrogen oxides at 1987 levels until 1994 and by 1996 to agree to further reductions based on critical levels.

The proposed agreements can be described as aiming at uniform reductions in emissions from each country relative to an arbitary starting date, and without regard to the costs and benefits at the country level.

3.2 The economics of designing a policy for acid rain

Welfare economics has developed a range of theories to deal with externalities and public good (or bad) problems such as acid rain, which hardly need rehearsing here. Acid rain differs from 'greenhouse gases' like CFCs and carbon dioxide in that it is not a pure public good (or bad) in the Samuelson sense. Acid rain causes damage where it is deposited, and there appears to be a reasonably linear relationship between emissions and depositions, as Table 2 suggests for the aggregate. Moreover, the EMEP is predicated on there being a predictable and stable relationship between the location of the source and the deposition, at least averaged over a year.[17] Detailed work, reported in Derwent (1988, 1990), suggests that both SO_2 and NO_x depositions can be statistically predicted, given the location of the source, and that depositions at specified sites can be traced back to their originating source. The main differences between the two pollutants is that a higher fraction of SO_2 is deposited within 100 km of the source, and that a larger fraction of SO_2 can be accounted for by individually identified large stationary sources,

[16] The details are reported in *Our Common Future: A perspective by the UK on the Report of the World Commission on Environment and Development*, Department of the Environment, July 1988.

[17] In the short run, the location of the deposition will depend on the current weather conditions and wind direction. Over a year or possibly longer these will follow predictable patterns in aggregate.

predominantly power stations. Nitrogen oxides are produced both by large stationary sources and by numerous small mobile sources—road vehicles.

Technically, acid rain appears to be a depletable or rival good in consumption, in that if one tonne of sulphur falls on a given local area, then that tonne cannot fall elsewhere, and reduces the amount which will harm others by that amount. It might appear to be a simple bilateral externality of the kind considered by Coase (1960). But there are two important differences from the simple case of bilateral externality in which a well-defined polluter deposits pollution on a single well-defined recipient. On a given day, the plume of pollution from a source can be tracked, and the amounts of acid rain deposited along its flight path estimated. The plume may pass over many different cells, and thus deposit pollutants on a number of different countries. Even over a short period of time, then, more than one recipient will be affected by the single pollutor. Over longer time periods the number of recipients will be larger as the wind direction and strength vary, even if the average deposit received by a given country from a given source is well-defined over some time period. If the recipients are to bargain over reductions in acid rain, they will have to agree among themselves how to coordinate their bargaining, and how to share any costs involved.

The second complicating fact is that reducing emissions requires a fixed durable investment in FGD equipment, as well as variable costs (largely additional energy costs to operate the FGD). Even if there were only one recipient at any date, there will be a number of recipients over the life of the FGD equipment, all of whom benefit from its installation. Again, there are joint beneficiaries, and the FGD plant itself is like a public good, available to all potential recipients once installed. Third, there are several alternative ways of reducing the damage caused by the main source of SO_2, power stations. One is to reduce emissions by investment in FGD, another is to reduce emissions by reducing the amount of electricity generated by a given station. In an integrated electricity grid, power stations are ranked by merit order, with the cheapest variable cost plants given the highest rank, and the most expensive the lowest. The despatcher calls for power from the highest ranked stations and moves down the merit order until total demand is satisfied. On a given day the wind direction and strength may cause the plume from one power station to deposit acid rain where it does high damage, while that of another power station may fall largely on the sea or on areas where damage costs are low. If the power despatcher includes in the costs of generating electricity the social cost of the damage done by emissions, and if these costs are varied with changing weather patterns, then the position of the power station in the merit order will change with the weather, and the damage done by the pollution will be reduced, even if the total amount of pollution remains roughly the same.[18] Much the same result can be obtained by holding stocks of low-sulphur coal, which are burned when the wind direction is adverse in preference to the use of normal (higher sulphur) coal.

Finally, if the public good and coordination problems facing the recipients of pollution from a given country can be overcome, then it should be possible for these recipients to bargain

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[18] Estimates presented by Sir Sam Edwards of the Cavendish Laboratory, Cambridge, suggest that this may be a very cost-effective way of reducing pollution costs for UK generators.

towards an efficient level of pollution, as argued by Coase (1960). How this might occur is discussed below.

A number of very important economic consequences flow from these facts. First, the damage done by acid rain depends critically on where the deposition occurs. If it occurs over the sea, it is unlikely to have much harmful effect, as the sea is naturally alkaline. If it occurs over sparsely populated areas with class 5 (acid tolerant) soils, then again the damage may be low. If it falls on rivers and lakes, it may be very damaging ecologically, but financially the costs look trivial compared to property damage. If it falls on densely populated areas, the property damage will be substantial. As a consequence, the benefits of reducing emissions will vary significantly from source to source, since different sources will have very different deposition patterns. Second, the costs of reducing SO_2 emissions will vary significantly with the source. The cost per tonne SO_2 reduction will depend on whether an integrated FGD system can be installed at the time of construction of a new station, or whether it must be retrofitted. If it is retrofitted, the cost will depend on the number of Gwhours the plant will produce over the remainder of its life. For a base load plant with a long expected life, the cost will be low, but for a plant near the end of its life, or one which is primarily used for peaking, the cost will be high.

If the object is to maximise the Europe-wide net benefit of pollution reduction, then three types of relationships are needed. EMEP divides Europe up into 150km square cells and measures depositions in each cell. This is probably be the finest degree of disaggregation practical, at least in the medium run. For each of these cells one needs a damage function, relating total damage to depositions. If q_i is total quantity of pollutant deposition in EMEP cell i, then the total damage is $D_i = D_i(q_i)$. For each source, j, one needs a cost function describing the cost of emission abatement. If e_j* is the uncontrolled or initial level of emission, and if e_j is the actual level of emissions then the amount of abatement is $a_j = e_j* - e_j$. The cost borne by j is $C_j = C_j(e_j*-e_j)$, with $C_j(0) = 0$, and the marginal cost of abatement, $dC_j/da_j > 0$.

The final component is a transport matrix, relating emissions at various sources to depositions at various destinations. If t_{ij} is the amount of deposition at location i per tonne of pollutant emitted at source j, then the vector of depositions $\mathbf{q} = \mathbf{Te}$, where $\mathbf{T}$ is the matrix whose ij-th element is t_{ij} . Total social costs to be minimised by the choice of $\mathbf{e}$ are

$$S = \sum_j C_j(e_j * - e_j) + \sum_i D_i(\sum_j t_{ij} e_j).$$

The first-order conditions are

$$\sum_i D_i^{/} . t_{ij} = C_j^{/}, \qquad j = 1, 2, ..., n.$$

Detailed figures on the damage functions are not readily available, but it is interesting to consider the implications of three different measures of damage. First, we need to measure the intensity of pollution produced by a given source, and the natural measure is SO_2 per square mile of the recipient country per tonne emitted. The total damage done will then depend on what is damaged. Given the dominance of property damage one possibility is to make the marginal damage proportional to capital stock times pollution intensity. Property (and possibly other relevant components of damage) are likely to be proportional to GDP, and so if we multiply tonnes deposited per tonne emitted by GDP per square mile, then we have a rough measure of property damage. Given the unreliability of estimates of GDP from socialist countries, the measure of GDP finally adopted are the purchasing power parity measures for 1985 given in Summers and Heston (1988). The **T** matrix can be derived from the fuller version of Table 1, given as table A2 of the Appendix, and is presented as Table A3. The resulting marginal damage costs are shown in Appendix Table A4. It is more useful to compute an index of these costs and Table 6 below gives these figures for various source countries, ranked by the damage they export, relative to Britain. The index of property damage in the column headed 'prop'.

