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## Efficiency and Distributional Aspects of Market Mechanisms in the Control of Pollution: An Empirical analysis

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Efficiency and Distributional Aspects of Market  
Mechanisms in the Control of Pollution: An Empirical  
analysis

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**Efficiency and distributional aspects of market mechanisms**  
**in the control of pollution: an empirical analysis**

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**Abstract** The cost-saving advantages of market mechanisms such as tradeable permits and emission taxes have been promoted by economists since the early 1970s. The research reported in this paper simulates their application to the Forth Estuary in Scotland, comparing such policies with regulation of output levels for emissions. Potential resource cost savings are identified under two alternative targets. However, there are serious obstacles to the attainment of these savings. We also consider the distributional aspects of the policy alternatives.

**Keywords:** water pollution control; tradeable permits; emission taxes;  
Biological Oxygen Demand

## 1. Introduction

Since the seminal papers of Baumol and Oates (1971) and Montgomery (1972), economists have stressed the comparative advantages of 'market mechanisms' for the control of pollution, over uniform standard setting approaches<sup>1</sup>. Market mechanisms include tradeable (pollution) permits, emission taxes and subsidies. Uniform standard approaches are typically of two types: specification of maximum emission levels (performance standards), and requirements to install a particular type of plant (design standards). Briefly stated, market mechanisms out-perform uniform standards since they allow firms greater flexibility in meeting overall pollution reduction targets, and are capable under certain conditions, of achieving a least-cost outcome. This outcome is characterised by either an equivalence of marginal control or abatement costs (MAC) across all discharges for pollutants which are uniformly mixed; or an equivalence of the marginal costs of damage reduction across sources for non-uniformly mixed pollutants.

These theoretical findings were reinforced by simulation studies comparing market mechanisms with command-and-control approaches. Most work was done on air pollution control in the U.S. (Krupnick, Oates and van der Verg, 1983; McGartland and Oates, 1985; Krupnick, 1986). A combination of the cost-savings forecast by these studies, and conflicts between clean air goals and economic growth, led to the adoption of a limited system of tradeable pollution permits in

<sup>1</sup> Sometimes referred to as 'command and control' approaches.

the U.S.A. in the mid- and late 1970s. This system has resulted in significant cost-savings to discharges (see, for a discussion, Liroff (1986) and Tietenberg (1987)).

Less simulation work was done for water pollution control, the focus of this paper. This was due in part to the greater problems envisaged in applying tradeable permits to water, due principally to the smaller number of emission points relative to the situation normally encountered for air. Rowley *et al.* (1979) reported large cost savings under a system of emission taxes for the attainment of water quality targets in the Tees Estuary, England, whilst O'Neil *et al* (1983) found large resource cost savings when tradeable permits were used in place of command-and-control for the Fox River in Wisconsin, U.S.A. This finding encouraged the state government to introduce a permit trading scheme for the Fox River, but cost-savings failed to materialise with only one trade being completed. Reasons for this apparent policy failure are discussed by Hahn (1989): we draw some parallels with the Forth Estuary later on in this paper.

Market mechanisms for water pollution have not been taken up on anything like the scale that the strength of economic arguments in their favour might predict. This point has been made by Opschoor and Vos (1989) and ERL (1990)<sup>2</sup> Reasons why this may be so are discussed at the general level by Hahn (1989) and Hanley, Moffatt and Hallett (1990). This paper explores why market mechanisms may

<sup>2</sup> The closest parallels are the emission charges schemes in the Netherlands and Germany: see Opschoor and Vos (1989) for details.

be less attractive than might be supposed for a particular case-study area: the Forth Estuary in central Scotland.

The rest of this paper is laid out as follows. Section Two gives a brief account of water quality and current management practices in the case-study area. Section Three describes the modelling approach used, and how data was assembled. Some results from the model are presented in Section Four, concentrating on a tradeable permit system for organic pollutants. Section Five offers some conclusions regarding this study, and regarding the take-up of market mechanisms in the UK.

## 2. Water Quality in the Forth Estuary

The Forth Estuary is located in central Scotland. Its tidal limit is 2 km upstream of Stirling Bridge, whilst from 56km seawards it becomes the Firth of Forth. The estuary is a multi-use resource. It provides inputs (cooling water, process water and waste disposal) to a large number of industries located along its shoreline; acts as a waste sink for discharges from many sewage treatment works (STWs); is a migration route for salmon and a location for commercial and sport fishing; and is a habitat for both migratory and resident birds (Bryant, 1987).

