Economic Analysis of Tradeable Emission Permits for Sulphur Dioxide Emissions in Europe

Sonja Kruitwagen

STELLINGEN

1. Handel in emissierechten voor niet uniform verspreide emissies moet op één of andere manier gerestricteerd worden om hoge lokale vervuilingsconcentraties te voorkomen.

2. De depositiedoelen voor verzuring in Nederland zijn impliciet gebaseerd op kosten baten analyse.

3. Er is sprake van een opmerkelijke paradox in internationale milieu overeenkomsten: naarmate internationale samenwerking noodzakelijker is, is deze moeilijker tot stand te brengen.
(Scott Barrett, 1990, The problem of global environmental protection, Oxford Review of Economic Policy, 6, pp. 68-79)

4. Om consumptiepatronen in een milieu-vriendelijker richting om te buigen moet de overheid in haar beleid veelvuldiger gebruik maken van financiële prikkels.

5. Indien Nederland streeft naar een substantiële participatie van vrouwen in het parlement, is het beter om het kiesstelsel van evenredige vertegenwoordiging te handhaven in plaats van over te stappen op een districtenstelsel.
(Joyce Outshoorn, 1995, Het hardnekkige verschil: de ondervertegenwoordiging van vrouwen in politiek en besluitvorming. In: Goldschmidt et.al., 1995, Feminisme en Wetenschap, Teleac/Prometheus, Utrecht/Amsterdam)

6. Flexibilisering van de arbeidsmarkt is in veel gevallen een eufemisme voor verslechtering van arbeidsvoorwaarden.

7. Voor het economie onderwijs aan 1e jaars studenten aan de Landbouwuniversiteit die gekenmerkt worden door een uiteenlopend kennisniveau met betrekking tot economie, is een individueel studiesysteem een geschikte onderwijsvorm.

8. Hardlopen bevordert zowel de lichamelijke als de geestelijke gezondheid.

9. Om de verruiming van winkelsluitingstijden op werkdagen ten volle te kunnen benutten, dient het sluitingstijdstip van de fietsenkelder van het LUW-gebouw de Leeuwenborch aangepast te worden.

10. Voor wie niet wil trouwen maar wel een feest wil geven, biedt een promotie uitkomst.

S. Kruitwagen
An economic analysis of tradeable emission permits for sulphur dioxide emissions in Europe
Wageningen, 10 december 1996

An Economic Analysis of Tradeable Emission Permits for Sulphur Dioxide Emissions in Europe

Sonja Kruitwagen

Promotoren: dr. H. Folmer
hoogleraar in de algemene economie
dr. L. Hordijk
bijzonder hoogleraar in de milieusysteemanalyse

Sonja Kruitwagen

An Economic Analysis of Tradeable Emission Permits for Sulphur Dioxide Emissions in Europe

Proefschrift
ter verkrijging van de graad van doctor
op gezag van de rector magnificus
van de Landbouwuniversiteit Wageningen,
dr. C.M. Karssen,
in het openbaar te verdedigen
op dinsdag 10 december 1996
des namiddags te vier uur in de Aula.

ISBN 90-5485-584-3

Ontwerp omslag: M. Vojt

VOORWOORD

Dit proefschrift is het resultaat van het NWO onderzoeksproject 'Bestrijding van grensoverschrijdende luchtverontreiniging als internationaal allocatievraagstuk' (450-230-006). De start van dit onderzoek vond plaats in september 1990 en de finish in de zomer van 1996.

Een aantal mensen heeft een bijzondere bijdrage geleverd aan dit onderzoek. Mijn promotoren, Henk Folmer en Leen Hordijk, wil ik bedanken voor hun begeleiding. Ondanks het feit dat concept teksten vaak op een laat moment werden toegestuurd, maakten ze ruimte in hun drukbezette agenda's om deze teksten van commentaar te voorzien. Ekko van Ierland wil ik hier in het bijzonder bedanken. Hij was vanaf het begin betrokken bij dit project en wist mij te motiveren en te stimuleren dit proefschrift tot een goed eind te brengen. Eligius Hendrix was bereid om - toen het op modelleren aankwam - mee te denken. Zijn hulp was onontbeerlijk voor de totstandkoming van de laatste hoofdstukken van dit proefschrift en hij voorzag me van raad over hoe de bevindingen te presenteren.

In de zomer van 1992 heb ik 3 maanden doorgebracht bij het IIASA in het kader van het Young Scientist's Summer Program. Hier werd een belangrijke stimulans aan dit onderzoek gegeven. Ger Klaassen, destijds werkzaam bij IIASA, wil ik bedanken voor zijn begeleiding.

Mevrouw Rigg-Lyall wil ik hartelijk bedanken voor het corrigeren van de engelse tekst. Haar aanwijzingen hebben het geheel leesbaarder gemaakt; blijvende onjuistheden zijn uiteraard voor mijn verantwoording.

Verder een woord van dank aan iedereen die de afgelopen jaren belangstelling toonde voor mijn onderzoek. Tenslotte een speciaal woord van dank aan Paul, die altijd bereid was mijn problemen aan te horen en waar mogelijk te helpen oplossen.

Sonja Kruitwagen
Wageningen, juli 1996

CONTENTS

1. INTRODUCTION

1.1	Introduction	1
1.2	Causes of acid rain	4
1.3	Atmospheric transport	7
1.4	Damage caused by acid rain	10
1.5	History of international acid rain policy	11
1.6	Aim of the study	18
1.7	Outline of the study	19

2. THEORETICAL PRINCIPLES FOR ACID RAIN POLICY

2.1	Introduction	21
2.2	Economic theory of environmental policy	21
2.3	International dimension: cooperation - non cooperation	26
2.3.1	Cooperation	26
2.3.2	Full cooperation with sidepayments	33
2.3.3	Uniform emission reduction	35
2.4	Choice of policy instruments	36
2.4.1	Introduction	36
2.4.2	Regulatory instruments	37
2.4.3	Economic instruments	38
2.4.4	Other instruments	42
2.4.5	Instruments compared for non-uniformly mixing pollutants	42
2.5	Conclusions	45

3. THEORY OF TRADEABLE EMISSION PERMITS

3.1	Introduction	47
3.2	Tradeable emission permits	48
3.3	Tradeable deposition permits	50
3.4	Alternative permit systems for non-uniformly mixing pollutants	52
	3.4.1 An overview	52
	3.4.2 Trading rules and trading ratios	57
3.5	Bilateral permit trading	63
3.6	Initial permit allocation	65
3.7	Permit trading in the U.S.	68
3.8	Conclusions	70

4. ACID RAIN MODELS

4.1	Introduction	73
4.2	The RAINS model	74
	4.2.1 Introduction to the RAINS model	74
	4.2.2 Energy and emissions	76
	4.2.3 Control technology and costs	82
	4.2.4 Atmospheric transport	89
	4.2.5 Environmental impacts	91
4.3	The ASAM Model	94
	4.3.1 Emissions and costs	94
	4.3.2 Effective emission reductions	95
4.4	The CASM Model	97
	4.4.1 Emissions	97
	4.4.2 Costs	98
	4.4.3 Critical loads	99
4.5	Non European acid rain models	100
4.6	Conclusions	103

5. PERMIT TRADING FOR NON-UNIFORMLY MIXING POLLUTANTS

5.1 Introduction 107
5.2 Cost effective emission abatement 110
5.3 Simultaneous permit trading 114
5.4 Bilateral permit trading 116
5.4.1 One-to-one emission trading 116
5.4.2 Offset rates 124
5.4.3 Optimum condition for bilateral trade 126
5.5 Guided bilateral trade 127
5.5.1 Introduction 127
5.5.2 Deriving guided bilateral trade 129
5.5.3 Extension: non-linear abatement cost functions 140
5.5.4 Guided bilateral trade and administrative procedures 144
5.6 Conclusions 145

6. SIMULATING GUIDED BILATERAL TRADE FOR SO_2 REDUCTIONS IN EUROPE

6.1 Introduction 147
6.2 The model 148
6.2.1 General description 148
6.2.2 Abatement cost functions 150
6.3 The Base Case 152
6.3.1 Basic assumptions 152
6.3.2 Results of guided bilateral trade 159
6.3.3 Alternative abatement strategies 169
6.3.4 Comparison of deposition levels 173
6.3.5 Distribution of abatement costs 173
6.4 Variant 1: Excluding current reduction plans 177
6.5 Variant 2: The Second SO_2 Protocol 181
6.6 Conclusions 192

7. SUMMARY AND CONCLUSIONS

7.1 Introduction 195

7.2 Economic theory of pollution control 195

7.3 Tradeable permits for non-uniformly mixing pollutants 196

7.4 Integrated assessment model 198

7.5 Guided bilateral trade 199

7.5.1 Theory 199

7.5.2 Simulation results 200

7.6 Guided bilateral trade revisited 201

7.7 Suggestions for further research 204

APPENDIX I 209

APPENDIX II 213

APPENDIX III 215

REFERENCES 217

SAMENVATTING 229

CURRICULUM VITAE 237

1 INTRODUCTION

1.1 Introduction

Increasing production and consumption have, in many cases, resulted in increased emissions of various pollutants which have led to large-scale damage to the environment. One of the major environmental problems in the 1980s and 1990s in Europe is acidification. Large-scale political and scientific awareness of this problem dates back to the early seventies. For years scientific research had indicated the seriousness of this problem, and abatement strategies were initiated both at the national and international level. In the past 25 years progress has been made, but acidification is still causing substantial environmental damage in parts of Europe, North America and Asia, and further reduction of acid rain seems necessary.[1]

Acid rain is caused by the emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3). The emissions of SO_2 and NO_x are mainly caused by burning fossil fuels, and emission of NH_3 is largely due to agricultural production. Once these acidifying pollutants have been emitted into the atmosphere, they are transformed and dispersed. A substantial portion of the pollutants is transported by the winds up to 1500 kilometres before being deposited. As a result, a country's deposition partly originates outside its geographical area. The extent to which this occurs differs between countries, because countries differ in size, wind patterns are not uniform and emission sources are not uniformly spread over countries. This non-uniformity is a main characteristic of acidifying emissions. Damage caused by acid rain not only depends on the amount of emission but also on the location of emissions.

Environmental damage caused by acidification depends on the level of acid deposition and the natural sensitivity of the environment. In recent years various

---

[1]The term 'acid rain' is used, although acidification can be caused by dry, wet or occult deposition.

types of damage have been represented by one indicator: the critical load (Nilsson and Grennfelt, 1988; Hettelingh et al., 1991). A critical load is defined as the deposition level below which no significant damage to the environment is expected to occur according to present knowledge (Nilsson and Grennfelt, 1988). These loads differ from location to location. Currently, deposition largely exceeds the critical loads in Europe. Simulation with the Regional Acidification Information and Simulation (RAINS) model indicates that, given countries' planned emission reductions, the critical loads will still be exceeded in the year 2000 in large parts of Europe (Amann et al., 1993).

This thesis deals with an economic analysis of acidification in Europe in as far as it is caused by SO_2 emissions. Particular emphasis is placed on cost-effective abatement of SO_2 emissions. This single pollutant approach is used in the actual international policy on acidification and the choice for SO_2 follows from the fact that, Europe wide, SO_2 is the main contributor to acid rain.

Because of the transboundary transport of acidifying emissions, acidification in Europe is an international environmental problem. Since countries are impacted by deposition originating abroad and because of reasons of effectiveness and efficiency, countries should cooperate in reducing acidifying emissions. In Europe this cooperation has resulted in two international sulphur protocols. The first dates from 1985 and required all signatories to reduce SO_2 emissions by 30% of their 1980 emission levels by 1993. The second dates from 1994 and is based on critical loads for sulphur the attainment of which would require a considerable emission abatement effort. This would result in high costs, since the cost of controlling acidifying emissions increase exponentially with increasing emission reductions beyond a moderate level of emission abatement (Alcamo et al., 1990). To minimize the total European abatement costs, the second sulphur protocol aimed at cost effective emission abatement. This resulted in differentiated national emission reductions.

To determine the cost-effective emission abatement allocation to reach critical loads, full information is needed on national control options and costs, on the atmospheric transport and deposition of emissions and on the critical loads. In general, countries with low marginal control costs should abate more emissions than countries with high marginal abatement costs to reach cost-effective abatement. Because of the atmospheric transport, emissions of upwind countries may cause more damage, implicating higher reductions for these countries. Also differences in critical loads require differences in deposition reductions and consequently differences in emission reductions among countries. In calculating the cost effective allocation all influencing factors should not be considered separately. What is needed is an integrated analysis. Therefore integrated assessment models, like the RAINS model can be used. The RAINS model was used in the negotiations on the second sulphur protocol. However, in spite of the fact that the protocol was partly based on cost-effective runs with the model, because during the negotiations the agreed emission reductions deviated from the simulated cost effective abatement allocation, the actual cost effectiveness can be improved. This deviation varies between countries to a maximum of 60% in Bulgaria (see further section 1.5).

Tradeable emission permits can be used to improve the cost effectiveness of SO_2 abatement. This is an interesting option since the second protocol allows countries to trade their emission reduction commitments. Moreover, in the United States a SO_2 permit trading system has already been introduced and based on first experiences, the system has been indicated as successful (Burtraw, 1996).

A major issue of research in this thesis is the question whether a system of tradeable SO_2 permits can contribute to the cost effectiveness of SO_2 emission reduction.[2] In the literature several systems for permit trading for non-uniformly mixing pollutants have been discussed, but for different reasons none of these is suitable for SO_2 permit trading in Europe. Unrestricted emission trading results in violation of deposition targets (Klaassen, 1995). For emission trading subject to rules, it is generally not clearly indicated how such rules should be implemented in separate trade transactions (Krupnick et al., 1983; Atkinson and Tietenberg, 1982; McGartland and Oates, 1985). Emission permit trading applying offset rates is untransparant and complex (Klaassen and Amann, 1992). Therefore, in this thesis a new system of tradeable emission permits, indicated as guided bilateral permit trading, is developed. This system aims at reaching deposition targets and

[2]Dispersion can be in water, soil and air. This thesis focusses on air pollution. For an economic analysis of tradeable permits for water pollution see for example Netusil and Braden (1995).

cost-effective emission abatement simultaneously, and trade restrictions are straightforward. Based on simulation results of guided bilateral permit trading for SO_2 emissions in Europe, this thesis indicates the consequences of this trading system.

Before considering tradeable emission permits as a policy instrument for the acid rain problem in Europe, I first look at the characteristics of this problem. In section 1.2 attention is paid to the causes of acid rain. Section 1.3 deals with the atmospheric transport of acidifying emissions, while section 1.4 discusses the resulting damage. As this thesis deals with international acid rain policy, the history of this policy is discussed in section 1.5. The aim of the study is formulated in section 1.6. The Introduction concludes in section 1.7 with a detailed outline of the study.

1.2 Causes of acid rain

The term acid rain refers to the transboundary environmental problem caused by the emissions of sulphur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3). After being emitted, these compounds are transported through the atmosphere and deposited in both wet and dry form on the earth's surface. The deposition of these compounds causes acidification of soils and surface waters. This section only takes account of man-made acidifying emissions, since they constitute the majority of the total emissions (and deposition) in Europe. Natural sources only have a minor share of total emissions in Europe and are considered to be exogenous because they cannot be influenced by environmental policy.[3] Emissions of both SO_2 and NO_x are largely related to energy use. The emissions of NH_3 originate largely from the agricultural sector.

SO_2 is formed during the combustion of fossil fuels (oil and coal) and emitted from stacks and exhausts. The main responsibility for SO_2 emissions lies with power stations, refineries and other industries. A small source of SO_2 emissions is

[3]Although these sources cannot be influenced, the negative effects can be mitigated for example by liming.

traffic.[4] The largest increase in emissions came after the 1950's as a result of the sharp increase in oil consumption. Figure 1.1 illustrates the European SO_2 emissions from 1950 to 1992. Since 1980, SO_2 emissions have been decreasing as a result of SO_2 reduction policies mainly in West European countries. Although SO_2 emissions have been falling in Eastern Europe recently, Eastern Europe is still a main contributor to European SO_2 emissions. In 1990 its share of total European emissions amounted to 70% (RIVM, 1991).

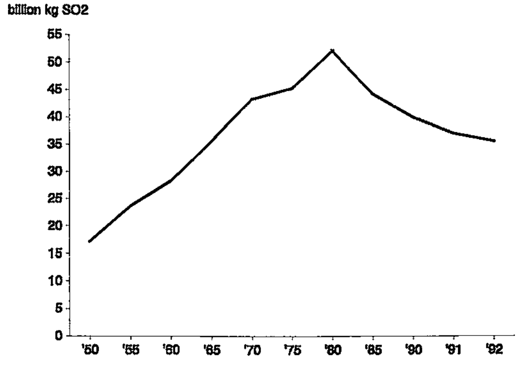

Figure 1.1 *Historical emissions of SO_2 in Europe (source: Thomas et al., 1988; Tuovinen et al., 1994).*

NO_x emissions are formed in all types of combustion. The amount of emission depends upon the N-content of fuels, boiler characteristics and the combustion temperature. At higher combustion temperatures, the NO_x formation increases exponentially. NO_x emissions mainly originate from traffic, power plants, industries and households (Lübkert, 1987). In Western European countries, traffic is the main source of NO_x emissions, followed by power plants. In Eastern European countries, however, power plants are the main source of NO_x emissions (Amann, 1990a). Figure 1.2 illustrates the European NO_x emissions from 1950 to 1992. Like SO_2 emissions, the NO_x emissions increased greatly between 1950 and 1980. In the period 1980 - 1992, the sharp increase of NO_x emissions levelled off. The increase in traffic in this period was counterbalanced by the introduction of catalytic converters (RIVM, 1991). In contrast to SO_2 emissions that mainly

---

[4]For the sake of completeness it should be noted that SO_2 is also emitted by the paper, pulp and iron melting industries.

originate in Eastern Europe, NO_x emissions mainly originate in Western Europe. In 1990, 60% of the European NO_x emissions originated in Western Europe (RIVM, 1991).

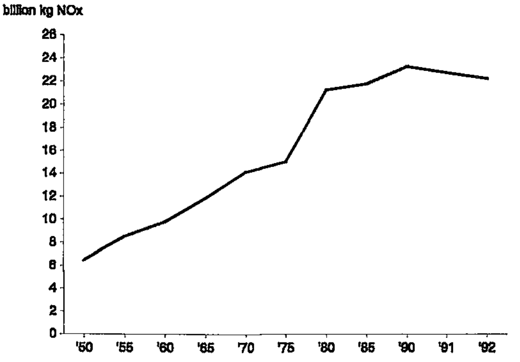

Figure 1.2 *Historical emissions of NO_x in Europe (source: Thomas et al., 1988; Tuovinen et al., 1994).*

Unlike NO_x and SO_2 emissions, NH_3 emissions are not related to energy use but originate mainly in the agricultural sector. NH_3 is released during the storage and application of manure, during the grazing period and through the application of nitrogen fertilizer. A minor source of NH_3 emissions (less than 2%) is the fertilizer and ammonia industry itself (Klaassen, 1991). Figure 1.3 illustrates the emissions of NH_3 in Europe between 1950 and 1992. Until 1970, NH_3 emissions increased only slowly. After that, they increased more sharply, as a result of the growing number of livestock.

