Instrument Choice in Environmental Policy:

A Comparative Study of Pollution Taxes and Tradable Pollution Rights in British Columbia, Germany, and the United States

by

ROBERT SEELIGER

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF

THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF ARTS

in

THE FACULTY OF GRADUATE STUDIES

DEPARTMENT OF POLITICAL SCIENCE

We accept this thesis as conforming

to the required standard

THE UNIVERSITY OF BRITISH COLUMBIA

December 1992

Abstract

Current theories of instrument choice in environmental policy suggest that policy makers choose policy instruments based on the distribution of costs and benefits in society. It is postulated that policy makers will select those policy instruments which confer concentrated benefits on interest groups that will return favours. Others suggest that instrument choice is an outcome of interest groups struggles. However, these approaches do not explain the variation of instrument choices across countries.

This thesis is a comparative investigation of instrument choice situations, and actual instrument choices. Combining propositions of public choice theory, blame avoidance theory, and policy style analysis it studies the factors that determine the choice between pollution taxes and tradable pollution rights in three countries (Canada, Germany, and the United States).

It is found that institutional structures, particularly the distribution of policy making authority among the various jurisdictions of a state, is the single most important variable. Federal states with centralized authority, e.g. the U.S., have a greater choice among policy instruments, unless other factors, such as ideology, exclude instrument options from the choice set. Federal states in which authority is functionally shared between federal and state governments based on historical precedent, such as Germany, are more likely to adopt pollution taxes. The implementation of tradable pollution rights would require substantial legislative changes at all levels of government. Although eswtablished patterns of interjurisdictional cooperation may facilitate such changes, it would still be a major undertaking that may not be justified by the potential macroeconomic gains associated with emission trading. In federal states with authority overlapping between federal government and provinces, such as Canada, the adoption of either pollution taxes or tradable pollution rights is extremely difficult. Unilateral actions could, in the case of pollution taxes, damage a province's competitive position vis-a-vis the other provinces, and thus the adoption is unlikely. Emission trading, however, is subject to the constraints of international agreements by the federal government, and consent by all other jurisdictions on the fixing of an emission cap. Provincial governments guard their jurisdictional authority and are more likely to choose policy instruments that resemble pollution taxes but will not disadvantage the provincial

economy. Such taxation schemes will likely be means to raise revenue rather than serving to achieve specific environmental quality objectives.

A preliminary survey of instrument choices in other countries suggests that this pattern is not restricted to federal states but possibly is indicative and representative of jurisdictional relationships within states in general.

Aside from the distribution of policy making authority among jurisdictions, other factors influencing instrument choice are: agenda setting, ideology and political culture, and the structuring of the decision making process. In none of the case studies did the distribution of costs and benefits among stakeholder groups significantly affect policy instrument choice.

Table of Contents

Abstract	ii
Table of Contents	iv
List of Tables	v
List of Figures	v
Glossary	vi
Acknowledgements	vii
Introduction	1
I. Theory	
1. What are policy instruments?	2
2. 'Economic instruments': Pollution taxes and tradable pollution rights	3
2.1. Pollution taxation	4
2.2. Emission trading	5
2.3. Pollution taxes vs. tradable pollution rights: A preliminary appraisal	7
3. Instrument choice in the policy process	8
4. Who chooses, and why?	9
5. Instrument choice in environmental policy	14
II. Case studies	18
1. Emission trading in the United States	
1.1. Framework and policy style of U.S. environmental regulation	19
1.2. EPA's 1976 offset rule	20
1.2.1. Political context	20
1.2.2. Policy content	22
1.2.3. Patterns of conflict and conflict resolution	23
1.2.4. Policy effectiveness	25
1.2.5. Case analysis	26
1.3. SO_2 trading under the 1990 Clean Air Act Amendments	27
1.3.1. Political context	27
1.3.2. Policy content	29
1.3.3. Patterns of conflict and conflict resolution	30
1.3.4. Policy effectiveness	33
1.3.5. Case analysis	33
1.4. The discourse over pollution taxes and emission trading in the U.S.	35

2. Germany's 1976 Effluent Charge Law 37
2.1. Framework and policy style of German environmental regulation 38
2.2. Political context 39
2.3. Policy content 40
2.4. Patterns of conflict and conflict resolution 42
2.5. Policy effectiveness 44
2.6. Case analysis 45
2.7. The German discourse over pollution taxation and emission trading 46

3. British Columbia's 1990 Waste Discharge Fee Regulations 49
3.1. Framework and policy style of B.C. environmental regulation 49
3.2. Political context 50
3.3. Policy content 51
3.4. Patterns of conflict and conflict resolution 52
3.5. Policy effectiveness 54
3.6. Case analysis 55
3.7. The Canadian discourse over pollution taxes and emission trading 56

III. Complements to a theory of instrument choice 58
1. How much do redistributive instrument properties matter? 59
2. Do political culture and values matter? 60
3. How do policy styles affect agenda setting affect instrument choice? 61
4. How do institutions influence instrument choice? 63

IV. National styles of instrument choice? 67

Conclusion 69

Bibliography 72

Appendicies
Appendix 1: Tables 86
Appendix 2: Figures 89
Appendix 3: B.C. Waste Managemnet Permit Fee Regulation Fee Schedules 93
Appendix 4: A comparision of B.C. and German effluent fees 94

List of tables
Table 1: Arenas and political relationships: a diagrammatic summary 86
Table 2: Authority over air and water policy in selected OECD countries 87

List of figures
Figure 1: Incidence of a pollution tax if demand is price inelastic 89
Figure 2: Incidence of a pollution tax if demand is price elastic 90
Figure 3: Incidence of emission trading if demand is price elastic 91
Figure 4: Incidence of emission trading if demand is price inelastic 92

Glossary

Abbreviations of periodicals and journals

BGBl	*Bundesgesetzblatt* (Federal Law Circular, Germany)
BTD	*Bundestagsdrucksache* (Parliament documents, Germany)
BW	*Business Week*
CQA	*Congressional Quarterly Almanac*
CQWR	*Congressional Quarterly Weekly Report*
DS	*Der Spiegel*
ER/CD	*Environmental Reporter, Current Developments*
GM	*Globe and Mail*
MG	*Montreal Gazette*
NS	*New Scientist*
NYT	*New York Times*
TS	*Toronto Star*

Abbreviations of government organizations

B.C.MoE	British Columbia Ministry of Environment, Lands and Parks
BVerwG	*Bundesverwaltungsgericht* (Federal Administrative Court, Germany)
EPA	United States Environmental Protection Agency
OMB	Office of Management and Budget
OPPE	Office of Policy Planning and Evaluation (EPA)

Technical abbreviations

Btu	British Thermal Unit
DM	Deutsche Mark (German Marks)
NAAQS	National Ambient Air Quality Standard
PTs	Pollution taxes
SIP	State Implementation Plan
TPRs	Tradable Pollution Rights
WMPFR	Waste Management Permit Fee Regulations

Acknowledgements

I am indebted to George Hoberg for nurturing my interest in environmental policy, and for his guidance and patience in my writing of this thesis. I also thank Alan Siaroff for valuable comments on the ideas developed in this study.

Thanks are due to Ellen Baar and Stephen Phillips for suggesting interesting literature, and for proofreading parts of the thesis.

Throughout my graduate studies the Department of Political Science and the Sustainable Development Research Institute provided me with teaching and research assistantships, and I gratefully acknowledge their support.

Most of all, the loving support of my parents enabled me to pursue my academic goals. They also made great efforts to provide me with legal texts of German environmental regulations.

Lisa A. and Camel K. were always there when I needed them. Without them I would not have lived through the darker hours of this project. My gratitude to them.

Lastly, I would like to thank the co-inhabitants of the Zzyzzy commune: Dick, Joe, Kathy, Liza, and Shen Hua. None of them share any of my research interests. Neither were they particularly helpful with the research or writing of this thesis. But they are great companions, sharing jokes, ice cream, and stories of epic hiking trips. May it remain this way.

I. Introduction

In this thesis I examine patterns of environmental policy instrument choices in the United States, Germany, and British Columbia. These jurisdictions chose to augment their standard-based regulatory systems with tradable pollution rights (TPRs) or pollution taxes (PTs). Why did some jurisdictions choose PTs, and others TPRs? What are the underlying factors determining these choices? These are the central questions I will address in this study.

Over the past several years scholars and practitioners pushing for environmental policy reform have argued that standard-based environmental policies are ineffective and impose substantial, unnecessary costs on the economy (Ackerman and Stewart 1988; Jorgenson and Wilcoxen 1990; Portney 1992). Economists endorsed 'economic instruments' as more efficient and effective means of implementing environmental policies.

Since the 1970s, the United States and various European countries have included PTs and TPRs in their arsenals of environmental policy instruments. Subsequent national and international policy studies explored the possibilities of applying these instruments further, but did not question their theoretical virtues in the context of political and institutional constraints (see Cassils 1991; Environment Canada 1990; *Project 88*). Suggestions for reforming environmental policy remained overly economistic, politically naive, or both. Essential questions have yet to be answered. For example, Hahn and Hester (1989: 365) observe that the literature has little advice to offer as to which of the presumably more efficient policy instruments should be used to address a certain environmental problem. Similarly, I am not aware of published research on the relationship between instrument choice and institutional and political context.

This thesis attempts to fill some of this void. I discuss the question of how policy makers in different institutional settings are likely to decide if they had to choose between PTs and TPRs.[1] I will argue that the redistributive properties of PTs and TPRs and concomitant interest group pressures cannot explain the variation of instrument choice across different countries. Instead, instrument choice appears to be significantly influenced by the institutional fundaments of

---

[1] It must be emphasized that theory and data regarding instrument choice are not thoroughly developed in the literature. This is particularly true for the choice between PTs and TPRs. Thus, this work is, by necessity, experimental and various hypothesis generated may not be testable to the desired extent.

environmental policy making. In particular, the decentralization of policy making authority among various levels of governments seems to inhibit the adoption of emission trading. Pollution taxes, on the other hand, can be designed to compensate, to a certain extent, for jurisdictional fragmentation. In situations of overlapping jurisdiction, and intense jurisdictional competition over policy making authority neither instrument is likely to be adopted.

The study takes the following course. Chapter one reviews theoretical aspects of instrument choice. In particular, I define policy instruments, and discuss the theoretical properties of PTs and TPRs. Furthermore, various theories of instrument choice are introduced and criticized. Chapter two presents four case studies on instrument choice in the United States, Germany and Canada (British Columbia): the 1976 EPA offset rule, the SO_2 emission trading provisions in the 1990 Clean Air Act Amendments, Germany's 1976 Effluent Fee Law, and the 1992 B.C. Waste Management Permit Fee Regulations. In chapter three I analyze the cases and develop some hypotheses that ought to be added to existing theories of policy instrument choice. Chapter four places the findings into a wider comparative context and poses research questions on a possible link between policy style and policy instrument choice.

I. Theory

1. What are policy instruments?

A theory of instrument choice has to build on a definition of policy instruments. Hahn (1990: 40), however, observes that policy instruments have not been sufficiently defined in the literature, and that this lack of conceptual clarity poses an obstacle to theory building. This section briefly addresses this concern.

Simply put, policy instruments are the means by which policy objectives are pursued. Trebilcock (1982: 25), however, notes that the means - end relationship is not clear. One policy, e.g. the reduction of air pollution by 50 per cent over five years, can be the means to lowering the incidence of respiratory diseases, which, in turn, may be instrumental in reducing public health expenditures. To complicate matters further, one policy instrument can be used to pursue multiple policy objectives. Or, several policy instruments may be employed to deal with just one policy problem. This intellectual conundrum is solved by thinking of related polices as hierarchically ordered action alternatives. Those of lower order are or can be instrumental in implementing policies of a higher order.

The literature on policy implementation offers a complementary and more specific concept of policy instruments. Mazmanian and Sabatier (1982: Ch. 1) find that a "valid technical theory and technology" to achieve given policy objectives should be incorporated in legal statutes to enable implementors to effect necessary changes in human behaviour.

The characterization of policy instruments as those parts of legal statutes that are intended to change human behaviour appears to be very appropriate in the investigation of 'economic instruments' in environmental policy. It helps to differentiate between policy objectives and the means of achieving them. In addition, it helps to focus on the actual mechanisms used to influence peoples' behaviour.

2. 'Economic instruments': Pollution Taxes and Tradable Pollution Rights

The terms 'economic incentives' or 'economic instruments' do not adequately describe the 'new' or 'innovative' policy instruments in environmental policy. Even command and control instruments (i.e. design and performance standards) can provide economic incentives if noncompliance results in fines or other forms of punishment. 'Economic instruments' are better understood as a variety of charge or tax schemes (deposit - refund schemes, compliance bonds, user charges, pollution taxes, etc.), and market creation (emission trading).[2]

As this study is concerned with the analysis of instrument choices between pollution taxes (PTs) and tradable pollution rights (TPRs), the economic theory behind the two instruments will be briefly reviewed below. In particular, I shall look at the distributive properties of PTs and TPRs, and a few other characteristics that may influence the choice between them.

---

[2] Moral suasion, compulsory environmental audits, environmental impact assessments, as well as changes in the legal system (e.g. liability provisions) may also be potent additions to regulators' tool chests. For a discussion of these policy instruments see Cassils (1991), Environment Canada (1992), and Paehlke (1990).

2.1. Pollution taxation

A pollution tax (PT) is a price to be paid per unit of discharged pollutants into the environment, or for each increment of other environmentally destructive activities.[3] Polluters are expected to maximize profits by reducing production, or by installing pollution controls. Pigou first proposed such a tax in his classic *The Economics of Welfare* (1932: 27-31).

PTs attach a price tag to discharges. The tax rate should be set high enough so that a sufficient number of polluters reduce their discharges, and given environmental quality objectives are achieved (Baumol and Oates 1975; Bohm and Russel 1985). PTs make environmental protection more efficient because they force firms to determine the quantity of their discharges based on their marginal abatement costs. It is difficult to calculate the 'correct' tax rate in one step, so that given environmental quality objectives will indeed be achieved. To do so regulators would have to know the internal cost functions of all companies. This is clearly impossible. If the tax is set too low the number of firms reducing discharges may not be large enough to reach environmental quality targets. If the tax is set too high, an unacceptably large number of companies may be forced to reduce production or even shut down. Cost functions will vary widely, even within the same industry sector, and good guesses are difficult. Frequent PT adjustments, however, may be counterproductive as industry will withhold pollution control investments not only until they are advantageous but also until they are cost effective in the long run. Thus, predictability of future PT increases may be a key factor in inducing polluting control, and in making PTs politically acceptable.

The impact of PTs on companies will depend on their cost functions, and the price elasticity of their demand and supply (Anderson et al 1990). Assuming that the discharge of pollution is proportional to the level of output[4] the imposition of a tax increases a firm's marginal costs by the tax rate. The effect of the tax on the firm's revenues is contingent on the shape of

[3] Various authors use the term pollution charge instead of pollution tax (e.g. OECD 1989). I will use the terms pollution tax, pollution fee, and pollution charge synonymously, with the understanding that the amount increases in relation to discharge quantity.

[4] If the level of pollution is not directly related to a firm's output the imposition of a tax does not change the marginal cost structure, at least in the short run. In this case, the additional costs will cut into profits, or may be (partially) passed on to workers, stockholders, or suppliers. This case shall not be considered here further because levying a tax on such pollution, obviously, does not induce pollution control.

the demand curve. If demand is perfectly price inelastic the cost increase can be fully passed on to consumers, resulting in higher product prices, unchanged production, and constant profits (Figure 1).[5] If demand is perfectly elastic, a pollution tax cannot be passed on to consumers and will decrease production, profits, wages, or other investments by the firm (Figure 2). For intermediate assumptions the firm will lose revenues and profits will decline. But the steeper the demand curve, the more can the tax be passed on to consumers.

If a PT is levied companies will have to pay for using the environment. Payments will have to continue even after companies install pollution controls, as discharges are unlikely to be eliminated completely. In a standard-based system this was not the case. There, companies could discharge for free as long they complied with the terms of their permits.

Taxes generate revenues, and government may choose to redistribute and reallocate these revenues for pollution control projects, or to compensate victims of pollution, for example. (Baumol and Oates 1975, 1979). Polluters usually favour the earmarking of revenues to subsidize research and development programmes into pollution control, or financing of communal pollution control projects, for example waste water treatment plants. This way, they could recoup at least a part of their tax payments.

2.2. Emission trading

In a TPRs system a maximum level of pollution or environmental degradation from a specific pollutant in a jurisdiction or ecosystem is determined. This total is then divided into shares that are allocated to polluting companies by grandfathering or by auction.[6] TPRs would then trade at a price that equals marginal pollution control costs at the plant where abatement is cheapest. In theory, this would guarantee the efficient allocation of pollution control expenditures. Those industries with the highest cost of pollution control will bid for TPRs while those

---

[5] The figures are contained in Appendix 1.

[6] The Canadian economist John Dales (1968) is generally credited with first conceiving markets for pollution rights. Bohm and Russel (1985: 419), however, point out that Dales only elaborated on the ideas of Thomas Crocker laid out in H. Wolozin (ed). 1966. *The Economics of Air Pollution*, New York: W.W. Norton.

with lower costs will choose to clean up. If the number of circulating TPRs is reduced by government, the market price for TPRs will increase and higher-cost firms will reduce pollution.

The establishment of a TPRs market requires serious thought on a variety of design issues.[7] Location and interaction of discharges are crucial in determining environmental quality and thus the overall cap on emissions (e.g. smog or low atmospheric ozone levels). Also, in the course of trading, TPRs, and thereby discharges, may concentrate in one locality and environmental quality standards may be violated. The design of an appropriate market area may be difficult if political jurisdictions are not congruent with a particular air or watershed. In addition, without a sufficient number of companies of similar size but with different pollution control cost functions the volume and benefits of trade are likely to be low.[8] Transaction costs for trades may be high, possibly prohibitive for small companies.[9]

The effects of a TPR system on firms are contingent on the initial distribution of the TPRs and the shapes of cost, supply and demand functions of the companies (Anderson et al 1990). If TPRs are grandfathered according to existing pollution permits companies can reap an enormous windfall. In some circumstances the value of TPRs may even exceed the value of industry production (Anderson et al 1990: 57). In addition, grandfathering would create market barriers similar to emission standards that discriminate between old and new sources (Dewees 1983), and companies wishing to locate in the market or trading area would have to buy permits from existing companies. The other allocation option would be to auction TPRs. Companies would incur an one-time expense but would not have to make continuous payments as in the case of PTs.

After trading commences, companies can reduce costs if suitable TPRs are available (if trades do not lower costs, no trades will occur). With decreasing marginal costs and perfectly

---

[7] Design problems are discussed, among others, by Atkinson (1981), Baumol and Oates (1975), Dales (1968), Hahn (1983, 1986), Hahn and Hester (1989a,b), and Hahn and Noll (1982).

[8] The design of a pollution permit market with oligopolistic or monopolistic features poses special problems and is discussed in Hahn (1984).

[9] Companies willing to trade have to search for suitable business partners in the various markets of pollutants it discharges. Opportunity and transaction costs may be high. In California, for example, hydrocarbon emission rights are traded at $500 per ton, while pollution abatement costs between $2,000 and $5,000 per ton (Anderson et al 1990: 18). The difference is accounted for by transaction costs.

price elastic demand the company can expand production and increase profits (Figure 3). On the other hand, if the company faces a price inelastic demand curve then decreasing marginal costs will increase company profits (Figure 4). Depending on the market structure the divergence of marginal costs and market price may lead to the intrance of competing companies into the market, and decreasing profits in the long run. For demand curves between these extremes the effects are indeterminate.

As in the case of PTs, the distributional effects of emission trading would be difficult to predict beforehand. However, if the government recognizes existing pollution permits as legitimate 'rights' and grandfathers TPRs, then companies have nothing to lose.[10] Industry can only gain from emission trading, whereas the imposition of PTs would almost always reduce profits. Thus, industry will prefer TPRs over PTs (cf. Dewees 1983).

2.3. Pollution taxes vs. tradable pollution rights: A preliminary appraisal

As described above, the key characteristic of PTs and TPRs are that, compared to standard-based regulation, government leaves the determination of either quantity of discharges or the price for pollution control to the individual or company.[11] Weitzman (1974) demonstrated that the information requirements are the same to determine either the correct PT level or to calculate the production cut backs resulting from the imposition of an emission cap. In neither case perfect information will exist, especially not concerning which company will bear the brunt. This uncertainty introduced by TPRs, and to a slightly lesser extent by PTs, weakens industry efforts to organize and lobby for a *common* position.

Neither standards, PTs, nor TPRs can dispense with pollution monitoring, enforcement, and administration. There is no evidence that total costs associated with any of the instruments would be significantly lower compared to others.[12] For example, in each case government

[10] Rolph (1983) and Welch (1983) show that the U.S. government customarily recognizes existing property rights.

[11] The question whether standards are preferable over PTs or TPRs is not at issue here. For a discussion see, among others, Baumol and Oates (1975, 1979); Bohm and Russel (1985); Freeman et al (1976); Kneese and Schultze (1975).

[12] Rather, the adding of a new policy instrument to the existing regulatory system would likely entail additional costs, regardless which one is chosen (Downing 1983; Hahn and Stavins 1991).

would have to set environmental quality standards to protect public health.[13] Such standards would also be a prerequisite for the operation of a TPRs system as so called 'hot spots, that is the concentration of too many TPRs and thus emissions in one area, have to be avoided. Similarly, even if a company were to pay the appropriate PTs, discharges beyond allowable environmental quality standards would have to be outlawed to protect public health. Thus, neither TPRs nor PTs are 'a license to pollute'. However, the moral stigma of polluting beyond the prescribed standard may be perceived as being stronger under standard-based regulations (Kelman 1981).[14]

In summary, I find that there are few technical differences between PTs and TPRs. The key distinction is that a tax system, in most cases, would be more expensive for polluters. Naturally, industry will favour policy instruments that are less expensive. Politically, the introduction and operation of a PT system would appear to be more onerous than emission trading. Interest group struggles over the allocation of revenues may be an additional burden.

3. Instrument choice in the policy process

Although problems of instrument choice have been discussed in the literature for more than a decade, textbooks on public policy do not identify instrument choice as a distinctive stage in the policy process (see, for example, Jones 1984). It is helpful, however, to distinguish instrument choice from policy choice and to identify the institutional setting of instrument choice.

Jones' description of policy formulation comes closest to what I characterize as instrument choice. For him, policy formulation includes the development of a plan, and the identification of an implementation mechanism to address a certain policy problem (Jones 1984: 77). Without a promising "solution" a policy issue may not progress from the political agenda to the "governmental agenda", or even the "decision making agenda" (Jones 1984: 28, Ch. 5). To achieve this strategic advance, Jones points out, intelligent policy formulation "must give

---

[13] Hahn (1989a,b) observes that PTs and TPRs are usually grafted onto existing standard-based regulations, and are not used to substitute them.

[14] I do argue that all three approaches need environmental quality standards for the same reason, and that there is no technical difference between them in this respect. However, I readily admit that in politics technical detail often matters less than perception. Indeed, environmental groups will try to exploit this perceptional difference. For an eloquent argument on this point see Wildavsky (1979: Ch. 8).

thought to the attitudes, rules, and demands that circumscribe the behaviour of legitimators" (1984: 89).[15]

Similarly, Kingdon (1984) sees instrument choice as the crucial stage in the life of a policy proposal. According to Kingdon, without an answer to 'how will this policy actually work?' a policy proposal will not advance from the "primeval policy soup" to a "short list of ideas." At this stage, policy proponents have to find technically feasible solutions that are compatible with the values of the relevant policy community, and that anticipate future constraints such as the impact of negotiations in legislative committees, or decision making rules (Kingdon 1984: 138-147). It is this *initial* but crucial choice that I will refer to as instrument choice. Details in the design of instruments are probably altered later on in the policy process. But to get that far, the 'fundamentals' of the instrument have to be acceptable in the policy community.