Water quality in the estuary is monitored and managed by the Forth River Purification Board (FRPB). Monitoring is undertaken with regard to both the chemical and ecological condition of the estuary. Management is undertaken using a three stage process. First, Environmental Quality Objectives (EQOs) are set. These are qualitative statements describing the desired condition of the estuary regarding the uses to which it is put. This desired

condition is to be achieved by setting Environmental Quality Standards (EQS). These are quantitative upper limits on ambient concentrations of polluting substances. These EQS levels are in turn achieved by imposed 'consents' on discharge conditions, for all point-source discharges (under the River's (Prevention of Pollution) (Scotland) Acts). Consents set both quantitative and qualitative limits on discharges, and are legally enforceable. EQOs may vary regionally according to existing water quality and intended uses of the water body. They differ fundamentally from the standard emission limits approach widely used in the rest of the EC.

Water quality may thus be judged by comparing observed levels for a range of determinants against target, EQS levels. We show this below in Table 1 for a selection of determinants, including two "Red List" substances (mercury and cadmium). Red List substances are subject to compulsory monitoring under EC Directive 76/464, and are the object of a national target reduction in discharges to the North Sea by 50% by 1995 (HMSO, 1990). As may be seen, measured levels for all determinants shown are within EQS levels. However, the figure for Dissolved Oxygen (DO), with a mean of 8.03 and a minimum value of 6.51 in 1989 is misleading, for the reason discussed next.

Discharges of many inputs exert a Biological Oxygen Demand on the receiving water, using up dissolved oxygen as they break down. The impact of a unit (kg/day) of such an effluent (which might originate, for example from a paper mill or distillery) depends crucially on two factors: the volume of receiving water, and temperature (TM).

**Table One: EQS versus levels for water quality determinands  
Forth Estuary, 1989 (annual values)**

DETERMINAND	UNIT	EQS	RANGE	MEAN
pH	-	6.0-8.5	7.19-7.93	7.5
DO	mg/1	4.5	6.51-10.23	8.3
Cadmium	µg/1	5.0	0.025-0.052	0.036
Mercury	ng/1	500	0.9-19.6	8.0
Copper	µg/1	5.00	0.51-1.15	0.87
Arsenic	µg/1	25.0	2.1-4.6	3.4

SOURCE: FRPB, various.

Volume determines dilution, and in the estuary is crucially dependent on flow (F), which depends in turn on surface drainage; and on tidal influx TD. The impact of a unit of discharge on dissolved oxygen levels at some point k (=1...n) in the estuary is given, in simplified form, by

$$a_{jk} = V(TM, F, TD) \tag{1}$$

for a source j (j=1...n). The function V(•) is likely to be stochastic and non-linear. We approximate V(•) below through a water quality model (see section 3).