To analyse the relative contribution of the three compounds to acid rain, both the emission amounts and the difference in acidifying potential of SO_2, NO_x and NH_3 have to be taken into account. This latter is done by using the concept of acid equivalents. One acid equivalent is defined as 1 mol H^+ potential acid with 1 tonne SO_2 being equivalent to 31,500 acid equivalents, 1 tonne NO_x equivalent to 21,500 acid equivalents and 1 tonne NH_3 equivalent to 59,000 acid equivalents (Erisman and Heij, 1991). The relative contribution of the three compounds in the total European acidifying emissions for the period 1950 - 1992 is represented in Figure 1.4. This figure is based on the emission numbers of Figures 1.1 - 1.3. On the basis of Figure 1.4 it can be concluded that the main cause of acid rain is the

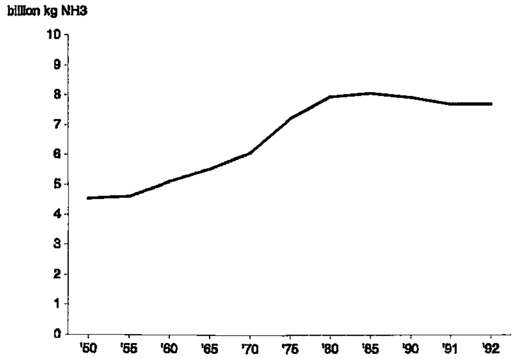

Figure 1.3 *Historical emissions of NH_3 in Europe (source: Thomas et al., 1988; Tuovinen, 1994).*

emission of SO_2. According to this figure, the contribution of SO_2 in total acidity increased between 1950 and 1970, after which it slowly decreased. In 1992 the contribution of SO_2 to total acidity amounted to 54%.

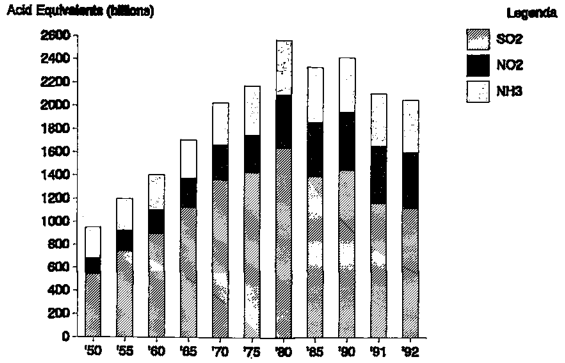

Figure 1.4 *Overview of total acidifying emissions in Europe between 1950 -1992.*

1.3 Atmospheric transport

As I indicated in the introduction to this chapter, acid rain is a transboundary environmental problem. Once the acidifying pollutants have been emitted into the atmosphere they are transformed and dispersed. Via gaseous or liquid phase reactions in the atmosphere, SO_2 can be transformed into sulphuric acid (SO_4^{2-}), and NO_x and NH_3 can partially be transformed into nitric acid (NO_3^-). The availability of ozone (O_3) is of crucial importance in the transformation process (Environmental Resources Limited, 1983).

A substantial portion of acidifying emissions is transported up to 1500 kilometres by the winds. As NH_3 is emitted just above the ground, this pollutant disperses far less than SO_2 which is emitted mainly through high smoke stacks. Because of this atmospheric transport, many countries are confronted with export and import of acidifying emissions. Table 1.1 illustrates the domestic share of the deposition and the export of emissions for European countries in 1992. Obviously, the sum of the domestic and export shares is not equal to 100%, since the denominator for both shares differs as the domestic share is related to deposition and the export share is related to emission. Because of the atmospheric transport, a country's deposition is not equal to its emission. Note also that since the data in Table 1.1 are presented in percentages, no information is given on the pollutant-trade balance of the countries.

Table 1.1 shows that the domestic share of deposition varies between the different compounds. It is clear that NH_3 is less transboundary in character than SO_2 and NO_x. Moreover, great variation exists between different European countries in the domestic share of any one compound. Consider for example SO_2. In Scandinavia, the domestic share is relatively low, and because of the prevailing winds that blow in Northern Europe from the South West to the North East, these countries have a high import of acid rain. Moreover, the source strength in upwind countries is important. Countries like Germany and Poland, for example, have many large emission sources. Other countries like the United Kingdom, Spain and Italy have a relatively large domestic share in depositions and suffer less from the import of acid rain. This can be explained by the geographical location (the upwind outskirts of Europe) and by the size of these countries. Since the precursors of acidification are transported in the atmosphere over hundreds of kilometres, acid rain can be classified as a continental environmental problem, requiring a continental (i.e. European) acid rain policy.[5]

[5]According to the categorization of environmental problems in the study *Concern for Tomorrow* (RIVM, 1989), continental scale problems are governed by the air circulation in the atmospheric boundary layer (0-3 kilometers altitude). Other categories that are distinguished in the RIVM study are: global scale, fluvial scale, regional scale and local scale.

Table 1.1 The domestic share in deposition and the export share in emissions of SO_2, NO_x and NH_x in 1992 (source: Tuovinen et al., 1994).

Country	Domestic Share in Deposition(%)			Export Share in Emissions(%)		
	SO_2	NO_x	NH_x	SO_2	NO_x	NH_3
Albania	34	3	66	83	97	60
Austria	7	5	44	78	95	51
Belgium	40	11	57	80	97	57
Bulgaria	56	13	82	77	94	57
Czechoslovakia	51	19	66	74	93	54
Denmark	30	9	82	89	98	60
Finland	19	14	36	70	88	44
France	43	36	83	66	84	42
Germany	67	39	69	67	85	43
Greece	30	32	55	84	94	61
Hungary	52	9	67	74	93	57
Ireland	39	14	91	82	94	48
Italy	67	47	79	72	88	49
Luxembourg	32	0	55	84	100	63
Netherlands	17	14	81	84	96	55
Norway	4	7	35	76	91	44
Poland	51	19	69	66	87	47
Portugal	57	27	82	78	93	56
Rumania	53	24	75	68	90	49
Spain	76	35	79	72	87	47
Sweden	8	12	30	73	88	48
Switzerland	14	10	51	71	93	46
Turkey	23	10	74	68	65	62
United Kingdom	86	60	78	72	91	46
Belorussia	34	9	66	71	91	50
Ukraine	47	18	6*	65	84	48
Moldavia	19	2	61	80	97	63
Russian Federation	43	31	71	52	65	33
Estonia	35	3	54	86	98	59
Latvia	14	3	54	81	96	58
Lithuania	22	3	61	75	95	55
Former Yugoslavia	40	7	50	75	94	55

* This is caused by very low national emission.

Because of this atmospheric transport, the acid rain problem differs from global environmental problems like the greenhouse effect. Acidifying emissions are called non-uniformly mixing pollutants since their environmental impacts depend not only on the total (European) volume emitted, but also on the location of the emission. This has main consequences for policy making on acidification since a restriction to total emissions only may result in substantial local damage caused by this locational aspect. For greenhouse gasses like CO_2, however, a policy on total emission reductions is satisfiable because the resulting damage depends only on total emissions and consequent concentrations and not on the emission location.

1.4 Damage caused by acid rain

The wet and dry deposition of acidifying pollutants has various negative effects. Some damage is caused directly by the pollutants. However, they can also have indirect effects on flora and fauna by causing changes in soils or aquatic systems. The effects of acid rain are surrounded by uncertainty. It is, for example, difficult to estimate the effect of acid rain in relation to other environmental damaging factors, like climatic changes or other pollutants. Moreover, observable effects can take a long time to become apparent. In this section a short overview is given of the most important impacts of acidification.

Acidification of lakes and streams involves extensive chemical and biological changes. Ultimately, acidification results in the death of fish and other aquatic life. Increase in aluminium concentration as a result of acidification is believed to play a major role in the disappearance of fish. The increase in acidification in thousands of lakes in Norway and Sweden has reduced or eliminated fish populations (Wetstone and Rosencranz, 1983; Alcamo et al., 1990). Not only surface water but also ground water is affected by acidification, which causes the release of heavy metals, thus affecting drinking water supplies (Environmental Resources Limited, 1983).

Acidification of soils has various effects. If the soil is acidified and unable to neutralize acid deposition, sulphate and nitrate may leak through the soil and contribute to water acidification. Acidification increases the concentrations of aluminium and other (toxic) heavy metals in the soil which have harmful effects on ecosystems (Berdén et al., 1987). Acid rain also contributes to the deterioration of forests. Large-scale deterioration and death of forests is a serious problem in many parts of Europe. Apart from the negative effects of soil acidification on forests, the precipitation of acidifying pollutants on leaves and needles erodes the waxy, protective layer, affecting the vitality of trees. It has also been suggested that forest dieback is related to excess nitrogen (Alcamo et al., 1990). Not only are forests negatively affected by acid precipitation; agricultural crops and other plants also suffer (Van der Eerden et al., 1987). Another impact of acid rain is damage to buildings. The effect of acidifying pollutants on (historic) buildings depends on climatic conditions, the specific properties of exposed materials and the mix of pollutants. Examples of this kind of damage are the deterioration of the Acropolis in Athens and of the cathedral in Cologne (ECE, 1985).

Since it is very difficult to provide a reliable quantitative estimate of monetary costs of the damage by acidification, this damage may be expressed in physical terms. For this purpose the concept of 'critical loads' can be used. Critical loads are defined as the deposition levels below which no significant damage to the environment is expected to occur according to present knowledge (Nillson and Grennfelt, 1988). Critical loads maps have been developed for Europe by the Coordination Center for Effects (Hettelingh et al., 1991, Posch et al., 1995). These maps show the critical loads for acidification in Europe at receptors of 150 x 150 km. The difference between the current deposition and the critical loads can be used as a measure for damage. The larger this difference, the larger the damage is assumed to be. However, no information is available on the relevant shape of the damage function (Amann et al., 1994).[6]

1.5 History of international acid rain policy

As acid rain has a transboundary character, countries are subjected to the pollution of other countries. It is generally recognized that an international acid rain policy is therefore required. Cooperation within the United Nations Economic

[6]It should be noted that acidification is directly linked with two other environmental problems: eutrophication and ozone formation. Both NO_x and NH_3 contribute to eutrophication and NO_x also plays a role in the formation of tropospheric ozone. However, these linked problems are outside the scope of this study.

Commission for Europe (ECE) has resulted in international reduction protocols. This section discusses the history of international European negotiations on the acid rain problem.

The UN Conference on the Human Environment, held in Stockholm in 1972, might be regarded as a starting point for the international attention to transboundary environmental pollution. Article 21 of the Declaration of this conference has played a central role in discussion and negotiations on transboundary air pollution. According to this Article, states have the sovereign right to exploit their own resources, but they also have the responsibility to ensure that activities within their jurisdiction or control do not cause any damage to the environment of other states or of areas beyond the limits of national jurisdiction (UN, 1973).

Initially, in the early seventies, the Organization for Economic Cooperation and Development (OECD) was a centre for the study and discussion of international environmental issues. In 1972, two months prior to the UN conference, the OECD started an international study of the long range air pollution entitled *The Cooperative Technical Programme to Measure the Long Range Transport of Air Pollutants*, in which 11 countries participated.[7] This study acknowledged the existence of the long range transport of sulphuric particles, as had already been pointed out by Swedish and Norwegian researchers. Although these findings were not supported by all the participating countries, the study did have an important impact on international policy discussions (Alcamo et al., 1990). However, since not all European countries are members of the OECD, this was not the appropriate organization for further discussion and negotiations. Therefore, the multilateral discussion and negotiation shifted to the United Nations Economic Commission for Europe (ECE), in which both Western and Eastern European countries participate.[8] On the initiative of Norway, Sweden and Canada, discussions were started within the ECE in 1977, with the aim of reaching an agreement between ECE members to cut back emissions by a fixed percentage.

[7]These countries are Austria, Belgium, Denmark, Finland, France, Germany, the Netherlands, Norway, Sweden, Switzerland and the United Kingdom.

[8] Canada and the United States also participate in the ECE, and thus in the negotiations on acidification.

In 1979 the Convention on Long-Range Transboundary Air-Pollution was signed. All European countries and two Soviet Republics signed, as well as the United States, Canada and the European Community. The Convention should be seen as an important step towards improvement of the environment in both Europe and the United States. It stipulates that governments protect their people and the environment against air pollution, and that they restrict and, if possible, reduce it as far as possible. Moreover, policies and strategies should be developed to abate the emission of air pollution. The Convention also calls for further research and development. It entered into force in 1983, when 24 countries had ratified the Convention (Alcamo et al., 1990). Between 1979 and 1983 the Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (EMEP) was initiated. The main task of the EMEP programme is to compile emission data, to measure air and precipitation quality and to develop atmospheric dispersion models. Nowadays, the EMEP models are widely used, and its annual publications on country-to-country transport matrices of sulphur, nitrogen and ammonia are generally accepted.

Ten years after the Conference of Stockholm, the signatories of the Convention came together again in Stockholm, this time for a conference on acidification of the environment. This meeting was a breakthrough in international efforts to reduce acidification. Firstly, Germany changed its attitude towards acidification as a result of pressure of the publicity in the media about the damage to forests. Secondly, a concerted international abatement programme was brought under the auspices of the ECE. As a follow-up to this conference, a number of countries proposed concrete steps for reducing SO_2 emissions. During a ministerial meeting in Ottawa (Canada) in 1984, ten countries decided to impose a 30% reduction of the 1980 amount of SO_2 emission by 1993. These ten countries were the Scandinavian countries, Finland, France, West Germany, Austria, Switzerland, Canada and the Netherlands. They were called the '30% club'. In July 1985, 21 countries signed a protocol in Helsinki to reduce SO_2 emission by at least 30% of the 1980 figures, to be reached by 1993.[9] Among the countries that did not sign were the

[9]These countries include the countries of the 30% club plus Belgium, Bulgaria, Belorussia, Czechoslovakia, West Germany, Hungary, Italy, Liechtenstein, Luxembourg, Ukraine and the USSR.

United Kingdom and Poland, two major emitters of SO_2. The United Kingdom's argument for not signing is worth mentioning. It criticized the arbitrary nature of the protocol, citing both the choice of 1980 as the base year and the equal percentage reduction of 30% as arbitrary choices. Moreover, the United Kingdom stressed the lack of connection of deposition levels with environmental impact.

After the completion of the SO_2 Protocol, a similar protocol was drawn up for NO_x which, however, met the criticism of the United Kingdom. The NO_x Protocol was signed by 25 countries, including Poland and the United Kingdom, in Sofia in October 1988. This protocol did not specify a reference year for reduction to relate to, but insisted that by 1994 NO_x emissions would not exceed the 1987 levels. Countries could also choose a reference year previous to 1987 provided the average yearly NO_x emission per ten-year period (1987-1996) did not exceed the emission level of 1987. The NO_x Protocol also said that countries should start negotiations for more stringent measures to push back NO_x emissions, thereby taking into account internationally accepted 'critical loads'. A number of countries found that a 'stand-still' principle did not go far enough as a first step towards an NO_x emission reduction. In an additional non-binding declaration eleven countries announced that, by 1998, they would achieve a 30% emission reduction compared to the level of 1987.[10] To realize a stand-still or indeed an emission reduction, national governments would have to adapt their policies. The protocol drew up national emission standards for new stationary, and new mobile emission sources. Suggestions for possible abatement measures for existing emission sources were given in a 'technical appendix'. The protocol indicated that these emission standards should be based on the best available technologies that were economically feasible. The NO_x Protocol also indicated that future negotiations on emission reductions should be based on the effects of acid rain.

In the early nineties, a further European SO_2 emission reduction was negotiated, finally resulting in the Second SO_2 Protocol in June 1994 (Oslo). This protocol was signed by most European countries. The flat rate reduction approach as applied in the First SO_2 Protocol and the NO_x Protocol was no longer favoured by

[10]These 11 countries were Austria, Denmark, Finland, France, West Germany, Italy, Liechtenstein, Netherlands, Norway, Sweden, Switzerland. Notably, in 1992 the average emission of NO_x of these countries had increased by 2.3% compared to 1987 (source: Tuovinen et al., 1994).

the parties of the Convention. Alternatively, the allocation of emission reduction was based on the environmental effects of emissions. The long term aim was to reach critical loads. However, it is not possible to achieve critical loads everywhere in Europe since these imply very low deposition in some parts of Europe and such low deposition cannot be realized without drastic reductions in energy consumption (Amann et al., 1992a). Therefore, an alternative deposition target was aimed at. This deposition target, indicated by 'gap closure', reduces the difference (the gap) between current deposition and critical loads. The deposition target finally aimed at in the negotiations on the Second SO_2 Protocol amounted to a gap closure of 60%. This target aims at bringing all ecosystems in Europe equally closer to the full achievement of critical loads, taking into account both regional differences in the environmental sensitivities and some notation of economic efforts for achieving these targets, by considering the current gap (Amann et al., 1994). In practice however, this deposition target will only result in a gap closure of 60% at binding receptors and a gap closure of more than 60% at non-binding receptors, because realizing this gap at binding receptors requires such emission reductions that the gap closure is more than 60% at non-binding receptors.

The cost-effective emission reductions to reach a gap closure of 60% were calculated by the RAINS model. However, this cost-effective emission abatement allocation was not complied within the protocol. The cost-effective abatement allocation was used as a basis in the negotiations, but for different reasons countries deviated from this allocation. One reason was that the associated costs of the cost-effective allocation are considered too high by countries. As a result the actual emission ceilings agreed on in the protocol deviate from the cost-effective emission allocation (Wüster, 1994). Table 1.2 shows the cost-effective emission allocation and the emission allocation agreed on for the year 2000 in the protocol.[11] Some countries agreed on further emission reduction for the years 2005 and/or 2010. Belorussia, Belgium and Greece would reduce emissions even

[11]The cost-effective allocation is calculated using the RAINS model, version 6.0. This version of the model does not yet take the new borders in Europe into account. The country classification in Table 1.2 is taken from the RAINS model. This classification is not limited to countries but sometimes considers regions.

below the cost effective emission allocation. Italy and Portugal will achieve their cost effective emission level in 2005, Poland and the United Kingdom in 2010. Bulgaria, the Czech Republic, Slovakia, Russia, France, Germany and Hungary will reduce their emissions further, but will not achieve their cost-effective emission level in 2010. Table 1.2 shows that only 9 countries will achieve the cost-effective emission level in 2000.

The Second SO_2 Protocol provides two references on cost effectiveness. First, Article 2.6 explicitly states that parties are allowed to apply economic instruments to encourage the adoption of cost-effective emission reduction. Although economic instruments do not by definition result in cost-effective emission abatement, their use may contribute to cost-effective abatement. Second, Article 2.7 opens up the opportunity for a joint implementation scheme that is intended to work similarly to an emission trading scheme (UN/ECE, 1994). The Second SO_2 Protocol was a major improvement in the international reduction policy on acidification compared to the flat rate approach in the former protocols on acidifying pollutants. The improvement was two-fold: emission reduction was based on the environmental impact of emissions and attention was paid to the cost effectiveness of emission reduction. However, not all European countries signed the protocol.

Table 1.2 Cost-effective SO_2 emissions (kt) and emissions according to the Helsinki Protocol for the year 2000 (source: Amann et al., 1993 and UN/ECE, 1994).