An alternative measure might be the number of population in the country affected, again multiplied by the pollution intensity. This index is given in the column headed 'Pop'. Finally, as a measure of willingness to pay for reductions in pollution, the third measure multiplies pollution intensity by GDP per head, and is given in the column headed 'GDP'.

Four countries appear to be as or more damaging than the UK by the property damage and by the population criterion, and eleven by the GDP measure, though one should remember that the estimated damages have been very crudely estimated. As SO_2 does not travel very far, and as the damage is weighted by the GDP density, countries like Belgium which are situated near wealthy neighbours yield high damage costs per tonne SO_2 released. At the other extreme, the Soviet Union, which produces a vast amount of SO_2, deposits most of it within its own borders, and even here the damage cost is low because of the low population and GDP density.

It is interesting, largely as an illustrative exercise, to see what kinds of SO_2 reduction would be justified given the marginal costs of abatement estimated by Amann and Kornai (1987). That source gives plots of total and marginal abatement costs by tonnage abated by country for the year 2000 (ie allowing adequate time to undertake adjustments, and predicting future levels of emissions without abatement). These plots have been used to determine marginal costs of abatement and the levels of reduction which would be justified—the level at which the marginal cost of abatement is equal to the marginal damage which would be appropriate for a 30% overall reduction in emissions.[19] If we had absolute measures of the damage then it would be possible to determine the appropriate total and individual reduction which is cost-justified, but as we only have an index of relative damage, this alternative finds the least-cost method of making the 30% reduction.

Table 6 gives the implied reductions which minimise costs. In some cases the estimate is reasonably robust, assuming the correctness of the underlying abatement cost figures. In other cases, indicated with a single star, the marginal cost of abatement was flat at a level close to the estimated marginal damage, and so the appropriate level of reduction is poorly identified. Figure 1 illustrates the problem for the UK. When the damage is measured by the third measure based on ability to pay (GDP/head) the marginal damage and marginal costs almost coincide over a

[19] The calculations were done iteratively, first linearising the estimated marginal abatement costs, then finding the level of marginal abatement cost corresponding to the index number of 100 in the damage index at which the overall reduction of 30% was achieved. This cost was then multiplied by the damage index for each country to give the country level abatement cost, and this figure was used to read off the appropriate level of abatement from the detailed abatement cost schedules. The overall level of reduction was then calculated, and is as shown in Table 6—close to 30% except for the last column.

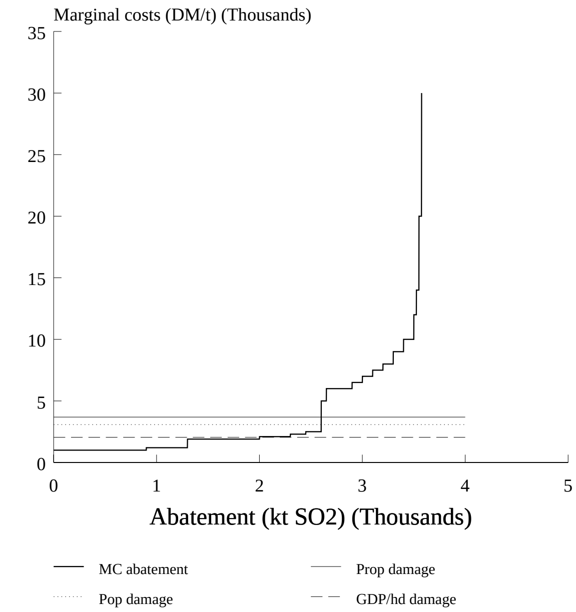
range, whereas for the other cases the marginal damage intersects the MC schedule at a vertical step.

The reductions proposed vary significantly by country and also according to the measure of damage. They are not simply related to the damage index, as abatement costs vary considerably by country. Thus although Luxembourg has a rather high damage index, the marginal abatement costs are so high that very low reductions are justified. In some cases the abatement costs of further reductions are high relative to the benefits of the reduction and for these countries zero reduction is proposed.

One obvious conclusion is that a uniform reduction in pollution from an arbitrary date like 1980 is most unlikely to maximize social net benefits. Instead, abatement should take place in an order determined by the size of the excess of social benefits over abatement costs, continuing until the net benefit falls to zero. In the illustrative case shown in the table, Britain appears to warrant a more radical abatement programme than most of her EC partners, primarily because abatement costs are quite low and damage costs appear to be high. It should be stressed that this conclusion is only as good as the data on which it rests and the methodology for estimating damage costs.

The exercise reported in Table 6 makes strong assumptions about costs and benefits, and it is therefore important to look at other similar exercises based on an economic approach to costs and benefits. Mäler (1989) has estimated cost and damage functions for the same set of European countries and used the 1984 EMEP transmission matrix to compute the efficient level of abatement. He assumed the costs of abatement were quadratic (ie the marginal costs were linear, which is a poor assumption for many of the countries)[20]. As in Table 6 the cost schedules were calibrated to cost estimates made at IIASA for the year 2000 using the schedules in Amann and Kornai (1987). He also assumed that damage costs were constant per tonne of sulphur deposited in each country (as was done in Table 6), but that the damage 'represents the evaluation of the damage that respective governments make today.' These estimates are made as follows. If each country takes the level of emissions of all other countries as given (the non-cooperative solution), and if each country then balances

Figure 1
Abatement cost function for UK



Amann and Kornai (1987)

[20] In an earlier draft this assumption was employed as a first approximation and it resulted in substantially lower reductions for the UK in particular.

the marginal cost of domestic abatement with the marginal benefits of reduced domestic deposition, then it will solve the equation

$$D_i' . t_{ii} = C_i' .$$

Table 6 Index of damage per tonne SO_2 emitted

	Index of damage			Reduction %		
Source	prop	pop	GDP	prop	pop	GDP
Belgium	175	160	149	39	30	26
Netherlands	164	154	121	65*	40	18*
GDR	156	139	155	83	80	75
France	100	92	146	29	28	28
UK	100	100	100	60	60	40*
FRG	95	100	136	36	28	23*
Luxebourg	84	69	88	6	6	6
Switzerland	78	68	97	25	25	25
Czecholsovakia	68	80	116	55	54	52
Italy	65	76	82	39	29	22
Denmark	46	41	111	31	21	35
Hungary	46	69	104	60*	65	70
Austria	45	51	92	30	30	30
Poland	43	65	95	26	33	33
Yugoslavia	35	55	77	63	75	75
Spain	30	37	69	10	11	36*
Ireland	27	33	51	11*	11*	12*
Portugal	25	49	53	10*	40	30*
Bulgaria	18	33	50	17	26	27
Greece	13	27	38	10*	60*	60*
Sweden	10	9	109	0	0	3
Albania	9	26	17	0	10	0
Romania	8	20	45	0	7*	15*
Turkey	8	25	30	0	0	0
Norway	7	5	104	0	0	6
Finland	7	7	105	0	0	18
USSR	5	6	73	2	3	20
Average	54**	59**	89**	30***	30***	33***

Notes: Damage is deposition per tonne emitted times damage measure, which is GDP/sq. mile (prop), population, or GDP/head, taking damage by UK as 100. Countries are ranked by total damage using the first measure of damage. Reductions are those required to achieve an overall reduction of 30% at minimum cost according to the three measures of cost. Damage index for USSR are related to population densities in the EMEP area alone.
* estimate unreliable as MC of abatement flat
** unweighted average
*** weighted average

If C'_i and t_{ii} are known, then D'_i can be deduced. On the strong assumption that the Governments of European countries were rational, selfish and non-cooperative, the revealed marginal willingness to pay for abatement can thus be deduced. (Mäler, 1989; 1990, p90).