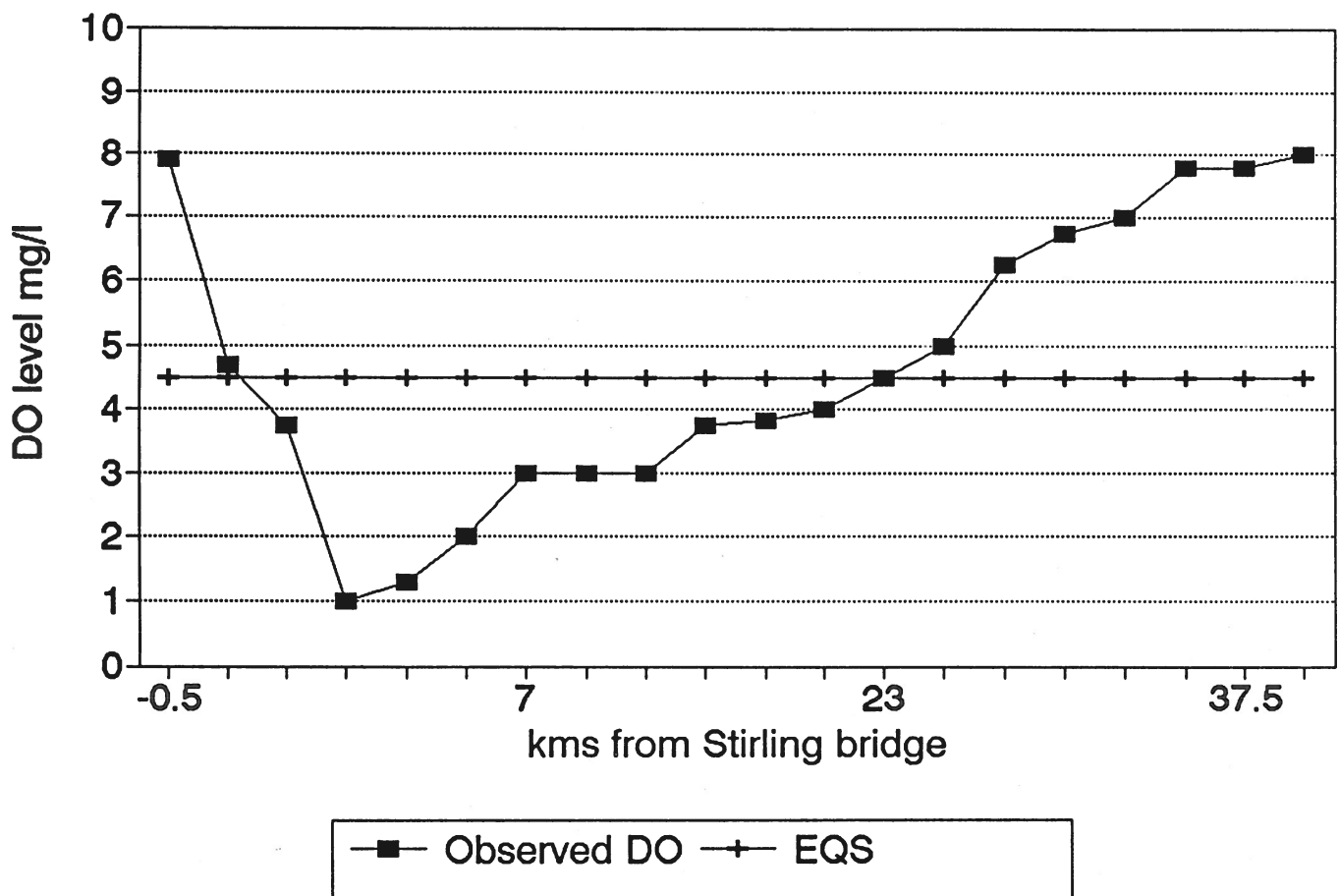
The implication of (1) is that DO levels, for a given total impact of pollutants exerting a BOD, will be lower during periods of low flow and high temperatures. This is what we find for the Forth Estuary. During the early summer, a sag develops in the level of DO, putting it below the EQS level for a significant portion of the estuary. This is shown in Figure 1.



FIGURE 1

# DO LEVELS IN ESTUARY

July 1989



## FIGURE 1

The rest of this paper restricts consideration to this Dissolved Oxygen problem, and the regulation of BOD inputs. Significant point sources of BOD were identified, showing that 81.2% of BOD "loading" (the product of dilution and input flow) in the estuary originates from seven industrial sources. These comprise an oil refinery, a fine chemicals and paint works, a yeast factory, a distillery, and two paper mills. Labelling these sources A to G, current discharge levels are shown in Table 2. Sewage Treatment Works (STWs) account for 15.5% of total loading, with the remaining 3.3% coming in from the eight rivers draining into the estuary. These rivers carry back-ground loading from non-point sources such as farm and forestry land; and industrial and STW discharges up-stream of their confluence with the estuary. Incoming river water quality is, on average, high.

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**Table Two: Details on total BOD loading  
(point sources only)**

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SOURCE	BOD loading kg/day	As % of total
1. Industries:		
A	25,000	27.7
B	1,990	2.2
C	10,000	11.1
D	3,000	3.3
E	20,000	22.1
F	2,000	2.2
G	11,374	12.6
2. STWs	13,964	15.5
3. Incoming Rivers	2,960	3.3
Total	90,288	100.0

Note: calculated from consented maximum daily discharge levels, 1990.

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Modelling efforts were centred on the seven industrial sources, and the STWs. Thirteen STWs discharge to the estuary, operated by three Regional Councils: Central, Fife and Lothian.