Country	Cost-effective SO_2 emissions	Protocol SO_2 emissions
Albania	132	*
Austria	78	78
Belgium	190	248
Bulgaria	533	1374
Czechoslovakia	868	1465
Denmark	58	90
Finland	116	116
France	670	868
West Germany	520	} 1300
East Germany	230	
Greece	595	595
Hungary	523	898
Ireland	131	155
Italy	1026	1330
Luxembourg	10	10
Netherlands	106	106
Norway	34	34
Poland	1397	2583
Portugal	294	304
Rumania	1062	*
Spain	1493	2143
Sweden	88	100
Switzerland	60	60
Turkey	2887	*
United Kingdom	1028	2449
Yugoslavia	1014	**
Kola Karelia	271	*
St. Petersburg	105	*
Baltic Region	89	*
Belorussia	456	456
Ukraine	1696	2310
Moldavia	231	*
Russia	4440	4440

* Did not sign the protocol. ** Only Croatia and Slovenia signed the protocol.

Although it took more than a decade after the 1972 UN Conference, European cooperation in reducing acidification eventually consisted of single pollutant protocols. The first protocols on reducing NO_x and SO_2 could be described as first steps in the right direction. The 1994 Protocol on further reducing SO_2 emissions, however, implied an improvement in reduction policy in both environmental and economic terms since the reduction of acidifying pollutants is based on their environmental effects and the cost effectiveness of emission abatement is taken into account. However, in this study it is shown that further cost savings can be reached.

1.6 Aim of the study

To improve the cost effectiveness of SO_2 emission reduction, the 1994 Protocol opts for the use of economic instruments and a joint implementation scheme. In principle this offers an opportunity for introducing a system of marketable permits in Europe. In the United States, such a system has already been introduced for SO_2 emission control of utility companies. In this thesis I discuss the possibility and the implications of an international system of marketable SO_2 permits among the European countries. The central question raised is whether a system of tradeable emission permits can contribute to a cost-effective European reduction of SO_2 emissions, taking into account prespecified deposition targets, and the conditions under which this is likely to happen. Before I can answer this question, three other research questions have to be examined.

(I) What are the main economic aspects of a European acid rain policy?

(II) What are the advantages and disadvantages of using a system of tradeable permits to implement a European acid rain policy?

(III) What should a system of tradeable emission permits for a non-uniformly mixing pollutant look like in order to take deposition targets into account?

The first of these questions, on the main economic aspects of acid rain policy, consists of three sub topics: (i) the cost effectiveness in an international context; (ii) the need for cooperation and the difficulties in achieving this and (iii) the choice of instruments for bringing about emission reductions. The second research question focuses in detail on the theory of tradeable permits. The difference between emission permits and deposition permits, and the need to restrict trading of non-uniformly mixing pollutants is discussed. The initial permit distribution is also discussed. Elaborating on the second research question, the third question concerns the implementation of a system of tradeable emission permits from an empirical point of view.

To answer the central question of whether a system of tradeable emission permits can contribute to a cost-effective European reduction of SO_2 emissions, taking into account prespecified deposition targets, I have made use of both a literature study and simulation. First, by means of a literature study economic aspects and the theory of tradeable emission permits in particular are examined. Next, taking the results from this into account, a theoretical concept of what I call 'guided bilateral permit trading' is formulated. Finally, guided bilateral permit trading is analysed by means of a simulation study.

1.7 Outline of the study

The central theme of this thesis, the analysis of the applicability of tradeable emission permits for cost-effective SO_2 reduction in Europe is dealt with as follows. First an economic theoretical background is presented by reviewing the relevant literature. Second, integrated assessment models are presented and compared. One of these models in selected for simulation. Third, after presenting these 'tools', a new permit trading systems is developed and analysed by detailed simulations of trading schemes, including their economic and environmental implications.

Chapter 2 discusses some general economic aspects of pollution control. Attention is paid to the cost effectiveness of pollution control, to the international dimension of acid rain policy and to the need for cooperation. Here I come close to the area of game theory but only some general game theoretic concepts are reviewed. Another topic discussed in this chapter concerns the alternative policy instruments for emission control. In this thesis I focus on tradeable emission permits. Given this general economic background, the theory of tradeable emission permits is elaborated on in Chapter 3. In this chapter permit trading for pollutants that are non-uniformly mixing is thoroughly discussed and illustrated by some empirical studies. After discussing both emission permit and deposition permit trading systems, alternative systems of tradeable permits for this kind of pollutants are examined. Two main aspects in examining permit trading systems concern (i) the kind of trading process assumed, involving the distinction between simultaneous multilateral permit trading versus bilateral sequential permit trading, and (ii) the initial distribution of emission permits. This thorough discussion on tradeable permits contributes to a better understanding of this policy instrument and sheds light on the implications of permit trading for non-uniformly mixing pollutants. The findings of this chapter indicate that a new permit trading system has to be developed. Chapter 4 describes and compares integrated assessment models for simulation of acid rain control. First, three integrated assessment models for the European acid rain problem are reviewed. To provide a complete overview of models, non-European models are reviewed as well. Secondly, the advantages and drawbacks of the different models are weighed up against each other, concluding in a selection of one of the European integrated assessment models to be used in the analysis of this thesis.

The first four chapters provide all elements necessary for developing and subsequently simulating a new permit trading system for SO_2 emissions. Building on Chapter 3, Chapter 5 develops in steps the concept of guided bilateral permit trading. This system aims at cost-effective emission abatement given prespecified deposition targets, assuming that permit trading is a bilateral and sequential process. The need to restrict (guide) trade is extensively discussed and the implications of trade restrictions for a bilateral and sequential trade process are explained. In Chapter 6 guided bilateral permit trading will be simulated for SO_2 permits in Europe. It is assumed that permit trading takes place among countries. First the simulation model is described. Next the simulation results for the base case and two variants are successively discussed. Since I assume that permit trading is a bilateral and sequential process special attention is paid to the trade sequence. By simulating different trade sequences the effect of the sequence of transactions is examined.

Finally, in Chapter 7 conclusions on the use of tradeable emission permits for reducing SO_2 emissions in Europe are summarized and discussed, based on the theoretical and empirical findings in this thesis.

2 THEORETICAL PRINCIPLES FOR ACID RAIN POLICY

2.1 Introduction

International acid rain policy stipulates to what extent and, occasionally, in which way countries should reduce their emissions. This allocation of emission reduction can take place according to several principles. These principles are dealt with in this chapter. As the allocation of emission reduction is an environmental economic question, the chapter begins with a brief review of environmental economic theory, in which I mainly focus on the environmental criteria approach.[12]

The transboundary transport of acidifying compounds means that international aspects have to be taken into account. Section 2.3 shows how important international cooperation is if optimal emission abatement is to be reached. Strategic behaviour, however, may hinder optimal coordination of abatement strategies. In discussing the international dimension of emission abatement allocation, attention is paid to some game theoretic concepts.

Finally, section 2.4 raises the question of which instruments could be used by policy makers to reach a desired abatement allocation.

2.2 Economic theory of environmental policy

The interfaces between environment and economics are often called 'environmental functions', referring to functions of the environment that are used for the satisfaction of human needs (e.g. Siebert, 1987; de Groot, 1992). In this interpretation, Siebert distinguishes four functions. Firstly, the environment provides public consumption goods like fresh air and the recreational function of nature. Secondly, the environment is a supplier of resources like fuel, minerals and water that are used in the production process. Thirdly, the environment is a receptacle of emissions, the undesired joint outputs of production and consumption activities being absorbed by different environmental media: water, air and soil. Finally, the

---

[12]This approach is also indicated by ecological approach.

environment provides space for location of industrial activity, agriculture, housing, infrastructure and recreation sites.[13] Environmental functions compete with one another. Excess use of the environment as a receptacle of wastes negatively affects the environment as a provider of public consumption goods. Environmental degradation arises if the use of an environmental function exceeds the environmental endowments. The competition for the use of the environment is a problem of scarcity. Therefore environmental problems can be interpreted as economic problems: how to allocate the environment between the competing uses.

This question can be dealt with from several economic approaches. The neo-classical economic approach explains the occurrence of environmental degradation by the discrepancy between the private costs and social costs of pollution. In the neo-classical line of thought, optimal pollution control occurs if marginal abatement costs equal marginal benefits of pollution abatement. If so, maximum social welfare is achieved (see e.g. Siebert, 1987). In practice a main difficulty related to define optimal pollution control levels, is the measurement of environmental benefits of pollution control. An alternative view of the problem of environmental deterioration is the institutional approach. This approach stresses the importance of specific institutions and the historical framework for the functioning of an economy. Property rights play a main role in this point of view (see e.g. Bromley, 1991). Environmental pollution emerges in situations in which property rights are not (or ill) defined (Dales, 1968). The analysis of Coase (1960) shows that if property rights are well defined, negotiation between the polluter and the victim will result in an optimal pollution level. If acid rain in Europe is regarded as an international allocation problem, then the concept of property rights is applicable at the country level. If a country is entitled to a certain environmental quality, negotiating with countries affecting the environmental quality is principally possible. However, as the acid rain problem is a reciprocal externality, negotiating will be complex, resulting in substantial transaction costs.

[13]De Groot (1992) also distinguishes four functions. However, his differ from those of Siebert (1987). De Groot distinguishes a production function comprising both the provision of public consumption goods and the provision of resources. As a fourth function he adds the information function, for example, scientific and educational information.

In this thesis the environmental criteria approach is followed since this approach is well suitable to the problem of acidification. In the environmental criteria approach, the level of emission abatement results from the criteria for the quality of the environment which are defined by the authorities. These criteria might be interpreted as limiting conditions for economic activity. These limits are based on scientific information as well as on prevailing social standards. In general, a higher priority is given to environmental goods by countries with higher welfare (Komen and Folmer, 1995). Using environmental criteria avoids the problem of monetary valuation of the environment. In the case of non-uniformly mixing pollutants, it is not the emissions but the deposition and the concentration caused by emissions are decisivė for the environmental damage. Therefore it is appropriate to define deposition targets for acidification. The assessment of a criterion for acidification in the Netherlands might be an illustrative example. From a scientific point of view, criteria are available for defining the maximum amount of annual acid deposition per hectare per year that does not cause any harm to the environment (that is critical loads, see section 1.4). A target load has been derived from this purely scientific criterion. This target load takes into account social economical circumstances. It allows more deposition than the critical load but, compared to the current deposition, it requires considerable deposition reduction.[14]

Given environmental target loads, the aim is to meet these loads at minimum abatement costs. The resulting abatement allocation is said to be cost effective with regard to the specified target. This abatement allocation is no longer optimal since it is not known whether the environmental criteria determined result in maximized social welfare.

Generally, environmental criteria can be expressed in terms of an emission target or in terms of a deposition target. If the environmental target is expressed

---

[14]Numbers for the Netherlands are as follows. The critical load varies from 200 to 500 acid equivalents/ha/yr. The deposition in 1989 amounted to 4800 acid equivalents/ha/yr. The long term deposition target for the year 2010, the target load, amounts to 1400 acid equivalents/ha/yr. This target is based both on environmental and on policy considerations. A deposition of 1400 acid equivalents/ha/yr will prevent the most serious damage to vital forest ecosystems. To realize this target the emission reduction in the Netherlands should amount to 70% (compared to 1980 levels). This reduction percentage is considered to be acceptable. Further emission reduction would result in extremely high abatement costs (IMP, 1984). In fact, an implicit cost benefit analysis is applied to derive this target load.

as a maximum amount of tolerated emission (E^*), the cost-effective allocation is the solution to the following optimization problem (Tietenberg, 1985):

$$Min\ \Sigma_{j=1}^{J}\ C_j(r_j) \quad (2.1)$$

$$s.t. \quad \Sigma_{j=1}^{J}\ (E_j - r_j) \leq E^* \quad (2.2)$$

$$0 \leq r_j \leq E_j \quad (2.3)$$

where:

$C_j(r_j)$: abatement cost function of source j

r_j: emission reduction of source j

E_j: initial emission of source j

E^*: emission target

The conditions for a cost-effective allocation follow from the Kuhn-Tucker conditions:

$$dC_j(r_j)/dr_j - L \geq 0 \qquad j = 1,...,J \quad (2.4)$$

$$r_j\ [dC_j(r_j)/dr_j - L] = 0 \qquad j = 1,...,J \quad (2.5)$$

$$\Sigma_{j=1}^{J}\ (E_j - r_j) \leq E^* \quad (2.6)$$

$$L\ [\Sigma_{j=1}^{J}\ (E_j - r_j) - E^*] = 0 \quad (2.7)$$

$$0 \leq r_j \leq E_j \qquad j = 1,...,J \quad (2.8)$$

L is the Lagrangian multiplier. The economic interpretation of L is as follows: it reflects the change in the optimal value of the objective function *(2.1)* if the environmental constraint *(2.2)* is relaxed by one unit. The Kuhn-Tucker conditions show that in a cost-effective abatement allocation, the marginal costs of abatement for each source equal L.[15] Consequently, marginal costs for each source are equal. In practice, condition *(2.8)* automatically follows from the abatement cost functions. The emission reduction is always less than the current emissions.

If the environmental target is expressed in a deposition target and if emissions of different sources do not affect receptors equally, as in the acid rain problem,

---

[15]This holds for those sources that reduce emissions, for other sources ($dC_j/dr_j > L$) reduction will be zero.

the optimization problem has to be reformulated. It is assumed that the relation between emissions and deposition is linear (Tietenberg, 1985):

$$Min\ \Sigma_{j=1}^{J} C_j(r_j) \tag{2.9}$$

$$s.t.\quad \Sigma_{j=1}^{J} a_{ij}(E_j\text{-}r_j) \leq d_i^* \qquad \forall\ i \tag{2.10}$$

$$0 \leq r_j \leq E_j \tag{2.11}$$

where:

d_i^*: deposition target at receptor i

a_{ij}: a transport coefficient which translates the emission of source j to deposition at receptor i.

The Kuhn Tucker conditions of this optimization problem are:

$$dC_j(r_j)/dr_j - \Sigma_{i=1}^{I} a_{ij}L_i \geq 0 \qquad j = 1,...,J \tag{2.12}$$

$$r_j\ [dC_j(r_j)/dr_j - \Sigma_{i=1}^{I} a_{ij}L_i] = 0 \quad j = 1,...,J \tag{2.13}$$

$$\Sigma_{j=1}^{J} a_{ij}(E_j\text{-}r_j) \leq d_i^* \qquad i = 1,...,I \tag{2.14}$$

$$L_i\ [\Sigma_{j=1}^{J} a_{ij}\ (E_j\text{-}r_j) - d_i^*] = 0 \qquad i = 1,...,I \tag{2.15}$$

$$0 \leq r_j \leq E_j \qquad j = 1,...,J \tag{2.16}$$

According to these conditions, abatement is cost effective if the marginal costs of emission reduction for each source equal the weighted sum of the shadow prices (L_i) for each receptor. Weights are the transfer coefficients from source j to the affected receptor i. The shadow price of a receptor reflects the marginal costs of a change in the deposition target for that receptor.

A final remark considering the use of environmental standards relates to the issue of valuing environmental damages. Although valuing environmental damage is avoided in the environmental criteria approach, costs and benefits are implicitly balanced by policy makers in establishing environmental criteria, since in the establishment of these criteria, considerations on costs to reach the criteria play a role in the establishment. Criteria are set so that the resulting environmental damage is limited, while abatement costs are not excessive.

Theoretically, using environmental criteria implies a special functional form of the damage function. One might regard these criteria as the upper limit for allowed pollution. Translating this into a damage function suggests that as long as the environmental standard is not violated, the environmental damage equals zero whereas, if the standard is exceeded, the damage is infinite. This suggests that there is little damage associated with introducing pollution activities into previously clean areas as long as pollution levels do not exceed the standard. Accordingly, in terms of an optimization problem, it holds that if a particular pollution constraint is not binding, the implied cost of a marginal environmental degradation is zero (see e.g. McGartland and Oates, 1985).

2.3 International dimension: cooperation versus non-cooperation

2.3.1 Cooperation

As I explained in section 2.2, according to the neo-classical theory, abatement of emission is optimal if marginal abatement costs equal marginal benefits of emission abatement. However, acidifying pollutants are transported via the atmosphere to other countries in substantial measure (see section 1.3).

A main distinction can be made for transboundary externalities between unilateral and reciprocal externalities (see e.g. Mäler, 1990, Helm and Pearce, 1990). A unilateral externality exists if country A affects country B, but country B does not affect country A. A classical example of a unilateral externality is the pollution of a river in which a downstream country suffers from the pollution caused by the upstream country. A reciprocal externality exists if a group of countries are at the same time the source and the victim of pollution. Generally, acid rain in Europe can be classified as a reciprocal external effect. Because of the transboundary transport of emission, emission abatement of one country can benefit other countries. This results in a discrepancy between national benefits of emission abatement and total benefits of emission abatement. Figure 2.1 illustrates this discrepancy and the consequences for optimal emission abatement.

In Figure 2.1 marginal abatement costs *(MC)* are assumed to increase as emission abates whereas marginal benefits are assumed to decrease as emission abates. Marginal national benefits (MB_j) are defined as: $MB_j = a_{jj}\ MB_c$. In this equation MB_j is the marginal benefit to a country whereas MB_c is the total marginal benefit for all countries. The transfer coefficient a_{jj} represents the relative contribution of

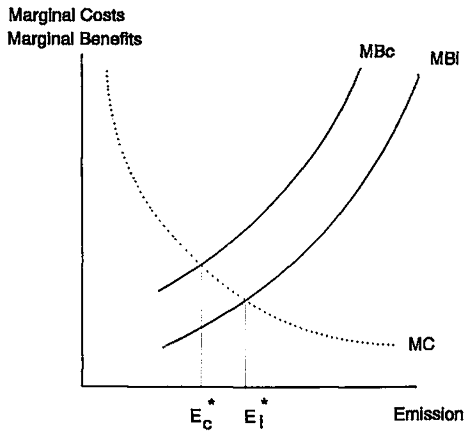

Figure 2.1 Optimal emission abatement

the emission of country j to its national deposition. The total marginal benefits *(MB_c)* of emission abatement by country j are the sum of marginal benefits at all receptor countries: $MB_c = \Sigma_{i=1}^{I} a_{ij} MB_j$. The number of affected countries is I. If a country only takes national benefits resulting from emission abatement into account, the optimal level of emission abatement is represented by E_I^*. This emission level is characterized by equalized marginal abatement costs and marginal national benefits. If countries take all damage caused both at home and abroad into account, the optimal emission abatement level results from the intersection of the marginal cost function *(MC)* and the total marginal benefit function *(MB_c)*. In Figure 2.1 this is at E_c^*. Obviously, if countries internalize the benefits of emission reduction abroad, the optimum level of emission abatement is higher than national benefits only.