His estimates of the efficient level of abatement involves an aggregate reduction of 8.9 million tonnes sulphur, or a reduction of 39%, yielding a net benefit on DM6.3 billion. The main conclusion to be drawn from this estimate is that the efficient solution requires very different reductions in each country, and that a uniform percentage reduction would yield much lower social benefits. His basic table gives the percentage reductions and net benefits for selected countries shown in Table 7. For comparison the percentage reductions for the three different measures of damage from Table 6 are presented next to Mäler's estimates. The relation between the different proposed reductions is also graphed in the scatter diagram in Figure 2. In interpreting the figure it should be remembered that Mäler's estimated overall level of reduction is 39 percent, rather than the 30 percent used in the calculations of Table 6, so that his figures will be on average higher. It is interesting (and rather surprising, given the very different approach adopted) that the order and the degree of pollution reduction in Mäler's calculation is not so different from those appropriate to some measures of Table 6. Table 8 gives the correlation matrix between the various proposed reductions. As expected, Mäler's measure has a lower correlation with those proposed here than each of the three measures have with each other.

Figure 2
Relation between proposed reductions

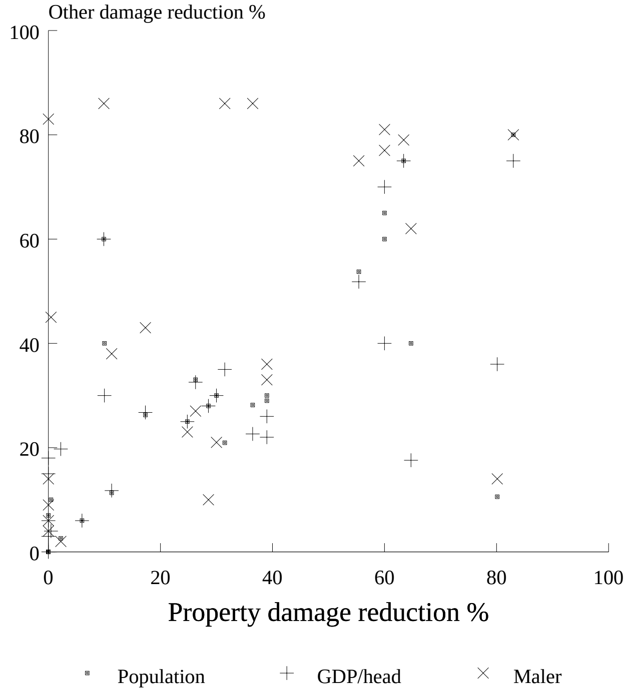


Table 7 shows, on Mäler's figures, that not all countries are net beneficiaries of the efficient plan, and the losers might not agree to join the agreement. The fact that most countries gain from a move to the efficient plan is also rather misleading, for if countries have to agree to the percentage of the efficient level of pollution reduction to jointly undertake, then many countries will lose from the final increments—say, in going from 90% to 95% of the plan. Mäler argues that in a repeated game with appropriate side payments, supporting the efficient solution could be made individually rational—deviating countries would trigger punishments in the form of increased depositions, and would lose any side-payments.

Table 7 Net Benefits from efficient solution

Country	Percentage reductions Prop	Pop	GDP/hd	Mäler	Net Benefit DM m.
GDR	83	80	75	80	11
Netherlands	65	40	18	62	565
Yugoslavia	63	75	75	79	344
Hungary	60	65	70	77	5
UK	60	60	40	81	-365
Czecholsovakia	55	54	52	75	152
Belgium	39	30	26	36	191
Italy	39	29	22	33	-81
FRG	36	28	23	86	328
Denmark	31	21	35	86	119
Austria	30	30	30	21	324
France	29	28	28	10	879
Poland	26	33	33	27	599
Switzerland	25	25	25	23	192
Bulgaria	17	26	27	43	-7
Ireland	11	11	12	38	71
Spain	10	11	36	14	-29
Portugal	10	40	30	19	10
Greece	10	60	60	86	53
Luxembourg	6	6	6		
USSR	2	3	20	2	1505
Albania	0	10	4	45	24
Romania	0	7	15	83	422
Finland	0	0	18	14	-2
Norway	0	0	6	6	272
Turkey	0	0	0	9	68
Sweden	0	0	3	4	606
Total	30	30	33	39	6290

Sources: Mäler (1989); Table 6 above

Note: Net benefits as estimated by Mäler.

Table 8 Correlations between different proposed reductions

R^2 percentages

	Prop	Pop	GDP/hd
Prop			
Pop		72	
GDP/hd	52	82	
Mäler	38	46	39

3.2.1 Bargaining over abatement Let us return to consider the efficacy of bilateral or multilateral bargaining over pollution abatement, in the spirit of Coase. One can imagine two possible allocations of property rights. The status quo is one in which each country is free to pollute its neighbours. The natural alternative is one in which each country agrees to a certain annual level of emissions—eg 70% of the 1980 measured level. Emissions above that level are only acceptable with the agreement of the recipient countries. In the second case, the polluter would have to pay for increased emissions, and a natural offer would be an amount between the marginal cost of abatement and the marginal damage done to the recipient. This principle is termed the Polluter Pays Principle or PPP. The important question of how this should be decentralized within the polluting country will be considered below, but in principle any new pollution source would have to buy the right to release a given number of tonnes of pollution from an existing source, who would then reduce its emissions by that amount.

In the status quo situation, new polluting sources could set up and pollute at no apparent cost. The costs of damage would be borne by the recipients (the Victim Pays Principle, or VPP). Does this mean that the status quo would favour entry by more polluters? Ignore for the moment the problem of coordination among the recipients, and suppose that there is one representative recipient who is willing to bargain with the polluting country, and one polluter within the country, such as the CEGB in Britain. Suppose also that there are no market failures in selling electricity, of the kind considered in the next section. Then it is a standard Coase result that the allocation of property rights does not affect the efficiency of the final outcome. The first task would be to agree a reference path of pollution in the absence of any cleanup—presumably based on a forecast of energy demand and emissions per GWhr. The recipient representative would then offer to pay the CEGB for reductions below this level. In both cases the social cost of producing electricity would be the same and would include the social cost of releasing pollutants. In the first case (PPP) the CEGB would pay the recipient for extra emissions, and in the second case (VPP) the CEGB's total profits would decrease for each extra tonne released—the cost of emissions would be the reduction in profit.