### **3. Methodology and model construction**

The purpose of the modelling exercise was to simulate alternative approaches for the control of BOD discharges to the estuary. The policy alternatives were:

1. The existing consent system - which we term "flexible regulation";
2. A system of uniform emission reduction requirements. This is closest to the uniform emission standard approach favoured by most EC countries except the UK, which is likely to be forced on the UK to an increasing degree (Hallett *et al.* 1991);
3. A system of tradeable emission permits;
4. A charge on BOD emissions.

In line with previous work, our modelling approach was to combine a mathematical programming model of dischargers' behaviour, with a hydrological model of water quality. Extensive data collection was required in each case. We now describe (i) the economic model, and (ii) the water quality model.

#### **3.1 The economic model**

The economic model assumes that dischargers' behaviour can be captured by a linear programming (LP) approach, where pollution abatement costs are minimised subject to a series of constraints. Most importantly, firms are assumed to minimise costs. Given the lumpy nature of investments in pollution control technology, a step-wise LP framework is used. For each firm, there are as many pollution reduction activities in the model as there are steps in

the firm's abatement cost function. The economic model in its basic version may be represented as follows:

$$\text{Min } Z = \sum_{j=1}^n \sum_{i=1}^p C_{ji}(X_{ij}) \quad (2)$$

$$\text{s. t. } X_{ij} \geq 0 \quad \forall i, j \quad (3)$$

$$\text{and } \sum_{i=1}^m \sum_{j=1}^n a_{ijk} X_{ij} \geq \Delta D_k \quad (4)$$

$$\text{and } S_{ij}(X_{ij}) = 0 \quad \forall i, j \quad (5)$$

Equation (2) minimises total abatement costs  $C$ , which depend for each discharger  $j$  on its level of emission reduction  $X$  and the step of the abatement cost function,  $i$ . Firms may cut emissions by installing new treatment plant, increasing the performance of their treatment plant, reducing output levels or changing production inputs. Dischargers are assumed to have identified all these options, and to have selected the least cost alternative for each level of emission reduction (firms were asked how they would achieve discharge cuts in the survey to collect the data for this model: see below). Equation (3) prohibits any increase in emissions by any discharger. Equation (4) requires that the total of emission reductions, weighted by the transfer coefficients  $a_{ijk}$ , are sufficient to meet the reduction in BOD loading at any stretch  $k$  necessary to just attain the water quality objective. Finally, equation (5) represents a sub-matrix of 'tie lines', which ensure that the maximum discharge reduction potential associated with any step of any cost function is not breached by the minimisation process.

Data for the model was collected from 6 of the 7 industrial sources identified to being significant, and for 6 of the 13 STWs. For the industrial sources, one firm refused to co-operate (firm G in table 2). All other sources provided the research team with data on (1) their production and pollution treatment processes; and (2) the costs associated with reducing emission levels by increasing target amounts over two time periods - 12 months and 5 years. We thus generated a 'short-run' and 'long-run' abatement cost data set. For many sources, some reduction targets were impossible to achieve within 12 months by any means other than output reduction, or significant changes in production methods. The results reported below relate only to the 'long run' data set. In each case, firms were asked to describe exactly how they would meet the targets. Capital costs were annualised at a 6% real rate over 4 years, then added to variable costs to give annual abatement costs per kg. BOD reduced for each step on the abatement cost function.

The data collected showed that the costs of emission reduction currently vary significantly across dischargers at the margin. The marginal cost of discharge reduction for the lowest-cost source was more than ten times less than the marginal cost of the highest-cost source. This implies that the current allocation of control responsibility is certainly not efficient if one ignores variations in impact coefficients. What is more, the source with lowest marginal cost is also the source with the largest impact on DO levels at the sag point in the estuary. For some sources, there are economies of scale in discharge treatment, with average abatement costs falling with the level of pollution treatment. This feature

would cause modelling problems if a step-wise approach had not been used - see Rowley *et al.* (1979).

For STWs, data on abatement costs was only obtainable for a subset of all STWs where improvements have been considered. Upgrading costs turn out to be highly site-specific. Almost no potential exists for improving the performance of any works on the estuary without capital upgrading. Most frequently, this involves going from primary treatment to secondary-stage treatment. Fortunately, all the major 'problem' works were sites for which data was available, with one exception. However, given the relative unimportance of STWs as a whole to BOD loading in the estuary (Table 2), this was not felt to be a serious weakness in the study. Data on capital and operating costs were collected from Regional Council staff. Again, capital costs were annualised and combined with running costs to give an annual cost per kg BOD reduction per day.