Full cooperation in emission abatement, which implies minimizing total abatement and damage costs for all European countries collectively, results in optimal European emission abatement. However, although full cooperation results in minimized European costs, individual countries may not act accordingly. This can be explained by using the concept of the 'prisoners' dilemma' (see e.g. Folmer and Musu, 1992; Mäler, 1991). Consider two countries A and B both emitting acidifying emissions. The emissions are transboundary: acid deposition in country A partly originate in country B and vice versa. Both countries may or may not decide to cooperate in emission abatement. In this context, cooperation means that both countries take the damage caused in the neighbouring country into account. Cooperation results for both countries in an advantage compared to a situation in which countries do not cooperate. However, if country A (B) internalizes the damage done in the neighbouring country but country B (A) does not, then country B (A) has an advantage over country A (B). These results are summarized in the pay-off matrix in Figure 2.2. The numbers in this matrix represent the fictive net benefits of emission abatement in country A and B for both strategies C and NC. The letter C stands for cooperation, indicating that a country takes the damage caused in the neighbouring country into account. The letters NC stand for non-cooperation, indicating that a country only takes its national damage into account. If both countries choose to cooperate, they both have a net benefit of 6. If one of the countries follows a cooperative strategy and the other does not, then the latter has an advantage as it benefits from the emission reduction in the neighbouring country, whereas it has fewer abatement costs. In this example, as presented in Figure 2.2, the benefits of cooperation are equally distributed between the two countries. In the real world, however, the benefits of cooperation will be very unequally distributed and can even be negative for some countries. A model study on acid rain in Europe by Mäler (1989) indicates that the United Kingdom, Italy and Spain would loose from participating in a full cooperative abatement strategy.

In the absence of a binding agreement, both countries will decide not to cooperate. If country A follows strategy C, then country B will follow strategy NC, as this maximizes the net benefits of country B. If country A follows strategy NC, then country B will follow strategy NC too, as this maximizes the net benefits of country B. Likewise, country A will always follow strategy NC, independent of the strategy choice of country B, as strategy NC maximizes the net benefits of country A. This analysis illustrates that individual rationality hampers cooperation. Each country has an incentive to free wheel on the abatement efforts of other countries. If countries do not cooperate, the optimal international abatement allocation will not be reached. Instead, a country will abate its emissions so that, given the emissions reductions of the other countries, its deposition target is fulfilled exactly.

Although the theoretical arguments of the prisoners' dilemma indicate that individual rationalism does not result in cooperation, in practice, countries do cooperate in the abatement of environmental pollution. For example, international agreements exist for the abatement of CFCs (Montreal Protocol), of NO_x (Sofia Protocol) and of SO_2 (Helsinki and Oslo Protocol). However, international agreement does not automatically result in optimal emission abatement.

Country B

Country A		C	NC
	C	6,6	1,8
	NC	8,1	2,2

Figure 2.2 The Prisoners' Dilemma

Unlike the solution to the prisoners' dilemma, countries may have several reasons for cooperating. One reason is that in the real world the game is played repeatedly. This opens up the opportunity for punishing countries that do not cooperate. The punishment consists of increases in the emission of cooperating countries to increase the environmental damage in the non-cooperating countries (Mäler, 1991). This threat provides an incentive for cooperation. Another reason is the occurrence of so-called interconnected games (Folmer et al., 1993). This term refers to a situation in which countries are involved in both environmental interdependencies and, for example, trade interdependencies. These linked interdependencies provide the opportunity for imposing sanctions, thus providing countries with an incentive to cooperate. A final reason to cooperate is the opportunity of introducing side payments. This opens up the chance for countries that benefit from cooperation to compensate countries that are disadvantaged by cooperation. If these countries are compensated for their losses they may agree to follow a cooperative strategy.

Mäler (1990) studied non-cooperation in the context of the acid rain problem. In a non-cooperative setting, each country tries to minimize its present value of future abatement costs and damage costs. Mäler (1990) indicates that there is a Nash (non-cooperative) equilibrium whereby emissions will, in the long run, approach the levels that are compatible with the deposition target, for example critical loads. This conclusion is based on the assumption that each country formulates an abatement policy that it will follow for ever. That is, countries receive no new information on the emission abatement of other countries or on the deposition in their own country. However, this is a rather unrealistic assumption. It is more realistic to assume that countries will adjust their emissions according to the information available on deposition levels. In this case, Mäler (1990) indicates that if side payments are allowed to compensate countries that would lose from cooperation, an equilibrium exists that corresponds to countries choosing cooperating strategies. In the long run, this cooperation results in emissions that are in accordance with critical loads. In conclusion, if it is assumed that the damage depends on the stock of emission, both the non-cooperative and the cooperative abatement strategy result in emissions that are compatible with critical loads. However, Mäler indicates that in the cooperative strategy the convergence towards the critical load emission level is faster and the resulting stock of sulphur is smaller (Mäler, 1990). It is worth noting that side payments are required for reaching this cooperative strategy. But formal cooperation is not necessary as long as each country aims at critical loads. In a special case, the Nash non-cooperative solution coincides with the cost effective solution. This depends on the shape of the marginal damage function underlying these models. If it is assumed that the marginal damage costs is infinite for depositions exceeding the critical load and zero below that load, then the cost- effective solution coincides with the Nash non-cooperative solution (Mäler, 1990).

By assuming that environmental damage is a function of the cumulated depositions, an economic basis is provided for using critical loads. If it is assumed that actual environmental damage depends on the cumulated depositions, it is possible to show that along the optimal path, the deposition will converge to the critical load. In other words, in the long run, countries should aim at emissions that correspond to deposition levels not exceeding the critical load. This means that there is an economic basis for using critical loads as a foundation for international agreements (Mäler, 1991).

Barrett (1990; 1991) demonstrated that if both the marginal damage function and the marginal abatement cost function have a steep slope, then there is an urgent need for international cooperation in pollution control. Although the framework developed by Barrett is not specifically applicable to acidification, and makes some rough simplifications on the real world, it is very suitable for illustrating the issue of cooperation in international pollution control. Barrett considered mondial environmental pollution. He assumed a world with $i = 1, ..., N$ identical countries, emitting the same amount of pollution. All N countries have identical benefit functions. The benefit function of a country i is given by:

$$B_i(R) = b[aR - R^2 / 2N] \tag{2.17}$$

where:

$B_i(R)$	: abatement benefit of country i
R	: global abatement
N	: number of countries
a, b	: positive parameters

Parameter a can be interpreted as the amount of pollution in absence of abatement. Parameter b is the slope of the global benefit function. Each country's abatement costs depend on its own abatement. The abatement-cost function of a country i is given by:

$$C_i(r_i) = cr_i^2 / 2 \tag{2.18}$$

where:

r_i	: abatement of country i
c	: parameter representing the slope of the abatement cost curve

The full cooperative solution aims at maximizing global social welfare. As indicated and explained in Figure 2.1, the social optimum occurs if each country's marginal abatement costs equal the global marginal benefit. The full cooperative abatement levels are:

$$R_c = aN^2 / (N + (c/b)) \quad r_c = aN / (N + (c/b)) \tag{2.19}$$

In the non-cooperative solution, countries aim at equalizing national social welfare. The national optimum will occur if a country equalizes its marginal abatement costs to the national marginal benefit. The non-cooperative abatement levels are:

$$R_{nc} = aN / (1 + (c/b)) \quad r_{nc} = a / (1 + (c/b)) \tag{2.20}$$

The mathematical difference between global net benefits for the cooperative and non-cooperative levels defines the potential gains of cooperation. The cooperative solution demands greater abatement but gives to every country a greater net benefit. For a given size of N, the difference between R_c and R_{nc} can be shown to depend on the slopes of the marginal abatement benefit and cost curves (parameters b and c). The difference will tend to be small, indicating that the need to cooperate is less urgent when the ratio c/b is either 'large' or 'small' (Barrett, 1990). Pollution that does not cause very large damage but is very costly to control, and pollution that is very hazardous but very cheap to control do not cause problems. For the former, even cooperation will not call for large abatement levels. For the latter, a non-cooperative strategy will result in substantial abatement. Cooperation is most needed if the marginal abatement-cost curve and the marginal benefit curve are both steep or both flat; i.e. damaging pollutants that are costly to control, and pollutants not causing serious damage that are inexpensive to control. Obviously, the former type of pollution causes the greatest concern as the cost of failing to cooperate in this case is very high (Barrett, 1990). Basing his conclusions on model simulations with large N and varying values for the parameters b and c, Barrett stated that it is difficult to reach an agreement if b and c are both large; i.e. if the need to cooperate is substantial.

The situation for acidification can be typified as follows. The costs of abatement are relatively small (as percentage of GDP) whereas the benefits of abatement are substantial. In accordance with Barrett's analysis, it was not very difficult to reach agreement on international cooperation. Particularly at first, the agreement on cooperation was a confirmation of already planned national emission reductions.

2.3.2 Full cooperation with side payments

In the preceding section, I outlined the need for cooperation on pollution control, and the related difficulties. To encourage countries to participate in a cooperative emission reduction policy, financial incentives may be introduced. As the benefits of full cooperation can be very unevenly distributed among countries or even be negative, some countries might not be willing to sign an agreement on international pollution control. To increase the number of signatories of an agreement, it might be necessary to introduce side payments. By reallocating the benefits of cooperation by means of side payments, both signatories and intended non-cooperators might be better off. Signatories want non-cooperators to sign the agreement because this increases their net benefits, even if they have to pay compensation. For non-cooperators, receiving of compensation makes signing an agreement beneficial.

Several criteria are available for introducing a system of side payments to reallocate the benefits of cooperation. For example, side payments can be based on the principle 'the victim pays'. This principle assumes that the contribution of a country to global abatement costs is based on the deposition in that country in a base year. Next to this, different kinds of shared responsibility are conceivable as principles for side payments. For instance, both the victim and the polluting country could pay an equal share of the global abatement costs or countries could contribute to the abatement costs according their level of economic development. Klaassen and Jansen (1989) studied the consequences of four cost distribution criteria and compared the results with the reduction plans of several countries for the year 2000.[16] The criteria considered were: (i) the civil liability principle; (ii) the polluter pays principle; (iii) the victim pays principle and (iv) the equally shared responsibility principle. It was assumed that countries prefer a criterion for

---

[16]The study is based on the efficient SO_2 abatement allocation in Europe given predetermined target loads. The abatement costs are based on the RAINS model.

side payments if this results in fewer abatement costs and a lower deposition level compared to their current reduction plans. If the cost sharing scheme results in higher costs and no deposition reduction, countries will resist such a scheme. It appears that there is no one criterion that is preferred by all the countries. Klaassen and Jansen (1989) argued that agreement on an acceptable criterion for side payments is partly hampered by the fact that realizing the target loads implies high abatement costs. If an exceedance of these targets is allowed at some receptors, abatement costs are substantially reduced.

Bergman et al. (1992) developed a system of side payments that, in contrast to the systems described above, is agreeable to all countries and consequently results in full cooperation. According to this system, countries contribute to the global abatement costs that correspond to their relative benefit of cooperation. Firstly, the minimum abatement costs $(\hat{C}_i)$ of a country i are calculated in order to realize a deposition target (d_i^*). This calculation assumes that countries act independently of each other. However, the abatement measures necessary to achieve d_i^* at minimum costs can be taken both at home and abroad. The total costs to realize d_i^* in each country i amount to $\sum_i \hat{C}_i$. Secondly, to realize d_i^*, the minimum abatement costs (C_i^*) are calculated assuming that countries do cooperate. The total costs assuming cooperation amount to $\sum_i C_i^*$. Since cooperation results in cost efficiency, these total costs are lower than $\sum_i \hat{C}_i$. According to the side payment scheme, the contribution of each country to the total abatement costs (the fund) depends on $\hat{C}_i$, namely $\alpha\hat{C}_i$, where α equals $\sum_i C_i^* / \sum_i \hat{C}_i$. The parameter α is the same for all countries. Given this side payment scheme the benefit of cooperation compared to non-cooperation, amounts to $(1-\alpha)\hat{C}_i$.[17] The amount collected, $\sum_i \alpha\hat{C}_i$, can be redistributed among the countries to bear part of the abatement costs of the countries. If each country receives C_i^*, namely the abatement costs under cooperation, the amount collected equals the payment, since $\sum_i \alpha\hat{C}_i = \sum_i C_i^*$. The net contribution of each country to the abatement costs amounts then to: $\alpha\hat{C}_i - C_i^*$. Obviously, countries that abate a relatively large amount of emission under cooperation, thus causing high abatement costs receive a substantial compensation. This makes it attractive for them to participate in the cooperative abatement

[17] As $\sum_i \hat{C}_i$ exceeds $\sum_i C_i^*$, α is smaller than 1.

scheme. These countries are net receivers. Countries that are not required to abate a large amount of emission are net payers as for these countries $\alpha\hat{C}_i$ exceeds C_i^*. However, cooperation is attractive for these countries as unilateral abatement of emission would be more expensive (Bergman et al., 1992).

2.3.3 Uniform emission reduction

Cooperative emission reduction can take place in different forms. This section discusses uniform emission reduction as it has been applied in SO_2 and NO_x emission reduction protocols, and its implications are illustrated.

An international emission reduction scheme based on uniform emission reduction implies that all countries have to reduce their emissions with an equal percentage, given a certain base year. For example, the Helsinki Protocol on SO_2 reduction aimed at an equal percentage reduction of 30% compared to the emission levels in 1980, to be reached in 1993. A benefit of an equal percentage approach is that it seems to imply equity for all the countries concerned. As a result, this supposed equity will not disturb the differences in production possibilities and costs. Given this equity, countries are more inclined to participate in a cooperative abatement scheme. However, this equity only appears to be real. The costs and benefits of an equal reduction scheme can vary substantially among countries. Costs may differ because of differences in energy use patterns and because of differences in implemented reduction measures in the past. The benefits of emission reduction differ between countries as a result of differences in the initial deposition level and the deposition percentage caused abroad.

A main disadvantage of an agreement based on uniform emission reduction is that it does not result in cost effective emission abatement (Hoel, 1992). Since uniform emission reduction does not take the differences in marginal abatement costs among countries into account, an equal amount of emission could be abated at lower costs. Saving in costs will be achieved if there is a reallocation of emission reduction measures so that countries facing low marginal abatement costs increase their emission reduction, and countries facing high marginal abatement costs decrease their emission reduction. However, in the case of non-uniformly mixing pollutants, the reallocation of reduction measures influences the deposition pattern. It is possible that deposition will increase in some countries compared to the uniform emission reduction, and this will cause the countries concerned to resist (von Weizsäker and Welsch, 1991). Another disadvantage of an equal reduction abatement scheme is that such a scheme is not attractive enough for all countries to participate. This mainly holds for countries not suffering from substantial imports of emission and facing high marginal abatement costs (Hoel, 1992). Finally, a disadvantage of a uniform emission reduction agreement is the lack of an incentive for increasing emission reductions if the agreed uniform reduction has been implemented. In conclusion, a uniform emission reduction may look attractive because of its simplicity and its putative equity. However, for efficiency reasons this approach is not to be preferred.

2.4 Policy instruments for controlling acid rain

2.4.1 Introduction

Having discussed several approaches for emission abatement allocation in the previous section, in this section I want to raise the central issue of which environmental policy instruments policy makers can use to reach a desired emission abatement allocation. Environmental policy instruments aim at influencing the decisions of economic agents in an environmentally sound direction. Several kinds of instruments are available for environmental policy. In general three are distinguished: (I) regulatory instruments, (II) economic instruments and (III) instruments aiming at moral persuasion. Regulatory instruments aim at directly influencing polluters' behaviour, e.g. by setting standards, by regulating processes, or by abandoning or limiting discharges. Economic instruments aim at influencing decision making indirectly by means of financial incentives. Instruments of moral persuasion aim at increasing awareness of environmental problems in order to stimulate consideration of the environmental consequences of individual behaviour. In fact, this category of instruments aims at influencing the preferences of economic agents.

The choice of instruments can take several criteria into account. Summarizing, the main criteria are: economic efficiency, environmental effectiveness, incentives, flexibility, simple mode of operation, cost of implementation, integration in sectoral policies, distributional effects, political acceptability, and conformity with international agreements (see e.g. Bohm and Russell, 1985, Siebert, 1987, Opschoor and Vos, 1989, Bovenberg et.al., 1991, Barde, 1995). In the past environmental policy was, to a large extent, based on direct controls, where each polluter had to abide by rules that specified the allowable levels of pollution and/or abatement technology to be used (see e.g. Bohm and Russell, 1985, Meister, 1990). However, economists stressed that substantial cost savings would be achieved if economic instruments were to be introduced. In the next sections the advantages and disadvantages of different environmental policy instruments are reviewed.

2.4.2 Regulatory Instruments

Regulatory instruments directly influence the behaviour of economic agents. Regulation can take different forms. First, the amount of pollutant can be regulated. Regulation can prescribe the maximum quantity of total allowed emissions, or can force a reduction of a certain amount of emission in absolute or relative quantities. Second, specific technology standards in abatement or production can be required by regulation. Third, product norms may define the quantity of pollutants which are contained in goods.

Regulatory instruments are characterized by inflexibility. Polluters are forced to comply with the regulation. Within the category of regulation, performance standards are more flexible than technology standards, because performance standards allow the polluter free choice in how to fulfil these standards. The main advantage of regulation is that, if the instrument is properly used, the environmental effect is certain. The impact on emissions is known in advance. This makes regulatory instruments very attractive. There are other arguments in favour of regulation. There is already a long standing experience with regulation in other fields of public concern. Regulation may provide an effective means of preventing hazards an irreversible effects (Barde, 1995). Regulation has no direct financial consequences for the government, and firms seem to prefer regulation to economic instruments (Bohm and Russell, 1985).

A main disadvantage of regulatory instruments, however, is that in general they are not cost effective. This can be easily clarified by the example where different polluters with the same environmental effect per unit of pollutant discharge have different marginal abatement costs for the performance standard assigned to them. Consequently, the environmental target is not reached at minimum abatement costs (see e.g. van Ierland, 1993). Only if marginal abatement costs are equalized for all polluters is emission reduction cost effective. Another drawback of regulation is that standards do not reflect a scarcity price of environmental goods (Siebert, 1987). Third, a drawback of regulating instruments is that they do not provide incentives to develop and adapt new abatement technologies. Finally, a drawback of regulations is that they may too easily be subject to bargaining and negotiations between public authorities and the private sector (Barde, 1995)

2.4.3 Economic instruments

Economic instruments aim at influencing the decision making of economic agents by offering financial incentives. Financial incentives can take different forms. In this section the economic instruments discussed are restricted to charges, subsidies and tradeable emission permits, as these are the key policy instruments.[18]

Charges

According to the neo-classical theory, applying policy instruments to control pollution should be aimed at reaching the optimum pollution level where social welfare is maximized. This optimum level can be reached by a Pigouvian tax, which must be equal to the marginal social damage in the Pareto optimal allocation. To reach the optimum pollution level, a unit tax must be attached directly to the polluting activity. However, as I explained in section 2.2.1, it is not possible to define this optimum without adequate knowledge of the monetary damage caused by pollution. One alternative in the absence of a measurable optimum is to determine policy targets for environmental quality (see section 2.2.3).