At the risk of repeating myself, this point deserves some emphasis. To conceptualize instrument choice as a continuous process that pervades the whole policy cycle would conflate instrument choice with policy choice. The reason *why* a certain policy problem should be addressed is different from *how* the problem should be addressed. Instrument choice does not just emerge from interest group struggles. Instrument choice is a conscious act by an identifiable set of people in an identifiable institutional setting. Jones (1984) and Kingdon (1984) emphasize that proposed solutions are strategic choices by policy proponents. Whether they will prevail, or how much they will have to negotiate away under the onslaught of competing interests, is a question of whether such policy instrument proponents made intelligent instrument choices, not a question of instrument choice itself.

4. Who chooses, and why?

The institutional setting of instrument choice can vary a great deal and with it how values in the policy community, and subsequent decision making processes, are considered by instrument choosers. Possible locations are staff offices of legislators, legislative committees, execu-

---

[15] In Jones' terminology "legitimators" are all actors in the policy process that push further discussion of a policy proposal (1984: 138-147).

tive agencies, lobby groups that wish to present a finalized policy proposal, or combinations of the above (Jones 1984: Ch. 5).

A number of scholars have analyzed such choice situations, and the key motivations of instrument choosers. Trebilcock (1982) suggests that legislators will try to allocate benefits to "marginal ridings" in which a change in the voting intentions of a small number of voters would have a pronounced effect on election outcomes. Doern and Wilson (1974) argue that politicians would tend to choose policy instruments that do not provoke the resistance of interest groups or other orders of governments. Thus, less coercive policy instruments would always be preferred. However, they did not define coerciveness, and perceptions of coerciveness may vary across different political cultures and systems. Niskanen (1971) contends that bureaucracies exploit their leverage over legislators to secure bureaucratic interests such as agency size, or budget allocations. Students of public administration emphasize that instrument choice crucially depends on past choices (Lindblohm 1959), or that instruments with poorly understood effects will rarely be incorporated in comprehensive long-term programmes (Braybrooke and Lindblohm 1963).

Lowi (1964) and J.Q. Wilson (1984) propose that instrument choices depend on how the distribution of costs and benefits of policy outcomes mobilizes interest groups in society. In their typologies of policies and politics, air and water pollution would be classified as imposing concentrated costs on polluters and spreading diffuse benefits of cleaner air or water throughout society. Wilson (1984: 429) calls the politics over such issues "entrepreneurial politics", while Lowi (1964: 690) refers to it as "regulative politics."

This general pattern of political interaction can, however, change under the impact of instrument choices. Policy implementation by a PT with an allocative mechanism can lead to interest group struggles (concentrated costs and concentrated benefits), or client politics (diffuse costs and concentrated benefits), depending on how the tax system is structured. An emission trading system with grandfathered TPRs (distributed benefits) would distribute costs and benefits according to regulatees' cost structures, not according to the organizational lines of industry lobbies. Although some firms will pay a high price, the market mechanism may obscure the severity and timing of the regulatory impact (for better or worse). In this case, political interaction would exhibit reduced conflict levels, approaching what Wilson (1984: 432) called "majoritarian politics" (diffuse costs and diffuse benefits). Such interaction would be accompanied by client politics (concentrated benefits and diffused costs) as individual firms or jurisdictions seek special treatment in the allocation of TPRs. Thus, whether policy makers choose PTs or TPRs, patterns of political interaction may deviate from the intense interest group struggles typical for standard-based regulations.

Interestingly, Lowi (1964) associates each policy type with a particular kind of interaction, in a characteristic political setting (see Table 1 in Appendix 2). Generally, the more fragmented the decision unit, the less likely the adoption of redistributive or coercive policy instruments. However, from this we cannot conclude that integrated decision units will not prefer distributive instruments as well.

A major shortcoming of explanations focusing on distributional aspects of instrument choice is their lack of explanatory power in situations of choice between instruments with similar distributional impacts, or if choices are unrelated to such considerations in the first place.

Also, Lowi's and Wilson's typologies are not necessarily transferable to the analysis of other political systems, as they are contingent on specific institutions that guide such interaction.

Institutionalists like Evans et al. (1985), Hall (1986), or March and Olsen (1984) would downplay the importance of instruments' distributional impacts. Instead they would stress that the structure of institutions and the rules that govern political interaction strongly influence instrument choice.

Weaver's (1986) blame avoidance theory also helps to address this problem. Weaver (1986: 385) argues that politicians basically pursue three strategies. Firstly, they may follow a "good policy orientation" and be interested in the substantive policy outcome "rather than the political credit or blame that is associated with it." Secondly, politicians may exercise leadership and maintain discretion over future policy instrument adjustments "to make more credible claims for credit from their constituents." And thirdly, politicians may avoid policy choices that entail "opportunities" for substantial blame but uncertain political credit. Weaver (1986: 372) claims that politicians ordinarily are "blame minimizers and credit-claiming and good policy satisficers." Among the strategies for avoiding blame Weaver (1986: 385) sees (1) issue redefinition, to "prevent blame generating by developing new policy options which diffuse or obfuscate losses", (2) passing the buck to "deflect blame by forcing others to make politically costly choices", and (3) by 'circling the wagons' to "diffuse blame among as many policy makers as possible." Weaver argues that politicians in a parliamentary system are more prone to follow blame avoiding strategies than are those in the decentralized U.S. congressional system. In the former system, blame concentrates on the party or government as a whole, and there is 'no one to pass the buck to.'

Harrison (1991) points out that blame avoidance is exacerbated by federalist state structures and division of authority over policy issues among jurisdictions. Taking Canada as a case in point, she argues that federal governments have little incentive to aggressively tackle environmental problems that are politically risky (backlash from cost bearers, possibility of policy failure), and in addition may generate conflicts with provincial governments. The pattern of blame avoidance and claim seeking may vary across policy areas. Generally, we can expect provincial governments to posture and underscore *their* jurisdictional authority by choosing policy instruments that minimize federal interference (Chandler and Bakvis 1989: 77).

Canada, however, is only one out of many different possible federalist arrangements, and blame avoidance may take different forms in other federalist states. Chandler (1989) distinguishes federalist systems by how power is shared between jurisdictions. In Canada authority is shared along jurisdictional, but in some policy areas, e.g. environmental policy, spheres of authority overlap. This results in duplication of governmental functions at the federal and provincial levels. The inadequacy of ties between political elites at the federal and provincial levels, and lack of consensus on a legitimate political process encourage adversarial federal - provincial relations. In the U.S. authority is also shared along jurisdictional lines, but federal - state cleavages are of low salience because the influence of interest groups effectively ties representatives to constituent interests, and enforces accountability. Also, states are directly represented in national policy making through the Senate. In Germany authority is divided by function. In some policy areas the federal government has exclusive authority. In some areas, such as environmental policy, the federal government legislates but the states implement policy. And some policy areas are left to the states. Strong party bonds between federal and state politicians, and the direct involvement of state governments in federal policy making through the Upper House encourage cooperation. Thus, blame avoidance in the U.S. and Germany are unlikely to resemble the Canadian pattern. Rather, Congresspersons will try to protect constituent interests by shaping legislation, and being part of the winning legislative coalition. In Germany conflicts are either resolved cooperatively (otherwise there may not be a policy output in the first place), or, if this is not possible, issues may be consensually redefined to fit the established patterns of interaction.

This brief survey indicates that instrument choice is embedded in a larger setting that includes, among others, the specific institutional features of state organization, and the impact of political parties in the policy process. In the crossnational analysis we should expect a wide variation of how these factors affect instrument choices. However, current theories of instrument choice, and available data on environmental policy formulation in general, are not sophisticated enough to accommodate these important but relatively specific factors.

Instead I will use the concept of policy style, that combines similar notions of institutional and ideological factors (Hoberg 1991; Richardson 1982; Simonis 1992, Vogel 1986). Hoberg (1991: 2) characterizes policy style along two axes: (1) Specific patterns of interaction between actors in the policy process, and (2) guiding norms or philosophies which lend those patterns legitimacy. The basic contention of this approach is that "the characteristics of a political regime are more important than the nature of the particular policy area itself in explaining policy processes" (Vogel 1986: 195). Different policy styles may produce different patterns of instrument choice over time, but such a pattern is not yet evident in environmental policy. A recent OECD report indicates that all member states predominantly regulate pollution sources with process or performance standards (OECD 1989). However, there appear to be varying inclinations to experiment with PTs and TPRs, and possible reasons for this will be investigated in the context of the case studies below. A more detailed examination of instrument choice theories in environmental policy in the next section will help to focus this exercise.

5. Instrument choice in environmental policy

Economic modelling has highlighted theoretical properties of policy instruments, and pointed out their effects under various market conditions.[16] Based on such modelling, a number of authors have made suggestions which policy instruments would be preferred by different actors. Buchanan and Tullock (1975) argue that firms would favour the assignment of company-specific pollution quotas over an industry-wide PT. Dewees (1983) compares the desirability of emission standards, PTs, and TPRs from the perspectives of industry and labour interests. He finds that policy instruments creating market entry barriers for new companies will be preferred: Companies would favour effluent standards over PTs, and would still prefer standards or pollution quotas over no regulation at all. The cost-free allocation of TPRs has a similar effect as quotas, but leaves more flexibility to firms. Thus, firms would favour TPRs over quotas, standards, or no regulation at all (Dewees 1983: 70).

Economic modelling, however, has its limitations. As soon as institutional aspects of instrument choice are considered "normative economics enters an Alice-in-Wonderland world in which policies that are desirable in the truncated model lose much of their appeal" (Goldberg 1974: 461). Most economists, perhaps with some frustration, recognize that economic theory cannot explain policy instrument choice without extending beyond its normative boundaries. Dales (1968: 39) admits that once the economist abandons his "artificial problems" and "artificial solutions" he is "quite unable to say that one policy is demonstrably superior to all others" (also see Baumol and Oates 1979; Bohm and Russel 1985). Freeman et al (1973: 104) make suggestions for more efficient and effective environmental policies but voice doubts about "the ability of a pluralist political system to make wise choices in issues of that sort."[17]

---

[16] Mäler (1974) discusses effects of policy instruments in a general equilibrium model, while Bohm and Russel (1985) address problems of instrument design in a partial equilibrium approach. Burrows (1981) and Misolek (1980) deal with policy instruments for pollution control in monopolistic markets. Linder and McBride (1984) and Malik (1990) address the influence of enforcement costs and effectiveness on the behaviour of firms and regulatory agencies.

[17] Some economists left their traditional territory and either entered the spheres of public choice theory or made suggestions to change institutions. Dales (1968), for example, conceptualized markets for pollution permits, which represented an institutional innovation in environmental policy at his time.

Empirical studies show that actual policy outcomes differ from theoretical predictions of actors' preferences and market behaviour (Hahn 1989a,b; Hahn and Hester 1989a,b). Hahn (1987, 1989a,b), and Hahn and Stavins (1991) offer more comprehensive explanations of instrument choices in environmental policy. They try to explain why market creation would be preferred by certain actors, and which political developments explain their increased adoption in the U.S.

Hahn (1987, 1989a,b, 1990) derives a public choice model of instrument selection by ascribing standard preferences to environmentalists and industry as the principal contenders in environmental policy. According to Hahn, industry would generally prefer less environmental quality or more market-oriented policy instruments as both would increase profits. Environmental groups would prefer more environmental protection, and would distrust market-oriented policies as they may be perceived as legitimizing pollution. Consequently, according to Hahn's argument, an increase in industry influence would result in either a decrease in environmental quality or an increased application of market-oriented policy instruments. Similarly, industry would resist higher fees for emissions and discharges, while environmental groups would advocate them. Again, with increased industry influence in the policy process, pollution charges would either not be adopted or would be expected to decline.[18]

Hahn's model reflects a pluralist vision of politics in which interest groups compete and the legislature "referees the group struggle, ratifies the victories of the successful coalition ... in the forms of statutes" (Latham 1952: 390). The model does not account for institutional features of the instrument selection process, the influence of players beyond two interest groups, and reasons for instrument choice unrelated to economic or political pay-off. More fundamentally, Hahn's model suffers from his dominant concern that more efficient policy instruments *should* be chosen.

Hahn (1989a,b) also studied the performance patterns of PTs and TPRs. He observes that "the major motivation for implementing emission fees is to raise revenues" which are then ear-

---

[18] Hahn also shows that industry and environmental groups would agree that existing pollution sources should be less stringently regulated than new ones. The existence of such a minimum common denominator would help to explain why the movement towards market-oriented policy instruments is fairly slow.

marked for pollution control (1989b: 48). He finds little evidence that fee levels had a systematic relation to expected behavioural changes. Yet, PTs "are widely perceived to have had a positive impact on environmental quality", presumably because of revenue reinvestment (1989b: 49). However, there are no data to support that environmental quality actually improved because of PTs (1989b: 48). Hahn's observations regarding TPRs are, again, interesting. "The primary motivation behind marketable permits is to provide increased flexibility in meeting prescribed environmental objectives", and environmental quality is rarely improved (1989b: 49). This is surprising as TPRs could easily be devaluated over time which should lead to environmental quality improvements. Why would the, in Hahn's view, more effective instrument lead to smaller environmental quality improvements? Lastly, Hahn notes a strange pattern of PT and TPRs application. Only the U.S. operates a number of emission trading schemes, primarily for air emissions. PTs are used in a number of countries, including Canada (British Columbia), France, Germany, and the Netherlands. PTs are usually used to control effluents, and not to reduce air emissions.

Hahn's model clearly would not explain this pattern. Hahn may object to taxation on ideological grounds. He dismisses PTs as a means to raise revenues. If governments were really interested in raising revenues, they should prefer more revenues to less, and also tax air emissions. Yet, with the exception of B.C., we do not observe this as a general trend. PTs are such a pervasive phenomenon that they ought to be studied seriously, and the erratic patterns of PT levels, alleged environmental improvements, etc. ought to be explained. In addition, the possible function and effects of PTs as revenue raisers should be studied within the wider context of environmental policy.

Studying the case of hydrocarbon reductions in Wisconsin Hahn (1987) finds that legislators prefer policy instruments that facilitate the creation or dispersion of highly visible benefits to their constituents (e.g. jobs, new factories). In the Wisconsin case policy instrument choice was subordinate to a broader social conflict usually couched as 'jobs vs. environment.' This is nothing new for students of public policy, and Hahn (1987) suggests that legislators may try to manipulate pollution charges or emission trading schemes to make them compatible with established practices like pork barrel politics. In view of the choice between PTs and TPRs the question would then be which of the two instruments is more adaptable to pork barrel politics. Also, pork barrel politics depend on the legistators' direct influence on policy implementation in their riding. Jurisdictional fragmentation between federal, state and local government may determine the degree of legislators' influence and their incentive to pursue this strategy. This shows that instrument choice may be contingent on institutional structures. Comparative studies may help to clarify the relationship between instrument choice, pork barrel polictics, and jurisdictional fragmentation.

Hahn and Stavins (1991) examine the socioeconomic and political developments that facilitated greater acceptance of economic instruments in the U.S. over the past decade. They argue that increasing pollution control costs, a decline in international competitiveness, and persisting budget deficits fostered more positive attitudes towards instrument innovation in the Executive, in Congress, and among environmental and industry interest groups. Budget constraints focused the debate on 'how much environmental quality do we get for our money.'[19] Key actors adjusted their political strategies and cooperated on issues like phasing out lead in gasoline, and CFCs (see Hahn and Hester 1987; Hahn and McGartland 1988) - without sacrificing their fundamental goals.

The changes described by Hahn and Stavins should not be interpreted as sufficient conditions for the adoption of economic instruments. Rather, changes in the regulatory arena create windows of opportunity for people promoting 'their solutions' to other peoples' problems (Kingdon 1984). The exclusive focus on facilitating conditions does not explain conclusively *why*, and more importantly, *which* among the many possible alternative economic instruments would be chosen. Following Kingdon's list of 'criteria for survival,' economic instruments may only offer the technically feasible solution to certain environmental problems. But are they also

---

[19] Ackerman and Stewart (1988) argue that this is a great strength of economic instruments which could help to make the debate over environmental policies more transparent for the general public.

compatible with the values held in the policy community? How do they perform under the constraints of decision making rules and other institutional characteristics?

A theory of instrument choice has to be flexible enough to explain instrument choices in a variety of institutional and cultural settings. But are the facilitating conditions described by Hahn and Stavins (1991) present in all political systems where PTs or TPRs were considered? Are some political systems more responsive to innovative policy instruments than others? Or do some arrangements of political institutions make it easier to adopt a well defined subset of 'economic instruments' (e.g. emission trading)?

This thesis will address these questions by examining four cases in which policy makers chose, or conceivably could have chosen, between PTs and TPRs to achieve given environmental quality objectives. The cross-national comparisons shall help to discover "empirical relationships among variables" (Lijphardt 1971: 683). It does not intend to postulate correlations or functional relationships between instrument choice, cultural values and institutions.

II. Case studies

This chapter describes and analyzes four cases in which regulators chose TPRs or PTs to achieve their policy objectives. I will examine emission trading in the United States, particularly EPA's 1976 offset rule, and the SO_2 trading provisions in the 1990 Clean Air Act Amendments. A third case study looks at Germany's 1976 Effluent Charge Law (ECL) (*Abwasserabgabengesetz*). The fourth case study describes the 1992 revision of waste discharge permit fees in British Columbia (Canada). The case studies are introduced by a brief review of the regulatory framework and dominant policy styles in the respective countries. They are followed by a brief analysis that will serve as a starting point for the comparative analysis in chapter three. In addition, I will briefly review the discourse over PTs and TPRs in the respective countries.

The case studies are guided by four blocks of questions:

(1) What was the political context of instrument choice? What kinds of policy windows opened? What was the institutional locus of policy instrument choice?

(2) What was the policy content? How did the instrument choice correspond with the values held in the policy community? How how did the instrument perform in the institutions of the decision making process?[20]

(3) What patterns of conflict and conflict resolution emerged? And how did these relate to the fundamental properties of the policy instrument? Were political conflicts resolved in a way that made future adoption of the same instrument likely?

(4) How effective was the policy? How were or will subsequent policy instrument choices be influenced by this experience?

1. Emission trading in the United States

1.1. Framework and policy style of U.S. environmental regulation

The United States is a federal country. Law making powers are divided and shared between the states and the federal government. The constitution gives Congress power to "regulate Commerce with foreign nations, [and] among the several States" (Art. 1(8)(3)). Furthermore, Congress has authority over federal lands (Art. 1(8)(17), and it can levy taxes on pollution by invoking Article 1(8(1). In *Sporhase vs. Nebraska* (457 US 273, 1982) the Supreme Court ruled that the Commerce Clause empowers Congress to regulate everything pertaining to health and the environment (Friedman 1987: 2). States have to comply with these laws but they may enact more stringent or additional legislation (Jessup 1990; Portney 1990b; Rehbinder and Stewart 1985).

[20] The technical feasibility and applicability of the instruments will not be evaluated. This would require technical expertise and access to information at least equal or greater to that of the actual instrument choosers.

The regulatory process is marked by intense competition in, and among, interest groups, Congress, executive agencies, and the courts (Hoberg 1991b, 1992; Melnick 1983). Horizontal fragmentation of power and mutual distrust leave little room for cooperative bargaining over either writing regulations, or over enforcement of regulations. Although EPA embarked on a few experiments with negotiated rulemaking throughout the 1980s, these were largely unsuccessful (EPA 1987). In addition, all major pieces of environmental legislation since the 1970 contain clauses that allow citizens to take EPA to court if it fails to meet Congressional deadlines, and to sue companies for breaking regulations. According to Stewart (1985), the propensity to litigate has made the U.S. regulatory system extraordinarily adversarial, rigid, and inefficient (also see Portney 1990b, 1992).

1.2. EPA's 1976 offset rule

In December 1976, EPA announced that companies could locate in areas which did not meet National Ambient Air Quality Standards (NAAQS) if they would ensure that other sources would reduce their emissions by at least an equivalent amount (offsets). The offset policy highlighted the enormous economic pressures the stringent technology-based standards of the 1970 Clean Air Act Amendments exerted on the economy. More fundamentally, the introduction of emission trading into U.S. environmental policy can be seen as a consequence and an extension of the pluralist regulatory regime emerging in the 1970s.

1.2.1. Political context

Popular concern over environmental pollution steadily rose throughout the 1960s and early 1970s, and figured increasingly prominent on the political agenda. Environmental interest groups mushroomed and gained significant influence in legislative and executive politics (Hoberg 1992: Ch. 3). In 1969, the National Environmental Protection Act passed Congress; in 1970, the Occupational Health and Safety Act passed, and the Clean Air Act was amended; in 1972, the Federal Water Pollution Control Act was enacted. The barrage of federal legislation signaled Congress assertion of power over lower jurisdictions, but, more important, over executive agencies (Hoberg 1992: Ch. 3).[21] Environmental groups had become a powerful actor in the policy process. Due to increased public awareness their membership had grown. And numerous progressive court judgements even granted standing in court to environmental groups.[22]

The Clean Air Act Amendments of 1970 (Public Law No. 91-604) set the agenda for air pollution control in the 1970s.[23] Congress directed EPA to formulate National Ambient Air Quality Standards (NAAQS) (Sect. 7410(a)(2)(A)), and to ensure that states meet primary standards, those "requisite to public health", by 1975 (Sect. 7409(b)(1)).[24] States had to formulate state implementation plans (SIPs) which outlined how areas that violated the NAAQS (nonattainment areas) would achieve compliance (Sect. 7410(a)(1)). Noncompliance would "prevent the construction or modification of any new source ... which the state determines will prevent the attainment or maintenance ... of a national ambient air quality primary or secondary standard" (Sect. 7410(a)(4)).

The legislation did not achieve its intent: By 1975 about half of the country's 247 air pollution control regions did not meet NAAQs (Landau 1979: 578), and the law severely constrained industrial development in nonattainment areas. Industry pressured EPA to stop sacrificing economic growth for overambitious environmental goals. Already in 1973, EPA was forced to relax standards for SO_2 emissions by allowing the construction of tall smoke stacks for better pollutant dispersion (Kneese and Schultze 1975: 59). In 1975, EPA proposed to exempt new or modified sources in nonattainment areas from the stringent new performance standards if emissions from the entire facility did not increase (40 FR 58416, Dec. 16, 1975).[25] In 1976,

---

[21] This is particularly evident in Congress' determination of policy objectives, policy instruments, and time frames for implementation that pervade the legislation of the time.

[22] See *Association of Data Processing Service Organizations, Inc. vs. Camp*, U.S.C. 150 (1970); *Sierra Club vs. Morton*, U.S.C. 727 (1972). For an in depth treatment of the role of courts in U.S. regulatory policy see Melnick (1983, 1985).

[23] For a legislative history and the politics of the Clean Air Act see Crandall (1983), Friedlaender (1978), Hoberg (1992), Lave and Omenn (1981), and Melnick (1983).

[24] Secondary standards, protecting "public welfare" were to be met within "reasonable time" (42 U.S.C.A. Sect. 7410(a)(2)(A)(ii)). On the role of technology-based standards in Clean Air Act regulations see M.L. Wilson et al (1990).

[25] The Carter administration supported the "netting" policy (Crandall 1983: 83). But netting was not fully established until 1984 when the Supreme Court overturned previous court rulings prohibiting netting as a means to

EPA officials tacitly allowed internal emission trade-offs on a case-by case basis in nonattainment areas with poor regional economies (Yandle 1978: 24). Meanwhile, EPA maintained, in part forced by court judgements,[26] it would not allow public health to be sacrificed for economic gains:

> Some have argued that a new source should be allowed to worsen existing NAAQS violations if a "cost-benefit" analysis indicates that the economic costs of necessary emission controls ... are excessive in relation to the resulting air quality benefits. ... The Clean Air Act simply does not allow such an approach. Application of such a policy could allow further delay in achieving already-overdue standards. ... Particularly with regard to NAAQSs, Congress and the Courts have made clear that economic consideration must be subordinated to NAAQSs achievement and maintenance (41 FR 55527, Dec. 21, 1976).

This tough statement, however, was already part of EPA's "interpretative ruling" which outlined the features of the so-called off-set policy (41 FR 55525, Dec. 21, 1976).