The output of the model is (i) the optimum (ie. cost-minimising solution); (ii) sensitivity analysis on abatement costs; and (iii) calculation of shadow prices for all constraints that are binding. The model was altered slightly to simulate a permit market, by introducing activities for the buying and selling of permits; by adding tie-lines to oblige firms to hold enough permits to validate their discharges; and by constraining the total supply of permits (see below). For the emissions charge runs, activities are introduced to force a tax payment for each unit (kg/day BOD) discharged by any discharger; and by an 'adding up' row which calculates total tax revenue.

The model is solved in every case by the simplex algorithm using the LINDO LP package (Schrage, 1989), and like the water quality model, runs on any IBM-compatible PC.

### 3.2 The water quality model

Estuaries are complex environments to model. This is because water quality is dependent on a large number of parameters, many stochastic. Relevant parameters include tidal influx, salinity, river inflow, temperature, pollution inputs and re-suspension of sediments. The Forth has been defined as a partially-mixed estuary, in terms of the inter-action of fresh and saline waters (McLusky, 1989). Estuaries are dynamic systems – tidal inflows occur twice daily whilst rainfall varies stochastically over time.

We model water quality in the estuary using a one dimensional, dynamic, point-specific approach. The Forth Estuary Dynamic Simulation model (FEDS) predicts DO levels for 20 reaches of the estuary over a 24-hour time period. Discharges by point sources can be varied individually in order to establish transfer coefficients, the  $a_{ij}$  terms in equation (4). The model was calibrated on 1988 data. Diffusion along the estuary axis (56 km long) depends on water velocity (which changes during the tidal cycle) and salinity. Reaeration increases DO levels, whilst biodegradation of pollutants reduces it. The model, which consists of over 300 active equations, is written in DYNAMO, and runs on any 286 IBM compatible PC. A full description of the model is given in Moffatt, Hallett and Hanley (1990). Data for the model was derived from the TCHEM (Tidal waters CHEMical survey) database maintained by the FRPB.



## **4. Results**

In this section, we present some of the more interesting results obtained from running the two models described above. We concentrate on comparisons between a tradeable permit system and command-and-control: both of the uniform emission limits variety, and the current flexible regulation approach employed by the FRPB. Two sorts of environmental objective are considered. Firstly, we look at a series of reductions in total BOD loading to the estuary. This type of objective is consistent with the North Sea conference agreements, which have so far been reached for Red List substances (section 2) and for nutrients such as nitrate and phosphate. The second type of objective relates to local, ambient water quality levels, within the spirit of the UK's EQO/EQS approach. Here, the target is taken to be the removal of the DO sag at low flow conditions, so that the EQS of 4.5 mg/l. is not breached. In all cases, we consider results from the 'long-run' data set only on the grounds that this is more complete, and more realistic in that it is unlikely that firms would have to meet large improvements in performance in such a short time period.

### **4.1 Reductions in total BOD loading**

#### **4.1.1 Least cost calculations**

Three target reductions in the total loading of BOD discharged to the estuary were considered: 10%, 25% and 50%, all relative to the current total discharge of those firms and STWs included in the model - six firms plus the STWs.<sup>3</sup> Given that we are interested in a

<sup>3</sup> In the short-run version of the model, there are six firms and only two

reduction in total loading, then all transfer coefficients (ie. for the nine sources over 20 stretches) are set equal to unity. Firm "B" is actually treated as 2 distinct sources in the model, as it has 2 distinct plants discharging at 2 different locations.

which is, however, not crucial due to the small loading contribution from this source. Table 3 shows the implications in resource cost terms for achieving these cuts by uniform discharge reductions relative to the least-cost solution.

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**Table Three: Resource costs of three target reductions**

TARGET REDUCTION (%)	ACTUAL REDUCTION (Kg/BOD/day)	in total BOD loading	
		RESOURCE COST UNDER UNIFORM REGULATION (£/YEAR)	RESOURCE COST UNDER LEAST-COST SOLUTION (£/YEAR)
10 <sup>a</sup>	6,499	1,340,560	275,455
25 <sup>a</sup>	15,497	7,382,976	692,365
50 <sup>b</sup>	37.379	16,128,660	3,075,989

Notes: a Minimum improvement from upgrading of the three STWs on which cost data is available is > 50% reduction in BOD.  
 b Least-cost solution for the 50% cut includes minimum improvements At all 3 STWs.