Emission charges can be applied to reach these desired targets. Although charges do not result in optimal control levels anymore, they do result in the lowest cost method for achieving a desired reduction in total emission (Baumol and Oates, 1988). A simple example may clarify this. Assume environmental

[18]For a complete overview of economic instruments see e.g. Bohm and Russell (1985) or Opschoor and Vos (1989).

policy aims at reducing sulphur dioxide emissions by 30% (performance standard). A possible way to achieve this goal is to require each source to reduce emission by 30%. However, if the marginal costs of reduction for some sources are much higher than for other sources, emission reduction will be much cheaper if sources with low marginal costs reduced more emission than sources with high marginal costs. In response to a tax, a cost-minimizing source will reduce its emission until the marginal cost of reduction is equal to the tax. Consequently, sources with low marginal costs will reduce more emission than sources with high marginal costs. A tax placed on emission at a level which results in a reduction of 30% would automatically result in the lowest cost solution (Baumol and Oates, 1988). This lowest cost feature is an attractive property of charges.

Another advantage concerns dynamic aspects. Charges provide a continuous incentive to search for the adoption of new environment-saving technology. Such technology results in lower emissions, which, in the long run means a saving in cost through a lower tax. Nevertheless, charges have some serious drawbacks as well. Although they result in the lowest cost solution from a polluter's point of view, total costs for emission sources might be higher than the costs of regulation because of them. By using charges, the total costs for the polluter consist of the control costs plus the tax which has to be paid on residual emission, whereas regulation only results in control costs. However, from the point of view of society, the charge is a transfer payment.

Another important difficulty of the use of charges is that the effect is uncertain. It is difficult to predict accurately at what level a charge has to be introduced in order to bring about a certain emission reduction. The environmental authority has no direct control over the quantity of emissions. Moreover it is doubtful whether efficient charges will ever be found. Inefficient charges result in additional costs (Bohm and Russell, 1985). A third drawback to the use of charges is that taxes have to be constantly adjusted because of economic growth and inflation. Inflation results in a lower real charge. In order to effect a similar emission reduction, the nominal charge has to be raised. At the same time, charges can add to inflation. Finally, in certain circumstances a charge might not be the most adequate instrument. Environmental problems do not always develop smoothly and gradually. The uncertainty associated with environmental conditions means that short term adaptations might be necessary. Under the assumption that tax reform is a difficult and slow process, flexible direct controls are a main additional instrument (Baumol and Oates, 1988).

An argument sometimes raised against the introduction of charges is that, generally speaking, charges distort. Charges alter relative prices and consequently change the allocation of resources, causing excess burden. According to traditional welfare theory, fiscal policy should generally aim at influencing allocation as little as possible. However, as the aim of charging pollution is just to change resource allocation, a distortion in allocation is justified. A distortion (e.g. pollution) can be improved by adding distortions of the right kind (i.e. the charge) (see e.g. Musgrave and Musgrave, 1984; Boadway and Bruce, 1984).

Subsidies

The opposite of charges are subsidies. A subsidy per unit of emission reduction could establish the same incentive as a charge of the same magnitude per unit pollution discharge. Like charges, subsidies result in cost-effective emission abatement, and provide an incentive to search for and implement new abatement technologies. The disadvantages of charges - uncertainty, sensitivity to inflation and economic growth - hold for subsidies as well. However, there are some main asymmetries between subsidies and charges. The first is the different implication for the profitability of production in a polluting industry. A subsidy increases profits whereas charges decrease them. Another asymmetry is that subsidies may cause an increase in the aggregated supply. Thirdly, subsidies may induce a firm to begin polluting more than it would otherwise have done, in order to qualify for a larger subsidy. Another issue is that, unlike charges, subsidies do not agree with the 'polluter pays' principle. For these reasons, subsidies are not a fully satisfactory alternative to emission charges.

Tradeable emission permits

Tradeable emission permits are environmental quotas that, once initially distributed can be traded between the polluters. An emission permit gives the right to emit one unit of a pollutant. Tradeable permits emerged from Coase's work

(1960) as tradeable permits for pollution problems institutionalizes property rights in an artificial market.

A system of tradeable pollution permits operates as follows. In order to emit a certain amount of pollution, a source is obliged to possess the appropriate number of permits. The environmental authority distributes a number of pollution permits according to the environmental standard agreed upon. Because of the tradeability of permits, a permit market develops and a market clearing price emerges. Cost minimizing polluters will buy permits as long as the marginal control costs exceed the price of permits. They will sell permits as long as the permit price exceeds marginal control costs. Consequently, like charges, tradeable pollution permits result in equalizing marginal control costs among sources. Equalized marginal costs is the first order condition for the lowest cost allocation of permits among sources (Baumol and Oates, 1988).

According to Bohm and Russel (1985), tradeable pollution permits amount to a dual between emission charges: emission quantities instead of prices are set by the environmental authority, and prices instead of discharge result from the free choices of those subject to the system. In contrast to pollution charges, because of the ceiling set on pollution by a system of tradeable pollution permits, a maximum pollution level is guaranteed. This certainty is the main benefit of tradeable pollution permits over charges. Another benefit compared to charges is that economic growth and inflation do not cause complications. Adjustments in the permit price will take place automatically without an increase in pollution (Baumol and Oates, 1988; Howe, 1994).

A third benefit discussed in the literature concerns costs. As I made clear in the previous section, charges might imply high costs on polluters. Consequently, tradeable pollution permits might be freely initially distributed (Baumol and Oates, 1988). However, free initial distribution has its drawbacks as well. First, there is the ethical question of whether to give polluters the right to pollute at zero costs. Moreover, it puts 'new' polluters at a disadvantage. Second, the initial distribution probably does not coincide with the lowest cost allocation. In order to reach this emission, sources have to trade pollution permits. Trading permits implies transaction costs. If pollution permits are initially auctioned, tradeable permits have no cost advantage over charges. In conclusion, tradeable pollution permits may have a potential cost advantage over charges.

In conclusion, both emission charges and tradeable emission permits result in cost-effective emission abatement. The crucial difference is that, from a policy point of view, permits give the environmental authority direct control over the emissions.[19]

2.4.4 Other instruments

Finally, some attention should be paid to a category of instruments which are neither economic instruments nor regulatory instruments. This category consists of the instruments education, propaganda and negotiation and is aimed at internalizing environmental awareness and responsibility (Opschoor and Vos, 1989). These instruments are rather important in environmental policy. Increased consciousness of environmental problems might stimulate producer and consumer decision making in an environmentally sound direction. Another instrument that needs to be mentioned is that of voluntary agreements. These can be defined as deals between government and industry whereby an industry or a group of individual corporations agrees to reach certain environmental objectives within a defined time frame. The main advantage of this instrument is the flexibility, transparency and incentive it gives to industry. However, there is also a risk that voluntary agreements will reduce government control over industry (Barde, 1995).

2.4.5 Instruments compared for non-uniformly mixing pollutants

In the previous sections I outlined the advantages and disadvantages of different instruments. The most promising instruments are charges and tradeable emission permits. However, no attention has yet been paid to the specific circumstances in which these instruments are applied to non-uniformly mixing pollutants. As shown in section 2.2, cost effective emission abatement of non-uniformly mixing pollutants is not characterized by equalized marginal abatement costs. Consequently, a uniform effluent charge is no longer an appropriate instrument. If charges are

[19]Comparative characteristics are schematically summarized in Howe (1994).

applied, they have to be differentiated among sources. A system of tradeable emission permits also needs adapting if it is to take emission location into account.

The established idea concerning the use of emission charges in the case of multiple receptors is that, to set emission charges which result in cost-effective emission abatement, the environmental authority has to know transfer coefficients and abatement costs of all sources (e.g. Bohm and Russel, 1985). This idea has been contradicted by Ermoliev et al. (1996). In their study they have shown that in the case of multiple receptors, by using an adjustment mechanism, the environmental agency is able to implement charges step by step which results in cost effective emission abatement, without having information on abatement costs. This mechanism uses the difference between the actual deposition level and the target deposition level as a signal for the adjustment of emission charges (Ermoliev et al., 1996).

If the environmental agency wants to set the appropriate emission charge at once, it needs to know the transfer coefficients and abatement costs of all sources in order to determine the shadow price of each receptor (λ_i). These shadow prices follow from the solution of the optimization problem:

$$Min. \quad \sum_j C_j(e_j) \tag{2.21}$$

$$s.t. \quad \sum_j a_{ij}\, e_j \leq d_i^* \quad i=1,\dots,I \tag{2.22}$$

$$e_j \geq 0,\ j=1,\dots,J \tag{2.23}$$

A vector of shadow prices at receptors which corresponds with the vector of cost effective emission standards does exist (Tietenberg, 1978). The linear structure of the pollution transport equations allows to calculate a vector of emission charges. The emission charge for a source j can be written as a function of transfer coefficients and the shadow prices λ_i, $i=1,\dots,I$ (Ermoliev et al., 1996):

$$u_j = a_{1j}\lambda_1 + a_{2j}\lambda_2 + \dots + a_{Ij}\lambda_I \tag{2.24}$$

However, if an adjustment mechanism is introduced, this prerequisite no longer applies. The adjustment mechanism uses a separation of the optimization problem as formulated by equation *(2.21)-(2.23)*. The first stage is that the environmental authority, given the information on the discrepancy between actual depositions and target depositions, has to decide on the initial levels of the emission charges and the adjustments of these levels. The second stage is that the individual cost-minimizing polluters choose their emission level, given the emission charge that is imposed by the environmental agency.

To explain the adjustment mechanism the Lagrange function of the optimization problem as formulated by equation *(2.21) - (2.23)* has to be considered (Ermoliev et al., 1996):

$$L(e,\lambda) = \sum_j C_j(e_j) + \sum_i \lambda_i (\sum_j e_j a_{ij} - d_i)$$
$$= \sum_j [C_j(e_j) + (\sum_i \lambda_i a_{ij}) e_j] - \sum_i \lambda_i d_i$$

The Lagrangean $L(e,\lambda)$ has a saddle point: a pair (e^*,λ^*) exists so that $L(e^*,\lambda) \leq L(e^*,\lambda^*) \leq L(e,\lambda^*)$. This is true when $C_j(e_j)$ are convex functions as is usually assumed (Ermoliev et al., 1996). Given this assumption, a vector of optimal emissions for all sources is achieved by solving the I independent subproblems:

$$Min.\ C_j(e_j) + (\sum_i \lambda_i a_{ij}) e_j, \quad e_j \geq 0 \tag{2.25}$$

where the values $u^*_j = \sum_i \lambda^*_j a_{ij}$ can be interpreted as optimal charges on emissions at sources. The subproblem which has to be solved by each emitter is how to minimize its individual costs on emission reduction and emission charges.

The question now is how the environmental agency can identify λ^* and u^* without having knowledge of the cost functions of the individual sources. The answer is by the use of an adjustment mechanism for emission charges, which step by step results in the optimal charges. This adjustment mechanism works as follows. First (step k=0), the agency chooses a vector of shadow prices λ^k, for example based on the initial exceedances. Equation *(2.25)* translates the shadow prices λ^k to emission charges u^k. Each polluter will adjust its emission level by minimizing total costs: $C_j(e_j^k) + u_j^k e_j^k$. Next, the agency observes the deposition resulting from the emission levels e^k and calculates the difference between actual depositions and target depositions. Then in a next step *(k+1)* the pollution charges are adjusted according to the formula:

$$\lambda_i^{k+1} = Max.\ \{0, \lambda^k_j + \rho_k (\sum_i e_j^k a_{ij} - d_i)\} \tag{2.26}$$

According to formula *(2.26)*, λ_i will increase in a next step if the actual deposition exceeds the deposition target. However, if the deposition target exceeds the actual emissions, λ_i will decrease in a following step. Obviously, if the deposition target is met, the shadow price will be in accordance with cost effective emission abatement and will not have to be adjusted. Ermoliev et al. (1996) have shown that *(2.26)* converges for any sequence of $\rho_0, \rho_1 \ldots$ so that $\rho_k \geq 0, \sum_{k=0}^{\infty} \rho_k = \infty$. These conditions guarantee a convergence independent of the initial vector of shadow prices λ^0.

Although it has been mathematically proved that it is possible to implement emission charges that will result in cost effective emission abatement, in practice, without having information on the abatement costs, this system will be very time consuming. Countries will resist this method as changing charge rates cause large uncertainty. Moreover, since the choice of the step multipliers $\rho_0, \rho_1, \ldots$ is based on excess concentrations (positive or negative) and not on the calculation of the total costs, the adjustment mechanism described does not lead to a monotonic decrease of total costs. Another argument against this charge is that the charges will differ among countries which can be regarded as injustice.

A system of tradeable emission permits must also be adapted to make it suitable for non-uniformly mixing pollutants. Several alternatives have been developed for taking emission location into consideration. One is the use of tradeable deposition permits instead of emission permits. A second option is trading emission permits in zones, within each of which all emissions can be treated as a single source. A third option is the so-called pollution offset systems (Tietenberg, 1985). In this thesis I explore the use of tradeable emission permits for non-uniformly mixing pollutants further.

2.5 Conclusion

In this chapter, I have discussed theoretical principles for acid rain policy, and have taken several issues have been taken into account. First, economic theory has been reviewed, next the need for cooperation has been outlined and finally, the choice of instruments has been discussed.

Neo-classical economic theory focuses on optimal pollution control. The optimal control level occurs at the intersection of marginal control costs and marginal benefits of pollution control. At this control level, total welfare for a society is maximized. Because of difficulties in establishing the marginal benefits of pollution control, it is hard to define an optimal control level in practice. In this thesis I follow the environmental criteria approach. This approach abandons the aim of an optimal allocation in the sense of achieving maximum welfare. Instead, it aims at cost-effective abatement. Once the environmental criteria set by the authority have been given, conditions for cost minimization can be derived. As this approach is well suited for empirical environmental problems, I have used it in this study.

To achieve cost-effective international emission abatement, cooperation is needed. However, Mäler (1990) indicated that if dynamics are taken into account, i.e., if the damage is assumed to depend upon the accumulated deposition corrected for the assimilative capacity of the environment, then both cooperative and non-cooperative emission abatement will result in critical load deposition levels. But, cooperative emission abatement converges to critical loads sooner and the total damage is smaller. Generally, individual countries do not have incentives to cooperate. According to the prisoners' dilemma, individual rationalism hampers the reaching of agreements on cooperative pollution control. However, the real situation is not as bad as this theory suggest, as there are actually several ways of applying sanctions. Moreover, side payments can be introduced to stimulate countries to participate in cooperative emission abatement.

The final question concerns the most suitable policy instrument for international emission abatement. For reason of cost effectiveness, economic instruments are to be preferred to regulation. Generally, these instruments will bring about an abatement allocation that minimizes total abatement costs. They allow more flexibility and provide a continuous incentive to search for and adapt new abatement technologies. For non-uniformly mixing pollutants systems of emission charges and of tradeable emission permits both have to be adapted to take emission location into account. This thesis elaborates the use of tradeable emission permits.

3 THEORY OF TRADEABLE EMISSION PERMITS

3.1 Introduction

In Chapter 2, I argued that tradeable pollution permits have some attractive properties which make them suitable for application in cost-effective emission-abatement strategies. Having discussed tradeable permit systems in general in the previous chapter, in this chapter I examine them more thoroughly. Permit trading for non-uniformly mixing pollutants is analysed in particular. The difficulties that occur in permit trading for this type of pollutant are shown and the alternatives put forward in the literature to solve these are discussed. The theory of tradeable emission permits has been well documented by Tietenberg (1985). This theory is elaborated on, illustrated and extended by reviewing some empirical studies.

Several issues on permit trading are dealt with. First a distinction is made between emission permits (section 3.2) and deposition permits (section 3.3). An emission permit gives the right to emit a unit of pollutant whereas a deposition permit gives the right to deposit a unit of pollutant at a certain receptor. Montgomery (1972) showed that for both kinds of permits, competitive markets result in cost effective abatement. In section 3.3, I argue that a system of tradeable deposition permits is, in theory, well suited to non-uniformly mixing pollutants. However, as there are important disadvantages to the implementation of a system of tradeable deposition permits, several alternative systems of tradeable permits for non-uniformly mixing pollutants can be found in the literature. These alternatives are discussed in section 3.4.

Atkinson and Tietenberg (1991) pointed out the difference between the actual trading process and the implicitly assumed trading process as modelled in existing empirical studies. It was generally assumed that permit trading is a simultaneous process. Atkinson and Tietenberg (1991), however, suggested that permit trading in practice is a bilateral and sequential process. I examine this issue of bilateral and sequential permit trading in section 3.5.

As well as the problem of which permit system to apply, another question that arises is the initial allocation of permits. Permits can be sold by auction or they can be distributed freely. In the latter case, the question arises of according to which principle permits should be distributed. As this thesis is concerned with an international environmental pollutant, the distribution of permits is a topic of international negotiation. Section 3.6 reviews the topic of the initial allocation of permits.

In the United States, a system of tradeable emission permits for SO_2 emissions has already been implemented (see e.g. Peeters, 1992; Chicago Board of Trade, 1995). Although the structure of environmental authority in the United States differs largely from that in Europe, it is of interest to discuss the US permit trading system. This is done in section 3.7. Finally, in section 3.8 I draw conclusions on permit trading for non-uniformly mixing pollutants.

3.2 Tradeable emission permits

An emission permit gives the right to its holder to emit a unit of a pollutant. By issuing a certain number of emission permits and subsequently requiring a polluter to posses the appropriate number of emission permits according to its emission level, authorities are able to control the total amount of emission. If permit markets are fully competitive, free trade in emission permits result in a cost-effective permit allocation among emission sources, provided the aim is to reduce the total quantity of emissions. The resulting allocation is characterized by emission abatement that meets the pollution target (defined by the total number of permits) at minimum abatement cost.

It is easy to illustrate how this allocation occurs. Initially, a source j is endowed with emission permits q^0_j. Assuming sources aim at minimizing costs, it is profitable for a source to sell emission permits so long as the price of emission permits exceeds the costs of additional emission abatement. However, if the cost of additional emission abatement exceeds the permit price, it is rational to buy permits. This can be illustrated mathematically for a source j (Tietenberg, 1985):

$$Min \ \{C_j(r_j) + P(q^1_j - q^0_j)\} \tag{3.1}$$

$$s.t. \quad r_j = E_j - q^1_j \tag{3.2}$$

$$r_j \geq 0 \qquad (3.3)$$

where:

$C_j(r_j)$: abatement cost of source j

r_j: abatement level of source j

E_j: initial emission of source j

q^0_j: initial allocated emission permits of source j

q^1_j: amount of permits of source j after permit trading

P: permit price

According to equation *(3.1)*, costs include abatement costs and the costs of buying permits or, in the case of selling permits, the benefits of selling. If emissions exceed the initial number of issued permits $((E_j - r_j) > q^0_j)$, a source has to buy emission permits or increase emission abatement. As long as the initial number of available emission permits exceeds the actual emissions $((E_j - r_j) < q^0_j)$, a source is able to sell permits or, if possible, decrease emission abatement. The choice between additional emission abatement and trading permits depends upon the price of the permit and the marginal abatement costs of the source. However, a source will only be able to buy permits from another source if the latter's abatement cost is less than the price at which permits are sold. Therefore, the source that does the cleaning is the one with lower abatement costs. Obviously, the total number of permits over all sources j remains constant during trading, that is: $\sum_j q^0_j = \sum_j q^1_j$.