There is a further similarity in the operation of the two forms of property right, in that recipient countries may find the benchmark figure of 70% of 1980 levels too high, in which case that becomes the new status quo point. Economic growth which raises the wealth density is likely to make further absolute reductions cost-effective, especially as technical progress should continue to lower abatement costs. The recipient would then wish to pay for reductions below the status quo, just as before. The only difference between the two cases would be in the size of the lump-sum transfer from recipient to polluter. In the second case the recipient would pay the polluting country the damage cost of the difference in 70% of 1980 levels and the reference emission path.

3.2.2 Complications caused by regulation and privatisation Suppose the electricity supply industry (ESI) is privately owned but regulated, as in the US. If regulation is based on actual costs (again as in the US), with the allowable price related to the average cost, then the *average* cost of electricity is lower under VPP than under PPP, and hence prices will be lower, demand higher, and the level of pollution will be inefficiently high. The appropriate method of price regulation would be to determine benchmark costs based on a pre-determined level of emissions control, and allow the generators either to keep payments for reductions, or force them to pay for emissions above the status quo level. Naturally this particular problem does not arise with price-based regulation of the RPI-X typified by the regulation of British Gas or British Telecom.

The present proposals for privatising the electricity supply industry in the UK involve splitting the CEGB into three components. All nuclear power stations will remain in public ownership (and produce no acid rain pollutants anyway). About two-thirds of the remaining capacity will be allocated to National Power, the rest going to Powergen. The grid will be owned by the twelve distribution companies (Discos). New entrants and existing private power suppliers will be free to sell to the grid or Discos. The price of electricity sold by the Discos will be regulated, but not that sold by the generators, who will be subject to normal market competition.

Suppose that the British government accepts a target level of abatement (eg 70% of 1980 levels), and must decide how to implement this target. The two standard instruments available are emissions taxes (per tonne of sulphur or NO_x emitted), or standards (eg tonnes SO_2 emission per GWhr). Standards applied uniformly are typically inefficient, and economists usually argue that their inefficiency can be eliminated by auctioning a fixed number of licenses to pollute 1 tonne SO_2 in a given year, equal to the total allowable emissions. If polluting firms are competitive, and the tax is appropriately set (at the auction price) then standard arguments show these two methods to be the same under certainty.[21] But the British ESI will not be competitive, at least in the near future until considerable entry takes place, and so this argument does not apply. In the Appendix a simple Cournot duopoly model is constructed to see how an auction market in license to pollute might work.

The model shows that if firms compete in the input market (for licenses to pollute) as well as in the output market, then it may pay them to raise the cost of inputs to each other, as a way of increasing the final product price. The idea that raising rival's costs may be a preferable competitive strategy to predatory pricing is not new, though the usual assumption is that firms are differentially affected by input costs (see eg Salop and Scheffman, 1983). The normal argument is informal, if intuitive, and ignores some of the subtleties in modelling competition in both input and output markets. The model in the Appendix considers two identical firms which can reduce emissions at a cost, and which are allocated or bid for licenses to pollute. If they behave as Cournot duopolists in the output market, then their joint profits will be higher under an asymmetric allocation of licenses than with the efficient symmetric allocation. The reason is that an asymmetric allocation raises marginal production costs of the smaller firm and allows the two firms to compete less intensely, to their joint benefit. What this implies is that there is a mutually beneficial transfer price for licenses between the two firms which compensates the firm with the smaller number of license more than its fall in profits. Unfortunately the increase in joint profits is smaller than the increased deadweight loss of the inefficient allocation of licenses, so net social surplus decreases.

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[21] Just as tariffs and quotas are equivalent under certainty, though not, as Newbery and Stiglitz (1981) show, under uncertainty.

Consequently, taxes and quotas or auctioned licenses are not equivalent in the presence of imperfect competition—for taxes can in principle mimic the competitive market provided the government knows that abatement cost function and can set the tax level appropriately.

There are additional arguments differentiating licenses and taxes. If there is uncertainty about the location of the damage and abatement cost functions, then it may be preferable (in the sense of lowering the expected social cost) to choose one instrument rather than another. In Appendix B taxes are shown to dominate quotas for a competitive industry in which here is uncertainty about the slope and level of the marginal cost of abatement schedule and similarly about the slope and level of the marginal damage cost schedule, provided the expected slope of the marginal cost of abatement exceeds the expected (absolute value of the) slope of the marginal damage schedule. This seems plausible, for the damage schedule is likely to be rather flat if damage is proportional to depositions, while the age distribution of large power plants will introduce an upward slope into the abatement schedule. The proportional relative advantage of taxes over quotas will be $(b - d)\sigma^2/(b + d)$, where b is the slope of the abatement schedule, $-d$ is the slope of the damage schedule, and σ is the coefficient of variation of the slope of the abatement schedule, a measure of the uncertainty about the marginal costs of abatement.

3.2.3 Choosing policy instruments Licenses to pollute which are tradeable and are auctioned off have several attractions. The first is that they are the natural instrument to meet international agreements couched in terms of total emission levels, typical in recent negotiations. Second, they can overcome organized resistance from the industry affected, since firms can be allocated licenses proportional to current emission levels. The costs of abatement fall on consumers (who forgo the revenue from auctioning off the licenses), and on new entrants, who have to buy licenses from incumbents. This anti-competitive feature has already been discussed, and is a reason for industries favouring stringent standards with appropriate grandfather clauses. Third, other countries can negotiate further reductions by buying up licenses, providing the issuing authority recognizes their property rights in such licenses.[22]

The limitations of licenses are that they allow anti-competitive behaviour, they are inferior to taxes in maximizing the expected net benefit of control, and if they are allocated rather than auctioned, they involve a lump-sum transfer to industry, of a kind comparable to giving away rather than selling state-owned assets upon privatisation. If instead, taxes are used (or licenses are sold) then the government can reduce other taxes by the amount of extra revenue, and so reduce deadweight losses elsewhere in the economy. Put another way, most taxes raised revenue at the cost of some inefficiency, but corrective taxes raise revenue while improving efficiency. The latter are highly attractive sources of revenue, not to be lightly foregone. There is a further limitation which has not yet been mentioned, arising because of the nature of the abatement technology. Flue gas desulphurisation or denitrification requires heavy fixed investment with a lifetime equal to that of the plant—perhaps up to 40 years. Its cost-effectiveness will depend on the costs of emission in the future, and these may be harder to predict if they

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[22] If the issuing authority interprets the international agreement as only limiting the total amount of emission, and not the total amount licensed, then foreign purchase with the intention of reducing the total emission would be thwarted by an increase in the number of licenses.

are determined by a spot market in licenses at each future date, than if they are determined by a tax which is announced and expected to remain relatively constant.[23] This objection can be overcome by issuing plants with an estimated (but non-negotiable) lifetime supply of licenses, so that the plant owner would expect neither to be a net seller nor net buyer of licenses for the expected lifetime of his plant. The difficulty with this solution is that licenses for new plant would have to come from the liquidation of allocations from existing plants, though one should not exaggerate this difficulty. Old plants will have to be retired, and would at the predetermined appropriate date release licenses for reallocation.

Licenses work well for large stationary sources which are the main source of SO_2, but are not immediately applicable to small mobile sources, like vehicles which are increasingly responsible for NO_x. Licenses could be issued or auctioned to car manufacturers who would fulfil their quota, T tonnes NO_2 equivalent per year, by selling n vehicles each of which emits no more than t tonnes per year of normal driving, where $nt \leq T$. Subsequent vehicle testing would ensure that emissions were kept down to t.