1.2.2. Policy content

The December 21 ruling allowed major modified or new sources[27] to locate in nonattainment areas under five conditions. (1) The new source had to meet the "lowest achievable emission rate", which was determined by considering "the most stringent emission limitation in any SIP and the lowest emission rate which is achieved in practice for such a type of source" (41 FR 55528). (2) All sources owned by the applicant already located in the same air quality control region had to be in compliance with all applicable regulations and standards (41 FR 55529). (3) The owner of the major modified or new source had to obtain emission offsets from existing sources exceeding its own projected emissions. Only intrapollutant trading was allowed (41 FR 55529). (4) The emission offsets had to provide a "positive net air quality benefit in the affected

---

locate or expand facilities in nonattainment areas (*Chevron U.S.A. vs. Natural Resources Defense Council, Inc.* 467 U.S.C. 837 (1984); also see *ASARCO vs. EPA*, 578 F.2d 319 (1978), *Natural Resources Defense Council vs. Gorusch*, 685 F.2d 718 (1982)).

[26] *American Petroleum Institute vs. Costle*, 667 F.2d 1176 (1981); *Lead Industries Association vs. EPA*, 647 F.2d 1131 (1980). Also see Melnick (1983: Ch. 8).

[27] Major modified or new sources were defined as those emitting more than 100 tons of particulate matter, sulphur oxides, nitrogen oxides, or non-methane hydrocarbons, or more than 1,000 tons of carbon monoxide. However, EPA proposed to reduce the limits to 50 and 500 tons, respectively (41 FR 55528, 55558, December 21, 1976).

areas" (41 FR 55529). And (5) the major modified or new source had to locate in an area where EPA had approved, or was in the process of approving, appropriate SIPs (41 FR 55529). This last condition was necessary because EPA chose to use SIP data as baselines for the calculation of emission offset credits. The ruling allowed industry to find the least costly way to achieve emission reductions in old sources, including input or process changes, installation of pollution control equipment, or temporary shutdowns. The ruling, however, explicitly prohibited the banking of emission offsets for later use (41 FR 555529).

EPA's offset rule was incorporated in the 1977 Clean Air Act Amendments (42 U.S.C.A. Sect. 7502). However, Congress altered condition no. (1) (see above) so that the new source had to comply with the lowest emission limit in either SIP or practiced technology (Sect. 7501(3)). Congress also left it to the states to use the offset policy or any other mechanism to facilitate economic growth while complying with NAAQSs (Sect. 7503(1)). Finally, the federal offset policy was to be phased out in mid-1979. After that the states were to assume full control over emission control in nonattainment areas (Sect. 7502).

The expansion of TPRs, and the introduction of PTs through the 1977 CAA Amendments was opposed by Senator Muskie, environmental groups, and industry association who, for a variety of reasons, did not endorse 'pollute for a fee' or 'let the market reign' approaches to pollution control (Kelman 1981). And the Treasury Department, and the Ways and Means Committee, did not want to tinker with the tax code (Marcus 1980: 171).

Unscathed by the developments in Congress, EPA still worked to improve the offset concept. In December 1978, the agency issued a revised emission offset rule (44 FR 3274, Dec. 29, 1978). Its central feature was the lifting of the banking ban to provide more flexibility to industry (44 FR 3274, 3285). The exact nature and the governing regulations of the banking scheme, however, were not established.

1.2.2. Patterns of conflict and conflict resolution

EPA's offset policy was accompanied by remarkably little conflict, for two reasons. First, the offset policy was geared towards resolving conflicts between industry and states' desire for economic development on one side, and Congress' legislative goals on the other. Secondly, EPA was aware that the offset policy constituted a radical departure from prevailing modes of regulation. It only published the interpretative ruling after extensive consultation with state and local air pollution control officers, the National Governors Conference, the National Conference of State Legislatures, the U.S. Conference of Mayors / National League of Cities, the National Association of Counties, all major labour unions, industry associations, and environmental groups (41 FR 55525). In addition, EPA held more consultations and hearings on the offset policy in numerous cities around the U.S. in the weeks following the rule's publication (41 FR 55527). The extensive efforts to build a political consensus at all levels of government and among interest groups paid off: the offset policy was not challenged in court.[28]

While interest groups did not oppose the offset policy they did not enthusiastically endorse it either. Environmentalists were generally aware of the potential cost savings associated with emission trading (Kelman 1981), and the stringent technology-based standards that accompanied it eased their discomfort about making air a tradable good (Meidinger 1985: 460). Industry groups were less familiar with emission trading and found the offset policy too complicated, and potentially entailing high transaction costs (Kelman 1981; Meidinger 1985: 459). Industry also distrusted any policy initiative from the Carter administration fearing some kind of Trojan horse. With only two more years to the next election (and the prospect of a Republican government bringing overall regulatory relief), and with no obligation to participate in the programme, industry saw no need to fight back (Meidinger 1985: 460).

Conflicts over the offset policy were more pronounced within EPA, where the top EPA administration advocated and the federal and state air programme offices opposed the policy (Meidinger 1985). The policy originated from the Office of Policy Planning and Evaluation (OPPE), a new EPA division cutting across all programme offices. OPPE was charged with evaluation and improvement of existing programmes. Unlike the programme departments, e.g. the Air Programme Office, OPPE had direct access to the EPA administrator and thereby more

---

[28] In 1979, however, the Manufacturing Chemists Association (MCA) filed a suit against EPA over the revised offset policy. It argued that the revised policy amounted to a substantiative "policy revision" which required notice and comment procedures of the Administrative Procedure Act (5 U.S.C.A. Sect. 553(b)(A)) be followed.

leverage over key policy decisions. Similar reorganizations took place in state environment departments. The tasks performed in the evaluation and planning offices necessitated the hiring of a new brand of public servants that differed from the law-oriented administrators in the programme offices. Meidinger (1985: 467) describes them as "typically young (under 40), policy rather than technically oriented, ambitious, ... entrepreneurs (who) effectively forced the discussion of market mechanisms." Many of these professionals were economists who saw the prevailing political science doctrine of interest group competition and pluralism best realized in a market-like regulatory scheme.

An emission tax with a redistributive mechanism may have achieved the same objective as the offset policy. Indeed, then Chairman of the Council of Economic Advisors Charles clearly favoured pollution taxes over emission trading (see Kneese and Schultze 1975; Schultze 1977). But EPA did not have the power to institute a taxation scheme, and it was unlikely that the Carter administration would have tried to push Congress to do it. In the early 1970s, several initiatives for effluent and air emission charges were defeated in Congress at the earliest stages.

OPPE's offset policy prevailed for three reasons: (1) the gridlock created by the Clean Air Act had to be dealt with and emission trading offered a theoretically attractive, and legally and politically feasible solution; and (2) the policy was pushed by professionals who could support it with rational argument, and who utilized their position of a relatively autonomous department office with direct access to top decision makers (Meidinger 1985).

1.2.4. Policy effectiveness

Offset credits clearly were TPRs. The market however was rather small. Transactions only took place between one seller and one buyer at a time. There was no competitive bidding for pollution offsets, because there was no institutional framework that could have facilitated such transactions. In addition, EPA's criteria for eligible buyers restricted the 'freedom of the market place.'

Due to the lack of comprehensive data it is impossible to evaluate the performance of the offset policy in its early years, and opinions differ widely. Crandall (1983: 97) states that by the end of 1980 only 32 offset transactions had taken place. Hahn and Hester (1989: 119) estimate that between 1977 and 1980 some 1500 polluters used offset transactions. Offset transactions seem to have dramatically increased after EPA augmented the offset policy with banking provisions, and related policies, such as netting and bubbling, became established, court-sanctioned policies. Dudek and Palmisano (1988) place the number of offset transactions between 1977 and 1987 at around 2500, 90 per cent of which were within the same company. They also argue that offset credit transactions resolved the conflict between economic growth and environmental quality standards in nonattainment areas.

1.2.5. Case analysis

By 1976, emission trading, the alleged solution to inefficient and growth-obstructing environmental regulation, had been discussed in the policy community for about 10 years. But policy makers did not have a lot of experience applying the instrument. In fact, the test case 'netting' had been a failure. Yet, policy designers in EPA's OPPE were convinced that emission trading was a good instrument; they waited and attached their solution to, literally, the next regulatory challenge.

The gridlock created by the 1970 CAA Amendments, and the apparent conflict between economic growth and environmental protection was *the* opportunity. Politically, the choice of offsets (a crude form of TPRs) was facilitated by three factors. Firstly, environmental groups were slightly weakened as the economic climate did not support a big fight 'against economic growth and jobs' in the second half of the 1970s (Hoberg 1992). Secondly, the offset policy weakened the business community as it only applied to major modified and new sources but subjected them to more stringent standards than applied to existing sources. Thus, industry was unlikely to mobilize through its peak organizations. And thirdly, the White House would not have supported pollution taxes, thus leaving few courses of action.

In the absence of a suitable institutional forum to arrive at a legitimate policy decision, EPA chose to extensively consult all stakeholders. Cognizant of the fact that another court ruling striking down an emission trading approach would spell doom on the future application of that favoured instrument, EPA explicitly recognized the core values of the stakeholders in the policy. The policy acknowledged standards as the baseline. Indeed, the insistence that only companies in compliance with all standards and regulations could engage in offset transactions enforced environmentalists' creed that pollution beyond the standard was illegal and immoral. Growth industries, on the other hand, were awarded with greater flexibility to site in nonattainment areas. Yet, market entry barriers created by standards remained in place.

Overall, the offset policy displayed distributional characteristics as described by Lowi (1964) and Wilson (1984). It allocated discrete benefits to diffuse and unorganized groups (here a subset of industry), without challenging environmental groups' achievements.

1.3. SO_2 trading under the 1990 Clean Air Act Amendments

In the 1990 amendments to the Clean Air Act (CAA), for the first time, the U.S. legislature chose emission trading as a policy instrument to solve a particular environmental problem. Why it did so, and how the instrument fared in the congressional bargaining process, gives important indications for the conditions of instrument acceptance, particularly how it fits into Congressional institutional procedures.

1.3.1. Political context

The 1970 CAA was up for reauthorization in 1982 but the persisting opposition of President Reagan and his congressional followers stalled any action until 1988. Congressional initiatives to revise and strengthen the Act were blocked by strategic committee chairpersons. Senate Majority Leader Robert C. Byrd (D-W.Vir.) prevented bills from going to the floor in fear that acid rain controls would hurt coal mines in his state. Likewise, House Energy and Commerce Committee Chairman John D. Dingell (D-Mich.) obstructed legislative progress to protect his state's auto industry from tougher tailpipe standards. The lack of legislative progress and the continued deterioation of air quality around the country forced EPA in 1989 to announce "draconian anti-industrial measures" against a number of cities lagging in their clean up efforts (The Economist, June 17, 1989: 29).

A group of nine democrats tried to break the gridlock with a compromise plan which excluded the controversial acid rain issue (CQA 1988: 144). Instead, Governors Mario M. Cuomo of New York and Richard F. Celeste of Ohio took up the acid rain issue. They carried a state-sponsored compromise plan on sulphur emission reductions to the House, but received only lukewarm acknowledgement and no reaction (CQA 1988: 144).

1989 was a year of change for air pollution legislation. The change of guard in the White House opened some room for new policy initiatives. George Bush was quite receptive to Canada's pressure to combat acid rain (NYT / Dowd 1989) because, domestically, he wanted to jump on the green bandwagon and establish himself as the 'Environmental President' (The Economist, June 17, 1989). During a state visit to Canada, Bush announced on February 10, 1989 that he would present a package of Clean Air Act Amendments within a few months (NYT / Dowd 1989). Changes in the Senate floor leadership made legislative action feasible: Senator Mitchell (D-Maine) replaced by Senator Byrd (D-Vir) as Senator floor leader. Mitchell's homestate Maine, one of the worst affected by acid rain, had little to lose from stringent Clean Air Act Amendments.

President Bush took advantage of this situation and presented a comprehensive clean air package to Congress on June 12, 1989 (ER/CD June 16, 1989). According to Secretary of State James Baker III the problem of the Clean Air Act Amendments was not so much the clean-up itself but "what has to be decided primarily is how much, what is the target for reduction, and who is going to pay the price ... [the decision] could affect different regions in different ways" (NYT / Dowd 1989: A3).

The White House proposal incorporated many features of the *Project 88*, a public policy study sponsored by Senators Timothy E. Wirth (D-Col.) and John Heinz (D-Pen.) that had brought together a wide range of policy analysts from government, interest groups, and academia. *Project 88* contained proposals on how market forces could be harnessed to solve the U.S. environmental problems. Among others, the Bush proposal included a system of tradable pollution allowances as suggested in the *Project 88*. Bush vehemently opposed any cost-sharing arrangements to soothe interregional conflicts over implementation and pollution control costs. The White House felt very confident that an emission trading scheme would perform very well in Congressional negotiations. In the words of one White House staffer emission allowances "were better than even money" to break the resistance of interest groups and Congresspersons (NYT / Possel 1989). The adoption of TPRs did indeed help to overcome Congress' divisions over acid rain and cost sharing, and facilitated the building of coalitions that were crucial for legislative progress. The 1990 Clean Air Act Amendments passed Congress on October 27 and President Bush signed them into law on November 15.

1.3.2. Policy content

The Clean Air Act Amendments (Public Law 1011-549 (1990)) provided for a phased drawdown of sulphur dioxide emissions by 10 million tons from 1985 levels in the 48 contiguous states and the District of Columbia. Phase I (Jan. 1, 1995 - Dec. 31, 1999) mandated standard-based controls on utility plants emitting at a rate above 2.5 pounds per million lbs./mmBtu (Section 7404). Utilities could avail of a 2 year extension if they installed scrubbers that removed at least 90% of sulphur emission. Phase II starts on January 1, 2000 and places a national emission cap of 8.9 million tons nationwide (Section 7405). In addition, plants have to reduce emissions to 1.2 lbs/mmBtu (with exemptions for small units). Plants were given substantial discretion on how to achieve these standards through the introduction of an allowance system (one allowance equals one ton of SO_2 emission per year). These allowances could be traded with any other SO_2 emitting source in the country. Plants burning high sulphur coal could earn up to a total of 3.5 million incentive allowances during phase one by reducing emissions below 1.2 lbs/mmBtu. Interregional differences were taken into account by granting 200,000 additional SO_2 allowances to plants in Illinois, Indiana, and Ohio during Phase I, and an additional 50,000 allowances to 10 Midwestern plants during Phase II (Section 7404(a)). The Act prohibited interpollutant trading (e.g. SO_2 against NO_x).

The Act required the EPA administrator to withhold 2.8 per cent of all yearly allowances for a Special Allowance Reserve (SAR) which would accommodate the needs of new sources. The administrator has to offer up to 50,000 allowances per year at a fixed price of US$1,500. In addition, the administrator has to organize yearly auctions of up to 150,000 allowances between 1995 and 1999, and 250,000 allowances after the year 2000 (Section 7416).

The Amendments also included several clauses that would allow the imposition of emission charges. However, all these provisions merely delegate the authority to do so to the states.[29] Only one provision could be interpreted as authorizing EPA to impose charges or taxes. Section 183(e)(4) authorizes EPA to use various systems of regulation to reduce pollution from consumer and commercial products, including "economic incentives (including marketable permits and auctions of emission rights)."[30]

1.3.3. Patterns of conflict and conflict resolution

During the bargaining over the 1990 Clean Air Act Amendments acid rain was the most contentious issue due to the difficulty of building majority coalitions to support SO_2 reductions. Actors split over the issue according to regional interest (East vs. Midwest vs. West), political ideology (for and against cost sharing; for and against tradable allowances), and, to a lesser extent, party allegiance. Initially, industry groups and environmentalists endorsed the Bush initiative (ER/CD June 16, 1989: 427; June 23, 1989: 468). The White House capitalized on these by initiating a discussion meeting with environmental and industry groups on the proposed Act on August 2. On that occasion, Environmental Defense Fund Executive Director Fred Krupp lauded President Bush for his "strong action ... on acid rain" and said he would lobby for it (ER/CD Aug. 4, 1989: 625). Other environmental groups, such as the National Resource Defense

[29] See Sections 118(a) (state implementation of EPA emission standards), 182(g) (federal facilities and agencies have to pay emission charges to states, where applicable), 183(e)(4) and 185(a),(b) (state implementation and enforcement of ambient air quality targets), 502(b)(3) (fees to cover administration expenses of the states).

[30] Westin and Gains (1991: 219) describe the evolution of the legal text.

> The original bill provided, non-inclusively, that fees and other incentives could be used. The House participants in the House - Senate conference attempted to strike the fee power altogether. What they got was a stand-off; the reference to fees was struck, but the House was rebuffed in its efforts to make the exclusion of fees explicit, with ambiguous legal results because implicitly fees are still an economic incentive.

Council (NDRC), did not endorse the trading provisions but simply stated that the sulphur reductions of 10 million tons envisioned by Bush were acceptable (NYT / Possel 1989). Low sulphur coal producers and miners, and railroads[31] supported strong acid rain regulation. But mining and industrial interests in the Eastern states rejected the Bush plan (ER/CD Aug. 4, 1989: 625).

In Congress, battlelines were redrawn as committee chairpersons repackaged the Act, and various members attempted to build coalitions around their pet issues. Senate negotiations stalled in January 1990. After weeks of backroom negotiations between the Senate factions under White House auspices (some) conflicts were resolved in March, without substantially changing the SO_2 provisions (CQA 1990: 232).[32] In the House, representatives accepted that some sort of allowance trading scheme with an emission cap had to be part of the Act to obtain consent from the Senate and President Bush. Bush also consistently threatened to veto any legislation with costs exceeding his own proposal by more than 10 per cent. And the lawmakers were keenly aware that any substantial deviation from the trading scheme would cost money, and diminish the chances to bring a pork barrel to their district.

Low sulphur coal burning clean states, and states already using effective scrubbers, adamantly opposed financing the dirty states' clean-up. On the other hand, dirty states simply wanted to preserve jobs in the coal mining industry and keep electricity rates competitive (NYT / Shabecoff 1990a). The allowance trading system based on grandfathered permits seemed best suited to resolve the conflict as it acknowledged rights to existing emissions, and enabled least cost solutions in the face of future emission reductions across the country. But clean states and states with projected utility expansion would have had to rely on allowance sales from dirty Midwestern states. It was feared that these would exploit their monopoly power and charge exorbitant prices. The Midwestern representatives argued that under a cost-sharing programme

---

[31] Railroads would have benefitted from transporting low sulphur coal to power plants using high sulphur coal.

[32] The emission cap of 10 million tons proposed by Bush remained untouched. Negotiations and compromises centered on the timetable for the emission reductions (NYT / Shabecoff 1990b).

these problems would not exist, and pushed for a cost-sharing scheme financed through a national utility tax.

In November 1989, Representative Dingell proposed a sulphur dioxide fee because the interregional conflict had "the potential to turn close friends into lasting enemies and divide this committee on many issues for many years to come" (NYT / Gold 1989: A21). However, the fee proposal proved to be a non-starter. George Bush had committed himself to 'no new taxes' and threatened to veto any taxes in disguise. Thus, the cost sharing initiative could not find enough supporters and was shelved (CQA 1990: 238).[33] Finally, after a marathon negotiation session, the Midwesterners, led by Philipp R. Sharp (D-Ind.), abandoned their cost sharing demand in March 1990, and settled for a manipulation of the allowance system that gave all states more pollution allowances (CQA 1990: 242).

The final House negotiations over SO_2 emission allowances saw a replay during the House - Senate conference in October. The conference dealt with the acid rain issue only at the very end because compromise was elusive: cost-sharing was on the agenda again. This meant that there were no other issues left to include in a give-and-take bargain. As before, the manipulation of the allowance scheme facilitated compromise: Conferees granted 200,000 more allowances to plants in Illinois, Indiana, and Ohio per year until the year 2000. Thereafter, 10 Midwestern states would get 50,000 allowances each year (CQA 1990: 278). In return, no cost-sharing programme was incorporated, and a provision allowing state governors to veto the import of out-of-state coal was dropped. Environmental groups later complained about the allocation of additional SO_2 allowances. But they had little choice than to complain after the fact: During both marathon negotiation sessions members of Congress secluded themselves from the public. Interest groups were left out on the hallways and had little, if any, influence on the manipulation of the allowance scheme.

According to the final statutes, sulphur dioxide emission will not exceed 8.9 million tons by the year 2000. Although this cap remained untouched, emission allowances during the phases

---

[33] The National Clean Air Coalition, an umbrella organization of environmental groups, chose to stay out of the haggling over cost-sharing (CQA 1990: 239).

leading up to and following the symbolic year 2000 will be above this level. Due to the inclusion of emission banking, bonus allowances for early installment of scrubbers, regular auctions of spare allowances, extra allowances for particular regions of the country, and various timetables to retire circulating allowances, the total amount of emissions will be closer to 12 - 14 million tons (NYT / Schneider 1990).[34]

The 1990 CAA Amendments passed the House in a 401-25 vote, with 9 dissenters coming from Midwestern states. Senate passed the bill 89-10, with 7 dissenters coming from Eastern and Midwestern coal mining states.

The provisions for auctioning emission allowances affirmed Congress' position that air is a resource corporations can own, and dispose of at their liking. What is more remarkable is how the features of emission trading helped to negotiate agreement in Congress. At two points, the final House negotiations and at the House - Senate conference, the number of emission allowances was increased to satisfy the stakeholders. The instrument emission trading proved to be extremely responsive to redistributive concerns in the policy process. It allowed that additional tangible benefits could be allocated in, literally, marginal increments with 'no one' having to pay for it.

1.3.4. Policy effectiveness

Considering that the legislation is barely two years old, and that phase one of the SO_2 reduction programme will only start in 1995, nothing can be said about policy effectiveness at this point in time.

1.3.5. Case analysis

The choice and success of TPRs in resolving the conflicts over acid rain was facilitated by a number of favorable conditions. As in the case of emission offsets, command and control instruments had failed to achieve the air quality targets set by previous legislation. A weak

---

[34] And the number could easily further increase if EPA and Congress decide that interpollutant trading between NO_x and SO_2 is good for the market and without appreciable consequence for the environment.

economy also contributed to a new openness among policy makers to experiment with new approaches. Most important, however, appeared to be Congressional initiatives to study the application of market-oriented policy instruments in a non-partisan manner. Some prominent environmental groups, especially the Environmental Defense Fund (EDF), also rode the wave of instrument reform. The EDF even cooperated with White House staff in writing Bush's Clean Air Act proposal (Hahn and Stavins 1991). Lastly, policy designers could build on valuable experiences gained through the emission trading programmes for the phase-out of lead in gasoline, and the phase-out of chlorofluorocarbons.

George Bush had appealed to anti-tax sentiments when he made the 'no new taxes' pledge ("read my lips") a centrepiece of his presidency. This elevated taxation not only as an ideological and cultural symbol, but also as the symbol of Bush's political survival. The political landscape of the time simply did not have space for PTs. Representative Dingell's cost sharing proposal may have been well-meaning, but, as Kingdon (1984) would say, it failed the survival criterion 'value compatibility.' In addition, any substantial PT proposal would have upset the ongoing legislative process of the Clean Air Act Amendments. Tax measures must be introduced through the Ways and Means Committee, and subsequent deliberations would have involved the Office of Management and Budget (White House), and the Treasury Department. The White House would have objected on ideological and political grounds, and the Treasury Department had long ruled out PTs because it could not administer such schemes without (unlikely) additional allocations to its operations (Westin and Gaines 1991).

Emission trading, on the other hand, performed extremely well in Congress, just as the instrument choosers in the White House had hoped. Pollution allowances are almost infinitesimally divisible, and *additional* TPRs could be allocated to regional markets and individual factories on an incremental basis. This shifted the focus from the redistributive consequences of pollution control to the distribution of tangible benefits to localized constituencies.

The explicit capping of emissions ("cut SO_2 emissions in half by the year 2000") appealed very much to Congress' habit to formulate highly ambitious legislative goals. Nevertheless, the emission trading programme could be manipulated by changing timetables, extra allowances, allowance banking, etc. so that the symbolic emission cap of 8.9 million tons in 2000 was left untouched, but emission allowances in the meantime could well exceed this limit. Thus, legislators were able to tell their constituents that they fought hard for them. In addition, TPRs concealed the future redistribution of wealth (and jobs) that would take place once trading commenced. Legislators would not have to take the blame for adverse redistributive consequences due to diffuse market dynamics.