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As may be seen, the least-cost solution in all cases is far cheaper than uniform cut backs, the ratio of uniform regulation to least-cost expenditure ranging from 4.9 (10% cut) to 10.7 (25% cut). This saving is explained by the fact that the least-cost solution targets big cut-backs on low cost-of-control firms, with lower or zero cut-backs falling on high cost-of-control firms. Uniform

STWs.

regulation forces all firms, on the other hand, to cut by the same amount.

The findings above are unsurprising given the large variation in MAC already mentioned. The interesting question, however, is whether either a tradeable permit system or a pollution tax could attain the least-cost solution in the way that theory suggests they should. It is also relevant to ask what the distributional consequences of the two policy alternatives are, since this will influence policy acceptability.

#### 4.1.2 A permit system

A tradeable permit system aimed at achieving a reduction in total loading can be of simple design, since in this case the spatial location of dischargers is unimportant. Permits can thus be specified in terms of emissions. Permits would be issued by the FRPB, by one of two means: by an auction of the fixed supply  $\bar{S}$ , where  $\bar{S} = E_C - E_T$  (for  $E_C$  = current total emissions and  $E_T$  = target reduction in total emissions); or by freely giving away (grandfathering) permits again up to a total of  $\bar{S}$ . Initial allocations under grandfathering would be pro rata with current consented discharges, so that any firm  $j$  would initially receive  $\left[ \bar{S} \left( E_C / e_C^j \right) \right]$ , where  $e_C^j$  is the current emission of the  $j$ th firm. Firms would then be allowed to buy and sell permits.

If all gains from trade are apparent to all firms, and if no firm is a large enough buyer or seller to influence the market price of permits, then the model will converge on the least cost solution. Similarly, if all firms are perfectly informed as to these MAC schedules (which form their buy and offer curves for permits), then

the model converges on the least-cost solution if the price that emerges is the competitive price. This will be equal to the marginal value product (shadow price) of the permit supply constraint in the LP model. This indicates that, for example, for a 50% target reduction, the competitive price is £357 per kg BOD/day for an annual permit. At this price, the permit market clears.

However, simulations of buying and selling under the grandfathering system revealed that two sources would dominate the market; one as a large buyer (firm E) and one as a large seller (firm A). For the long run, 25% target reduction scenario, A will wish to sell 76% of total permits offered, whilst E will wish to buy 80% of total permits offered. Firm A can reduce its financial burden (the sum of abatement costs and net permit receipts) by pushing up the permit price, by withholding permits. Firm E will wish to minimise its financial burden too, so will presumably try to bid down the offer price from A.

Unequal market power is also problematic, though perhaps less so, under an auction system. Here, firms will again seek to minimise their financial burdens by manipulating the price. However, any price setting firm is constrained in this respect by the steepness of its MAC function beyond the cost-effective allocation (since it can only push down the price it must pay by buying fewer permits, which it can only do if it controls more emissions). However, one firm (A again) will not be affected this way in our case, since it enjoys economies of scale in abatement. Firm A, as a major discharger, could reduce the price it must pay for permits in an auction by such action. However, using a second-price auction might

restrict such behaviour, whilst, as Tietenberg (1984) has pointed out, previous work in this area suggests that whilst uncompetitive behaviour can result in significant changes in financial burden, effects on resource costs (ie. abatement costs) seem less substantial: for example, Maloney and Yardle (1984) found that in their worst-case scenario (90% monopoly power), the permit market still achieved cost savings of 66% over uniform standards.

There are, however, three other potential barriers to a permit market hitting the least-cost solution. The first of these is hoarding. Firms may refuse to sell, even if they could make reductions in their financial burden, because (i) they feel that by this action they will increase their allocation of permits in the next round; or (ii) because the potential buyer is in product-market competition with them. Few trades under the EPA's Bubble policy in the U.S. have been between different firms. Second, if trading is sequential rather than instantaneous, then Atkinson and Tietenberg (1991) have shown that grandfathered permit systems may result in far lower cost savings than expected. Finally, if pollution control costs are low relative to total costs for a firm, then if transactions costs are greater than zero, busy managers may choose not to get involved in permit trading. This is more likely to be the case where more detailed trading rules are in place than in the case in this section (but see section 4.2), and has been suggested by David and Joeres (1983) as one reason for the 'failure' of the Fox River scheme.