Given the advantages and disadvantages of taxes and licenses, is there some compromise system which might do better than either pure instrument? The anti-competitive problem can be met by the government or its agency being willing to buy or sell licenses at a predetermined price. This would require an international agreement about the form in which abatement would be encouraged. If such agreements insist on quantity controls, then the agency would find it difficult to hold the price of licenses down to a satisfactory level needed to prevent the incentive to raise rivals' costs. If the international agreements merely defined the status quo for bargaining, then recipient countries could establish a market price for licenses, and might, if they overcame their coordination problems, be able to act as a quasi competitive force.[24] This, quite part from the increased efficiency of such a system, provides another argument for attempting to negotiate such treaties.

The main arguments against using pollution taxes are that recipient countries may fear that it has little effect on pollution levels, while the polluting companies see the tax as harming their profits. The first worry can be greatly alleviated by combining the tax with emission standards on new plant, designed to be cost-justified at the tax rate. (It may be enough to set limits which firms would choose to better given the tax, but which are sufficient to allay doubts on the part of recipients.) The second problem is primarily one of dealing with firms with existing inappropriate plant. It may be sufficient to draw up a reference emission plan for each large plant, specifying the maximum allowable emissions in each of the next 20 years (perhaps falling by 5% of the initial value each year, down to zero). The plant would be entitled either to licenses or a tax rebate to this amount, but would have to pay for emissions above this level.

[23] It may be sensible to index the tax to per capita income in the recipient countries to allow for increasing damage costs, and this will be no less predictable than most taxes.

[24] If damage per tonne is relatively independent of pollution levels, and relatively independent of the year in which it occurs, then recipient countries can arbitrage the price from year to year, and the price should be maintainable within moderate bands, reducing the ability of dominant polluters to manipulate the price of licenses.

3.2.4. Licenses and charges Hahn (1989) provides a useful survey of the experience in dealing with environmental problems in four countries. The instrument most economists find attractive is the marketable permit, and he describes two important applications in the US. In 1981 Wisconsin allowed firms to trade the rights to pollute the Fox River, but resulting cost-savings were minimal. Hahn finds this unsurprising, as the main sources were regulated municipal utilities and oligopolistic paper and pulp manufacturers. The arguments above suggest that in both cases the incentives to trade to achieve efficiency will be muted, and it is interesting to have this empirical confirmation.

A much more important experiment was the US Environmental Protection Agency's emissions trading policy, with its various components allowing firms to meet firm level or area level standards by either internal reorganization of their pollution activities or by external trading or bargaining. The resulting cost-savings have been substantial, and Hahn cites estimates of cost savings of between \$525 million to \$12 billion, the wide range reflecting very incomplete monitoring of the cost-savings achieved within firms. Most of the gains came from internal trading, and pressures from environmental groups appears to have lead regulators to downplay the property rights nature of permits, thus undermining their marketability. Hahn concludes that 'emissions trading is best viewed as an incremental departure from the existing approach. Property rights were grandfathered.' (Hahn, 1989, p101.)

The most successful example of permit trading was lead trading in gasoline, where refineries could buy or sell the right to lead levels in gasoline, based on an existing standard per gallon. About 15% of total lead rights used were traded, with an active spot market and an intertemporal market via 'banking'. The systems appears to have worked well because it is easy to monitor and there was widespread agreement about the objectives of the programme, which were to gradually phase out lead in gasoline. Perhaps the most difficult aspect of environmental regulation is this problem of reaching a well-defined consensus on what to do—much of the environmental opposition arises because of distrust as to the objectives of the various parties.

If marketable permits have been rather disappointing, there appears to have been reasonably widespread acceptance in Europe at least for charges for pollution (primarily for water borne and solid pollutants). The revenue from these charges is normally earmarked for environmental cleanup, so the link between the charge and the resulting clean-up is seen to be fair, given that it would have presumably cost the polluter more to clean up the pollution itself.

3.3 A possible strategy

To date, inter-government negotiations have been guided by the principles of uniform reduction in emissions from a benchmark level (usually 1980 actual emissions) for large combustion plants, or towards uniform standards for mobile sources (i.e. cars). The attraction of this is that it appears to impose equal costs on the participants, so that it gives no commercial advantage or disadvantage to producers in each country. In fact, of course, countries which have become heavily dependent on nuclear power (France), hydroelectricity (Scandinavia), or gas (Netherlands), can achieve these emission reductions at lower costs than those like the UK and/or Germany still heavily dependent on coal. There are two serious problems with this approach. The first is political, and may be sufficient to derail the negotiations—some countries are net losers from such negotiations. The second objection is economic—the reduction in aggregate pollution damage is done at higher than least cost. Both probelms can be solved by the natural solution which allows beneficial and efficient bargaining. This would involve first agreeing a benchmark trajectory of allowable emissions (not necessarily equal for all countries), and then facilitating bargaining over deviations from this level. This might best be done by first estimating the marginal damage costs (or willingness to pay for abatement, if higher) by EMEP cell, then calculating the appropriate cost sharing formula for the group of countries affected by polluter *j*. Thus for country *i* its share of the payments to polluter *j* would be

$$\alpha_i = \frac{D_i^{/} t_{ij}}{\sum_k D_k^{/} t_{kj}}.$$

The recipients would then appoint a negotiator with powers to levy charges on recipients proportional to these cost fractions, up to the total damage level. The negotiator could then bribe the polluting country to make additional reductions.

The main problem lies in decentralising this within the polluting country. One solution would be to compute cost-effective emission levels for new plants (i.e. the level at which the marginal cost of further abatement is equal to the marginal damage) and set this (or a somewhat more lenient standard) as the benchmark for new plant. Deviations from this could then be taxed (if excessive) or rebated (if lower). Old plant could be similarly dealt with, at least for the large combustion plants, which are not too numerous in any country. Again, the first step is to determine the benchmark, presumably starting near current emission levels, but falling to zero at the estimated economic date of plant retirement. Where retro-fitting is evidently cost-effective, then the benchmark might be the retro-fitted emission level. The emission tax would be uniform over all plants, new and old, but would vary with location.

The current tax rate would be the amount offered by the recipient negotiator. The main problem is that FGD requires a large fixed investment to amortise over up to 40 years, and thus requires the investor to calculate the future benefits (i.e. taxes avoided) of installation. If these future benefits are uncertain, then abatement may be deterred. It would pay the recipient negotiator to offer formal contracts, possibly indexed to recipient GNP, in order to reduce uncertainty and thus induce greater abatement for the same cost. This would not completely solve the problem, at least in the UK, for the following reason. The argument advanced above suggested that it might be in the interests of the duopoly electricity generators to face competition from a high cost fringe of producers. It might therefore pay National Power and Powergen to sell off their least efficient, most polluting plant to independents. The buyers would be uncertain of the likely future demand for their power (assuming they are unlikely to secure long-term supply contracts with the Discos for more than a small fraction of their output). Even if they knew the future payment per tonne SO_2 reduced, they would be uncertain of their likely future electricity (and hence SO_2) production. They are unlikely to install FGD, and their operating costs will thus be raised (by the emissions tax). This in turn will raise the equilibrium price of electricity, and increase the profits of National Power and Powergen. In short, the asymmetry in abatement costs for new base load plant and old coal-fired plant makes the case for raising rival's costs stronger than that given in the Appendix.