1.4. The discourse over pollution taxes and emission trading in the U.S.

Ever since the publication of Dales' *Pollution, Property and Prices* in 1968 American scholars and practitioners engaged in a vigorous debate over TPRs. Curiously, the incipient discussion and advocacy of TPRs at first vitalized the debate over PTs which dominated the political discourse in the early 1970s, and academic publications until the mid-1970s. Bohm and Russel (1985: 404) list numerous unsuccessful PT initiatives in Congress, among others,

- in 1970 a proposal for a national tax on lead in gasoline,
- in 1971 a proposal for a national effluent charge on BOD,
- in 1972 a proposal for a tax on SO_2 emissions.

Despite the political failure of PTs, the Congressional Research Service in 1977 published a 869 page "collection of articles, probably the most comprehensive ever published on the subject of incentives, pollution taxes, and other alternatives" (Marcus 1980: 182). In the academic literature, PTs were at least cautiously endorsed by Kneese and Schultze (1975), and Baumol and Oates (1979).

The subsequent discourse was influenced by EPA's implementation of various emission trading schemes starting with netting in 1974, moving to offsets in 1976, bubbles in 1979 (44 FR 71779), and finally a comprehensive trading programme in 1986 (51 FR 43814). While the 'netting' and 'bubble' policies faced numerous legal challenges and delays, offsets and the trading in lead and CFCs were and are fairly successful (Dudek and Palmisano 1988; Hahn and Hester 1989a,b).

These positive experiences coincided with the ideological changes that swept the U.S. during the Reagan and Bush years, soaring costs for environmental protection and a decline in the U.S. international competitiveness (Hahn and Stavins 1991). Subsequently, a number of academics systematically attacked the ideological foundations of standard-based regulation in favour of market based approaches. Dudek and Palmisano (1988: 218) estimate that between 1976 and 1988 more than 100 articles on market-based approaches were published in prestigious journals and magazines.[35] These efforts culminated 1988 in the *Project 88*, a non-partisan policy study sponsored by Congresspersons T.E. Wirth (D-Col.) and J.H. Heinz (D-Pen.) which endorsed emission trading for a variety of pollution problems. While most of the proposals have not translated into actual policies, market-oriented policy instruments are increasingly accepted, or at least not opposed by environmental and industry lobby groups (Hahn and Stavins 1991).

With few exceptions (e.g. Russel 1978), the debate in the U.S. focused on emission trading versus standard-based regulation, and did not play TPRs against PTs. And although PTs do not have the political viability record of TPRs, they have not been totally discredited. Indeed, since about 1990 a number of 'green tax' measures have been and are discussed in Congress, with increasing backing from House Ways and Means Committee Chairperson Dan Rostenkowski (D-Ill.) (BW / Gleckman and Cahan 1990). Some of the tax proposals intend to discourage environmentally destructive behaviour, others are to recoup expenses for administration, or to counteract indirect subsidy programmes which are difficult to change. With the explicit backing of green taxes by some individuals in the incoming Democratic administration one can expect a more vigorous debate over the merits of PTs, and possibly concrete PT policy initiatives. Foresightful academics have already jumped on the bandwagon: Stavins and Whitehead published the first article endorsing PTs in years in September 1992.

Beyond economic theory the American discourse over TPRs and PTs is rooted in the firm belief that social conflicts are best negotiated through the market. This corresponds very much

[35] However, the actual number of advocates appears to be much smaller. A selection of articles by the principal proponents are Ackermann and Stewart 1988; Dudek and Palmisano (1988); Hahn (1989a,b); Hahn and Hester (1989a,b); Portney (1988, 1992), and Stewart (1985).

with American views on the proper extent of state involvement in society (King 1973; Lipset 1989). Americans traditionally have opposed taxation in part because they oppose redistribution, but even more so because they associate taxation with big government, and attacks on individual freedoms (Lipset 1989: Ch. 2). Even proponents of PTs tried to appeal to this sentiment and argued that the instruments' application would, and should, diminish state intervention into individuals' resource allocation and redistribution decisions. In the words of Charles Schultze, once economic advisor to President Carter,

> ... Buyer-seller relationships of the marketplace have substantial advantages as a form of social organization. ... Market-like arrangements not only minimize the need for coercion as a means of organizing society; they also reduce the need for compassion, patriotism, brotherly love, and cultural solidarity as motivating forces behind social improvement (1977: 16, 17).[36]

The question here is whether anti-tax sentiments and opposition to redistributive government policy are intricately linked, or whether the latter could be overcome by moral suasion, zero-revenue tax reforms, or institutional innovations.

2. Germany's 1976 Effluent Fee Law

In 1976, the German *Bundestag* (federal Parliament) passed the *Abwasserabgabengesetz* (henceforth Effluent Charge Law [ECL]) with overwhelming majority: Only seven MPs dissented, and they wanted stricter regulation. The law required that all dischargers pay fees according to volume, chemical composition, and toxicity of effluents. The ECL abruptly broke with a tradition of cost-free licensed discharge which dated back into the 19th century.[37] The ECL marked government's unequivocal assertion of property rights over the environment, and its

---

[36] Schultze reasoned that there are basically three forces organizing human society: coercion, self-interest, and emotional forces (like those listed above). He argued that the population of the U.S. was too heterogeneous to appeal to the emotional forces, and that coercion is against the American ethos; thus, the government should harness private interest to order individuals' interactions (1977: 18).

[37] Komareck (1986) and Peacock et al (1984) give a concise account on the development of water management in Germany, its administrative structures and practices. For a comparison of German practices with those in neighbouring states see Bongaerts and Kraemer (1989).

determination to charge polluters on a continuing basis. Such a significant change would suggest that intense political conflict preceded and accompanied the ECL's formulation and implementation. However, the ECL's smooth sailing in Parliament indicates otherwise. This is all the more surprising as the law came at the height of Germany's most severe post-war recession. Throughout the 1970s industry and trade unions pressed hard on the governing Social Democratic Party (SPD) and Liberal Democrats (FDP) to relax or postpone environmental regulation to facilitate economic recovery (Miller and Miller 1988: 464).

2.1. Framework and policy style of German environmental regulation

Germany is a federal state. The directly elected federal Parliament (*Bundestag*) and the Federal Council (*Bundesrat*), whose members are appointed by state (*Länder*) governments and carry an imperative mandate, are the legislative organs of the Republic. Federal and state governments share the responsibilities for environmental policy. Under Articles 72 and 75 of the Basic Law (*Grundgesetz* [GG]) the federal government can issue framework legislation which states have to augment with their own legislation and implement accordingly. Water policy, for example, falls into this category. Other environmental policy areas, such as air pollution and waste management, fall into the category of "competing legislative authority" (Article 72). In these areas the federal government can issue binding legislation which states have to implement. In the absence of such laws states may pursue their own policies. The federal government has the exclusive authority to levy taxes and duties (Article 105(1)). States only do so if they are empowered by a specific federal law (Article 71). Thus, only the federal government can impose pollution taxes, and like all other federal environmental legislation, such a law would have to pass through both houses of Parliament. Since state governments can block legislation in the *Bundesrat*, the federal government is usually very sensitive to states' concerns. States have the reputation of following the federal government's lead on matters that clearly are within federal jurisdiction. Intergovernmental cooperation with the goal of preserving the economic union and a comparable quality of life throughout the country is among the guiding principles of environmental policy in Germany (Busch 1989: 255; GG, Article 72(3)).

German environmental policy making is characterized by intensive government (federal and state) - industry consultation throughout all phases of legislation and implementation (Brickman et al 1985; Busch 1989; Dyson 1982; Simonis 1992). The establishment of task forces through which government, industry and academia cooperate in writing regulations is a regular feature of the German regulatory process, even under conservative Christian democratic governments. Citizen groups, however, are rarely admitted, especially if they could challenge fundamental premises of government policy, or if it could undermine the bureaucracy's elevated status (Greiffenhagen 1984; Simonis 1992).[38]

2.2. Political context

Concern over surface water pollution mounted after chemical spills and a prolonged drought threatened drinking water supplies along the Rhine river in 1969 (DS 1976b: 62).[39] In 1970, Chancellor Willy Brandt instituted a cabinet task force to coordinate environmental policy development among the various ministries (Reese 1983: 140). A subsequent constitutional amendment in 1972 gave the federal government more power over air, noise, water, and waste policy and thereby laid the groundwork for federal regulatory initiatives. In the wake of the Oil Crisis in the early to mid-1970s, industry and labour unions pressured the government to retreat from its environmental policy plans. In response, the government made environmental policy part of the agenda of the "Concerted Action" conferences where representatives from industry, unions, and all political parties discussed issues of national concern (Reese 1983). The SPD/FDP government also studied and endorsed the utilization of economic incentive mechanisms, hoping they could reconcile environmental protection with economic growth objectives

---

[38] According to Manfred Stolz (1992), a senior policy advisor for waste disposal in the federal environment ministry, "in principle, participation is open to all social interest groups as long as they are there to work towards establishing regulations within the framework set by government. Cooperative processes don't work if you have a bunch of environmental fundamentalists in there."

[39] Water pollution had been a concern in Germany since the industrial revolution. But no decisive actions were taken until the late 1950s when water quality became so bad that it damaged industrial production facilities. The Federation of German Industry (*Bundesverband Deutscher Industrie*) lobbied for the adoption of a national permitting system in the 1957 Water Management Act (*Wasserhaushaltsgesetz*). In 1957, industry also supported a constitutional amendment to bring water management under federal control. However, the initiative failed in Parliament over the resistance of the *Länder* (Peacock et al (1984: 73).

(Miller and Miller 1988). Pollution taxes and marketable permits were among the instruments considered. But leading economists and law scholars at the time questioned the viability of TPRs to rapidly improve environmental quality. PTs in conjunction with standards were deemed to be more effective. More important, there was a general distrust that markets could allocate pollution permits in environmentally and politically acceptable ways (Cansier 1975; Rehbinder 1973).

2.3. Policy content

In 1973, internal documents of the Interior Ministry suggested that a fee equivalent to the average marginal abatement cost of an estimated DM 80 should be imposed (see DS 1976b: 64).[40] This clearly demonstrates that the policy makers who chose the policy instrument did so on the basis of the economic principles of PTs.

While the instrument survived the early stages of debate, it was clear that the fee had to be reduced. A first public draft of the ECL in 1974 proposed the imposition of a DM 25 fee per damage unit of discharge in 1976, which would then rise to DM 40 in 1980. However, paragraph 9 of the final law[41] only provided for a fee of DM 12 beginning January 1, 1981. The fee would be raised to DM 40 in 1986, and would reach DM 90 only in 1997. Fees are calculated based on the expected discharges of suspended solids, chemical oxygen demand (COD), cadmium, mercury, and fish toxicity.[42] Each substance is assigned a different unit value (e.g. mercury 20 g). Total discharge of the substance divided by the unit value multiplied by the fee results in the total fees payable for any one substance. Generally, discharge levels indicated in the permits are used to calculate the fees. However, polluters may submit certified monitoring data to base fee calculation on actual discharges (section 4). The fee applies to all direct discharges,

---

[40] DM 80 were approximately US $32 in 1980.

[41] The ECL was proclaimed on September 13, 1976 (BGBl. I, p. 2721). Since, it has been amended twice, first on December 19, 1986 (BGBl. I, p. 2619), and then on November 6, 1990 (BGBl. I, p. 2432). While the discussion will focus on the 1976 version, important points of the amendments will be mentioned.

[42] In 1986, the following categories were added: Chemical oxygen demand (COD), organic halogen compounds, chromium, nickel, and copper. The 1990 amendment expanded the list again by adding phosphorus and nitrogen. The initial omission of these substances was criticized by scientists, and even government advisers in 1976 (DS 1976b). The catalogue of chargeable effluents is expected to be expanded further as costs for sampling and analysis decline (Kloepper 1989: 660).

regardless of the quality or assimilative capacity of the receiving environment (BTD 7/2272: 27). The levy also applies to discharges from precipitation run-off. The fee also applies to all direct discharges from private households and municipal entities (paragraph 8).

Companies that are hit hard by the discharge fee can apply for a partial or full waiver under paragraph 10. Dischargers qualify for a discount on their fee assessment of up to 75 per cent for the portion of the pollution load which cannot be controlled although best available technology (BAT) has been installed. The discount is reduced to 20 per cent over eight years as new technology presumably would be available (section 9(5)). The discount system is restarted each time new BAT-based regulation for maximum discharge concentration come into effect. To further encourage pollution control investments the law allows companies to claim deductions for construction costs from their effluent charge bills up to three years in advance of actual operation. Alternatively, a refund can be claimed if fees have already been paid (section 10).

The ECL prescribes that revenues are to be used to finance the construction or maintenance of treatment plants, collection or distribution water works that facilitate the communal use of treatment plants, sewage sludge treatment plants, research, and water technology development (paragraph 13). However, states are authorized to cover their expenses of administering the law by retaining the appropriate amount.

The language accompanying the ECL emphasizes that the payment of effluent fees does not constitute an authorization to discharge (BTD 7/2272: 21-22). The state maintains full property rights over the waterways, and the levy underscores rather then diminishes this authority. Despite the charge payment polluters have to observe the effluent standards imposed by the Federal Water Act (FWA). After the enactment of the ECL, the government instituted industry-specific working groups to design a system of stringent technical standards. Thus, industry was not only subject to a charge but also obliged to limit maximum pollution loads in their discharges. On the other hand, the participation in standard setting gave industry some influence on determining its own average fee payments as section 9(5) of the ECL allows for discounts for pollution loads after treatment according to the standard.

The purpose of the ECL was to give an incentive to polluters to reduce discharges. The ECL shifted the burden of financing water pollution control away from the taxpayer and placed it (at least in part) on the polluters. Further, the ECL did not intend to redistribute wealth only among those who were obliged to pay, nor did it limit the circle of recipients of funding from the revenues to charge payers. Thus, the German law literature has come to classify the ECL as a 'special charge with guidance and redistribution function'.[43]

2.4. Patterns of conflict and conflict resolution

Leading economists and the independent Council of Experts on the Environment supported the effluent law, and did not foresee any negative effects on the economic development of the country.[44] The opposition Christian Democratic Party (CDU) and Christian Social Union (CSU) did not oppose the ECL. Public water utilities, the prime beneficiaries of potential tax reinvestment into waterworks, supported the introduction of the pollution tax. Environmental groups did not take a high profile in the debate as they had not yet developed strong peak associations. Local environmental citizens groups (*Bürgerinitiativen*), at the time, were more concerned about nuclear power plants, nuclear waste, and deployment of U.S. nuclear missiles than about water quality and the ECL (DS 1976a; Simonis 1992). Industry lobbies, however, made representations to Chancellor Helmut Schmidt and argued that the pollution tax was economically unacceptable. Industry emphasized that the pollution tax *per se* was wrong and against established practices of pollution management in Germany. During the last meeting of the Concerted Action in 1975 government and industry agreed in principle that environmental protection measures should not be delayed even if they conflicted with economic objectives (Reese 1983: 141). Nevertheless, industry lobbied against the ECL, particularly with local and state governments, because to pass in the German upper house, the law needed the support of the state representatives.

---

[43] For further discussion of this concept and judgements by the German Constitutional Court see Schroeder (1983).

[44] Brown and Johnson (1984: 942) cite the Council of Experts on the Environment as stating that the charge would make water pollution control about one-third cheaper than a uniform technology standard.

The power of industry lobbying was constrained by the fact that industry organizations also included medium and small companies.[45] More than 90 per cent of all industrial facilities discharged their waste water through public sewerage systems. Technical standards under the Water Act would have required them to achieve substantial, and expensive, pollution reduction at each plant if communal treatment plants could not be upgraded. This upgrading would be, in part, financed from revenues of the ECL. In addition, the indirect dischargers felt that they had substantial bargaining power with local water authorities if these wanted to pass on costs resulting from the ECL's implementation (Peacock et al 1984: 147). Thus, industry changed its attitude and focused its campaign on a reduction of the discharge tax, a postponement of implementation, and exemption clauses. Brown and Johnson (1984) do not mention this split of interests among industry lobbyists. They state that industry changed its position after substantial political support for the pollution tax had built up. From their interviews with German bureaucrats they conclude that

> there is a feeling in the FRG that once a consensus on the need for legislation is achieved the various parties are more inclined to work in cooperation towards the common goal in contrast to the United States where a more adversarial philosophy seems to operate (Brown and Johnson 1984: 932).

Labour unions initially supported the industry line. Given the severity of the economic crisis concern over jobs was paramount. On the other hand, the prospect of substantial investments into water works soothed union opposition to the ECL. Considering the long time horizon of the law (until the year 2000), and the possibility that more stringent effluent standards would necessitate future upgrading of water treatment facilities, made the ECL an attractive long-term employment programme for the construction industry.

The law that passed Parliament in 1976 was, literally, a piece of framework legislation. It lacked technical specifications; standards, sampling and analytical methods referred to in the text or accompanying legal commentary. To remedy this deficiency, the Interior Ministry estab-

---

[45] The terms small and medium should not be interpreted literally. They refer to industries with a low discharge level of pollutants which would incur substantial costs in constructing appropriate treatment plants. While discharge level and profits may be correlated, this may not always be the case.

lished some 60 working groups composed of representatives from the state governments, industry, universities, and laboratories to build a consensus around proper implementation norms for the ECL. Brown and Johnson (1984: 940) observe that these task forces were instrumental for the federal government to move the legislation through the upper house. In voting for the ECL, the states gave up substantial authority over water management without exactly knowing the technical standards, implementation costs, and overall economic impact of the law. Participation and cooperation through the task forces assured states and industry of substantial influence in writing the detailed implementation regulation.

In summary, the federal government prevailed in introducing the ECL, the first tax on environmental pollution in Germany, despite initial opposition from industry and, to a lesser extent, the states. A central element of the policy process was the movement of the issue into institutional settings which were designed to facilitate compromise and cooperation. Where such institutions did not exist they were deliberately created. By doing so the federal government accommodated industry pressures, as well as state concerns stemming from the jurisdictional fragmentation of Germany. While numerous details in the ECL were adapted after negotiations, the government did not reverse its policy instrument choice.

2.5. Policy effectiveness

No studies have been conducted to assess the precise impact of the ECL on the improvement of water quality in Germany. Generally, water quality did improve over the past two decades, but to what extent this was due to the ECL is uncertain. Revenues from effluent charges were estimated DM 800 - 900 million during the first two years of implementation (Komareck 1988: 13).[46] Brown and Johnson (1984), as well as Reese (1983) state that the introduction of the fee accelerated the planning and construction of private and municipal waste water treatment plants. Between 1975 and 1979 manufacturing industry investments into water

---

[46] Detailed revenue data are not available for two reasons. First, effluent charges are collected by state authorities, no aggregate statistics are available. Secondly, accounting practices often do not permit the separation of effluent charges from other charges collected by the water control authorities. It is not possible to say what percentage of the revenues were reinvested, and how charge paying polluters may have benefitted from reinvestments.

pollution control declined by an average of 7.9 per cent a year (constant 1980 prices). During the same period government investments into water pollution control rose about 4.3 per cent a year. After the introduction of the ECL, industry investments jumped almost 20 per cent between 1980 and 1982. Between 1980 and 1984 they grew an average of 2.8 per cent per year. Government expenditures into water pollution, however, declined by an average of 9.2 per cent between 1980 and 1984 (Leipert and Simonis 1989). The data clearly show that the ECL had the desired effect of shifting the burden of pollution control investment from the public to the polluters. The law also induced a number of cooperative arrangements in waste water management between large companies and surrounding communities along the Rhine river.[47]

Hahn (1989a,b) and the OECD (1989) describe the ECL as one of the most effective PT programmes in practice.

2.6. Case analysis

The ECL was a response to the rapidly deteriorating water quality of Germany's major waterways, as well increasing impacts of industrial pollution on basic human needs (i.e. drinking water). The instrument choice was a result of studies on 'economic instruments' commissioned by the Interior Ministry in the early 1970s to find ways to reconcile economic growth with environmental protection. To this end, environmental protection measures had to be made more efficient.

Pollution taxation proved to be the instrument most adaptable to a variety of pressures. Tax rates could be set federally, but the law could be administered locally. This helped to avoid federal - state conflicts because it ensured that no transfers between state jurisdictions would happen. It also separated the decision over fee levels from reinvestment decisions, and thus soothed industry resistance. Industry had more bargaining power at the local level over enforcement of the fee, it could better press for subsidies, and could constrain treatment plants' efforts to pass on the costs to indirect dischargers (Peacock 1984).

[47] Among the more remarkable of such cooperations was the construction of a large treatment plant at the BASF which also received the discharges of the 300,000 residents of nearby Ludwigshafen.

Once basic agreement from the stakeholders was secured, tax levels, phase-in schedules, and exemption clauses were negotiated with industry and state representatives in the established mode of cooperative policy formulation.

The conduct of these negotiations moved through numerous institutional settings, in part in strict observance of the law, but also to involve as many actors as possible to deflect blame if the 'new' policy instrument would not perform to expectation. Initially, consultations were held in the "Concerted Action" conferences, then in legislative committees in both Houses of Parliament. Later, the writing of implementation guidelines and auxiliary operational standards were referred to special committees made up of federal, state, and industry representatives, as well as independent scientists.

In summary, the federal government cautiously avoided undue confrontation with either industry or state governments. In addition, the PT was not set to reach a well defined water quality goal. Thus, if any controversy arose later, the government could blame it on the poor work of the technical committees, or improper implementation by state authorities.

2.7. The German discourse over pollution taxation and emission trading

As discussed in section 2.1., there are no constitutional obstacles to the imposition of environmental taxes by the federal government. Nevertheless, taxes or special duties like the ECL have been rather exceptional choices in German environmental policy. Standards have been and are the instrument of choice, and their implementation is often facilitated through subsidy schemes at the federal and state levels (OECD 1989; Zezschwitz 1991). Aside from waste water effluent charges, noise charges collected with landing fees at airports, and a tax surcharge on leaded gasoline are the only taxes with incentive character implemented to date (Zezschwitz 1991). By 1993, car taxes will be restructured to take carbon dioxide, exhaust volume, and noise emissions into account (The Economist, March, 17, 1990: 47). Similarly, industrial carbon dioxide emission will be taxed at a rate of about DM 10 (approximately CDN 8.50) per ton (NS / McKenzie 1990).

The advent of the Green Party in the *Bundestag* in 1982 fueled a more vigorous debate over environmental taxation and tax reform in general. The general theme of the debate is that pollution should be made so expensive that polluters change their behaviour. Since the mid-1980s all German parties have included some form of pollution taxation in their party programmes. So far, the most detailed proposals have been made by the SPD in 1989. The Social Democrats published a comprehensive programme to reform the whole tax system by reducing income and corporate taxes and substituting them with a wide range of taxes on environmentally harmful activities (Zezschwitz 1991: 101).

German environmental organizations and even the government have advocated environmental taxes on air emissions, and 'dirty technologies' on a European-wide scale (NS / McKenzie 1990; Weizsäcker 1989).

Emission trading has never been seriously discussed in Germany. To date, the Federal Environment Administration (*Umweltbundesamt*) commissioned only one major study on the *Possibilities and Limits to Transfer New Concepts in U.S. Clean Air Policy Into German Environmental Policy* (Rehbinder and Sprenger 1985).[48] The key findings of the study are that policy inconsistencies across the *Länder* prevent the expansion of the existing pollution compensation provisions in the Federal Emission Control Act[49] to full-scale emission trading. First of all, there are constitutional barriers. Federal legislative powers are limited and subject to *Länder* consent as described in section 2.1. Federal environment legislation is also implemented and enforced by *Länder* authorities. While Section 47 of the *BImSchG* mandates that states formulate air management plans, the federal government has no authority to prescribe what should be contained in such plans and what states should do with it (Rehbinder and Sprenger 1985: 281, 346). Thus, aside from emission standards there are no applicable federal ambient air pollution limitations. Instead, the federal government tried to ensure a nationally even level of air pollution by insisting that old pollution sources receive no special treatment in conforming with updated

---

[48] The original title is *Möglichkeiten und Grenzen der Übertragbarkeit neuerer Konzepte der US-amerikanischen Luftreinhaltepolitik in den Bereich der deutschen Umweltpolitik.*

[49] *Bundesimmissionsschutzgesetz* (BImSchG). For the latest version see BGBl. I, May 14, 1990, p. 880, and BGBl. II, September 23, 1990, p. 885.

technology standards, if need be with the aid of subsidies. Thus, theoretically, there should be no pollution credits available (although in reality limits are not effectively enforced on old plants) (ibid. 344).