The conditions for a cost-effective allocation for the set of all sources is (Tietenberg, 1985):

$$dC(r_j)/dr_j - P \geq 0 \qquad (3.4)$$

$$r_j[\, dC(r_j)/dr_j - P \,] = 0 \qquad (3.5)$$

$$r_j \geq 0 \qquad (3.6)$$

If the market for emission permits is fully competitive, then an equilibrium price P^* will emerge. The equations show that marginal abatement costs will be equal to P^*, and consequently will result in equalized marginal abatement costs between polluters, which yields the cost-effective allocation.[20] Abatement costs increase exponentially if r_j approximates E_j, therefore q^1_j is always positive. The equilibrium permit price is a function of the total number of permits issued, the total initial emissions before any reductions, and the marginal abatement costs of emission reduction (Bohm and Larsen, 1994).

Theoretically, for uniformly mixing emissions like greenhouse gases, tradeable emission permits are a suitable instrument for generating a cost-effective emission reduction, provided that control costs and transaction costs are not excessive. However, this does not hold for emissions that are non-uniformly mixing. For these, the damage caused depends on the amount of emission, on the location of the emission sources, and on factors of air chemistry and meteorology. For this type of emission, a system of tradeable emission permits will not generally automatically result in cost-effective emission abatement. To what extent a tradeable emission permit system deviates from the cost-effective allocation for non-uniformly mixing pollutants depends on the source-receptor matrix, the source's abatement costs and the deposition targets. Empirical research can provide insight in this matter (Tietenberg, 1995).

3.3 Tradeable deposition permits

An obvious alternative to emission permits are deposition permits which take into account the location of emissions. A deposition permit gives the holder the right to deposit a unit of pollutant at a certain receptor. Authorities are able to realize deposition targets by issuing the number of deposition permits for each receptor according to its deposition target, and by subsequently requiring sources to possess the appropriate number of deposition permits for each receptor affected. In this way, both the emission location and emission dispersion are taken into account.

In order to emit a pollutant, a source has to own the appropriate number of deposition permits for the receptors it affects. The appropriate number of deposition permits necessary for source j follows from the transport coefficient a_{ij},

---

[20]For a buyer it may be that: $dC_j(r_j)/dr_j > P$ if $r_j = 0$. In that case, the marginal abatement costs will not be equal to P^*.

representing the relative contribution of the emission of source j at the deposition at receptor i (see section 1.3). If deposition permits for each receptor are issued, a system of tradeable deposition permits results in separate markets for each receptor. Consequently, a source affecting several receptors has to operate in a number of markets. An individual source that minimizes costs can be represented by equations *(3.7) - (3.9)* (Montgomery, 1972):

$$Min\ C_j(r_j) + \Sigma^I_{i=1} P_i[q^1_{ij} - q^0_{ij}] \tag{3.7}$$

$$s.t. \quad \Sigma\, a_{ij}(E_j - r_j) \leq q^1_j, \quad for\ each\ i \tag{3.8}$$

$$r_j \geq 0 \tag{3.9}$$

where:

P_i: permit price for receptor i

q^0_{ij}: initially allocated deposition permits of source j for receptor i

q^1_{ij}: deposition permits of source j for receptor i after permit trading

According to equation *(3.7)*, total costs include abatement costs and costs of buying permits or, in the case of selling, the benefits of selling permits. A source requires permits for all receptors i it affects, which means the costs of permits are summed over i. If emissions exceed the initial number of issued permits for a receptor, a source either has to buy deposition permits for that receptor or has to increase emission abatement. So long as the initial number of available deposition permits exceeds the actual emissions, a source is able to sell permits or, if possible, decrease emission abatement. Unrestricted trade in deposition permits will result in a cost effective deposition permit allocation. This allocation is characterized by emissions which meet the deposition targets at minimum abatement costs. The allocation which minimizes abatement costs is characterized by equations *(3.10) - (3.12)* (Tietenberg, 1985):

$$dC_j(r_j)/dr_j - \Sigma^I_{i=1} P_i\, a_{ij} \geq 0 \tag{3.10}$$

$$r_j[dC_j(r_j)/dr_j - \Sigma^I_{i=1} P_i\, a_{ij}] = 0 \tag{3.11}$$

$$r_j \geq 0 \tag{3.12}$$

Atkinson and Tietenberg (1982) noted that a deposition permit system has the potentially undesirable characteristic that while it controls the deposition target at the receptors, it may do so while allowing increases in total emissions. They suggested that this increase is politically objectionable.[21]

Apart from political considerations, the deposition permit system also has practical problems. The difficulty for a source to operate under a deposition permit system is sketched in Tietenberg (1985). If a source wants to increase emissions, separate permits have to be bought for each affected receptor, in different markets. In general, the permit price will differ between receptors, which reflects the difficulty of meeting the deposition target at that receptor. In markets with few participants, prices may be uncertain. Moreover, the demand for permits in a market not only depends on permit prices in that market, but also on prices in all other permit markets. This interdependency among markets means that a source has to operate simultaneously in all relevant markets, since a source can not definitely buy in market A until it knows the price in market B and vice versa (Tietenberg, 1985). Although these arguments certainly hold, it is no novelty for firms to deal with uncertain prices or with dependency among markets. However, in a situation with a large number of receptors, trading becomes quite complicated resulting in high transaction costs. Because of these problems related to the implementation of a system of tradeable deposition permits, alternative permit systems have had to be developed.

3.4 Alternative permit systems for non-uniformly mixing pollutants

3.4.1 An overview

Several alternatives for deposition permit trading that take emission location into account are found in the literature. These alternatives range from the relatively simple to the more advanced. Four alternatives are discussed here: (1) emission permits, (2) zonal emission permits, (3) a "worst case" approach, and (4) trading subject to trading rules. As Tietenberg (1985) indicated the fourth option to be a

[21]Atkinson and Tietenberg (1982) pointed out in particular that while total emissions are clearly of secondary importance in the US Clean Air Act, strategies which allow large aggregate emission increases may not be allowed under the current Act.

promising one, I examine it in detail in the following section. The first three options are discussed in this section.

Emission permits

The crudest alternative is simply to ignore spatial complexity and to introduce a system of tradeable emission permits (Tietenberg, 1985). As I showed in section 3.2, such a system will result in equal marginal costs of emission reduction among sources. Nevertheless, there is no guarantee that deposition targets will not be violated. This equal marginal cost component is an advantage of the system since equalized marginal costs are a necessary condition for achieving a certain degree of required reduction at the lowest cost. However, the degree of required emission reduction needed to reach deposition targets cost-effectively is usually not the same for all sources, since what is important in determining how much emission reduction is needed for non-uniformly mixing pollutants is the location of sources and emission permits have no control over this.

Whether emission permits provide a suitable instrument for pollution control depends on the degree of required control component. The degree of emission reduction required by emission permits depends on the spatial configuration of sources. If a few sources affect a receptor requiring a large deposition reduction, these sources will need to control their emissions to a very high degree. This will result in high marginal abatement costs. This emission reduction can only be realized by issuing a limited number of emission permits. As these cause marginal abatement costs to be equalized across all sources, other sources also face high marginal costs of control, despite the fact that emissions from these sources may have very little impact on the receptors where the greatest environmental improvement is needed (Tietenberg, 1995). In conclusion, if a cluster of sources dominates a receptor, the emission permit system induces additional emission control in order to reach deposition targets. In that case, the degree of required control dominates the advantage of the equal marginal cost characteristic, causing the control costs to be too high with an emission permit system.[22] Where there is no

[22]It is assumed that policy makers aim at non-violation of deposition targets. Alternatively, issuing more emission permits implies lower abatement costs, but will cause a violation of these targets.

cluster of sources dominating a vulnerable receptor, the implementation of a system of emission permits leads to less overcontrol (Tietenberg, 1995). In this latter situation, an emission permit system might be a suitable instrument for non-uniformly mixing pollutants.

To judge whether emission permits are, indeed, a suitable instrument there needs to be an examination of the practical situation. For Europe, empirical research indicates that unrestricted emission trading results in a substantial loss in the ecosystem protection (Klaassen, 1995). In the USA, opinions differ on this subject. Kete (1992) stated that in the Sulphur Allowance Program in the USA, the expected reductions from an unrestricted emission trading system are not likely to lead to a worse environmental situation than that of a command and control policy. 'Sensitive ecosystems will be protected and improved by reductions in deposition regardless of which particular sources are controlled, or to what specific degree' (Kete, 1992:82). Kete's motivation was as follows. Acid rain involves a great number of sources, and emissions are transported long distances. Therefore deposits come from many dispersed sources. 'Thus no one source, not even the largest and dirtiest of the old power plants in the Ohio River Valley, dominates deposition at any northeast or mid-Appalachian region receptor.' This view is slightly contradicted and modified by the NAPAP (1991) research results. Although emission permit trading is not explicitly modelled, preliminary analysis indicates that deposition patterns will change with interstate trading. But the importance of geographical differences diminishes as the overall reduction goal increases. Three categories can be identified for SO_2 emission reductions below the 1980 levels (NAPAP, 1991).

1. Reduction goals up to 8 million tonnes: different geographical patterns of control are potentially important.
2. Reduction goals between 8 - 12 million tonnes: 'transitional'.
3. Reduction goals above 12 million tonnes: geographical patterns are relatively unimportant because virtually all major sources would need to be controlled to very low emission levels.

This classification assumes that very low emission control levels imply an equal percentage reduction. Otherwise geographical differences in deposition patterns will occur. Kete's view also contradicts with research results of Young and Shaw

(1986) who indicated that most sulphur emission reductions should take place within a limited number of regions if the aim is to reach deposition targets efficiently. On the basis of these research results I conclude that unrestricted trading in emission permits will generally not be a suitable instrument for controlling non-uniformly mixing emissions.

Zonal emission permits

A second alternative for deposition permit trading is zonal emission permit trading. In this, the control region is divided into a number of zones. Trading in emission permits on a one to one basis is only allowed within one zone, and trading permits between zones is prohibited. Creation of zones takes location crudely into account. The underlying idea is that sources in one zone are closely clustered with more or less the same transfer coefficients. The advantage of zonal permit trading is that it is a rather uncomplicated system that prevents the most serious damage at vulnerable receptors. However, the implementation of a zonal system places a larger burden on the control authority than the implementation of a pure emission permits system. The initial zonal allocation of permits is difficult. Allocating too much control responsibility to one zone and too little to another hampers the achievement of a cost-effective allocation (Tietenberg, 1985).

Trading permits that are restricted to zones obviously reduce the potential for cost savings. As the creation of zones reduces the number of sources that are allowed to trade with each other, trade opportunities are restricted. There is generally a trade off between the size of the zone (the number of sources) and non- violation of deposition targets. Non-violation of deposition targets requires small zones, but this consequently decreases cost savings. How sensitive the cost penalty is to the size of the market is an empirical question. Some empirical work has indicated that substantial savings can be achieved in emissions trading even when the trading areas are quite small. The suggestion is that even when small zones are necessary, emissions trading may still represent an improvement over traditional regulatory policies which do not allow any trading at all (Tietenberg, 1995).

Worst case approach

The third alternative, the "worst case" approach, involves a simple version of tradeable deposition permits. Instead of issuing permits for all receptors, only a single receptor deposition-permit market is created. In this all trades are completed on the basis of their effect on a single "worst case" receptor. Other receptors are presumed to benefit indirectly from the emission reduction needed to meet the deposition target at this single receptor. The main benefit of this single receptor permit market is that transaction complexity is avoided, and as permit trading is not restricted, the cost penalty associated with restricted trade opportunities is avoided. Single receptor dominance is a prerequisite in this approach to the guaranteeing of non-violation of deposition targets at all receptors. For a single receptor dominance to exist, the deposition constraint has to be binding on only one receptor in the cost-minimum abatement allocation. If more than one binding receptor exists, a single deposition-permit market no longer guarantees compliance by these other receptors. Additional emission reduction is needed to achieve non-violation of deposition targets at these receptors. The amount of excess emission reduction needed to assure compliance depends on the proximity of the binding receptors to each other (Tietenberg, 1985).

A second consideration regarding a single receptor permit market put forward by Tietenberg (1985) is that locational incentives are established over time. Sources tend to be situated at a distance, thus avoiding affecting the "worst case" receptor, because at a distance, sources affect this receptor to a smaller degree. This means that fewer permits are required to legitimize emissions. Consequently, new "worst case" receptors might emerge. However, locational incentives to move pollution activities might also have positive aspect if polluting activities are relocated so that fewer vulnerable receptors are affected. This consideration raises the question of to what extent firms take environmental aspects into account in their choice on location. As the cost of environmental policy is only a small fraction of total costs related to this choice, locational incentives will not be very strong (Komen en Folmer, 1995). A profound discussion on this topic is beyond the scope of this thesis.

If there is more than one binding receptor, an obvious alternative is to create separate permit markets for each of these receptors, thus combining zonal permit trading and a "worst case" type of permit trading. The idea is as follows. Trading emission permits is replaced by trading in deposition permits. In each established zone, deposition permits for one single "worst case" receptor are traded, it being assumed that the other receptors will benefit indirectly from the emission reduction needed for compliance by the "worst case" receptor. The establishment of zones has to be based on the effect sources have on binding receptors. By grouping all the sources that affect a "worst case" receptor in one zone, the non-violation of deposition targets is guaranteed. A difficulty occurs if sources affect more than one binding receptor. It is not easily to classify these sources in one receptor market while guaranteeing non-violation of deposition targets. Alternatively, these sources could operate in more than one market, but this would diminish the attractiveness of the system. As the number of binding receptors increase, so the complexity of this system approaches that of the deposition permit system where separate permits for each receptor are required.

Emission trading subject to rules

The fourth alternative for emission trading in non-uniformly mixing pollutants is to subject emission trading to rules regulating deposition. According to Tietenberg (1985), the introduction of trading rules is probably the most practical way for incorporating source location into permit based pollution control policies. If spatial complexity is to be taken into account, trading cannot be allowed to be unlimited, but has to meet certain constraints on the exchange of emission permits. Trading subject to rules sometimes implies the use of trade ratios in permit trading. Several studies applying these kinds of trade restrictions have been described in the literature. I discuss trading rules and trading ratios in more detail in the next section.

3.4.2 Trading rules and trading ratios

Emission trading subject to rules is a hybrid approach that combines certain characteristics of both deposition permit trading and emission permit trading. The basic idea is to define permits in terms of emissions and allow their sale among polluters, though generally not on a one-to-one basis. More specifically, transfers of emission permits are subject to the restriction that the transfer does not result in a violation of the deposition targets. This explains the hybrid character of emission trading subject to rules. It involves the purchase and sale of emission permits, and at the same time, it captures the spirit of a deposition-based system because the rate at which permits exchange with one another depends on the relative effects of the associated emissions on deposition level at receptors (Krupnick et al., 1983). Three trading rules have been suggested to account for deposition targets.

1. The pollution offset (Krupnick et al., 1983).
2. The non-degradation offset (Atkinson and Tietenberg, 1982).
3. The modified pollution offset (McGartland and Oates, 1985).

The pollution offset approach allows emission trading so long as deposition targets are not violated. The non-degradation offset rule ensures that trading does not violate deposition targets and, at the same time, prevents an increase in the total amount of emissions. According to the modified pollution offset concept, trading in emission permits is allowed so long as both deposition targets and pre-trade deposition levels are not violated. This approach does not directly control the total amount of emissions. The differences between the three pollution offset systems can be illustrated mathematically by equations *(3.13)-(3.24)*. All pollution offset systems aim at minimizing costs, but the constraints differ.

Pollution offset system (Krupnick et al., 1983):

$$Min\ \Sigma^{J}_{j=1}\ C_j\ (r_j) \tag{3.13}$$

$$s.t. \quad \Sigma^{J}_{j=1}\ a_{ij}\ (E_j - r_j) \leq d^*_i,\ \forall\ i \tag{3.14}$$

$$r_j \geq 0 \tag{3.15}$$

Since the pollution offset system does not put any restriction on permit trading except that of the non-violation of deposition targets, the formal statement of this system coincides with the definition of cost-effective emission abatement (see section 2.2).

Non-degradation offset system (Atkinson and Tietenberg, 1982):

$$Min\ \Sigma^{J}_{j=1}\ C_j\ (r_j) \tag{3.16}$$

$$s.t. \quad \Sigma\ a_{ij}\ e^1_j \leq d^*_i,\ \forall\ i \tag{3.17}$$

$$\Sigma\, e^0_j \geq \Sigma\, e^1_j \qquad (3.18)$$

$$r_j = E_j - e^1_j \qquad (3.19)$$

$$r_j \geq 0 \qquad (3.20)$$

where:

e^0_j: pre trade emissions of source j

e^1_j: post trade emissions of source j

Modified pollution offset system (McGartland and Oates, 1985):

$$Min\ \Sigma^J_{j=1}\, C_j\,(r_j) \qquad (3.21)$$

$$s.t. \quad \Sigma\, a_{ij}\,(e^1_j) \leq Min.\ \{d^*_i,\ d^0_i\}\ \forall\ i \qquad (3.22)$$

$$e_j \geq 0 \qquad (3.23)$$

$$r_j \geq 0 \qquad (3.24)$$

where:

d^0_i : pre-trade deposition at receptor i

d^*_i : deposition target at receptor i

The pre-trade deposition at a receptor i equals the summation over j of the transfer coefficients and the pre-trade emissions concerned.

To explain the trade opportunities defined by the three different approaches consider Figure 3.1 which is a graph illustrating the different trade rules (Tietenberg, 1985). This graph is restricted to two receptors and two emission sources. Line *R1* shows the emission combinations of source 1 and source 2 which do not violate the deposition target at receptor 1. Line *R2* shows the emission combinations of source 1 and source 2 which do not violate the deposition target at receptor 2. Point *E* is the starting point representing the initial distribution of emission permits. In the initial permit allocation only the second receptor is binding. Given the initial permit allocation, line *Rc* represents combinations of emissions that keep the depositions at receptor 1 at pre-trade level. The 45 degree line represents combinations of emissions that hold the total emissions constant by construction. Since, after trade allowance, costs are lowered as one moves away from the origin in either the horizontal or vertical direction (because less control is required), emissions would be represented by a point on the outer edge of the appropriate frontier.