One should not exaggerate this risk. Encouraging divestiture by the duopoly generators will at least increase the number of competitors, and should make entry-deterrent collusion more difficult, for the future benefit of consumers. It may also be that retro-fitting is not cost-effective for the kind of plant likely to be sold, in which case the problem does not arise. Nor does it seem likely that emissions standards and taxes would discriminate against new entrants, because the best practice generating technique for new entrants appears to be high efficiency gas turbine or gas-based combined heat and power systems, of modest capacity (130-300 MW). Emissions from such systems are naturally low, giving them an additional advantage over the existing large coal-fired conventional power stations.

If this seems utopian, it nevertheless appears likely that some international agreement on emissions will be reached, and it is in any case in each country’s own interest to reduce emissions, as such a large fraction is deposited within its own borders. This being the case, countries like the UK which have, or will have a privately owned electricity supply industry will still need a decentralised system of abatement. Standards, appropriately chosen to be cost-effective, may be satisfactory for new sources (and especially for small sources, like vehicles where the cost of individually monitored charges is likely to make them uneconomic). They are not so suitable for existing sources where some abatement is cost-effective. Here taxes (or charges) have obvious advantages, though bargaining between the relevant government agency (Her Majesty’s Inspectorate of Pollution) and the individual firm may be a satisfactory alternative. The attraction of an explicit tax or charge is two-fold—it signals to recipient countries the advantage of further negotiations (with side-payments) and it signals the advantage of developing more cost-effective abatement strategies. In their absence, the generators themselves have no such inducement, as they risk having the new technology mandated in more stringent standards (now cost-effective).

4. Summary and conclusions

In many ways, the debate on acid rain is quite encouraging from an economists’ perspective. EMEP has quantified pollution flows and provided the raw data for bilateral bargaining. The least satisfactory element so far is that of quantifying the damage costs (rather than the levels of deposition). Much commendable effort has been directed to identifying the ecological path-ways by which acid rain harms the ecological environment, but insignificant effort to quantifying the cost of this harm. It appears that property damage is more important than ecological damage, and that forest and crop damage is far greater than lake and river acidification, yet very little recent work is directed to quantifying property damage. No doubt this reflects the emotional level at which most environmental issues are discussed, but one might hope that the best way to lower the emotional temperature is to increase the factual and economic content of such debates.

The main findings reported here are that it is likely to be excessively costly to aim at a uniform reduction in emissions in all countries. Since some countries will gain while others lose from such proposals, it is likely to be politically difficult or costly to reach agreement. Instead, it seems more sensible to aim at reducing emissions where the benefits of reduced damage exceed the marginal costs of abatement, and to aim at methods of coordinating payments for abatement to overcome the ‘free-rider’ problem.

The other finding, less secure given the present state of scientific knowledge, is that reducing NO_x appears less urgent and less attractive than reducing SO_2. Certainly, the cost of reducing automobile emissions is high, the health gains small, and the ecological benefits uncertain. On the other hand, it may be cheap to reduce NO_x emissions from stationary sources, using better burner designs and chemical additives. If one priority is to be singled out, it is that of reducing the appalling pollution in the socialist countries in order to secure primarily health benefits. Given the inefficiency of energy use in these countries, it is likely that substantial reductions can be achieved at relatively low cost and high benefit, in many cases just by closing down unprofitable enterprises, and perhaps moving towards a more integrated energy market, relying more on trade and less on self-sufficiency.[25]

[25] See Hughes (1990) for a discussion of energy and environmental issues in Eastern Europe.

Appendix A

Table A1 Net exports of Sulphur in Europe

Thousands of Tonnes Sulphur per year

Net exports from		*Exports to* BG	CS	DK	FR	DD	DE	GR	HU	IT	NL	NO	PL	RO	ES	SE	TR	SU	GB	YU
Bulgaria	BG																			
Czechoslovakia	CS	-4																		
Denmark	DK	0	2																	
France	FR	0	8	0																
GDR	DD	-3	-44	-10	-27															
West Germany	DE	-1	19	-4	29	102														
Greec	GR	14	2	0	0	2	0													
Hungary	HU	-9	-14	0	-2	14	3	-3												
Italy	IT	-5	3	0	0	13	-5	-5	-1											
Netherlands	NL	0	1	-1	6	4	21	0	0	0										
Norway	NO	0	5	6	2	15	6	0	1	1	1									
Poland	PL	-4	50	1	0	278	24	-2	15	-4	2	-11								
Romania	RO	18	28	0	3	18	5	2	60	13	0	0	32							
Spain	ES	0	1	0	-54	4	-3	0	2	-8	-1	0	2	-1						
Sweden	SE	0	12	13	3	33	12	0	3	1	2	-1	32	0	0					
Turkey	TR	12	3	0	1	4	1	5	5	4	0	0	3	0	0	0				
Soviet Union*	SU	12	105	8	10	166	35	1	83	12	3	-9	319	-5	1	-16	-5			
Britain	GB	0	-2	-7	-29	0	-34	0	-1	-3	-17	-19	-12	-1	-4	-13	0	-15		
Yugoslavia	YU	4	13	0	5	17	7	-2	24	45	0	0	13	-38	3	-1	-5	-23	32	
N Africa**	NA	37	105	43	135	253	131	26	64	179	32	7	194	6	209	21	22	196	437	61

Source: *Acid Magazine* Sep 1989, from EMEP data, and Table A2.

Note: Sulphur dioxide figures will be about twice as large. Total wet plus dry deposition of sulphur for period 11.12.87 to 6.11.88

* European part of USSR within EMEP area of calculation.