The compensation clause of Section 7a *BImSchG* is reminiscent of the U.S. offset credit, however much more restrictive. Compensations only apply to old facilities and cannot be used to site major new facilities in nonattainment areas designated by state authorities.[50] So far the courts have prevented the expansion of compensations to netting or bubble transactions by very narrow interpretations of what constitutes a new facility. For example, the Federal Administrative Court ruled in 1973 that substituting an essential part of the production unit equals its new construction (ibid. 300).[51]

Rehbinder and Sprenger find that the decentralized nature of the German administrative system would prevent any coordinated federal emission trading programme. Implementation and enforcement differences across states would be unavoidable, and these may "enforce existing regional imbalances in air pollution control" (ibid. 349). *Länder* governments are prevented from implementing state-wide trading programmes without a federal law authorizing it. However, the 1990 *BImSchG* Amendments that liberalized the use of compensations may be used to justify a state trading initiative.[52] Yet, such an initiative could run into difficulties as could a federal trading programme. Conceivably, a state trading programme could result in substantial regional environmental or economic imbalances. However, Articles 71 and 72 of the *Grundgesetz* implies that *Länder* governments ought not to pursue policies with such consequences. Thus, emission trading in the *Länder* may be challenged in federal courts, or it may even trigger federal legislative action under *Grundgesetz* Article 72(3). Given limited federal administrative capacity in pollution control federal action may not remedy the situation. Instead it would put

[50] Minor new facilities that worsen air pollution by less than 1 per cent over no longer than three years are acceptable even without immediate compensations (Rehbinder and Sprenger 1985: 294).

[51] In the cited case a brick factory changed its furnaces without a permit for building a new brick factory, and was punished accordingly.

[52] The law now allows compensations for similar (instead of identical) pollutants from facilities which need not have an immediate spatial and temporal relation as long as overall air quality improves (Sections 7(2), (3)).

constitutional reform and redistribution of powers on the agenda, perhaps too big a price for modest monetary savings.

In sum, the German discourse over PTs and TPRs is only partially developed. While PTs are recognized as potent instrument and various proposals have been advanced, emission trading has not been and is not extensively discussed. PTs appear to be more adaptable to the federal structures than emission trading. It is also notable that the discourse appears to be dominated by political parties and government institutions. This may explain the low profile of the politically much more sensitive instrument emission trading.

3. British Columbia's 1990 Waste Discharge Regulations

In July 1992, the governing New Democratic Party (NDP) reformed British Columbia's Waste Management Permit Fee Regulation (WMPFR). The WMPFR superseded earlier regulations that based fees on type of discharge permit (air emission, effluent, or solid waste), industry group, and yearly production capacity (rather than quantity and quality of the discharge associated with production). This system was deemed inequitable, and lacking incentives to reduce pollution. The reform intended to enshrine the polluter pay principle in the province's waste regulations by levying discharge fees in proportion to volume and environmental impact of the discharge on the roughly 3400 permit holders (B.C.MoE 1992e). This action coincided with severe economic problems in several key B.C. industry sectors (e.g. pulp and paper, mining).

3.1. Framework and policy style of B.C. environmental regulation

In Canada, federal and provincial governments share environmental policy making authority. Section 109 of the 1867 British North America Act grants the provinces jurisdiction over natural resources. In addition, Section 92(13) empowers the provinces to legislate regarding property and civil rights in their jurisdictions. Provincial policy making powers over air, water, and waste management are derived from these constitutional clauses. However, federal authority, derived from its prerogative over criminal law, trade and commerce, and generally

"Peace, Order, and good Government", overlaps with provincial jurisdictions (Dwiwedi and Woodrow 1989). Although the federal government accumulated substantial powers through the 1970 Canada Water Act, the 1970 Fisheries Act Amendments, the 1971 Clean Air Act, and the 1987 Canada Environmental Protection Act (CEPA), it rarely used these powers to challenge provincial governments (Harrison 1991; also see Mcdonald 1991). These customarily oppose federal regulatory initiatives because they see them as interference with their governmental and economic affairs (Albert 1992; Schrecker 1990).

In B.C., the Ministry of Environment is in charge of pollution control, but in several areas it competes with the resource ministries (e.g. Forestry) for jurisdiction and influence. As at the federal level, environmental policy making in B.C. has been characterized by intensive government - industry bargaining (Dorcey 1986, 1987; J.R. Wilson 1990). More recently, this bipartite bargaining has been widened to multistakeholder consultations (Dorcey 1987; Hoberg 1991b).

Substantial administrative discretion pervades all pieces of provincial environmental protection legislation. Statues like the Fisheries Act, the Environmental Protection Act, or the Waste Management Act, and pertinent regulations, usually lack technical detail. Instead, ministerial officials are awarded the authority to set permit specifications, and to judge over necessary enforcement actions (Dorcey 1986: Ch. 5).

3.2. Political context

The November 1991 elections propelled the New Democratic Party (NDP) under Michael Harcourt into government. During the election campaign the NDP capitalized on popular dissatisfaction with provincial environmental management. Environmental groups were not numerous enough to be a key interest group for electoral success (unions are much more important to the NDP). But they were potent enough to cause political upsets for provincial governments in the past when groups prevented the siting of hazardous waste incinerators, or successfully campaigned against dioxin pollution from pulp mills and clearcutting by advocating a boycott of B.C. forest products in Europe. Thus, the NDP government was under pressure to make good on its environmental policy promises. Environment Minister John Cashore identified five key environmental policy concerns of the NDP government: improvement of environmental impact assessment procedures, preservation of biodiversity and natural areas, reduction of waste and pollution prevention, improved water management, and strengthened compliance enforcement (B.C.MoE 1992a,b). It also endorsed the adoption of economic incentives to resolve the perceived conflict between economic growth and environmental protection (B.C.MoE 1992a,b).[53]

Federal - provincial relations on environmental policy were cool as always, in part due to uncertainties over changes in jurisdictional authority in the context of constitutional reform. Among the specific conflicts at the time was the provincial governments' unwillingness to underwrite the federal governments endorsement of emission trading in the national NO_X-VO_X reduction consultations (Doern 1990b; also see CCME 1992). And B.C. was no exception. Although the WMPFR were geared toward revenue raising they also underscored that federal policy preferences had no relevance for provincial policy directions (Foley 1992).

3.3. Policy content

The WMPFR obligates permit holders to pay an annual $100 permit fee (Section 3(1), Schedule A). In addition, an approval fee is levied which is calculated by summation of the fees for each discharged contaminant over a period of one year (Section 4). Under the WMPFR, 15 types of emissions, and 21 types of water effluents are chargeable (see Appendix 3). Fee levels were calculated based on the relative risk posed by contaminants with suspended solids serving as reference. However, fees on several pollutants (e.g. AO_X, mercury) were capped to prevent extreme hardship for industry.[54]

Fees are based on permit levels, not on actual discharges. The fee schedule is phased in two steps. Until August 31, 1993, fees will be discounted by 50%, afterwards full charges apply.

---

[53] Economic incentives were already studied in the B.C.MOE under the Social Credit government (Stone 1990). Likewise, the federal government sponsored several studies of economic incentives and market-oriented policy instruments (see Cassils 1991; Environment Canada 1992; CCME 1992).

[54] For a comparison of current German effluent fees with B.C. effluent fees see Appendix 4.

Entities of the provincial and federal governments are excluded from the pollution fees. Municipalities are exempted from the fee for all discharges under permits issues on or before December 31, 1995 (Section 5(b)). After that date fees are waived for municipalities that have taken steps to reduce solid waste by 50% by the year 2000, and which implemented a volume-based solid waste user-pay strategy. The fee schedule for air emissions does not apply to companies in the Greater Vancouver Regional District (GVRD), or any other regional district which is implementing air emission fee schedules.[55]

The regulations do not outline the objective of the discharge fees, nor do they indicate the use of the revenues. A discussion paper preceding the regulations states that the fees should at least cover the $15 million the province spends every year on administering waste discharge permits. Revenues beyond that target would be used to finance pollution control measures or fund research through the Sustainable Environment Fund (B.C.MoE 1992e).[56] However, the Finance Ministry encouraged the B.C.MoE to raise more than $15 million, and move a portion of the revenue to the general budget "to cover other social costs of pollution, for example health care" (interview with staff in the Environmental Protection Division, B.C.MoE, anonymity requested).

3.4. Patterns of conflict and conflict resolution

Industry associations were informally informed in the fall of 1991 that a revision of the discharge fee schedule was being prepared. The revision came in the wake of a failed attempt by the B.C.MoE to recoup administration costs through a tax on chlorine discharges from pulp mills, and a general tax on chemicals produced in the province (Lockhart 1992; McCloy 1992). After the change in government, industry was notified of the new plans in December 1991, and received a discussion paper on the proposed fee system in January 1992. The Ministry solicited

---

[55] The GVRD passed the Air Quality Management Bylaw No. 725 on July 31, 1992. The bylaw establishes a $1000 permit application fee (Schedule B, B.1(b)). Fees for carbon monoxide are $1 per ton, and $60 per ton for sulphur oxides, nitrogen oxides, hydrocarbons, particulate, and other authorized air contaminants (Schedule B, B.3).

[56] The B.C.MoE proposed that the revenues from waste discharge permit fees be more directly attributed to waste management than the current distribution through the Sustainable Environment Fund (B.C.MoE 1992e: 40).

comments on the discussion paper. Industry made about 400 submissions, while about a handful of environmental and public interest groups responded (Austin 1992).

Industry complained bitterly about the additional burden of the PTs in a time of recession and intense international competition. Beyond this 'typical industry complaint' there is a surprising similarity in the criticisms brought forward by interest groups. In their submissions environmental groups, industry associations and individual companies supported the adoption of the polluter pay principle. Public interest groups and industries both criticized that fees be based on permit data rather than actual discharges, the exemptions for federal, provincial, and municipal permit holders, as well as monitoring and compliance issues. In addition, both sides raised concerns over lack of specificity of the regulations, above all the lack of well-defined environmental quality objectives (see, for example, Council of Forest Industries of B.C. 1992; West Coast Environmental Law Association 1992).

Much to the dismay of industry, and to a lesser extent environmental groups, the B.C.MoE did not respond at all, or only after the regulation's promulgation, to the submissions.[57] Industry reacted with frustration when the final regulations did not reflect any suggestions made in the submissions.[58] And since the regulations were passed as Order in Council interest groups had no opportunity to lobby their MLAs, or exploit parliamentary debate for their cause.

Together with the WMPFR, the B.C.MoE announced a workshop "to begin discussions with stakeholders on future changes to the regulation (e.g., use of actual data; environmental impact; etc.)" (Fast 1992a). Some 200 participants attended the workshop in September 1992.[59] During the workshop, and in a parallel consultation process on new approaches to environmental protection in B.C. (B.C.MoE 1992c), industry representatives advocated emission trading as a better pollution control mechanism. Industry not only pointed out that emission trading would

---

[57] Interviews with numerous company representatives during a stakeholder workshop on the WMPFR sponsored by the B.C.MoE September 24, 1992 in Delta, B.C.

[58] The submissions were not systematically evaluated by the B.C.MoE (Austin 1992).

[59] Roughly 175 industry representatives and industry consultants, five environmentalists and public interest group members, and about twenty representatives of regional districts, municipal and federal governments attended the workshop.

be more efficient, but also demanded that "such a system be allowed to operate independent of government control" (B.C.MoE 1992f: 14).

In response to intense industry criticism of the WMPFR the B.C.MoE convened a panel of technical experts from various industry associations to discuss technical issues, such as measurement methodology, instrumentation, and certification of private laboratories (Fast 1992b).[60] Industry, however, did not see this as a continuation or deepening of the consultation process but as "a smokescreen" and "as sidetracking the real issues of fee level, and control over revenues" (McCloy 1992).[61]

Industry's rage indicates that the imposition of the fee schedule interrupted the cosy relationship it enjoyed with the Social Credit Government. The B.C.MoE did not succeed in introducing a chemical products tax or a chlorine discharge tax because the Social Credit cabinet listened to industry criticisms. The Environment Minister's position in cabinet is weak because the resource sectors (forestry, mining) are governed by industry-specific ministries. In the case of the WMPFR, however, the B.C.MoE could harness the support of the cash-strapped Finance Ministry (Glen Clark is probably the most powerful cabinet member) to assert itself in cabinet, and over industry.

3.5. Policy effectiveness

Since the fee system has only be introduced in September 1992, no data are available concerning the environmental impact. However, industry and environmentalists concur that the fees are much too low to force the installment of pollution control equipment, or changes in production processes. For example, the construction of a secondary treatment sewage plant at the Port Allice pulp mill would cost about $100 million, but yearly fee payments would only be

---

[60] The following organizations were invited to join the committee: The Council of Forest Industries, the Union of British Columbia Municipalities, the Mining Association of British Columbia, the Canadian Chemical Producers' Association, the Canadian Petroleum Products Institute, the Fisheries Council of British Columbia, and a private refuse operator (Fast 1992b).

[61] Industry sees the fee primarily as a means to finance the expansion of the administering departments. The GVRD Air Quality and Source Control department hired 28 new staff since the introduction of the fee schedule. The B.C.MoE employed 12 additional people to develop the fee system and to assist in its administration (Lockhart 1992).

$0.5 to 1 million. The spatial distribution of industry is also not conducive to cooperative arrangements between firms or companies and municipalities to treat waste water. Only one such arrangement exists between Kariboo Pulp and the Municipality of Quesnel. Recently, the B.C.MoE opposed a similar cooperation between a Fletcher Challenge pulp mill and the Municipality of Campbell River (McCloy 1992).[62] Thus, most companies will choose to pay, and not to clean up. In addition, the provincial government indicated with the proclamation or the WMPFR that it planned to review the regulations within two years (Fast 1992a; McCloy 1992). It is unlikely that many companies are going to make major investments before then, especially since the regulation's review would already be close to the next election.

The fees, however, already had some unintended positive consequences. Since fees are based on permitted pollution discharge levels many companies hired certified laboratories to test their discharges. Based on these actual measurements companies are negotiating with the B.C.MoE the issuance of new permits. According to Hu Wallace (1992), many permits are many years old and "we didn't really know how much pollution was out there. ... Thus, the fees help to build a data base for better enforcement and any future policies, be it taxation or trading."

3.6. Case analysis

There can be little argument that the WMPFR are primarily intended to raise revenue. The B.C. MoE had considered two similar policies before (taxes on chemical products and taxes on chlorine discharges), without success. The change of government opened a new opportunity to attach the solution (PTs) to the political challenges of the day. The NDP was eager to establish a 'green image' and looked for a policy that would make good on the polluter pay principle. At the same time, budget constraints made any proposal that generated revenue welcome: It relieved budget pressures on the Ministry of Finance, it secured the staff positions and programmes in the B.C. MoE which otherwise would, at best, have seen no additional finance.

---

[62] The B.C.MoE feared that in the event of permit violations of the sewage plant it could not verify whether the municipality or the pulp mill were to blame for excess pollution.

Provincial PTs have the potential to appeal to regional identity in case of renewed federal - provincial conflicts over pollution control (particularly effluents into the Fraser River, and the Georgia Strait, or air pollution in the Lower Mainland). Similarly, the PTs may also serve to obstruct a federal emission trading demonstration project in the Lower Mainland. Notably, the WMPFR exempts regional districts which administer their own fee systems from the levy. However, district which would operate an emission trading scheme would still have to pay. Conceivably, the B.C. government can counter future federal regulatory initiative with the argument that 'we already have fees for that purpose', and that 'the federal government ought not to tell the province how to do things.'

The WMPFR's institutional path underscores the typical policy style of Canadian governments with one minor exception. As the government had to act fast, it passed the WMPFR as an Order in Council without intensive consultation. It was heavily criticized for this by industry and environmentalists. Thus, the government immediately opened a typical consultation process: conferences with diffuse goals and without mandate, so that any future policy changes would be left to government maneuvering and backroom bargaining with selected interests.

3.7. The Canadian discourse over pollution taxation and emission trading

Water use fees in British Columbia date back to 1859, however, as in other provinces, the system was never developed into a pollution tax scheme for discharges (Reese 1983: 57). In 1988, Ontario studied the possibilities for effluent charges to control water pollution. But Ontario did not implement such a scheme because it deemed the determination of the appropriate effluent charge impossible (Dewees 1992: 51).

The 1970 Canada Water Act is the only federal legislation that specifically authorizes the collection of pollution taxes (Subsection 13(1),(3)), but this provision was never implemented. More recently, *Canada's Green Plan* endorses emission trading (Government of Canada 1990: 55, 157), but does not specify a government position on pollution taxes.[63] Environmental

---

[63] During the consultations leading up to the *Green Plan* emission trading was extensively discussed, "leaving no time" to deliberate on pollution taxes (*Montreal Gazette*, August 22, 1990: B1)

groups had demanded a government commitment to pollution taxes, particularly a carbon tax on fossil fuels (TS / Spears 1989). But the *Green Plan* simply states that economic instruments would be studied further in the future (Government of Canada 1990: 157).

So far, only two initiatives are under way. In 1992, the Departments of the Environment and Finance released a report on how Canada should pursue its goal to stabilize carbon emissions at 1990 levels by the year 2000. The report looked at the effects of regulation, taxation, and emission trading. However, it is expected that the federal government will not use any of these instruments but rely on "moral suasion" instead (*Financial Post*, May 22, 1992: 17; *Financial Times of Canada*, May 25, 1992: 18).

A second initiative is the use of emission trading in the reduction of NO_x-VO_x emissions which is coordinated through CCME. Federal - provincial negotiations over a possible pilot project are slow. Provinces, particularly Ontario and B.C., are unwilling to commit to a pilot project in their area because they fear federal intrusion into their jurisdiction, and possible federal credit claiming taking if the programme is a success (Doern 1990b). According to Hu Wallace of the Air Resources Division in the B.C.MoE

> A suggestion at the federal level that emission trading in the Lower Fraser Valley is a good idea is just that: A suggestion. ... There is no question about who has jurisdictional responsibility. The province manages emission permits, so it's the province that institutes trading of these permits. If there is emission trading, it is the province which does it for good or for bad. The point here is that emission trading may be a good idea, but we want to avoid any doubt about whose good idea it was (Wallace 1992).

The B.C.MoE expects that emission trading would increase administration, monitoring and enforcement costs and that it "would highlight the weaknesses of the present, incomplete regulatory system" (Wallace 1992). Even if these constraints could be overcome, provinces could still not embark on emission trading on their own because the federal government committed the country to national emission caps in international treaties. The allocation of these national quotas among the provinces has yet to be negotiated which is "not an easy process" (Wallace 1992).

Opinion polls taken in 1990 indicate strong public support for 'green taxes:' Between 50 and 75 per cent of the respondents were willing to pay product or even income tax surcharges to protect the environment (GM / Miller 1990; *Montreal Gazette*, May 31, 1990: B8; *Toronto Star* May 31, 1990: A1).

The debate over emission trading and pollution taxation appears to be very much an insider discourse. As of November 1992, the *Canadian Current Affairs Index* does not list any articles on emission trading in a major newspaper or magazine. Aside from the C.D. Howe and Fraser Institutes, which strongly support TPRs, other leading research institutes have not formulated a final position on PTs or TPRs (Macdonald 1991: Ch. 13). In the academic community, only Olewiler (1990, 1992) appears to lend cautious support to PTs. No other scholars have put their position on record.

Interestingly, research papers on PTs and TPRs circulated by government agencies, academics, and private sector organizations mainly discuss the U.S. experience with these instruments (Cassils 1991; Environment Canada 1991). So far, there appears to be little willingness to openly identify and discuss the political and institutional features that will constrain the adoption of PTs and TPRs in Canada.

In sum, the Canadian debate over PTs and TPRs is not far advanced. In the two cases where concrete initiatives evolved (carbon tax, NO_x-VO_x), federal - provincial cleavages appear to strongly militate against the adoption of either policy instrument.

III. Complements to a theory of instrument choice

The patterns of instrument choice in all four case studies are consistent with and affirm the choice model advanced by Hahn (1989a,b, 1990). Industry favoured less costly policy instruments, and generally demanded and received special treatment. Thus, Hahn's theory would predict the same outcome for all cases, and is the adoption of emission trading. However, this was not the case. Yet, neither in Germany nor in B.C. was environmental interest group influence strong enough to account for the variance. This demonstrates the limitations of choice models based on game theory. While such models can generate predictions of individual actors' preferences they do not adequately take account of other factors that influence the choice and design of policy instruments.

In this section I will reflect on several key aspects of instrument choice and identify the most important variables that should be added to a theory of instrument choice.

1. How much do redistributive instrument properties matter?

Existing theories suggest that the distribution of costs and benefits among stakeholders is the overriding factor determining instrument choice. While the case studies would not fundamentally challenge this view they do propose that PTs and TPRs may restructure existing conflict alliances in environmental policy. Interests will not necessarily organize along the lines of existing lobby organizations but along regulatees' cost structures which are key to the incidence of the policy instruments in question. Of course, policy makers can exploit these divisions in the ranks of interest groups.

U.S. policy makers chose emission trading to diffuse regional conflicts. Likewise, German policy makers diffused conflicts with state governments and industry by including amiable clauses and administration procedures into the law. A similar strategy is apparent in B.C., although less pronounced, perhaps because the impact of the fees was much lower compared to the German fees (see Appendix 4). Generally, I conclude that PTs and TPRs can transform political interaction patterns even over redistributive issues. The choice of PTs or TPRs helps to redefine redistributive issues as distributive problems. Especially the SO_2 trading and the German effluent charges show that policy makers can separate allocative instrument effects from the redistributive ones by transferring the decisions over each aspect into different institutional arenas. In Germany the fee levels were determined at the federal level, but the redistributive questions of subsidies and reinvestment were left to local and state authorities. In the U.S. TPRs were allocated by Congress, but the ultimate redistributive impact will be decided through a myriad of market transactions.

Of course, these strategies also help politicians to shield themselves from blame. On the other hand, they give up credit claiming opportunities and possible electoral gains by not exploiting their power to allocate costs and benefits on marginal ridings (cf. Trebilcock et al 1982).

Institutional structures did not make a difference in how PTs or TPRs diffused opposition, or how governments adapted the different instruments to break resistance from industry or regional interests. Also, there are no indications that the German and Canadian policy styles of consensus-seeking bargaining made an appreciable difference in the application or political performance of the more redistributive PTs compared to TPRs in the U.S. I conclude that redistributional issues linked to PTs and TPRs cannot explain the variation of instrument choice across countries.

2. Do political culture and values matter?

An explanation based on political values might also be ventured. While it is true that Germans and Canadians may have less confidence in the invisible hand of the market than their U.S. counterparts, this did not translate into industry lobbying for PTs rather than TPRs. Clearly, industry is keenly aware of its international competitiveness and will oppose any policy instrument that would damage its position. The economies of B.C. and Germany are even more dependent on international trade than is the U.S. economy. Governments are aware of this as much as is industry, and would not unnecessarily impose burdens on industry. This does not mean that political values could not explain instrument choice within a single country, such as in the U.S. Clearly, ideological considerations figured prominently in Bush's choice of TPRs over PTs. But concern over political values was not as evident in the other two countries. In Canada and in Germany the public discourse is not far advanced and neither instrument has been claimed by a certain ideological stream in society or the political arena. The non-ideological treatment of PTs and TPRs may be advantageous as it may help towards the pragmatic application of these instruments in the future. On the other hand, the lack of public discourse fails to promote innovative ideas and active research into the possible integration of PTs and TPRs into the existing regulatory systems.