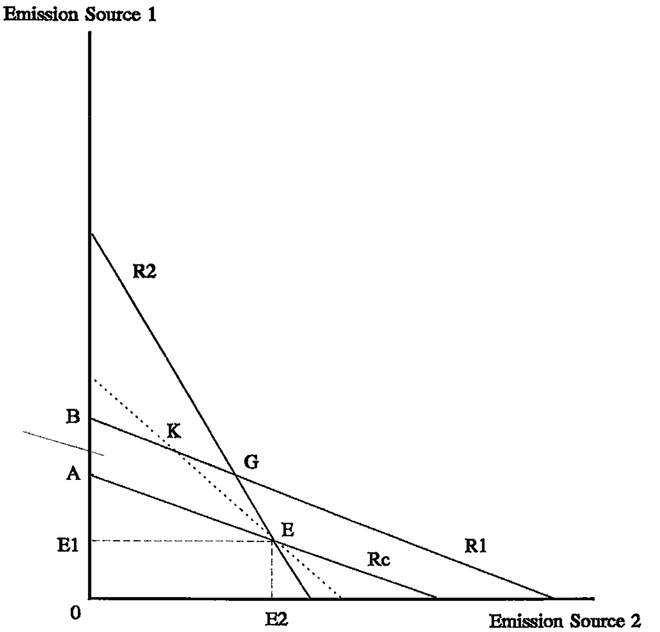

Figure 3.1 Trading Rules (Source: Tietenberg, 1985)

If source 2 is the first source to sell permits (which implies that source 2 decreases its emissions) starting from *E2*, the trade area using the pollution offset is represented by area E_2EGB0, because neither the deposition target at receptor 1 nor at receptor 2 may be violated. The pollution offset is the only rule that makes trading opportunities contingent to the pre-trade emission situation as *R1* and *R2* are only a function of deposition targets and transfer coefficients. If the non-degradation offset is applied, then the trade area is restricted to E_2EKB0, because, in addition to non-violation of deposition targets, total emissions are not allowed to increase, i.e. should remain to the left of and below the 45 degree line. The modified pollution offset restricts trade to the area E_2EA0. This rule allows trade only so long as neither deposition targets nor pre-trade deposition levels are violated. Therefore emissions should remain to the left of *Rc*. Both *Rc* and the 45 degree line depend on the initial allocation of emission permits of source 1 and 2 as both lines must pass through point *E*. This means that the degree to which the non-degradation and the modified pollution offset can be expected to diverge from the lowest cost allocation is sensitive to the initial allocation (Atkinson and Tietenberg, 1987).

Obviously, the pollution offset offers the most opportunities for trade, since it does not make trade dependent on the pre-trade emission level of the sources. Consequently, the pollution offset also offers the most opportunities for cost-savings, while guaranteeing the achievement of deposition targets. An underlying condition for this system to function well is the provision of additional emission permits by the environmental agency, provided deposition targets are not violated. This latter could cause a free rider problem where polluters may benefit from the transactions of others (McGartland, 1988). Consider the hypothetical situation as presented in Figure 3.2 where three sources, A, B and C, affect two receptors, 1 and 2.

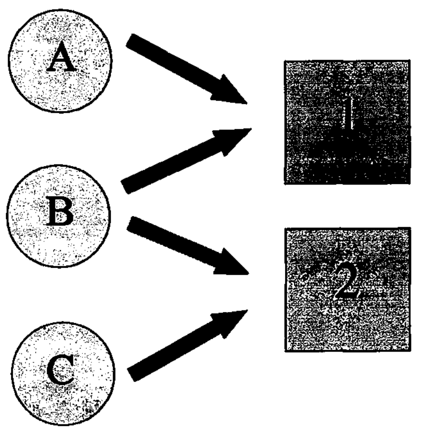

Figure 3.2 *Illustration of opportunities to free ride in case of a pollution offset system.*

Emissions of source A only affect receptor 1 and emissions of source C only affect receptor 2 whereas the emissions of source B affect both receptors 1 and 2. If source B sells emission permits to source C, then deposition at receptor 1 will always be reduced below the target. This provides source A with the opportunity to free ride, since it allows source A to increase emissions at no cost until the deposition at receptor 1 equals the target. Obviously, a free riding option may also exists for source A if receptor 1 is initially not binding.[23]

The modified pollution offset system excludes free riding initially since pre-trade deposition levels may not be violated in this system. However, after some trading has been carried out, the deposition at some receptors may be below the redefined targets. This would create an opportunity for free riding. Only the non-degradation offset will rule out free riding. It must be remembered that under this offset system, deposition targets may not be violated and total emissions may not increase. This latter requirement excludes free-riding opportunities since sources cannot increase emissions without another source decreasing emissions.

A study by McGartland and Oates (1985) for the Baltimore air quality control region indicates that permit trading subject to rules generates substantial cost savings compared to a command and control approach implying a equiproport-ional cut-back in emissions from a defined baseline, sufficient to attain the air quality standards. The study confirms that a pollution offset approach results in lower abatement costs than a modified pollution offset system. However, under the pollution offset, emissions increase dramatically compared to the command and control system, whereas under the modified offset system, total emissions are not much higher than emissions associated with the command and control approach. McGartland and Oates (1985) concluded that the modified pollution system is attractive on both economic and political grounds since it captures large cost savings while at the same time preventing any deterioration in environmental quality.

Atkinson and Tietenberg (1987) applied the non-degradation and the modified pollution offset to the St. Louis air quality region and the Cleveland area. Two air quality standards were considered for both areas. The permit trading systems were compared to the State Implementation Plan (SIP). In this plan, the control requirements for each source are in proportion to its weighted percentage degradation of regional air quality measured at all receptors. This study shows that the non-degradation offset trading rule yields a lower cost than the modified pollution offset. However, neither of these two rules can dominate universally (with respect

[23]This holds analogously for source B and C if receptor 2 is initially not binding.

to the lowest cost). If the starting allocation for trades controls emissions is sufficiently high, it is possible that the modified pollution offset trade system will result in lower control costs. In this case the amount of required emission reduction would be high, increasing the value of the flexibility inherent in trades which allow emissions to increase. Though modified pollution offset trades have this flexibility, non-degradation offset trades do not. The overall conclusion of Atkinson and Tietenberg (1987) was that the use of either trading rule represents a substantial improvement over the exclusive reliance on the SIP allocation.

3.5 Bilateral permit trading

Several authors have discussed possible impediments to the non-coinciding of emission trading with the solution of the mathematical optimization models, referring to cost-effective allocation of emission abatement. Hahn (1984) considered market power, Tschirhart (1984) the regulation of firms trading in permits and Zylicz (1993) the case of interacting pollutants. In this section I want to discuss losses in efficiency caused by bilateral sequential trading (BST) (Atkinson and Tietenberg, 1991).[24]

In the previous sections I have implicitly assumed trading in emission permits to be a simultaneous and multilateral process. This assumption on the trade process for non-uniformly mixing pollutants is a necessary condition for the coinciding of the market result with the cost-effective emission abatement. Atkinson and Tietenberg (1991) suggested that trading in emission permits is actually more likely to take place bilaterally and sequentially than simultaneously. This results in fewer cost savings than is suggested by the results of the optimization models. This is an important issue since it forces us to think about how to deal with trade restrictions on BST transactions as defined by trade rules. The cost effectiveness may also depend on the sequence in which trade takes place.

Atkinson and Tietenberg (1991) simulated bilateral sequential permit trading for air pollution in the St. Louis Air Quality region. Trading was restricted by the non-degradation offset, permit trading only being allowed if every bilateral trade

[24]For an overview of impediments see Munro et al. (1995).

agreed individually not to violate the environmental quality constraints and not to allow permit trading to increase total emissions. The simulation results confirm that bilateral sequential trades cannot reach the cost-effective emission allocation. Atkinson and Tietenberg suggested three reasons for this. First, some trades which could be completed in a multilateral, simultaneous trading environment could not be completed in a BST environment because they would violate the environmental standards. Second, for non-increasing emissions to reach the cost-effective allocation from the starting point, some trades need to be allowed to increase their emissions. This occurs when the purchasing source affects receptors whose deposition is below the targets. Third, the simulation model assumes that traders secure all the permits they can from their trading partner at once. Greater cost savings could result if some of these were reserved for subsequent trades (Atkinson and Tietenberg, 1991).

After Atkinson and Tietenberg (1991), other authors also used the concept of bilateral and sequential permit trading to analyse permit trading for non-uniformly mixing pollutants. Klaassen and Amann (1992) and Klaassen (1994a) studied a pollution offset for SO_2 emissions trading among European countries. The pollution offset can be implemented by means of an exchange rate which defines the volume of emissions that one source has to decrease (through selling permits) if another source increases its emissions by one unit (through buying permits). Two conditions have to be fulfilled to reach the cost minimum solution by this kind of permit system. First, the exchange rate between two traders has to equal the ratio of the marginal abatement costs in the cost minimum solution. Second, the initial permit allocation has to be on a line with a slope equal to the ratio of marginal cost in the optimum and this line has to go through the optimum. As this is a very restricting condition, Klaassen (1994a) concluded that the application of an exchange rate will not guarantee the attaining of the cost minimum solution, and will not ensure that deposition targets will be met. To analyse the extent to which exchange trading deviates from the cost-effective emission abatement, bilateral permit trading using an exchange rate must be simulated. The starting point for this simulation is the cost minimum allocation for attaining deposition targets based on a gap closure of 60%, assuming that countries at least carry out their current reduction plans. The simulation results indicate that exchange rate trading generates a cost saving of 5%. However, trading increases emissions by 3.5% causing a violation of deposition targets. To a certain degree these violations are small, but for a number of locations the deposition after trading is up to 40% higher than the deposition targets (Klaassen, 1994a).

Burtraw (1994) studied bilateral sequential permit trading in Europe by applying a pollution offset programme in which trade is allowed as long as any deposition constraint is not violated. As well as considering permit trading among nations, Burtraw (1994) also simulated trading between disaggregated stylized representations of economic enterprises by dividing national cost functions into two or three parts. Simulation results show that the performance of the trading programme improves significantly if trading takes place between enterprises rather than between nations. However, the cost minimum solution cannot be fully reached. Burtraw (1994) offered no explanation for this improvement. Klaassen (1995a), however, suggested that this improvement may simply follow from the increase in trading partners constructed by the dividing of the cost functions, since an increase in the number of trading partners increases the change that a trading partner can be found for a cost-saving trade that does not violate deposition constraints.

3.6 Initial permit allocation

Independent of what kind of permit trading system is used, emission permits have to be distributed initially before permit trading can actually start. The choice of an initial permit allocation is politically sensitive as this choice involves equity implications and may have strong economic consequences. Two questions regarding the initial distribution of emission permits at country level are addressed in this section: first, how the permits should be distributed among the countries; second, which principles are available to decide how many permits each country initially receives.

The first question considers whether permits should be initially distributed gratis or whether they should be auctioned. Auctioning of permits causes countries facing low marginal abatement costs to reduce emissions, whereas countries facing high marginal abatement costs will buy permits. If countries behave rationally, the auctioning of permits will lead to the cost-effective emission abatement allocation.

One drawback to the auctioning of permits is that it places a high financial burden on countries as they have to pay both control costs and permit costs (Atkinson and Tietenberg, 1984). Therefore countries will generally favour free initial distribution of permits. Another drawback is that country-specific circumstances, such as those resulting in extraordinarily large emission reductions cannot be compensated if permits are initially distributed by auction. Introducing an emission permit system in which permits are gratis provides a way to compensate countries.

This leads us to the second question addressed in this section: that of under which distribution rule permits should be initially distributed. Different rules for the distribution of permits result in differences in the distribution of gains from trade. Several distribution rules based on equity criteria are presented here. Rose and Stevens (1993) gave an overview of alternative equity criteria for global warming. These criteria are easily applicable to acidification as this involves the distribution of permits over a large number of countries. Table 3.1 summarizes the equity criteria, the general operational rule and the operational rule for allocating CO_2 (or SO_2) permits. The table illustrates that equity implies a normative evaluation that allows for more than one interpretation.

Table 3.1 Overview of equity criteria and the resulting distribution rule for CO_2 permits (Source: Rose and Stevens, 1993).

Criterion	General operational rule	Operational rule for CO_2 permits
horizontal	equalize net welfare change across nations	allocate permits to equalize net welfare gains (as proportion of GDP equal for each nation)
vertical	progressively share net welfare change	progressively distribute permits (net cost proportions inversely correlated with per capita GDP)
ability to pay	equalize abatement costs across nations	allocate permits to equalize abatement costs (as proportion of GDP)
sovereignty	cut back emissions proportionally across all nations	allocate permits in proportion to emissions
egalitarian	cut back emissions in proportion to population	allocate permits in proportion to population
market justice	make greater use of markets	auction permits to highest bidder
consensus	seek a political solution promoting stability	distribute permits so majority of nations are satisfied
compensation	compensate net losing nations	distribute permits so no nation suffers a net loss of welfare
Rawls' maximin	maximize the net benefit to the poorest nations	distribute large proportion of permits to poorest nations
environmental	cut back emissions to maximize environmental values	limit permits associated with vulnerable ecosystems

The 'ability to pay' criterion is based on abatement cost whereas the horizontal, vertical and compensation criterion are based on both the costs and the benefits of emission abatement. Although rough estimates are available for abatement costs, benefits of abatement however can hardly be measured. For this reason, the allocation of permits would, in practice, have to be based on relatively straightforward rules. In the first sulphur protocol on emissions reduction, the sovereignty criterion is applied as a rule for distribution of emissions reductions.[25] This criterion offers an operational advantage because it focuses on easily observable burden sharing. It is for this reason that international environmental agreements often take the form of a uniform percentage reduction. However, this rule minimizes the disruption of current production and works in favour of (heavy) polluters. Using population or GDP as a base favours other countries. An accept-

[25]This protocol, however, does not involve the use of tradeable permits.

able allocation rule might take account of historical emissions, GDP and population together (Zhang and Folmer, 1995).

The absence of an international institution that is able to enforce the initial allocation means that countries have to negotiate and it may be difficult to reach consensus amongst countries of widely differing levels of economic development (Collins, 1995). However, it was possible to negotiate a second sulphur dioxide protocol in which the agreed emission reductions differ substantially between countries.[26] The outcome of this protocol could be used as a starting point for trading permits.

3.7 Permit Trading in the U.S.

In the United States, air pollution control is laid down by the Clean Air Act (CAA). Among other things this act deals with SO_2 permit trading, but it also includes environmental quality objectives. In this section I turn to the permit trading system for SO_2 in the US, beginning with the environmental targets of the CAA.

Responsibility for the enforcement of the CAA is assigned to the Environmental Protection Agency (EPA), which has to establish national ambient air quality standards (NAAQS). Under the CAA, states are required to formulate a State Implementation Plan (SIP). This plan has to include enforceable emission limitations and other control measures for sources and timetables for compliance to reach the NAAQS. In addition to NAAQS and SIP provisions, the CAA imposes several types of technologically based standards on new and modified sources of emission. Standards for preventing significant deterioration are applied in areas which already have a better air quality than the NAAQS, (Peeters, 1992; Kete, 1994). The Clean Air Act Amendments of 1990 introduced the opportunity for large scale SO_2 permit trading among sources. This SO_2 trading involves a reduction of 10 million tons of SO_2 emissions in the 1980 levels[27]. Trading is restricted to the electric utility industry. The trading programme has two phases. Phase I which began in 1995, affects the 110 highest emitting utility plants and

[26]The Second Sulphur Protocol uses the environmental criterion.

[27]An American ton equals 907,19 kg.

involves a reduction of 5 million tons SO_2. Phase II, beginning in the year 2000, will tighten the annual emission limits and affect a total of 800 utility plants. The annual total allocated allowances in phase II amount to 8.9 million tons of sulphur dioxide (Peeters, 1992).

Beginning in 1995, allowances are allocated annually according to the grandfathering principle. Utilities that begin operating in 1996 or later will not be allocated permits. Instead they will have to buy allowances from the market or from EPA auctions. For this purpose, EPA will create a small allowances reserve. Each allowance is an authorization to emit 1 ton of SO_2 during or after a specified year. Obviously, a utility is only allowed to emit an amount of SO_2 if it holds the appropriate number of permits. If compliance is not achieved, a penalty of $ 2000 has to be paid per excess ton of emissions (Kete, 1994). An allowance may be bought, sold or banked. This latter implies that a source may keep the allowance for future use. However, regardless of the number of allowances a source holds, it may not emit at levels that violate federal or state limits (NAAQS and SIP). This requirement takes care of the emission location. Cronshaw (1994) indicated that firms are willing to bank permits if prices increase minimally with the rate of interest. However, if a firm is subject to profit regulation, it might be willing to bank permits, even though prices rise more slowly than the interest rate. Banking of permits is probably a good thing from the environmental perspective because it means that emission reductions occur sooner, rather than later (Kete, 1992).

Allowances can be bought from EPA's auctions and direct sales. In addition to the EPA allowances, private allowance holders may offer their allowances at the EPA auction. The auctions consist of a spot market, where allowances that can be used in the same year (or banked for future use) can be bought, and of an advance market where allowances usable in seven years can be bought. Apart from the auctions, EPA reserves allowances for direct sales at a fixed price of $1500. The main objective of these sales is to ensure that there is always an opportunity for new utilities to buy permits. Both the auctions and direct sales are administered by the Chicago Board of Trade, and are held annually. The first were held in 1993. Table 3.2 illustrates the auction results from 1993 to 1995.

Table 3.2 Overview of spot and advance auction results 1993-1995 (source: Chicago Board of Trade, 1995)

	Spot auction results			Advance auction results		
	1993	1994	1995	1993	1994	1995
offered allowances	95.010	58.001	8.306	30.500	47.000	7.000
offered allowances sold	10	0	600	0	800	400
total bid quantities	321.354	294.354	255.371	283.406	489.399	236.928
clearing price	\$131	\$150	\$130	\$122	\$140	\$126
average winning bid	\$156	\$159	\$132	\$136	\$149	\$128

The results from Table 3.2 indicate that SO_2 allowance trading is slowly increasing. As in phase I only 110 plants are involved in the trading programme, trading may be expected to expand in phase II, restricting the total emissions further and involving 800 plants.

A final comment on the US SO_2 trading concerns its emission-oriented character. The system itself is spatially indifferent and based on emissions only. Trading is not restricted by rules and no other modifications which take into account environmental targets apply. However, all the other CAA air pollution regulations continue to apply. Therefore, regardless of how many permits a unit holds, it cannot emit SO_2 above the SIP limits and other requirements of the CAA. This guarantees that environmental criteria are not violated.

3.8 Conclusion

In this chapter I have examined permit trading for non-uniformly mixing pollutants. Environmental degradation by these pollutants not only depends on the amount of emission but also on the emission location. Therefore the aim is to use permit trading for reaching deposition targets rather than emission targets. The difficulties occurring in permit trading for this type of pollutant have been illustrated and the alternatives found in the literature for solving these difficulties, have been discussed.

Using tradeable emission permits for non-uniformly mixing pollutants results in cost-effective emission abatement. However, it does not take the deposition targets being aimed at into account. The extent to which the resulting cost-effective emission allocation deviates from the cost-effective deposition allocation depends on the source receptor matrix and the marginal control costs of sources. Generally speaking, it cannot be concluded that emission permits are a suitable instrument for non-uniformly mixing pollutants, though in the US tradeable emission permits are actually used for SO_2. However, this emission trading does not aim at reaching a deposition target but is only emission-oriented. Environmental targets are taken care of by State Implementation Plans. Instead of using emission permits it may be more suitable to implement a system of deposition permits. Theoretically, the use of tradeable deposition permits results in a cost-effective allocation that meets deposition targets. However, a disadvantage of this system is that it may result in an increase in emissions, something which is politically delicate. Another argument put forward in the literature against this system is the trading complexity that occurs when sources have to operate simultaneously in many different markets.

Since the use of both emission and deposition permits is not considered a very suitable instrument, several alternatives permit systems for non-uniformly mixing pollutants have emerged in the literature. Relatively simple alternatives are permit trading within zones and the worst case approach. Unfortunately, however, these systems are no guarantee against the violation of deposition targets. A permit trading system that does not violate deposition targets, is perhaps a more advanced system where trading is subject to rules. There are three variants: the pollution offset, the non-degradation offset and the modified pollution offset. All systems guarantee non-violation of deposition targets. The pollution offset offers results in a cost-effective deposition allocation and offers the largest trade opportunities. However, it allows for an increase in emissions. The non-degradation offset system excludes an increase in emissions and the modified pollution offset sets no restriction on emissions but protects pre-trade deposition levels. Whether the non-degradation offset system or the modified pollution offset system results in the lowest cost is theoretically impossible to say.