** Deposition in North Africa within areas of calculation

Table A2 Origins of Sulphur Deposition in Europe

Thousands of Tonnes Sulphur per year

	Emitters																														
	AL	AT	BE	BG	CS	DK	FI	FR	DD	DE	GR	HU	IS	IE	IT	LU	NL	NO	PL	PT	RO	ES	SE	CH	TR	SU	GB	YU	NA	UI	Σ Receivers
AL	5	0	0	1	1	0	0	0	1	1	1	1	0	0	4	0	0	0	1	0	0	0	0	0	0	0	0	0	0	8	30 Albania
AT	0	18	2	0	27	0	0	12	24	18	0	12	0	0	30	0	1	0	15	0	0	2	0	0	0	0	4	8	0	27	207 Austria
BE	0	0	50	0	2	0	0	19	7	15	0	0	0	0	0	0	4	0	2	0	0	1	0	0	0	0	11	0	0	8	121 Belgium
BG	0	0	0	152	4	0	0	0	3	1	3	10	0	0	5	0	0	0	5	0	4	0	0	0	1	5	0	14	0	22	235 Bulgaria
CS	0	4	4	0	385	0	0	11	128	28	0	45	0	0	10	0	1	0	95	0	0	1	0	0	0	2	7	9	0	28	765 Czechoslovia
DK	0	0	1	0	2	31	0	1	12	7	0	0	0	0	0	0	1	0	4	0	0	0	0	0	0	1	7	0	0	9	83 Denmark
FI	0	0	0	0	4	1	48	0	8	3	0	2	0	0	0	0	0	0	12	0	0	0	3	0	0	57	3	1	0	62	210 Finland
FR	0	0	23	0	19	1	0	332	41	40	0	5	0	1	21	0	5	0	15	2	0	65	0	1	0	0	43	3	1	139	760 France
DD	0	0	7	0	84	2	0	14	725	61	0	2	0	0	2	0	4	0	32	0	0	1	0	0	0	1	15	1	0	24	979 GDR
DE	0	1	29	0	47	3	0	69	163	330	0	3	0	0	13	1	14	0	23	0	0	6	0	1	0	1	45	1	0	64	821 FRG
GR	0	0	0	17	2	0	0	0	2	0	45	3	0	0	5	0	0	0	2	0	0	0	0	0	1	2	0	6	0	28	119 Greece
HU	0	3	1	1	31	0	0	3	16	6	0	190	0	0	12	0	0	0	25	0	1	0	0	0	0	1	1	23	0	18	337 Hungary
IS	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	15	21 Iceland
IE	0	0	1	0	0	0	0	2	2	1	0	0	0	21	0	0	0	0	1	0	0	0	0	0	0	0	17	0	0	19	68 Ireland
IT	0	1	2	0	13	0	0	21	15	8	0	11	0	0	353	0	0	0	14	0	0	10	0	2	0	1	4	16	3	83	562 Italy
LU	0	0	0	0	0	0	0	2	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	7 Luxembourg
NL	0	0	15	0	2	0	0	11	8	35	0	0	0	0	0	0	32	0	2	0	0	1	0	0	0	0	20	0	0	10	139 Netherland
NO	0	0	1	0	5	6	2	2	15	6	0	1	0	0	1	0	1	13	11	0	0	0	5	0	0	10	19	0	0	91	194 Norway
PL	0	3	6	1	145	5	0	15	310	47	0	40	0	0	10	0	4	0	790	0	1	1	1	0	0	18	15	11	0	64	1492 Poland
PT	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	35	0	19	0	0	0	0	0	0	0	26	83 Portugal
RO	0	1	0	22	28	0	0	3	18	5	2	61	0	0	13	0	0	0	33	0	34	1	0	0	1	17	1	39	0	43	330 Romania
ES	0	0	1	0	2	0	0	11	5	3	0	2	0	0	2	0	0	0	3	15	0	523	0	0	0	0	6	1	1	97	674 Spain
SE	0	0	2	0	12	13	8	3	33	12	0	3	0	0	1	0	2	4	33	0	0	0	37	0	0	23	13	1	0	103	307 Sweden
CH	0	0	1	0	2	0	0	13	4	4	0	0	0	0	14	0	0	0	1	0	0	3	0	8	0	0	3	0	0	14	70 Switzerland
TR	0	0	0	13	3	0	0	1	4	1	6	5	0	0	4	0	0	0	3	0	1	0	0	0	61	12	0	5	1	86	210 Turkey
SU	0	3	5	17	107	9	22	10	167	36	3	84	0	0	13	0	3	1	337	0	12	1	7	0	7	2204	16	24	0	491	3584 USSR*
GB	0	0	5	0	5	0	0	14	15	11	0	0	0	6	1	0	3	0	3	0	0	2	0	0	0	1	571	0	0	60	702 Britain
YU	1	3	1	18	22	0	0	8	18	8	4	47	0	0	61	0	0	0	24	0	1	4	0	0	0	1	32	192	1	74	497 Yugoslavia
NA	3	5	39	37	105	43	18	136	253	131	26	64	0	17	182	0	32	7	194	13	6	210	21	1	23	196	437	62	11	810	3087 N Africa**
	3	5	6	4	5	7	3	8	7	4	4	3	0		2	3	8	3	5	2	5	5	5	4	1	5	-22	7	3	180	31 Rounding Errors
Σ	12	47	202	283	1064	121	101	721	2005	823	94	594	0	51	759	5	115	28	1685	67	65	856	79	17	95	2558	1271	424	21	2532	16695 Total
	25	75	244	570	1450	155	162	923	2500	1022	180	710	3	84	1252	6	141	50	2270	116	100	1581	116	31	177	5100	1840	588		0	21471 Production 1987

Source: *Acid Magazine* No. 8, Sep 1989, p8 Notes: As for Table A1

Table A3 Depositions per tonne emission

percent emissions deposited

Receivers	*Emitters* AL	AT	BE	BG	CS	DK	FI	FR	DD	DE	GR	HU	IS	IE	IT	LU	NL	NO	PL	PT	RO	ES	SE	CH	TR	SU	GB	YU
Albania	20	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Austria	0	24	1	0	2	0	0	1	1	2	0	2	0	0	2	0	1	0	1	0	0	0	0	0	0	0	0	1
Belgium	0	0	20	0	0	0	0	2	0	1	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	1	0
Bulgaria	0	0	0	27	0	0	0	0	0	0	2	1	0	0	0	0	0	0	0	0	4	0	0	0	1	0	0	2
Czechoslovakia	0	5	2	0	27	0	0	1	5	3	0	6	0	0	1	0	1	0	4	0	0	0	0	0	0	0	0	2
Denmark	0	0	0	0	0	20	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Finland	0	0	0	0	0	1	30	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	3	0	0	1	0	0
France	0	0	9	0	1	1	0	36	2	4	0	1	0	1	2	0	4	0	1	2	0	4	0	3	0	0	2	1
GDR	0	0	3	0	6	1	0	2	29	6	0	0	0	0	0	0	3	0	1	0	0	0	0	0	0	0	1	0
West Germany	0	1	12	0	3	2	0	7	7	32	0	0	0	0	1	17	10	0	1	0	0	0	0	3	0	0	2	0
Greec	0	0	0	3	0	0	0	0	0	0	25	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1
Hungary	0	4	0	0	2	0	0	0	1	1	0	27	0	0	1	0	0	0	1	0	1	0	0	0	0	0	0	4
Iceland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ireland	0	0	0	0	0	0	0	0	0	0	0	0	0	25	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Italy	0	1	1	0	1	0	0	2	1	1	0	2	0	0	28	0	0	0	1	0	0	1	0	6	0	0	0	3
Luxembourg	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	17	0	0	0	0	0	0	0	0	0	0	0	0
Netherlands	0	0	6	0	0	0	0	1	0	3	0	0	0	0	0	0	23	0	0	0	0	0	0	0	0	0	1	0
Norway	0	0	0	0	0	4	1	0	1	1	0	0	0	0	0	0	1	26	0	0	0	0	4	0	0	0	1	0
Poland	0	4	2	0	10	3	0	2	12	5	0	6	0	0	1	0	3	0	35	0	1	0	1	0	0	0	1	2
Portugal	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	30	0	1	0	0	0	0	0	0
Romania	0	1	0	4	2	0	0	0	1	0	1	9	0	0	1	0	0	0	1	0	34	0	0	0	1	0	0	7
Spain	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	13	0	33	0	0	0	0	0	0
Sweden	0	0	1	0	1	8	5	0	1	1	0	0	0	0	0	0	1	8	1	0	0	0	32	0	0	0	1	0
Switzerland	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	26	0	0	0	0
Turkey	0	0	0	2	0	0	0	0	0	0	3	1	0	0	0	0	0	0	0	0	1	0	0	0	34	0	0	1
Soviet Union*	0	4	2	3	7	6	14	1	7	4	2	12	0	0	1	0	2	2	15	0	12	0	6	0	4	43	1	4
Britain	0	0	2	0	0	0	0	2	1	1	0	0	0	7	0	0	2	0	0	0	0	0	0	0	0	0	31	0
Yugoslavia	4	4	0	3	2	0	0	1	1	1	2	7	0	0	5	0	0	0	1	0	1	0	0	0	0	0	2	33
N Africa	12	7	16	6	7	28	11	15	10	13	14	9	0	20	15	0	23	14	9	11	6	13	18	3	13	4	24	11
Errors	12	7	2	1	0	5	2	1	0	0	2	0	0	0	0	50	6	6	0	2	5	0	4	13	1	0	-1	1
Sum (%ident)	48	63	83	50	73	78	62	78	80	81	52	84	0	61	61	83	82	56	74	58	65	54	68	55	54	50	69	72
Product 1987	100	100	100	100	100	100	100	100	100	100	100	100		100	100	100	100	100	100	100	100	100	100	100	100	100	100	100