Value judgements emerge from discourse of an issue. Institutional structures in Germany and Canada do not favour discussion and the discernment of value judgements. Given the low level of public awareness the chances of creating the necessary momentum for the adoption of TPRs are low, and policy entrepreneurs may see more promising opportunities in other policy areas. It is also notable that the administrative structures in Germany and Canada would not allow the quick diffusion and implementation of new policy ideas as it was evident in EPA's introduction of emission offsets. Even if the federal environment ministries in Germany and Canada created a hothouse for new policy ideas like EPA's OPPE, their effect may be minimal. Any policy initiative would have to pass through numerous other institutions, and hurdle various veto points due to shared or overlapping jurisdiction, so that the chances for survival of new ideas in the bureaucracy are slim.

3. How do policy styles affect agenda setting and instrument choice?

There are key differences between the U.S. and Germany or B.C. in the policy context from which the instrument choice emerged. In the U.S., 'economic instruments', particularly emission trading, were a response to the enormous costs associated with rigid standard-based regulations (e.g. implementation, litigation). By contrast, in Germany and B.C. 'economic instruments' were seen as a means to make environmental protection more efficient, or to reallocate costs for environmental protection. They were not under immediate pressure to choose between economic growth and environmental protection (Rehbinder and Sprenger 1985). More to the point, the standard-based regulatory systems of Germany and B.C. did not experience the failure of their U.S. counterpart. While German regulatory standards are not less stringent than U.S. standards, they are produced through intensive government - industry consultation, often geared towards improving industry's competitive position rather than constraining it. In addition, permit-granting regulatory agencies are usually bestowed with discretion in applying these standards to avoid undue economic hardship on firms. And since federal law does not grant court standing to environmental groups, bureaucratic discretion is seldom challenged in the courts. The same is true for B.C., although government - industry consultations have not produced stringent regulatory standards. Rather, permit conditions are set on a case by case basis, often heavily favouring industry and sacrificing environmental protection.

Emission trading schemes could hardly emerge from a regulatory culture that is marked by bargaining and compromise. A well functioning TPRs market crucially depends on uniform environmental quality standards for all sources, i.e. the fixing of an emission cap. Neither Germany nor B.C. have *legally enforceable* environmental quality standards. And they are unlikely to develop them for two reasons. Firstly, legally enforceable environmental quality standards would undermine the discretion and power of their bureaucracies. Such standards or objectives would direct increased media attention to the closed bargaining between bureaucracy and industry over permitted pollution discharges. They may also invite legal challenges from concerned citizens. The second reason, jurisdictional constraints and blame avoidance and credit claiming in decentralized federal systems, will be discussed below.

Emission trading also depends on highly visible enforcement actions. Otherwise, confidence in the market is undermined, and TPRs prices would not reflect marginal abatement costs (plus transaction costs). Instead of buying TPRs, companies would simply pollute. In the absence of a comprehensive regulatory system, enforcement institutions, and punishment for violations, there is no credible enforcement threat (Kagan 1989). Canadian authorities' enforcement record is poor (Castrilli 1982; Schrecker 1990; Sproule-Jones 1980). Regulatory enforcement and punishment in Germany may be stricter, but because laws are executed through the states enforcement is uneven (Forschungsgruppe Umwelt 1991). While the record of U.S. agencies in enforcing the regulatory standards is not outstanding either, punishments are much harsher than in either B.C. or Germany. In addition, citizens can take companies and regulatory agencies to court for failing to meet regulatory standards, or for violating laws governing a TPRs market.

From these deliberations I conclude that a comprehensive framework of legally enforceable standard-based regulation is a prerequisite for the establishment of TPRs markets. Economically, such a system is onerous and government, industry, and even environmentalists may have an interest in reforming it. But it places all stakeholders on a legally equal footing: effective legal recourse is the basis of confidence in the market. Standard-based regulations produced and enforced through bargaining are much less rigid, and there is less economic pressure to change the system. And if change occurs it is more likely that PTs are chosen since a taxation system can accommodate continued bargaining without breaking down (Downing 1983).

4. How do institutions influence instrument choice?

The third key variance between the countries under study is their institutional frameworks for environmental policy. In the U.S., environmental policy powers are concentrated in the federal government. The 1970 Clean Air Act, and even more so the 1972 Water Pollution Control Act, explicitly transferred all primary management responsibility from the states to the federal government, thereby reversing power relations in environmental management (Brown and Johnson 1984: 953; Freeman 1990; Portney 1990b,1990c). Individual states, however, are constrained in their instrument choice by federal legislation because they can only implement stricter laws.[64] States have limited legal powers or political incentives to challenge or obstruct federal programmes through their own legislations. Consequently, Congress and EPA face no jurisdictional constraints when choosing among policy instruments.

The German constitution empowers the federal government to issue binding legislation on air pollution control, but only allows for framework legislation on water pollution control.[65] In either case, state authorities have to execute these laws. Generally, federal - state relationships are generally not adversarial because they are well defined in law. In addition, both orders of government have to cooperate in the *Bundesrat* to avoid legislative paralysis. Nevertheless, state governments cautiously guard their jurisdictional authority and the federal government hardly ever tests the legal limits. On issues that may impinge on state authority, state governments may not vote along party lines (i.e. SPD-governed states voting for legislation proposed by a SPD federal government, CDU-governed states voting against it). Such events are rare, and

---

[64] See, for example, Clean Water Act, 33 U.S.C. 1982, Sections 1314(1), 1316(c), and 1318(c). Brown and Johnson (1984: 958) note that under the Clean Water Act it is not clear whether 'stringency' here refers to more stringent standards, or would allow for PTs or TPRs in addition to standards.

[65] The constitution thereby recognizes the historically strong role of local and state governments in water management.

customarily interpreted as reflecting weak leadership on the part of the Chancellor, who usually also chairs the governing party. Federal water policy could not introduce emission trading without a coordinated legislative effort on the part of the states. Although the federal government has the authority to introduce emission trading into its air policy, it has cautiously avoided doing so. There are no federal institutions that could administer such a programme. Implementation through state agencies would require major bureaucratic reorganizations, and substantial changes to state administrative, environmental, and criminal laws to ensure the smooth operation of a TPRs market (Rehbinder and Sprenger 1985). This would amount to an enormous legislative reform programme which may even have to include constitutional amendments. An endeavour of this magnitude would absorb much of the federal and state governments' capacity for considerable time, and politicians would have to question seriously why this effort should be undertaken. Any reform would necessarily concentrate more powers and responsibilities either in the hands of state or federal government, thus removing the possibility of shifting blame to the other order of government for legislative or implementation shortfalls. Considering that the outcome of such legal reforms would be highly uncertain, and that there was and is no public demand to create the legal basis for emission trading, any major policy initiative to institute a trading programme would offer more blame opportunities than potential political payoffs. Even intensive federal - state cooperation could not ameliorate these dangers.

Canadian federal - provincial cooperation is not as clearly defined, and not as cooperative as in Germany. Although the federal government could exercise powers on water and air pollution control under the Fisheries Act, the Canada Water Act, and the Canada Clean Air Act (now part of CEPA), it has not asserted itself. All three acts made standard-setting and their enforcement conditional on federal - provincial cooperation. Provinces have consistently opposed intrusion by federal regulatory authority into their jurisdictions (Dwiwedi and Woodrow 1989). And the federal government usually resorted to the establishment of national minimum standards, often in the form of guidelines rather than laws, or encouraged inter-provincial agreements through the Canadian Council of Ministers for the Environment (CCME).

Yet, the capacity of the provincial government to implement either serious PTs or an emission trading programme is constrained by jurisdictional overlap and inter-governmental / inter-provincial competition. B.C. cannot fix an emission cap for a trading programme because for a number of pollutants national caps have been designated (i.e. SO_2, NO_x-VO_x). Unilateral action to appropriate a share of this national total would trigger strong protests from other provincial governments. Serious PT measures (without compensating tax cuts) could hurt the competitive position of B.C. businesses, and deter investment. At the same time, it is not in the interest of the provincial government to invite federal involvement to implement either PTs or emission trading as it would seem to chip away on provincial authority in environmental policy making. Although past patterns of federal non-action on environmental policy matters are not suggestive of this happening, Canadian non-compliance with international environment treaties may tarnish the federal government's image enough to induce selective intrusions into provincial affairs to fulfill its international commitments.

Strategically, B.C. emission and effluent discharges have to be understood in this context. The B.C. government could argue that their fees are 'equivalent' to a range of possible federal regulatory initiatives.[66] Accordingly, the fee schedule covers a broad range of contaminants for air and water, but imposes only very moderate fees.[67] Thus, the fees can be used as a jurisdictional defence mechanism. Similarly, B.C. is currently considering the adoption of the Californian system for classifying hazardous wastes as well as car emissions. This action would certainly embarrass the federal government which uses a different system, and it would obstruct the development of coordinated national policies.

If we analyze instrument choice from the perspective of jurisdictional fragmentation we arrive at the following pattern. Jurisdictions with relatively centralized authority over air or water management, as in the U.S., could move towards emission trading or pollution taxation.

---

[66] For example, the second draft of the federal NO_x-VO_x reduction programme included an equivalency test for provincial programme. However, Ontario and B.C. vigorously opposed any stringent tests for their NO_x-VO_x reduction policies. Subsequently, the clause has been dropped from the NO_x-VO_x management plan. Without factual tests for equivalency, provinces can make strong political claims which the federal government is unlikely to challenge.

[67] The WMPFR also include a 'catch all' fee category for "other contaminants not otherwise specified" (OIC 1264, 1992, Schedule C).

They could adopt either PTs or TPRs systems, if not for other factors that narrow the choice, such as political culture or institutions. The structure and processes of the U.S. legislature encourage members of Congress to build large coalitions around the distribution of discret benefits, and to join such coalitions to deliver benefits to their constituents. TPRs appear to be better suited in the formation of such coalitions.

The German case leads to the conclusion that jurisdictions with shared authority over policy areas will opt for PTs as they are easily adaptable to administrative fragmentation, and can even constitute an explicit recognition of historical local authority and institutions in environmental management. Consequently, the central government can claim credit for facilitating a national approach to pollution control, but leave other credit and blame opportunities to local authorities. Claims of improved environmental quality can easily be made in the absence of consistent state or national environmental quality surveys.

In states with fragmented and overlapping authority over environmental policy, such as Canada, central governments have few incentives or limited powers to establish either PTs or TPRs markets. Lower jurisdictions' adoption of emission trading may be constrained by international commitments of the federal government. Also, inter-provincial competition for investment may lead provincial governments to refrain from imposing PTs. Instead, lower orders of governments will, at best, use low PTs to raise revenues to maintain or improve their environmental management capabilities. Strategically, relatively low PTs can be used to occupy policy areas and block outside intrusion.

These deliberations on the effects of jurisdictional fragmentation call into question other explanations of instrument choice. Trebilcock et al (1982) argue that politicians will choose instruments so they can spread costs and benefits across ridings with the purpose of increasing electoral support. Such a strategy is viable in systems with undivided jurisdiction over a policy area. For example, TPRs allocation through Congress demonstrated that politicians will prefer policy instruments that allow them to allocate discrete benefits to their constituencies. In the German case, however, politicians did not have the opportunity and did not show interest in establishing control over the spread of reinvestments and subsidies at the implementation level. Thus, the proposition of Trebilcock et al only applies in selected cases.

In summary, the differences in instrument choice across the three countries under study are best explained by considering jurisdictional fragmentation, the history of standard-based regulation, and policy style. The final decision within each country then appears to be strongly influenced by prevailing values concerning the efficacy of the market to resolve social conflicts. These factors should be considered and systematically integrated into existing explanations for instrument choice. To do so successfully, further research will be necessary, and the following chapter outlines some of the challenges to future research.

IV. National styles of instrument choice?

The complements to a theory of instrument choice outlined above were based on four case studies in three countries. The question obviously is whether it holds for other cases as well. Aside from Germany, Italy, the Netherlands and France operate effluent charge schemes (Bongaerts and Kraemer 1989; OECD 1989; Reese 1983; Schleicher 1986). And as in Germany or Canada, bargaining and consultation are distinguishing features of the French, Italian, and Dutch policy styles. Likewise, their standard-based regulatory systems are not as rigid as the U.S. system. In all three cases, the effluent PTs are administered by local authorities which had been regulating water use and distribution for a long time. As in Germany, the Dutch, French, and Italian governments also have legislative authority to implement national air pollution control programmes with whatever instruments they choose. However, PTs and TPRs for air emissions are curiously absent.

Building on the German case I hypothesize that central governments in these cases do not want to directly or indirectly challenge traditional local administrative functions by imposing a regulatory scheme that may extend beyond local control.[68] PTs are better suited in such a

[68] The strength of local administrative institutions is particularly evident in The Netherlands where the local waterboards also administer some aspects of air pollution control policies.

situation because they can be structured to acknowledge and validate local environmental management functions.

This appears to be a promising direction for future research as it may explain why PTs are the preferred instrument choice for water issues. I suggest it is not because they are technically better suited than TPRs. Rather they are political responses to historically grown administrative responsibilities. In addition, fee levels appear to be set lower than necessary to induce to behavioural change so that the local economy is not damaged. To the same end, reinvestment decisions are left to local authorities.

It appears that governments throughout Europe face this situation and have, so far, refrained from setting a precedent in air policy that could lead to strenuous discussions over the allocation of authority in environmental policy. Discussions over the reallocation of power are always politically risky, and involve significant 'opportunities' for blame. Why should politicians chose this path, if their bargaining-style policy approach can deal with the challenges of the day?

Table 2 summarizes a rough survey of jurisdictional responsibility in water and air policy in selected OECD member states. The European countries appear to resemble, broadly speaking, the pattern of the German case, and in choice situations we should expect similar outcomes. Table 2 also indicates that the centralization of authority over water and air policy in the U.S. is rather unique. Thus, policy instrument innovations may not be easily transferred from the U.S. to other countries. It should be emphasized that these are very preliminary observations which need further investigation.

This raises the question how national governments in countries with shared or overlapping jurisdictions ensure national environmental quality standards. The German case suggested that, unlike in the U.S., emission standards for old and for new sources may be set at very similar levels to reach nationally uniform levels of environmental protection. Again, this would be an interesting research opportunity as it may validate or invalidate existing theoretical propositions for the politics of standard-setting (Hahn 1989a,b, 1990). Furthermore, this would help to place standards, and alternative policy instruments, into the contexts of wider national instrument choice patterns.

Such research should be conducted with explicit consideration of the institutional context of policy making and instrument choice. This will enable the evaluation of choices among different policy instruments in specific institutional settings. If the impact of institutions on instrument choice is as strong as suggested by the case studies presented in this thesis, then it is unlikely that the utilization of policy instruments across the industrialized world will become more and more alike. Instead we should expect the persistence of established national styles of regulation and instrument choice.

IV. Conclusion

In this thesis I argued that existing theories of instrument choice in environmental policy cannot adequately explain the variation of choices across different countries. In particular, I observed a lack of empirical research and theoretical conceptualization on choice problems involving 'economic instruments' for environmental protection. Pollution taxes and tradable pollutions rights were taken as representative examples from this group of policy instruments, and their distributional properties were analyzed. It was shown that TPRs place fewer financial demands on polluters, and that therefore TPRs should be the preferred policy instrument.

The analysis of emission trading in the U.S., the German Effluent Charge Law, and British Columbia's Waste Management Permit Fee Regulations, however, demonstrated that the choice pattern predicted by theory does not hold. The analysis explored the explanatory power of other reasons for instrument choice, such as political culture and values, agenda setting and policy styles, and institutions.

I found that institutional structures that allocate jurisdictional authority over policy issues are the single most important explanation for instrument choice. In particular, emission trading is only adopted in the U.S., the only country where the federal government exercises unconstrained jurisdiction over air policy. It is in this unique setting that emission trading found acceptance among administrators, interest groups, and legislators. In countries with functionally divided jurisdiction, such as Germany, or with overlapping authority over policy, such as Canada, the introduction of emission trading is inhibited by policy divergence and uneven enforcement across lower jurisdictions. The German case study has shown that PTs can be designed in ways that accommodate jurisdictional concerns. However, such an endeavour may be contingent on institutions and a spirit that facilitate interjurisdictional cooperation. These are less developed in Canada than in Germany.

Based on these research results I hypothesized that patterns of instrument choice similar to those in the case studies should be found in other countries that share key features of allocation of authority and regulatory style in environmental policy with Canada and Germany. An initial survey identified several such countries (e.g. France, Italy, The Netherlands), and research on instrument choice patterns in these countries should help to validate my propositions. So far, the literature on instrument choice in environmental policy almost exclusively builds on the U.S. experience. But the institutional setting of U.S. environmental policy is rather unique, and theory building must include the examination of choice patterns in other countries as well. This thesis endeavoured to contribute to this.

The comparative study of instrument choice should also be of practical value as policy makers think about ways to apply PTs, TPRs, or other 'economic instruments' in the framework of regulatory regimes that do not favour their adoption. The key to the utilization of innovative policy instruments may lie in reforming regulatory institutions. However, such reforms must build on a clear understanding of how the performance of a policy instruments is affected by institutional structures, or interacts with other policy instruments in such given institutional structures. These questions deserve further research.

This thesis restricted the focus of analysis to the choice between TPRs and PTs, and considered the allocation of jurisdictional as the major factor in the choice process. While this helped to expand the knowledge and the theoretical conception of instrument choices, the 'theory of instrument choice' is far from complete. A next step in theory building should expand the analysis to structures and actors in the decision making processes in various countries. For example, the nature and location of veto points in the decision processes will vary across countries and affect instrument choice in different ways. Strong national party organizations may obscure otherwise unsuitable institutional arrangements as they may help to bridge jurisdictional cleavages. Different legal systems may offer a widely different legal basis for the adoption and implementation of TPRs and PTs in different countries. These and other factors, of course, interact in the policy process. A theory of instrument choice would not only benefit of such more complex considerations. It would also enable comparative studies of instrument choice in environmental policy and other policy areas, such as industrial and economic policy, in which knowledge and analysis are further advanced and relatively more sophisticated.

Bibliography

Ackerman, B. A. and R. B. Stewart. 1988. "Reforming Environmental Law: The Democratic Case for Market Incentives." *Columbia Journal of Environmental Law,* Vol. 13, pp. 171 - 199.

Albert, K.H. 1992. *Federal Choice of Policy Instruments in the Canada Green Plan.* Unpublished M.A. Thesis, Department of Political Science, University of British Columbia, Vancouver.

Anderson, R.C., L.A. Hofmann and M. Rusin. 1990. *The Use of Economic Incentives in Environmental Management.* Washington, DC: American Petroleum Institute.

Atkinson, S. 1981. *The Design and Cost-Effectiveness of Marketable Emission Permits for Air Pollution Control.* Washington, DC: American Petroleum Institute.

Atkinson, S. and T. Tietenberg. 1991. "Market failure in incentive-based regulation: The case of emission trading." *Journal of Environmental Economics and Management,* Vol. 21, pp. 17-31.

Austin, Jason. 1992. Personal communications. Mr. Austin is assistant administrator for the Waste Permit Fee Project in the B.C. Ministry of Environment, Victoia.

Baar, E. 1989. *Positive Compliance Programs: Their Potential as Instruments for Regulatory Reform, Volume I.* Ottawa: Department of Justice. Mimeo.

Baumol, W.J. and W.E. Oates. 1975. *The Economic Theory of Environmental Policy.* Englewood Cliffs, NJ: Prentice Hall.

Baumol, W.J. and W.E. Oates. 1979. *Economics, Environmental Policy, and the Quality of Life.* Englewood Cliffs, NJ: Prentice Hall.

Beyme, K.v. (ed) 1985. *Policy and Politics in the Federal Republic of Germany.* Hants, England: Gower Publishing Company Ltd.

Beyme, K.v. 1990. "Economic and Social Policy in the Bundestag." In Thaysen, U., R.H. Davidson, R.G. Livingston. 1990. *The U.S. Congress and the German Bundestag: Comparisions of Democratic Processes.* Boulder, CO: Westview Press.

Bird, R. and Mintz, J. (ed). 1992. *Taxation to 2000 and Beyond.* Toronto: Canadian Tax Foundation.

Bohm, P. and C.S. Russel. 1985. "Comparative Analysis of Alternative Policy Instruments." in Kneese, A. V. and J. L. Sweeney (ed). 1985. *Handbook of Natural Resource and Energy Economics.* Vol I. New York: North-Holland Publishers.

Braybroocke, D. and C.E. Lindblom. 1963. *A Strategy of Decision*. New York: Free Press.

Brickman, R., S. Isanoff, T. Ilgen. 1985. *Controlling Chemicals: The Politics of Regulation in Europe and the United States*. Ithaca, NY: Cornell University Press.

British Columbia. 1992a. *Environmental Action Plan for British Columbia*. Victoria, B.C.: Ministry of the Environment, Public Lands and Parks.

British Columbia. 1992b. *British Columbia's Environment : Planning for the Future*. Victoria, B.C.: Ministry of the Environment, Public Lands and Parks.

British Columbia. 1992c. *New Approaches to Environmental Protection in British Columbia*. Victoria, B.C.: Ministry of the Environment, Public Lands and Parks.

British Columbia. 1992d. *Reforming Environmental Assessment in British Columbia*. Victoria, B.C.: Ministry of the Environment, Public Lands and Parks.

British Columbia. 1992e. *Revising British Columbia's Waste Discharge Permit Fee System: A Discussion Paper*. Victoria, B.C.: Ministry of the Environment, Public Lands and Parks.

Canadian Council of Ministers of the Environment (CCME). 1992. *Emission Trading : A Discussion Paper*. Winnipeg: CCME. CCME-EPC/ET-40E.

Busch, J.-D. 1989. "Environmental Policy: Its Amins and Instruments in the Federal System of West Germany." In W.M. Chandler and C.W. Zöllner. 1989. *Challenges to Federalism: Policy-Making in Canada and the Federal Republic of Germany*. Kingston, Ontario: Queen's University, Institute of Intergovernmental Relations.

Cansier, D. 1975. *Ökonomische Grundprobleme der Umweltpolitik*. Berlin: Erich Schmidt Verlag.

Cassils, A.J. 1991. *Exploring Incentives: An Introduction to Incentives and Economic Instruments for Sustainable Development*. Ottawa: National Roundtable on the Environment and the Economy.

Chandler, W.M. 1989. "Challenges to Federalism: Comparative Themes." In W.M. Chandler and C.W. Zöllner (ed). 1989. *Challenges to Federalism: Policy-Making in Canada and the Federal Republic of Germany*. Kingston, Ontario: Queen's University, Institute of Intergovernmental Relations

Chandler, W.M. and H. Bakvis. 1989. "Federalism and the Strong-State / Weak-State Conundrum: Canadian Economic Policymaking in Comparative Perspective." *Publius*, Vol. 19, Winter 1989, pp. 59 - 77.

Chandler, W.M. and C.W. Zöllner. 1989. *Challenges to Federalism: Policy-Making in Canada and the Federal Republic of Germany*. Kingston, Ontario: Queen's University, Institute of Intergovernmental Relations

Council of Forest Industries of British Columbia. 1992. Submission to Steven Martin, Acting Director, Evaluation Economics & Laboratory Services Branch, Environmental Protection Division, B.C. Ministry of Environment Lands and Parks, dated April 14, 1992. Vancouver: COFI.

Crandall, R.W. 1983. *Controlling Industrial Pollution*. Washington, DC: Brookings Institution.

Dales, J. 1968. *Pollution, Property and Prices*. Toronto: University of Toronto Press.

Dewees, D.N. 1983. "Instrument Choice in Environmental Policy." *Economic Inquiry*, Vol. 21, No. 1, pp. 53-71.

Dewees, D.N. 1992. "Taxation and the Environment." In R. Bird and Mintz, J. (ed). 1992. *Taxation to 2000 and Beyond*. Toronto: Canadian Tax Foundation.

Doern, B.G. 1990a. *Getting it Green: Case Studies in Canadian Environmental Regulation*. Toronto: C.D. Howe Institute.