Distributional aspects of permit systems depend on whether a grandfathering or auction approach is used; and on the extent of

price-setting behaviour. Under an auction system, transfer payments leave the industry en bloc as expenditure on permits. If the price that permits sell for is determined by the maximum willingness to pay of the marginal firm (which will be the case if the price is a market clearing price in the absence of price-setting behaviour), then a total transfer payment of £2,232,624/year leaves the industry if the 25% cut scenario is considered. This implies the following expenditures across firms:

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**Table 4: Financial Burdens Under Auctioned Permits**

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FIRM	PERMIT PURCHASES	PERMIT EXPENDITURE (£)	ADDITIONAL CONTROL COSTS (£)	TOTAL FINANCIAL (£)	FINANCIAL BURDEN PER UNIT OF POST POLICY EMISSION (£/Kg/day)
A	14,023	673,104	494,865	1,167,969	83.29
B	1,990	95,520	0	95,520	48.00
C	6,000	288,000	192,000	480,000	80.00
D	3,000	144,000	0	144,000	48.00
E	20,000	960,000	0	960,000	48.00
F	1,500	<u>72,000</u>	<u>5,500</u>	<u>77,500</u>	51.67
TOTALS:		£2,232,624	£692,365	£2,924,989	

**Notes:** total emissions under policy = 46,513 kg/day  
cut in total emission relative to baseline = 15,497 kg/day

---

Permit expenditure figures are calculated by multiplying, for each source, the cost-minimizing number of permits purchased by the market-clearing permit price. This optimal adjustment for each firm implies in some cases a combination of emission reductions relative to the baseline and permit purchases. Additional control costs relate to the former activity, and are computed by the model.

From the above table, it can be seen that the permit scheme might be judged 'fair' on the grounds that it makes payments per unit of original discharge roughly equal across firms, since those making large cuts in emissions benefit from reduced spending on permits in relative terms. However, this may not be 'fair' if one reason why

firms have low control costs is because they have not been tightly controlled in the past. There is certainly some evidence of this in the Forth.<sup>4</sup>

Finally in this section, it should be noted that all permit market simulations exclude STWs. This is because it is not clear that Regional Councils have the same objective functions as industry in terms of how their behaviour in a permit market is modelled. We are unsure how, or if, Regional Councils would behave if allowed to buy and sell permits. For firms, however, the cost-minimisation assumption is a reasonable one.

#### 4.1.3 A charge on BOD

This section will deal very briefly with findings on a pollution tax. This brevity is because it seems less attractive as a policy instrument for a number of reasons (relative to permits). Firstly, unless firms actually cost minimise, the least-cost tax rate will not achieve the desired level of control. This could have very damaging and long-term effects on water quality in the estuary. With permits, so long as firms do not cheat, then the quantity of emissions cannot exceed the fixed permit supply. Secondly, calculation of the least-cost rate requires detailed information on MAC schedules, which control authorities may not be able to obtain. Thirdly, the tax rate required to hit a given target changes whenever the industry MAC curve shifts.

We find two further problems with taxes. Firstly, they are incapable of achieving given target reductions in emissions if there

<sup>4</sup> We are indebted to John Pezzey for pointing this out to us.



are steps in the aggregate MAC function. This 'existence problem' was first noted theoretically by Walker and Storey (1977), and empirically by Rowley *et al.* (1979). It means that a tax rate of  $t_1$  will achieve some level of emissions reduction  $X_1$ , which is less than the target reduction  $X_T$ . However, raising the tax by one unit to  $t_2$  increases emission reduction to  $X_2$ , where  $X_2 > X_T$ . We find this to be the case in all scenarios studied. Secondly, a tax scheme extracts large transfer payments from dischargers, even larger than an auctioned permit systems (due to the existence problem). This further reduces the attractiveness of the tax policy, if a conventional interpretation of the Polluter Pays Principle is taken (Pezzey, 1988).