Source: Table A2

Appendix B

The choice between taxes and quotas under uncertainty

Suppose that the abatement schedule gives the marginal cost of abatement *p* in terms of the level of abatement, *x*, as

$$p = a + b\theta x + \varepsilon, \qquad E\theta = 1; \quad E\varepsilon = 0; \quad E\theta\varepsilon = 0,$$

where θ represents the uncertainty about the slope parameter *b* and ε represents uncertainty about the level of initial cleanup costs, assumed independent. If *w* is the willingness to pay for cleanup, and *X* is the current level of emissions, then suppose that

$$w = c + \eta + d\phi(X - x), \quad E\phi = 1; \quad E\eta = 0; \quad E\phi\eta = 0.$$

Again, ϕ represents uncertainty about the slope, η about the level when there is no pollution (ie $x = X$). Assume that all four random variables are independently distributed.

The net social surplus (consumer surplus plus profit) when the level of abatement is *x* is given by

$$S = \int_0^x (w - p)dx = S(x).$$

If the government must choose the level of quota, *x*, before knowing the parameters of the system, then *x* is chosen to max E*S*(*x*). The solution is

$$x = \frac{c+dX-a}{b+d}; \qquad ES = \frac{(c+dX-a)^2}{2(b+d)}.$$

If the government must choose a tax rate *t*, then $x = x(t)$, given by

$$x = \frac{t - a - \varepsilon}{b\theta}; \qquad \frac{dx}{dt} = \frac{1}{b\theta}.$$

The value of *t* is now chosen to max E*S*[*x*(*t*)], and the result is

$$ES = \frac{(c+dX-a)^2 (E1/\theta)^2}{2[bE(1/\theta)+dE(\theta^{-2})]} \approx \frac{(c+dX-a)^2 (1+2\sigma^2)}{2[b(1+\sigma^2)+d(1+3\sigma^2)]}$$

where $\sigma^2 = E(\theta-1)^2$ is the coefficient of variation squared of the abatement slope. Taxes will be superior to quotas if E$S(t)$ > ES, i.e if

$$b[(E\psi)^2 - E\psi^2] > d[E\psi^2 - (E\psi)^2], \quad \psi \equiv 1/\theta.$$

A sufficient condition for this to hold is that σ is small and $b > d$, ie if the slope of the abatement cost schedule is steeper than the slope of the damage schedule.

Duopoly model

Consider the following very simplified model of a duopoly electricity supply industry. Demand is linear, and the market clearing price is $p = a - Q$ when aggregate supply is Q. The technology is one of constant unit operating costs (set equal to zero without loss of generality) ignoring pollution abatement. The amount of pollution released when output is q and abatement equipment h is installed is q/h. The unit cost of abatement equipment is r, and the generator is allowed to release x units of pollution per year. Its profit is then $pq - rh = (p - r/x)q$.
Each generator behaves as a Cournot duopolist in the output market, given its entitlement to release pollutant, and chooses q to maximize profit, assuming the other generator's output is given. If the other generator can release y units of pollution, then

$$q = \frac{1}{3}\left(a - \left\{\frac{2}{x} - \frac{1}{y}\right\}r\right),$$

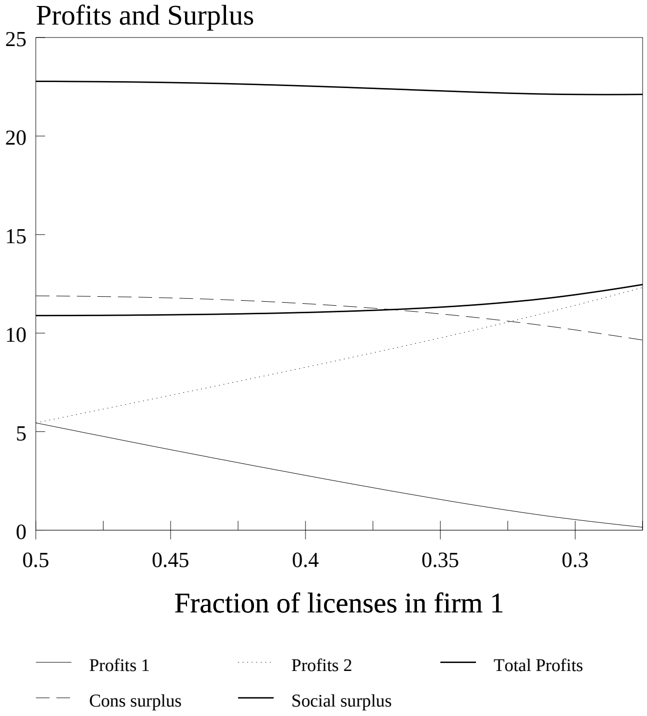
and the market clearing price is

$$p = \frac{1}{3}\left\{ a + r\left(\frac{1}{x} + \frac{1}{y}\right)\right\},$$

providing in both cases that the other generator produces a positive output. If not, then the generator is assumed to behave as an unconstrained monopolist. If the total pollution level is fixed at 1 unit, so that $y = 1 - x$, then individual profit, aggregate profit, consumer surplus, and net social surplus (the sum of consumer surplus and aggregate profit) are all functions of x. At the symmetric duopoly equilibrium $x = 0.5$, locally net social surplus is maximized, while net aggregate profits are minimized given the chosen parameter values ($a = 10$, $r = 1.5$). Fig.1 shows the resulting plot. There is a discontinuity as the duopoly collapses into a monopoly, and it is assumed that the two firms are not allowed effectively to combine to form a monopoly, and the graphs is terminated at the point at which monopoly would occur. It is an interesting observation that if the total licenses were to be allocated to the surviving firm, then the net social surplus may increase as the combined inefficiency of the duopoly and the misallocation of pollution licenses is replaced by the single inefficiency of a monopoly.

The fact that aggregate profits increases as the duopoly moves away from the social optimum towards an asymmetric equilibrium means that there are profitable exchanges of pollution licenses for cash between the two duopolists which leave each better off than in the symmetric equilibrium. It will typically pay one firm to buy all the licenses from the other firm in order to become an unconstrained monopolist. If this is not allowed by competition policy, then the firms will aim at the maximum allowable degree of asymmetry.

Fig. 1
Emissions license reeallocations



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