Doern, B.G. 1990b. "Regulations and Incentives: The NOx-VOCs Case." In B.G. Doern. 1990a. *Getting it Green: Case Studies in Canadian Environmental Regulation*. Toronto: C.D. Howe Institute.

Doern, B.G. and V.S. Wilson. 1974. *Issues in Canadian Public Policy*. Toronto:

Dorcey, A. 1986. *Bargaining in the Governance of Pacific Coastal Resources*. Vancouver: Westwater Research Centre, University of British Columbia.

Dorcey, A. 1987. " The Managemnet of Super, Natural British Columbia." *B.C. Studies*, No. 73, Spring 1987, pp. 14 - 42.

Downing, P.B. 1983. "Bargaining in Pollution Control." *Policy Studies Journal*, Vol. 11, pp. 577 - 586.

Dudek, D.J. and J. Palmisano. 1988. "Emissions Trading: Why is this Thoroughbred Hobbled?" *Columbia Journal of Environmental Law*, Vol. 13, pp. 217 - 256.

Dwivedi, O.P. and R.B. Woodrow. 1989. "Environmental Policy-Making and Administration in a Federal State; The Impact opf Overlapping Jurisdiction in Canada." In W.M. Chandler and C.W. Zöllner. 1989. *Challenges to Federalism: Policy-Making in Canada and the Federal Republic of Germany*. Kingston, Ontario: Queen's University, Institute of Intergovernmental Relations.

Dyson, K. 1982. "West Germany: The Search for Rationalist Consensus." In J. Richardson (ed). 1982. *Policy Styles in Western Europe*. London: George Allen & Unwin.

Environment Canada. 1992. *Economic Incentives for Environmental Protection*. Ottawa: Minister of Supplies and Services.

Evans, P.B., D. Rueschmeyer and T. Skocpol. 1985. Bringing the State Back In. New York: Cambridge University Press.

Fast, D.A. 1992a. Letter to permit holders dated July 31, 1992. Mr. Fast is Executive Director of the Environmental Protection Division in the B.C. Ministry of Environment.

Fast, D.A. 1992b. Letter to permit holders dated October 28, 1992.

Foley, Dermit. 1992. Interview in November 1992. Mr. Foley is a researcher with the Society for Protection of the Environment and Conservation (SPEC) in Vancouver.

Forschungsgruppe Umwelt. 1991. *Die Behördliche Praxis bei der Definition und Definition vom Umweltstrafsachen*. Bonn, Germany: Institute for Criminology, University of Bonn.

Freeman, A.M. 1990. "Water Pollution Policy." In P.R. Portney (ed) 1990a. *Public Policies for Environmental Protection*. Washington, DC: Resources for the Future.

Freeman, A.M., R.H. Haveman, and A.V. Kneese. 1973. *The Economics of Environmental Policy*. New York: John Wiley & Sons.

Friedlander, A.F. 1978. *Approaches to Controlling Air Pollution. Cambridge*, MA: MIT Press.

Gaines, S.E. and R.A. Westin. 1991. *Taxation for Environmental Protection*. Westport, CT: Quorum Books

Goldberg, V.P. "Institutional Change and the Quasi-Invisible Hand." *Journal of Law and Economics*. Vol. 17, No. 2, pp. 461 - 496.

Greiffenhagen, M. 1984. "Vom Obrigkeitsstaat zur Demokratie: Die Politische Kultur in der Bundesrepublik Deutschland." In P. Reichel (ed). 1984. *Politische Kultur in Westeuropa: Bürger und Staaten in der Europäischen Gemeinschaft*. Frankfurt/Main: Campus Verlag.

Government of Canada. 1990. *Canada's Green Plan*. Ottawa: Minister of Supplies and Services.

Hahn, R.W. 1983. "Designing Markets in Transferable Property Rights: A Practitioner's Guide." In E.Joeres and M. David (eds). 1983. *Buying a Better Environment.* Land Economics Monograph #6. Madison: University of Wisconsin Press.

Hahn, R.W. 1984. "Market power and transferable property rights." *Quarterly Journal of Economics*, Vol. 99, pp. 753-765.

Hahn, R.W. 1986a. "Tradeoffs in Designing Markets with Multiple Objectives." *Journal of Environmental Economics and Management*, Vol. 13, No. 1, pp. 1 - 13.

Hahn, R.W. 1987. "Jobs and environmental quality: Some implications for instrument choice." *Policy Sciences*, Vol. 20, No. 3, pp. 289-306.

Hahn, R.W. 1989a. "Economic prescriptions for environmental problems: How the patient followed the doctor's orders." *Journal of Economic Perspectives*, Vol. 3, No. 2, pp. 95 - 114.

Hahn, R.W. 1989b. *A Primer on Environmental Policy Design.* New York: Harwood Academic Publishers.

Hahn, R.W. 1990a. "The political economy of environmental regulation: Towards a unifying framework." *Public Choice*, No. 65, pp. 21 - 47.

Hahn, R.W. and A.M. McGartland. 1989. "The political economy of instrument choice: An examination of the U.S. role in implementing the Montreal Protocol." *Northwestern University Law Review*, Vol. 83, No. 3, pp. 591 - 611.

Hahn, R.W. and G.L. Hester. 1989a. "Marketable Permits: Lessons for Theory and Practice." *Ecology Law Quarterly*, Vol. 16, pp. 361 - 406.

Hahn, R.W. and G.L. Hester. 1989b. "Where did all the Markets go? An Analysis of EPA's Emissions Trading." *Yale Journal on Regulation*, Vol. 6, No. 1, pp. 109 - 153.

Hahn, R.W. and R. Noll. 1982. "Designing a Market for Tradeable Emission Permits." In W. Magat (ed). 1982. *Reform of Environmental Regulation.* Cambridge, MA: Ballinger Publishing Company.

Hahn, R.W. and R.N. Stavins. 1991. "Incentive-based Environmental Regulation: A New Era from an Old Idea?" *Ecology Law Quarterly*, Vol. 18, pp. 1 - 42.

Hall, P. 1986. *Governing the Economy: The Politics of State Intervention in Britain and France.* New York: Oxford University Press.

Harrison, K. 1991. *Federalism, Environmental Protection, and Blame Avoidance.* Paper presented at the Annual Meeting of the Canadian Political Science Association, Kingston, June 2 - 4, 1991.

Hoberg, G. 1991a. *Representation and Governance in Canadian Environmental Policy.* Paper prepared for delivery at "Governing Canada: Public Policy and Political Institutions", a conference at McMaster University, Hamilton, Ontario, October 25-26, 1991.

Hoberg, G. 1991b. *Governing the Commons: Environmental Policy in Canada and the United States.* Paper prepared for delivery at the symposium on "Canada and the United States in a Changing World" at Queen's University, November 1-2, 1991.

Hoberg, G. 1992. *Pluralism by Design: Environmental Policy and the American Regulatory State.* New York Praeger.

Hucke, J. 1985. "Environmental Policy: The Development of a New Policy Area." In K.v. Beyme. 1985. *Policy and Politics in the Federal Republic of Germany.* Hants, England: Gower Publishing Company Ltd.

Jessup, D.H. 1990. *Guide to State Environment Programs.* Washington, DC: The Bureau of National Affairs Inc.

Jones, C.O. 1984. *An Introduction to the Study of Public Policy.* Montery, CA: Brooks/Cole Publishing Company.

Jones, E. and M. David. 1983. *Buying a Better Environment.* Madison, WI: University of Wisconsin Press.

Jorgenson, D.W. and P.J. Wilcoxen. 1990. "Environmental regulation and U.S. economic growth." *Rand Journal of Economics*, Vol. 21, No. 2, pp. 314 - 339.

Kagan, R.A. 1989. "Editor's Introduction: Understanding Regulatory Enforcement." *Law and Policy*, Vol. 11, No. 2, pp. 89 - 119.

Kaufman, F.-X., G. Majone, and V. Ostrom. 1986. *Guidance, Control, and Evaluation in the Public Sector.* Berlin: Walter de Gruyter.

Kelman, S. 1981. *What Price Incentives? Economists and the Environment.* Boston, MA: Auburn House.

Kelman, S. 1983. "Economic Incentives and Environmental Policy: Politics, Ideology, and Philosophy." In: T. Schelling (ed) 1983. *Incentives for Environmental Protection.* Cambridge, MA: MIT Press.

King, A. 1973. "Ideas, Institutions and the Policies of Governments A Comparative Analysis." *British Journal of Political Science*, Vol. 3, No. 4, pp. 409 - 424.

Kingdon, J.W. 1984. *Agendas, Alternatives, and Public Policies*. Toronto: Little Brown Company.

Kloepper, M. 1989. *Umweltrecht*. Munich: Beck'sche Verlagsbuchhandlung.

Kneese, A.V. and C.L. Schultze. 1975. *Pollution, Prices and Public Policy*. Washington, DC: Brookings Institution.

Komareck, P. 1986. *European Environmental Policy in Practice: Federal Republic of Germany: Water and Waste*. London: Graham & Trotman.

Landau, J.L. 1979. "Who owns the air? The emission offset concept and its implications." *Environmental Law*, Vol. 9, pp. 575 - 600.

Latham, E. 1952. "The Group Basis for Politics." *American Political Science Review*, Vol. 65, pp. 376 - 397.

Lave, L.B. and G.S. Omenn. 1981. *Clearing the Air: Reforming the Clean Air Act*. Washington, DC: Brookings Institution.

Leipert, Ch. and Simonis, U. 1989. "Environmental Protection Expenditures: The German Example." *Rivista Internazionale di Scienze Economiche e Commerciali*, Vol. 36, No. 3, pp. 255 - 270.

Linder, S.H. and M.E. McBride. 1984. "Enfocrement costs and regulatory reform: The agency and firm response." *Journal of Environmental Economics and Management*, Vol. 11, pp. 327 - 346.

Lipset, M.S. 1989. *The Continental Divide: Values and Institutions in the United States and Canada*. New York: Routledge.

Lockhart, Brian. 1992. Interview on October 23, 1992. Mr. Lockhart is B.C. Regional Manager of the Canadian Chemical Producers Association.

Lowi, T. 1964. "American Business, Public Policy, Case-Studies, and Political Theory." *World Politics*, Vol. 16, pp. 667 - 713.

Majone, G. 1976. "Choice among Policy Instruments for Pollution Control." *Policy Analysis*, Vol. 2, pp. 589 - 613.

Mäler, K-G. 1974. *Environmental Economics: A Theoretical Inquiry.* Baltimore: Johns Hopkins University Press.

Malik, A.S. 1990. "Markets for pollution control when firms are noncompliant." *Journal of Environmental Economics and Management,* Vol. 18, pp. 97 - 106.

Marcus, A. 1980. *Promise and Performance: Choosing and Implementing an Environmental Policy.* Westport, CT: Greenwood Press.

Mazmanian, D.A. and P.A. Sabatier. 1982. *Effective Policy Implementation.* Toronto: D.C. Heath and Company.

McCloy, Brian. 1992. Interview October 23, 1992. Mr. McCloy is Environment Manager of the Council of Forest Industries of British Columbia.

March. J. and J. Olsen. 1984. "The New Institutionalism: Organizational Factors in Political Life." *American Political Science Review,* Vol. 78, No. 3, pp. 734 - 740.

Mcdonald, D. 1991. *The Politics of Pollution: Why Canadians Are Failing Their Environment.* Toronto: McClelland & Stewart Inc.

Meidinger, E. "On Explaining the Development of 'Emissions Trading' in U.S. Air Pollution Regulation." *Law and Policy,* Vol. 7, No. 4, pp. 447 - 479.

Melnick, R. 1983. *Regulation and the Courts.* Washington, DC: Brookings Institution.

Melnick, R. 1985. "The Politics of Partnership." *Public Administration Review,* Vol. 45, pp. 653 - 660.

Miller, J.R. and L. Miller. 1988. "Principles of environmental economics and the political economy of West German environmental policy." *Environment and Planning, Government and Policy,* Vol. 6, pp. 457 - 474.

Misolek, W.S. 1980. "Effluent taxation in monopoly markets." *Journal of Environmental Economics and Management,* Vol. 7, pp. 103-107.

Moe, T.M. 1991. "Politics and the theory of organization." *Journal of Law, Economics, and Organization,* Vol. 7, pp. 106-129.

Morley, T.J, N.J. Ruff, Swainson, R.J. Wilson and W.D. Young. 1983. *The Reins of Power: Governing British Columbia.* Vancouver: Douglas & McIntyre.

Muller-Rommel, F. 1990. "Interest Group Representation in the Bundestag." In U. Thaysen, R.H. Davidson, R.G. Livingston (ed). 1990. *The U.S. Congress and the German Bundestag.* Boulder, CO: Westview Press.

Niskanen, W. 1973. *Bureaucracy: Servant or Master?* London: Institute of Economic Affairs.

Noll, R. 1983. "The political foundations of regulatory policy." *Zeitschrift für die gesamte Staatswissenschaft*, Vol. 139, No. 3, pp. 377-404.

OECD. 1989. *Economic Instruments for Environmental Protection.* Paris: OECD.

Olewiler, N. 1990. "The Case for Pollution Taxes." In B.G. Doern. 1990a. *Getting it Green: Case Studies in Candaian Environmental Regulation.* Toronto: Howe Institute.

Olewiler, N. 1992. "Comment: Pollution Taxes - A Chance for a New Environmental Policy." In R. Bird and Mintz, J. (ed). 1992. *Taxation to 2000 and Beyond.* Toronto: Canadian Tax Foundation.

Paehlke, R. 1990. "Regulatory and non-regulatory approaches to environmental protection." *Canadian Public Administration*, Vol. 33, No.1 pp. 17-36.

Pigou, A.C. 1932. *The Economics of Welfare.* Fourth edition. London: MacMillan & Company.

Portney, P.R. 1988. "Reforming Environmental Regulation: Three Modest Proposals." *Columbia Journal of Environmental Law*, Vol. 13, pp. 200 - 215.

Portney, P.R. (ed) 1990a. *Public Policies for Environmental Protection.* Washington, DC: Resources for the Future.

Portney, P.R. 1990b. "The Evolution of Federal Regulation." In P.R. Portney (ed) 1990a. *Public Policies for Environmental Protection.* Washington, DC: Resources for the Future.

Portney, P.R. 1990c. "Air Pollution Policy." In P.R. Portney (ed) 1990a. *Public Policies for Environmental Protection.* Washington, DC: Resources for the Future.

Portney, P.R. 1992. "The Economics of Hazardous Waste Regulation." In *U.S. Waste Management Policies: Impact on Economic Growth and Investment Strategies.* Washington, DC: American Council for Capital Formation.

Rehbinder, E. 1973. *Politische und rechtliche Probleme des Verursacherprinzips.* Berlin: Springer Verlag.

Rehbinder, E. and R.-U. Sprenger. 1985. *Möglichkeiten und Grenzen der Übertragbarkeit neuer Konzepte der US-amerikanischen Luftreinhaltepolitik in den Bereich der deutschen Umweltpolitik.* Berlin: Umweltbundesamt / Erich Schmidt Verlag.

Rehbinder, E. and R. Stewart. 1985. *Integration Through Law: Europe and the American Federal Experience. Volume 2: Environmental Protection Policy.* New York: Walter de Gruyter.

Reese, C.E. 1983. *Deregulation and Environmental Quality.* Westport, CT: Greenwood Press.

Reichel, P. (ed). 1984. *Politische Kultur in Westeuropa: Bürger und Staaten in der Europäischen Gemeinschaft.* Frankfurt/Main: Campus Verlag.

Richardson, J. (ed). 1982. *Policy Styles in Western Europe.* London: George Allen & Unwin.

Richardson, J., G. Gustafsson, and G. Jordan. 1982. "The Concept of Policy Style." In J. Richardson (ed). 1982. *Policy Styles in Western Europe.* London: George Allen & Unwin.

Russel, C.S. 1979. "What Can We Get from Effluent Charges?" *Policy Analysis*, Vol. 5, No. 2, pp. 155 - 180.

Schleicher, H. 1986. "Building Coordination Structures." In F.-X. Kaufman, G. Majone, and V. Ostrom. 1986. *Guidance, Control, and Evaluation in the Public Sector.* Berlin: Walter de Gruyter.

Schröder, M. 1983. "Lenkungsabgaben am Beispiel der Umweltabgabe." *Die Öffentliche Verwaltung*, No. 16, August 1983, pp. 667 - 674.

Simonis, U.E. 1992. *Environmental Policy in the Federal Republic of Germany.* Berlin: Wissenschaftszentrum (Science Centre).

Stavins, R.N. 1990. "Innovative Policies for Sustainable Development: The Role of Economic Incentives for Environmental Protection." *Harvard Public Policy Review*, Vol. 7, pp. 13-25.

Stavins, R.N. and B.W. Whitehead. 1992. "Dealing with Pollution." *Environment*, Vol. 34, No. 7, pp. 7 -11, 29 - 42.

Stewart, R.B. 1985. "The Discontents of Legalism: Interest Group Relations in Administrative Regulation." *Wisconsin Law Review*, No. 3, pp. 655-686.

Stewart, R.B. 1988. "Controlling Environmental Risks Through Economic Incentives." *Columbia Journal of Environmental Law*, Vol. 13, pp. 153-169.

Stolz, M. 1992. Interview, May 1992. Mr. Stolz is Ministerial Director for Waste Policy in the Federal Ministry for Environment, Nature Conservation, and Nuclear Safety, Bonn, Germany.

Stone, M. 1990. *Environmental Excise Taxes: Options for British Columbia.* Victoria: Ministry of Environment, Public Lands, and Parks.

Tietenberg, T. 1980. "Transferable Discharge Permits and the Control of Stationary Source Air Pollution; A Survey and Synthesis." *Land Economics*, Vol. 56, pp. 391 - 416.

Tietenberg, T. 1985. *Emissions Trading: An Exercise in Reforming Pollution Policy.* Washington, D.C.: Resources for the Future.

Trebilcock, D.G., R.S. Hartle and D.N. Dewees. 1982. *The Choice of Governing Instrument.* Ottawa: Minister of Supply and Services.

U.S. Environmental Protection Agency. 1987. *An Assessment of EPA's Negotiated Rulemaking Activities.* Washington, DC: EPA (Program Evaluation Division, Office of Management Systems and Evaluation, Office of Policy Planning and Evaluation).

Vogel, D. and V. Kun. 1987. "The Comparative Study of Environmental Policy." In M. Dierkes, H.N. Weiler, and A.B. Antal. 1987. *Comparative Policy Research: Learning by Experience.* Berlin: WZB Publications.

Wallace, H. 1992. Interview. Mr. Wallace works with the Air Resources Division of the B.C. Ministry of Environment, Lands and Parks, Victoria.

Weaver, K. 1986. "The Politics of Blame Avoidance." *Journal of Public Policy*, Vol. 6, No. 4, pp. 371 - 398.

Weitzman, M.L. 1974. "Prices vs. Quantities." *Review of Economic Studies.* Vol. 41, No. 4, pp. 477-491.

Weizsäcker, E.-U. 1988. "Regulatory Reform and the Environment: The Cause for Environmental Taxes." in Majone, G. 1988. *Deregulation or Re-regulation? Regulatory Reform in Europe and the United States.* New York: St. Martin's Press.

Welch, W.P. 1983. "The political feasibility of full ownership property rights: The cases of pollution and fisheries." *Policy Sciences*, Vol. 16, pp. 165-180.

West Coast Envrionmental Law Association. 1992. *Comments on "Revising British Columbia's Waste Discharge Permit Fee System: A Discussion Paper"*, dated March 31, 1992. Vancouver: WCELA (mimeo).

Westin, R.A. and S.E. Gaines. 1991. "Environmental Taxes in the United States." In S.E. Gaines and R.A. Westin. 1991. *Taxation for Environmental Protection.* Westport, CT: Quorum Books

Wilson J. 1990. "Wilderness Politics in B.C.: The Business Dominated State and the Containment of Environmentalism." In: W. Coleman and G. Skogstad (ed). 1990. *Policy Communities and Policy Networks.* Missasauga: Copp Clark Pitman.

Wilson, J.Q. 1980. *The Politics of Regulation.* New York: Basic Books.

Wilson, M.L. L.M. Martins, and D.M. Friedland. 1990. "A Critical Review of the Environmental Protection Agengy's Standards for "Best Available Control Technology" Under the Clean Air Act." *Environmental Law Reporter, News and Analysis,* Vol. 20, No. 2, pp. 10067 - 10076.

Wildavsky, A. 1979. *Speaking Trith to Power: The Art and Craft of Policy Analysis.* Toronto: Little Brown and Company.

Wirth, T.E. and J. Heinz. 1988. *Project 88 - Harnessing Market Forces To Protect Our Environment: Initiatives for the New President.* Washington, DC: mimeo.

Wolozin, H. (ed). 1966. *The Economics of Air Pollution,* New York: W.W. Norton.

Woodside, K. "Policy Instruments and the Study of Public Policy." *Canadian Journal of Political Science,* Vol. 19, No. 4, pp. 775 - 793.

Yandle, B. 1978. "The Emerging Market in Air Pollution Control." *Regulation,* July / August 1978, pp. 21 - 29.

Zezschwitz, F.v. 1991. "Environmental Taxes in Germany." In S.E. Gaines and R.A. Westin. 1991. *Taxation for Environmental Protection.* Westport, CT: Quorum Books.

Newspaper articles

Business Week

Gleckman, H. and V. Cahan. 1990. "Will Eco-Tax Fervor Sweep Congress Off its Feet?" *Business Week,* April 30, 1990, p. 28

Der Spiegel

Der Spiegel. 1976a. "Hohe Schwermetallwerte im Bodensee." *Der Spiegel,* Vol. 30, No. 49, p. 246.

Der Spiegel. 1976b. "Ablasshandel zu Billigstpreisen." *Der Spiegel,* Vol. 30, No. 27, pp. 62 - 68.

Globe and Mail

Miller, D. 1990. "Gung-ho for Green." *Globe and Mail*, December 11, 1990, p. A15.

Montreal Gazette

Montreal Gazette. 1990a. "Most would Dig Deep for a Green Canada." *Montreal Gazette*, May 31, 1990, p. B6.

Montreal Gazette. 1990b. "Federal Advisers Silent on Green-Tax Recommendation." *Montreal Gazette*, August 22, 1990, p. B1.

New Scientist

MacKenzie, D. 1990. "...as Europe's Ministers Fail to Agree on Framework for Green Taxes." *New Scientist*, September 29, 1990, p. 17.

New York Times

Dowd, M. 1989. "Bush in Ottawa, Now Vows to Pusue Acid Rain Pact." *New Tork Times*, February 11, 1989, p. A3.

Gold, A.R. 1989. "Acid Rain Measure Could Divide Cost." *New York Times*, November 3, 1989, p. A21

Possel, P. 1989. "Sale of Air Pollution Permits is Part of Bush Acid Rain Plan." *New York Times*, May 17, 1989, p. A1.

Schneider, K. 1990. "Lawmakers Agree On Rules To Reduce Acid Rain Damage." *New York Times*, October 22, 1990, p. A1.

Shabecoff. P. 1990a. "Several Governors Oppose Plan on Acid Rain." *New York Times*, January 3, 1990, p. A19.

Shabecoff, P. 1990b. "Senators Approve Clean Air Measure by a Vote of 89-11." *New York Times*, April 4, 1990, p. A1.

The Economist

The Economist. 1989. "The Greening of George Bush." *The Economist*, June 17, 1989, pp. 29-32.

The Economist. 1990. "Green Taxes: Where there' Muck there's Brass." *The Economist*, March 17, 1990, pp. 46 - 47.

Toronto Star

Spears, T. 1989. "$40 Billion Fuel Tax Urged to Cut Pollution." *Toronto Star*, June 23, 1989, p. A1.

Toronto Star. 1990. "$100 Environment Tax Backed by 54% in Poll." *Toronto Star*, May 31, 1990, p. A1,A13.