#### 4.2 Removing the Dissolved Oxygen sag

In this section, we report results from a series of simulations designed with a different type of target to those of the preceding section. Here, the sole objective is to improve DO levels at the sag point (which stretched for some 22 kilometres in July 1989 and takes in reaches 2→8 of the FEDS model). Here, given that we are concerned with achieving an improvement in ambient water quality levels (to  $> 4.5$  mg/l DO in reaches 2→8 under low flow conditions), it is no longer appropriate to work with transfer coefficients set to unity. Output from the FEDS model shows that only discharges in reaches 1 to 10 have impacts on DO levels over stretches 2→8. This means that reductions in BOD levels by, for example, dischargers in Grangemouth, have no significant effect on the sag problem, due to axial distance and the tendency of discharges to move sea-wards. Given this, all discharges outside stretches 1-10 were allocated a

transfer coefficient of zero. This reduces the effective number of activities in the economic model to discharge reductions from three firms (A, B and F) and two STWs (Stirling and Alloa).

The comparison reported here is between the least-cost solution and flexible regulation, embodied as the FRPB's current plan to remove the DO sag. This involves targetting large reductions at one industrial source and one STW. This programme was costed, and compared to the least-cost solution obtained from the economic model:

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**Table 5: Policy Alternatives to remove the DO sag**

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POLICY	ANNUAL RESOURCE COST (£)	REDUCTION IN BOD (KGS/DAY)
Flexible Regulation	1,184,000	17,700
Least Cost	996,885	17,700

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As may be seen, the flexible regulation approach gets very close to the least cost solution (the cost saving is only 15.8%). This is because in this instance, the control agency has targetted large reductions on dischargers with relatively low MACs and relatively high impacts on DO levels, due to their position. In fact, given the problems noted earlier in getting a tradeable permit system to achieve the least-cost outcome, the difference may be judged insignificant. However, this neglects two points. First, that the control agency's plan in this case may have fortuitously got very close to the least-cost outcome, which situation would only re-occur by chance. We have no further comments on this except that (i) the FRPB is very aware of relative impacts on DO due to location, but

that (ii) FRPB officials had only the intuitive impression that these dischargers could meet big target reductions at relatively low cost: dischargers have no incentive to honestly reveal their true abatement costs to a regulator under flexible regulation, since their financial burden is unambiguously reduced by over-stating these costs. Secondly, flexible regulation does not provide the continual incentive for firms to improve the efficiency of pollution treatment through time, which market mechanisms provide.<sup>5</sup>

With regard to the equity properties of the two schemes considered for the DO sag case, under flexible regulation the only firms to suffer losses are those on whom discharge reductions are targetted. The other three sources are asked to bear no additional control costs. Under an auctioned permit scheme, financial burdens are incurred by all firms, since dischargers buy permits and, in some cases, cut emission levels. Flexible regulation might therefore be argued to be less equitable. However, if the reason discharge reductions are targetted on the two sources is because the low level of their marginal abatement costs relative to other sources is due to a lower level of control being applied in the past, then as we argued in the section on total loading reduction, our conclusion is altered for non-static definitions of "equity".

<sup>5</sup> Although there will be some incentive if improved residuals handling has incidental benefits, such as the re-use of raw materials or energy conservation.

## 5. Conclusions

Our research provides further evidence that for large target reductions in water pollution levels, the least cost solution represents a very large saving over uniform regulation. Yet there are significant problems relating to the ability of either tradeable permits or emission taxes (particularly) to achieve these cost-savings. The relatively small number of traders in a permit market sets up the possibility of price-setting behaviour, with a consequent partial erosion of efficiency gains. This result was obtained in what was thought to be the most favourable setting, a priori, for a tradeable permit system for water pollution in Scotland. All other case study areas studied suffered from much greater conceptual difficulties than the Forth estuary.

These factors may explain partially why tradeable permit systems have been adopted much more at the policy level for air pollution control than for water, since typically the number of sources is far greater, whilst modelling diffusion in an air shed is no more difficult than is the case for estuaries. However, permits may, under certain circumstances, possess a second attraction, namely that they can produce outcomes which appear equitable. This is in addition to their incentive-for-innovation property, which we have not explicitly considered. With regard to near-term future policy developments in the UK, however, it appears that a tradeable permit system is more likely for certain forms of air pollution control (such as sulphur dioxide emissions from large power stations) than for water.

With regard to the model presented here, work is continuing on improving the accuracy of FEDS predictions, so that a range of target DO levels can be considered. Also, the economic model is to be extended to explicitly model price-setting behaviour amongst dischargers.

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