Appendix 1: Tables

Table 1: Arenas and political relationships: a diagrammatic summary (Lowi 1964: 713)

Arena	*Primary Political Unit*	*Relation Among Units*	*Power Structure*	*Stability of Structure*	*Primary Decisional Locus*	*Implementation*
Distribution	Individual, firm, corporation	Log-rolling, mutual non-interference, uncommon interests	Non-conflictual elite with support groups	Stable	Congressional committee and/ or agency**	Agency centralized to primary functional unit ("bureau")
*Regulation**	Group	"The coalition," shared subject-matter interest, bargaining	Pluralistic, multi-centered, "theory of balance"	Unstable	Congress, in classic role	Agency decentralized from center by "delegation," mixed control
Redistribution	Association	The "peak association," class, ideology	Conflictual elite, i.e., elite and counterelite	Stable	Executive and peak associations	Agency centralized toward top (above "bureau"), elaborate standards

* Given the multiplicity of organized interests in the regulatory arena, there are obviously many cases of successful log-rolling coalitions that resemble the coalitions prevailing in distributive politics. In this respect, the difference between the regulatory and the distributive arenas is thus one of degree. The *predominant* form of coalition in regulatory politics is deemed to be that of common or tangential interest. Although the difference is only one of degree, it is significant because this prevailing type of coalition makes the regulatory arena so much more unstable, unpredictable, and non-elitist ("balance of power"). When we turn to the redistributive arena, however, we find differences of principle in every sense of the word.

** Distributive politics tends to stabilize around an institutional unit. In most cases, it is the Congressional committee (or subcommittee). But in others, particularly in the Department of Agriculture, the focus is the agency or the agency *and* the committee. In the cities, this is the arena where machine domination continues, if machines were in control in the first place.

Table 2: Authority over air and water policy in selected OECD countries

Canada:
Air: Federal and provincial legislation, uneven regulation, implementation, and enforcement.
Water: Federal and provincial legislation, uneven regulation, implementation and enforcement.

Denmark
Air: National laws, implementation and administration by local authorities; local decisions can be reviewed by National Agency for Environmental Protection (NAEP).
Water: National laws, implementation and administration by local authorities; no NAEP reviews; enforcement is lenient due to strong business pressure on local authorities.

France
Air: Some national laws, implementation and administration by local or regional governments; extensive regulatory discretion at lower government levels; policies between governments are not well coordinated.
Water: National legislation, implementation through River Basin Agencies, strong local control; enforcement problems

Germany
Air: Federal laws and regulations, implementation through state bureaucracies; no national strategy to combat air pollution.
Water: Federal framework legislation augmented by state laws, problems in policy coordination, uneven enforcement.

Italy
Air: Some national laws and decrees; monitoring and enforcement done by local and regional authorities; no comprehensive national policy.
Water: Some national laws augmented by policies of local and regional governments; ineffective inspections and enforcement.

The Netherlands
Air: National legislation, regulatory authority spread among numerous ministries, implementation through local governments and waterboards.
Water: National legislation, augmented by provincial and local regulations; administration through waterboards which also have rulemaking powers.

United States

Air: Federal legislation, additional state legislation; federal legislation preempts state legislation, implementation through EPA, or through state agencies upon EPA approval.
Water: Federal legislation, additional state legislation; federal legislation preempts state legislation, implementation through EPA, or through state agencies upon EPA approval.

Appendix 2: Figures

Figure 1: Incidence of a pollution tax if demand is price inelastic

The tax increases marginal costs by the tax rate *t*. The marginal cost curve MC_1 shifts left to MC_2. Since demand is price inelastic the price increase can be fully passed on to consumers. Prices rise by the tax rate from P_1 to P_2. Production does not change ($Q_1 = Q_2$). The firm's profits remain constant at OBA. The state collects tax revenue equal to $t * Q_2$.

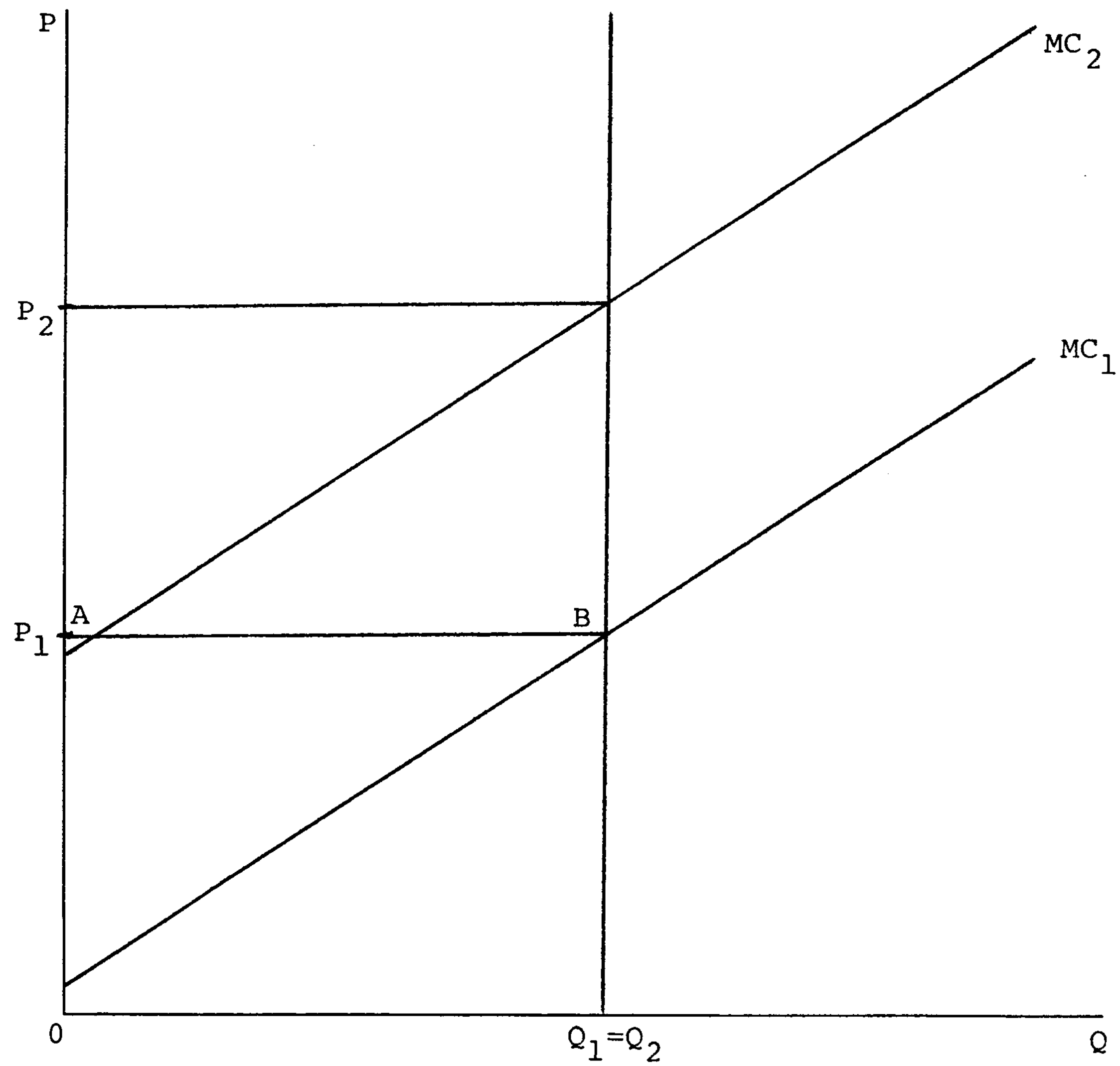


Figure 2: Incidence of a pollution tax if demand is price elastic

The tax increases the firms marginal cost by the tax rate. The marginal cost curve MC_1 shifts left to MC_2. Since demand is perfectly price elastic the producer cannot pass on the cost increase to consumers ($P_1 = P_2$). Production declines from Q_1 to Q_2. Profits decline from OCE to OBA. Tax revenues amount to $t * Q_2$.

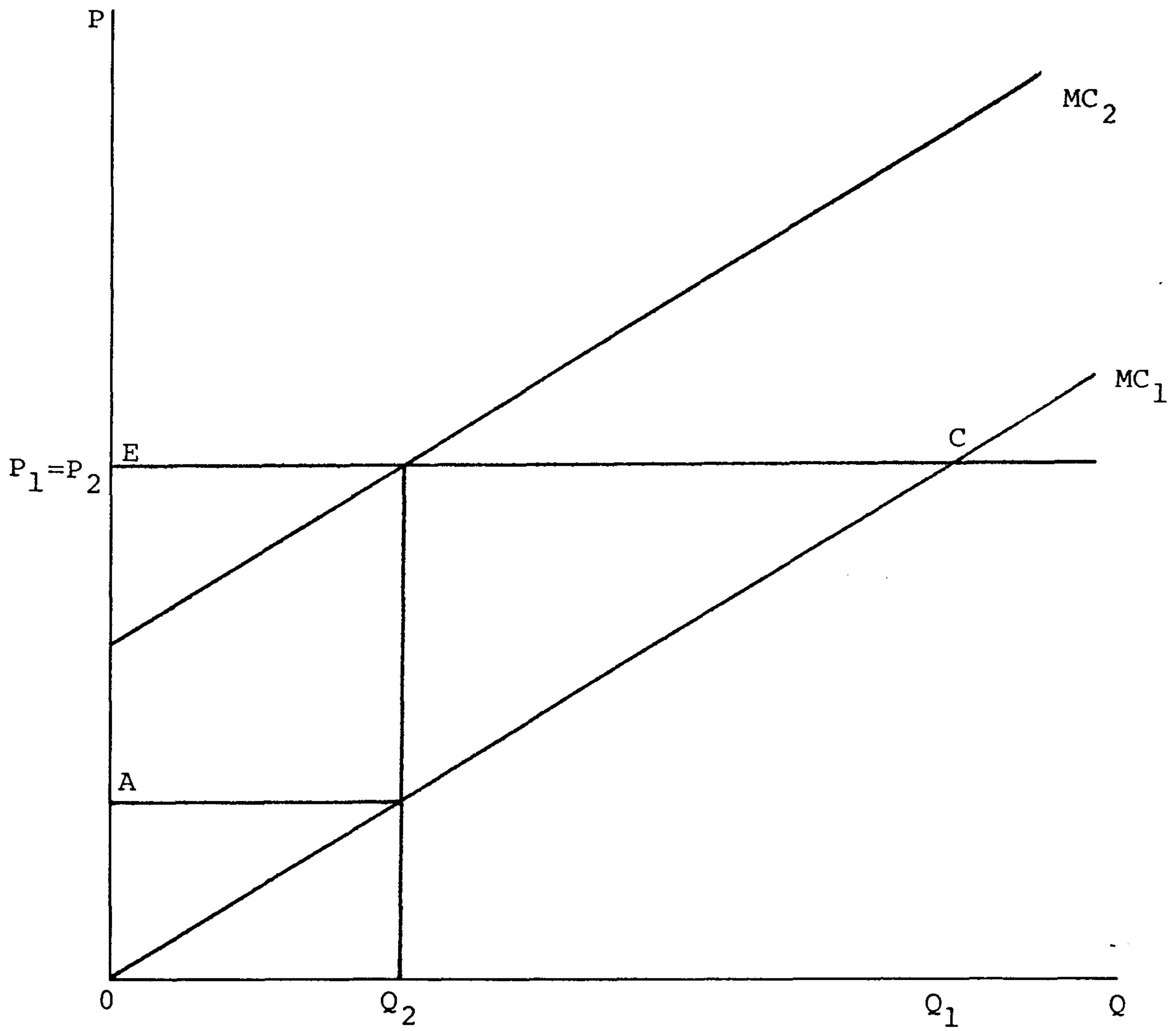


Figure 3: Incidence of emission trading if demand is price elastic

Trading in pollution rights lowers marginal costs, otherwise no trades are made. The marginal cost curve MC_1 moves to the right to MC_2. Since demand is perfectly price elastic, prices will not change ($P_1 = P_2$). However, production can be expanded from Q_1 to Q_2. Consequently, profits increase from ECD to ABD.

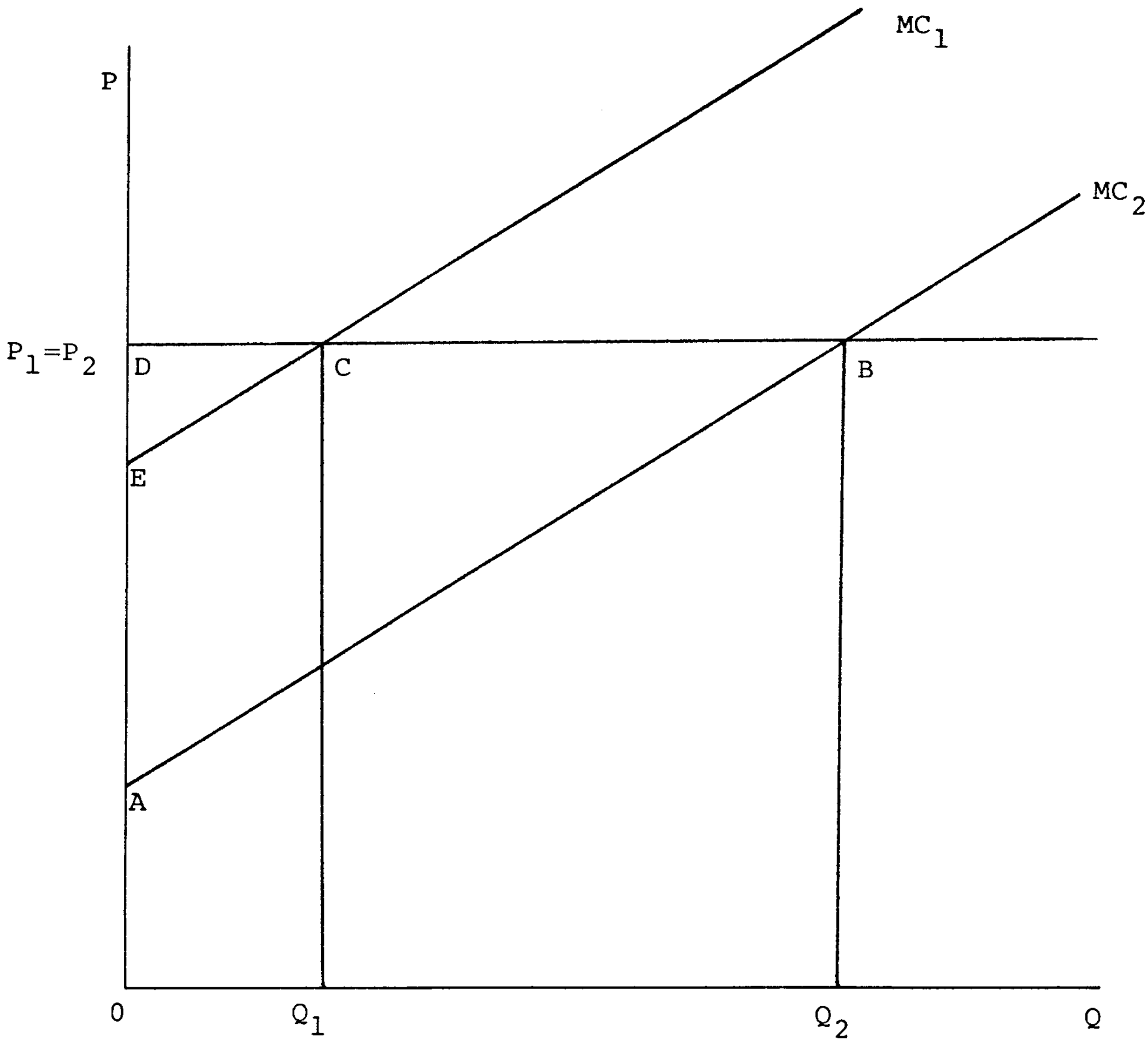
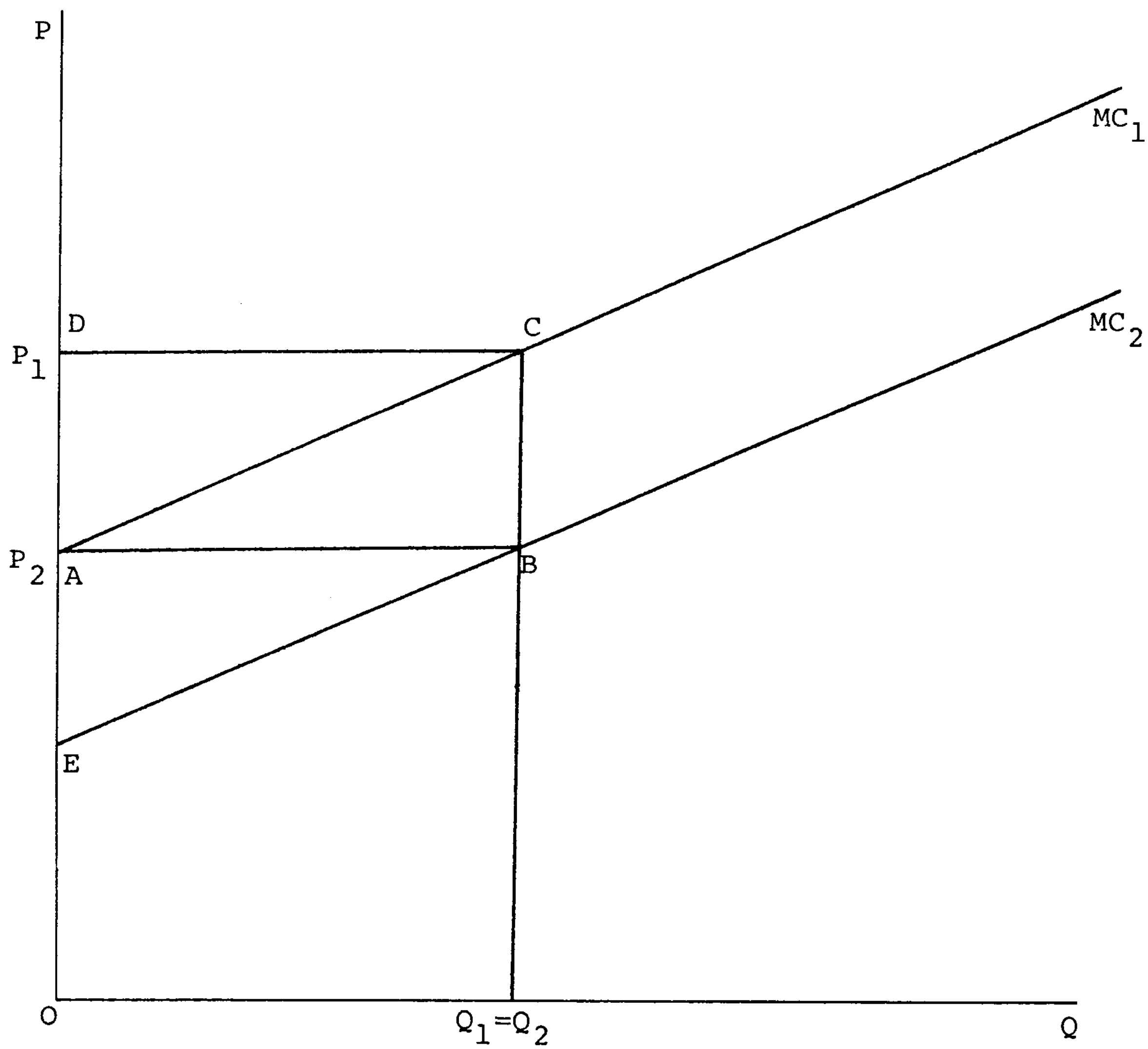


Figure 4: Incidence of emission trading if demand is price inelastic

Trading in pollution rights lowers marginal costs, otherwise no trades are made. The marginal cost curve MC_1 moves to the right to MC_2. Since demand is perfectly price inelastic production remains constant at $Q_1 = Q_2$. Production costs drop but the firm may not have to let consumers benefit and can continue to charge prices at P_1. Thus, profits increase from ACD to ABCD. However, in competitive markets the divergence of price and marginal costs will attract new firms into the market. In the long run, prices will fall from P_1 to P_2, and the company's profits will decline to ABE.



SCHEDULE "B"

Contaminant Fees For Air Emission Permits or Approvals

Column 1 Contaminant	Column 2 Fee Per Tonne Discharged	
	Sept. 1/92 - Aug. 31/93	After Aug. 31/93
Ammonia	$5.65	$11.30
Asbestos*	$5.65/Unit	$11.30/Unit
Carbon Monoxide	$0.15	$0.30
Chlorine and Chlorine Oxides	$3.80	$7.60
Fluorides	$226.80	$453.60
Hydrocarbons	$5.65	$11.30
Hydrogen Chloride	$3.80	$7.60
Metals	$226.80	$453.60
Nitrogen Oxides	$3.80	$7.60
Phenols	$5.65	$11.30
Sulphur and Sulphur Oxides	$4.40	$8.80
Total Particulate	$5.65	$11.30
TRS	$189.00	$378.00
VOCs	$5.65	$11.30
Other contaminants not otherwise specified	$5.65	$11.30

*Units of Asbestos are equivalent to 5 cubic metres of air emissions per minute at a concentration of 2 fibres per cubic centimeter.

SCHEDULE "C"

Contaminant Fees For Effluent Permits or Approvals

Column 1 Contaminant	Column 2 Fee Per Tonne Discharged	
	Sept. 1/92 - Aug. 31/93	After Aug. 31/93
Acute Toxicity*	$5.05/Unit	$10.10/Unit
Ammonia	$34.65	$69.30
AOX	$92.00	$184.00
Arsenic	$92.00	$184.00
BOD	$6.95	$13.90
Chlorine	$92.00	$184.00
Cyanide	$92.00	$184.00
Fluoride	$34.65	$69.30
Metals	$92.00	$184.00
Nitrogen and Nitrates	$13.85	$27.70
Oil and Grease	$23.10	$46.20
Other Petroleum Products	$23.10	$46.20
Other Solids	$4.60	$9.20
Phenols	$92.00	$184.00
Phosphorus and Phosphates	$34.65	$69.30
Sulphates	$1.35	$2.70
Sulphides	$92.00	$184.00
Surfactants	$23.10	$46.20
Suspended Solids	$4.60	$9.20
Other contaminants not otherwise specified	$4.60	$9.20

* Units of Acute Toxicity are determined using the following formula:

$$\textit{Units of Acute Toxicity} = \frac{\left(\frac{\textit{Average Daily Flow}}{20}\right) \cdot \left(100 - LC_{50}\right)}{100}$$

Appendix 4: A comparison of B.C. and German effluent fees

This table compares effluent charges in B.C. and Germany. The coverage of both fee schemes is not congruent. Here only those substances that occur in both schedules are listed. The German fees are the fees as of January 1, 1993. The B.C. fees are as of September 1, 1993. Until then, the B.C. fees are discounted by 50 per cent. All prices are in Canadian dollars based on an exchange rate of DM 1,00 = CDN 0,85.

Substance	German limit value[1]	German fee/t	German fee/t with max. discount[2]	B.C. fee/t
phosphorus & phosphates	0.1 mg/l 15 kg/y	11,600	2,900	69.30
nitrogen & nitrates	5 mg/l 125 kg/y	2,040	510	27.70
AO_x	100 ug/l[3] 10 kg/y	25,500	6,375	184.00
mercury	1 ug/l 100 g/y	2,550,000	637,000	184.00
cadmium	5 ug/l 500g/y	510,000	127,500	184.00
chromium	50 ug/l 2.5 kg/y	102,000	25,000	184.00
nickel	50 ug/l 2.5 kg/y	102,000	25,000	184.00
lead	50 ug/l 2.5 kg/y	102,000	25,000	184.00
copper	100 ug/l 5 kg/y	51,000	12,750	184.00

[1] The limit value demarks maximum concentrations and amounts below which the effluent charges do not apply. No similar provision exists in the B.C. regulations. However, the B.C. regulations only apply to waste discharge permit holders, whereas the German fees are payable by all entities that discharge directly into surface or groundwater.

[2] Dischargers employing best available technology are eligable for a fee reduction of up to 75 per cent of the pollution load that cannot not be controlled despite BAT. The fee reduction decreases to 20 per cent over eight years and than ceases to apply unless new technology has been installed.

[3] ug = microgram