In judging permit trading systems, special attention has to be paid to the nature of the trading process. Following Atkinson and Tietenberg (1991), I have assumed that permit trading is a bilateral and sequential process. Combining permit trading subject to rules with the bilateral and sequential trading concept sheds new light on the cost effectiveness of these permit trading systems. Empirical studies indicate that bilateral permit trading applying offset rules will not result in cost-effective abatement.

Assuming a bilateral and sequential trading process, the question whether an emission permit trading system can result in both minimum abatement costs and non-violation of deposition permits cannot yet be answered affirmatively. However, bilateral and sequential trading has only recently been applied in studies on permit trading and needs to be further examined. It has become obvious that permit trading has to be restricted in some way to take deposition targets into account. Restrictions are needed that will result in non-violation of deposition targets, while at the same time, will not hamper the achievement of the cost-effective allocation.

4 ACID RAIN MODELS

4.1 Introduction

In Chapter 2, I examined theoretical principles for emission abatement strategies. There are various models available for quantifying these strategies, both in terms of costs and in terms of environmental benefits. This chapter discusses integrated assessment models for acid rain. First, however, I want to briefly introduce these models.

Integrated assessment models assemble and integrate information for analysing the problem of (in this case) acidification. Integrated acid rain models provide a scientific basis for developing and evaluating alternative acid rain abatement strategies and are therefore a useful tool in decision-making. The major aim of these models is to assist in international policy-making. In this chapter three European models are discussed which can be characterized as integrated assessment models since they cover a wide range of aspects of acidification by combining information from different disciplines. These models are (i) the Regional Acidification INformation and Simulation (RAINS) model, developed at the International Institute of Applied Systems Analysis in Laxenburg, Austria (Alcamo et al., 1990), (ii) the Abatement Strategies Assessment Model (ASAM) developed at Imperial College London (ApSimon and Wilson, 1991), and (iii) the Coordinated Abatement Strategy Model (CASM) developed at the Stockholm Environmental Institute in York, United Kingdom (Rosemarin, 1991).

Integrated assessment models have a similar format. The starting point is economic activity, causing emissions that are linked to energy use. The spatial scale on which emissions are considered may differ. Emissions may be reduced by several abatement techniques or by changes in the fuel mix or through energy conservation. To calculate the costs of emission reduction strategies, abatement cost functions are specified. This enables comparison of the cost effectiveness of the alternative abatement options. Since acid rain is an international (continental) transboundary problem, the models described in this chapter include all European countries. As a result the level of detail that can be analysed by these models as far as emission control options are concerned is limited to the national level. No sub-national information is provided. To calculate the deposition resulting from an emission allocation a source-receptor matrix is used which describes the atmospheric transport of acidifying emissions. The deposition resulting from a certain emission pattern may be compared with deposition targets. This comparison gives an indication of the environmental damage caused by acidifying emissions.

The organization of this chapter is as follows. As the RAINS model is the most complete and the most extensively documented integrated assessment model I discuss it first. After giving a general description of the model (4.2.1) I go on to discuss specific topics and critically review them in the following subsections. In discussing energy use, attention is given to the Energy Flow Optimization Model - Environment (EFOM-ENV), which was used in combination with RAINS. This model focuses on alternative options for the future development of entire energy systems, taking into account technical, economic and environmental aspects of energy supply and emission generation (Rentz et al., 1994). The ASAM model is discussed in section 4.3 and the CASM model in section 4.4. Both models are examined in relation to the RAINS model. In section 4.5 I look at non-European acid rain models, and in section 4.6 I put forward conclusions.

4.2 The RAINS model[28,29]

4.2.1 Introduction to the RAINS model

The Regional Acidification Information and Simulation (RAINS) model was developed at the International Institute of Applied Systems Analysis (IIASA) (Alcamo et al., 1990) and focuses on acidification in Europe. As the aim is to provide a temporal spatial overview of acidification in Europe, the regional and temporal scales of RAINS are accordingly large (Alcamo et al., 1990). The RAINS model has been used in European emission reduction negotiations since 1988 (Hordijk, 1995).

---

[28]This section refers to the RAINS model version 6.0.

[29]This section draws heavily on Alcamo et al., 1990.

The model combines information on future energy use and agricultural activity, (using emission coefficients for SO_2, NO_x and NH_3 to determine regional emission levels) with information on costs and effects of pollution control strategies, long range atmospheric transport and sensitivity of ecosystems. The model simulates the flow of these acidifying pollutants from source regions in Europe to environmental receptors. The current model version (6.0) covers 38 source regions: 26 countries, 7 regions in the former USSR and 5 sea regions (for ship emissions). Analysis of depositions is carried out for 547 land-based receptor sites with a regular grid size of 150 x 150 km. The model has three parts: (1) the energy, emissions and costs of pollution control submodel, (2) the submodel describing the atmospheric transport of emissions, and (3) the submodel describing the impact of acid deposition on the environment. Figure 4.1 is a schematic diagram of the RAINS model.

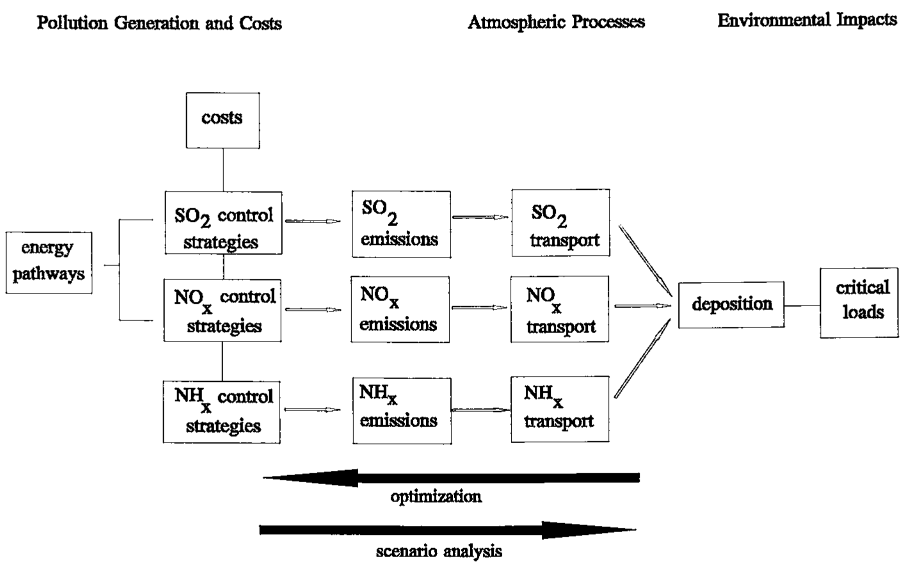

Figure 4.1 *Schematic diagram of the RAINS model (based on: Alcamo et al., 1990)*

There are two basic ways of using the model: scenario analysis and optimization analysis. Scenario analysis examines the implications of an energy pathway and a control strategy. As indicated in Figure 4.1, the model is used from left to right for scenario analysis. Optimization analysis uses RAINS from right to left through the model, starting with goals for environmental protection. The model determines the lowest cost emission reduction strategy needed to accomplish the specified goals.[30] In the following subsections I describe the main parts of the model in more detail.

4.2.2 Energy and emissions

The RAINS model computes country scale acidifying emissions. Emissions of sulphur dioxide (SO_2) and nitrogen oxides (NO_x) are based on data on national energy use (Amann, 1990). Emissions of ammonia (NH_x) are based on the number of livestock animals, fertilizer use and industrial emissions (Klaassen, 1994).

Energy

National energy use is subdivided into 6 economic sectors and each sector is broken down into different fuel types according to their characteristics of producing sulphur and nitrogen emissions. In total 12 fuel types are distinguished. In addition, energy production processes are introduced so as to have all necessary information available for creating alternative energy scenarios, while maintaining a country's energy balance. Historical data on energy consumption are taken from the UN-ECE (United Nations European Commission for Europe - Geneva) energy database, which is the only consistent database available for all European countries.

For future energy use, an Official Energy Pathway (OEP) was created. This pathway is based on national projections of energy trends and descriptions of energy policies up to the year 2000, as submitted to international organizations by individual governments. However, these data are not guaranteed to be either consistent or plausible, in some cases simply reflecting the desired goal of governments. Nevertheless, the data do represent the most official views on future energy use for Europe (Amann, 1990). According to this OEP, total energy consumption in Western Europe is stabilizing. It is assumed that the use of natural

---

[30]In RAINS version 6.0 only cost minimization is implemented. In earlier versions of the model emission minimization and sulphur removal maximization, given a limit on expenditures were implemented as well.

gas will increase sharply. Eastern European countries show a continued increase in energy consumption. Table 4.1 illustrates the energy use for the Netherlands in the year 2000 according to the OEP.

Table 4.1 Energy use per sector and fuel type (PJ) for the Netherlands, OEP, year: 2000 (source: RAINS 6.0).[31,32]

	CON	PP	DOM	TRA	IND	OTH	SUM
BC	0	0	0	0	0	0	0
HC	25	325	0	0	47	0	397
DC	0	0	0	0	100	0	100
MD	0	0	81	224	13	0	318
HF	50	4	11	0	35	0	100
LF	38	0	0	237	11	293	579
GAS	63	214	528	0	285	105	1195
OS	0	19	0	0	0	0	19
NUC	0	29	0	0	0	0	29
HYD	0	25	0	0	0	0	25
ELE	13	-260	142	8	130	0	33
DH	0	-54	54	0	0	0	0
SUM	189	302	816	469	621	398	2795

Sulphur Dioxide Emissions

Sulphur dioxide (SO_2) emissions are caused by combustion of fossil fuels (mainly coal and oil) and by industrial processes. Since there is a direct relation between the sulphur emissions and the sulphur content of fuel, sulphur emissions originating from fuel combustion are computed by using data on the energy consumed in several sectors in each country, together with information on the

[31]Sectors are: conversion (Con), power plants (PP), domestic (Dom), transport (Tra), industry (Ind), other sector (Oth).

[32]Fuel types are: brown coal (BC), hard coal (HC), derived coal (DC), medium destillate (MD), heavy fuel oil (HF), light fuel oil (LF), Gas (Gas), other solids (OS), nuclear (NUC), hydro (HYD), electricity (ELE), district heating (DH).

sulphur content of fuel in each country. The sulphur content of fuels is compared to the energy content of the fuel by using information on the heat values of the fuels. The fraction of sulphur contained in ash is also taken into account. Since coal is a rather heterogeneous fuel its sulphur content is the most sensitive parameter for estimating national SO_2 estimates. Estimates of sulphur content are more accurate for liquid fuels as they are more homogeneous. Future emissions are estimated by using the 1980 fuel parameters values. The user of RAINS might alter these values when developing abatement scenarios.

Because consistent quantitative information is not available, computation of sulphur emissions by industrial processes is less accurate than that of sulphur content of fossil fuels. Process emissions are only included for those countries that explicitly report these emissions, assuming that in countries not reporting them, process emissions are of minor importance. In general, process emissions will be about 10% of the national total (Alcamo et al., 1990).

Nitrogen Oxides Emissions

Nitrogen oxides (NO_x) are formed during combustion mainly in two ways: (i) by oxidation of the nitrogen contained in the fuel (fuel NO_x) and (ii) by the high temperature combination of nitrogen and oxygen in the combustion air (thermal NO_x).[33] Formation of fuel NO_x depends on the nitrogen contained naturally in the fuel as well as on the burner type and the firing mode used in the combustion process. Thermal NO_x formation depends on the combustion temperature, increasing exponentially at higher temperatures (Lübkert, 1987). Because of the variability in NO_x formation, computing NO_x emissions is complex if all factors are separately taken into account.

However, there is an OECD emission inventory for 11 countries.[34] This inventory provides information on NO_x emissions per sector for 1980. Average emission coefficients for sector/fuel combinations are derived by combining these

[33]A third way is the so called 'prompt' NO_x, produced in the flame reaction zone. However, this NO_x is an order of magnitude lower than the thermal and the fuel NO_x and is therefore considered negligible (Lübkert, 1987).

[34]These countries are: Austria, Denmark, Finland, France, (former) West Germany, Italy, the Netherlands, Norway, Sweden, Switzerland and the United Kingdom.

emission data with data on total energy consumption per sector in 1980. Only fuels of major importance in a sector are taken into account. The resulting emission coefficients are extrapolated to countries for which no detailed information is available, i.e the eastern European countries. By multiplication of NO_x emission coefficients with data on energy consumption, total NO_x emissions per sector and per country are calculated for all of Europe. Although the method of calculating NO_x emission at a country level is rather inaccurate, comparison of the estimated emission numbers with other emission inventories indicates that the estimates are within an acceptable range (Lübkert, 1987).

Ammonia Emissions

Ammonia (NH_x) emissions originate from livestock farming, fertilizers, industries and other anthropogenic sources. Of these, livestock farming is the most important one. Generally NH_x emissions are calculated as a product of emission coefficients and level of activity (Klaassen, 1994).

Emission coefficients for livestock farming are animal and country specific. However, lack of information for most European countries means the starting point is the information on emission coefficients in the Netherlands.[35] This information is combined with country specific circumstances like nitrogen fertilizer application and meadow and stall periods. For example, to calculate the emission factor for sheep, the stable and meadow period are country specific, but the data on N-excretion and volatilization factors are based on Dutch data. Emission coefficients for fertilizer use are based on the type of fertilizer used in each country. This information allows average N-losses to be determined. For countries where no information is available, an average N-loss of 5% is assumed. Ammonia emissions from industries mainly originate from ammonia production and fertilizer plants. Using the literature, an emission coefficient of 5.8 kg NH_3 per tonne fertilizer produced has been used for every country. The emission coefficient for human population (respiration) equals 0.3 kg NH_3 per person.

---

[35]National data are used in only a few cases. For the United Kingdom national data are used for the emission coefficients of pigs, laying hens, broilers, horses and sheep. For Finland national data are used for pigs, laying hens, broilers, horses and 'other cattle'.

Estimating future ammonia emissions requires projections for livestock population and fertilizer use. These forecasts are based on national forecasts from various institutes. Where no country specific data are available, trends are extrapolated. For a projection of fertilizer production a trend extrapolation is also used. The emission projections are based on the assumption that the emission coefficients remain constant. According to this projection, total ammonia emissions in Europe will increase. This trend differs for European regions: in Northern EC countries and in Scandinavia, emissions are expected to decline or to stabilize, while in Southern EC countries and in Eastern Europe an increase is generally expected.

Discussion on Emissions

Having described the way emissions are treated in the RAINS model, some points of discussion have to be dealt with. Relating SO_2 and NO_x emissions to energy use seems to be a flexible approach, since this offers the user of the RAINS model the opportunity of defining an alternative energy pathway, which is translated by the model into SO_2 and NO_x emissions. However, in the RAINS model, energy use is an exogenous input and the related emission coefficients are based on 1980-data. As a result, the way in which sulphur and nitrogen emissions are computed seems to be rather rigid. To meet this rigidity to a certain extent, the RAINS model has the option of modifying energy scenarios. By means of this option, alternative energy scenarios can be implemented. However, energy remains an exogenous input. Several alternative energy scenarios which affect the emissions of SO_2, NO_x and CO_2 are provided in a study by the Institute for Industrial Production (IIP) and IIASA (Rentz et al., 1992). The Energy Flow Optimization Model - Environment (EFOM-ENV) was used to construct these scenarios. This model focuses on alternative options for the future development of entire national energy systems. It takes technical, economic and environmental aspects of energy supply and emission generation into account. EFOM-ENV describes the energy flows from the primary energy supply over several conversion stages to the final energy consumption.

Another consequence of using exogenous energy data in the RAINS model is that the option of energy conservation is not explicitly modelled. Energy conservation not only reduces SO_2 and NO_x emissions, but it also plays an important role in reducing CO_2 emissions, the most important greenhouse gas. Since global warming is a major environmental problem, this link is very interesting, but it remains outside the scope of the RAINS model. The EFOM-ENV model, however, does include the option of energy saving. It follows that a reduction of CO_2 by means of energy conservation coincides with a reduction of SO_2 and NO_x. In contrast to this influence of CO_2 reductions on SO_2 emissions, there is no significant influence on CO_2 reductions by abatement of SO_2 (Rentz et al., 1994). In fact, because installation of emission reduction equipment causes an increase in energy use, SO_2 reduction requires a moderate energy increase. Other things being equal, this increase will result in increasing CO_2 emissions. The same holds for the use of a catalyst for reducing NO_x. Since energy use in the RAINS model is mostly an exogenous input, there is no direct link with economic activity. Because energy use is generally correlated with economic activity, this relation is only indirectly included in the model. Therefore the RAINS model can be classified as a partial model. Moreover, it can also be characterized as partial because it only covers acidification and does not take into account other related environmental problems like climate change and pollution by heavy metals.[36,37]

In the RAINS model the computation of ammonia emissions is treated differently from nitrogen and sulphur emissions since the first is not related to energy use. Because information on other European countries is lacking, the estimation of emission coefficients is largely based upon Dutch data. The estimation of emission factors for ammonia was the subject of discussion. First, Buijsman et al. (1985) estimated emission factors. Next, research by Asman (1990) indicated that these factors underestimate actual emissions. Emission estimates by the Cooperative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (EMEP) are based on Buijsman et al. (1985) multiplied by a correction factor 1.2. In view of the large differences in agricultural practices

---

[36]It should, however, be noted that in 1993 IIASA started to include photo-oxidants in RAINS. The emission database now includes emissions on Volatile Organic Compounds. Work on a long-range transport model for O_3 formation is underway.

[37]An application of the atmospheric submodel of RAINS to heavy metals can be found in Bolo et al. (1991).

in Europe, the estimation of ammonia emissions which is mainly based on Dutch data is not very accurate. However, given the available information it seems to be the best possible.

4.2.3 Control technology and costs

Emissions not only depend upon energy use, but also on the availability and implementation of control technologies. In this section I discuss control strategies and their costs. In accordance with the scale of analysis, control costs are country specific. The requirement to assess the abatement costs for all European countries (regions) limits the level of detail that can be maintained in the RAINS model. Emphasis is put on the international consistency and comparability of abatement cost functions. Given the available control options, RAINS can establish national control functions which are used in the optimization procedure. However, it is also possible to specify certain abatement measures (scenarios) and to calculate the resulting emissions and the related costs. Since the RAINS model is used for assessing the efficiency of different pollution control strategies, control costs form an essential part of the model.

Sulphur Removal

Sulphur reduction can take place in various ways. There are generally five abatement options: (i) energy conservation; (ii) fuel substitution; (iii) use of low sulphur fuels or desulphurized fuels; (iv) desulphurization during the combustion process and (v) desulphurization of flue gasses after combustion.[38] Energy conservation as such is not included in the RAINS model as a sulphur removal option. Nevertheless, it is possible to define an energy scenario based on energy conservation and to calculate the resulting emission. However, RAINS does not calculate the costs of that energy conservation, nor does it check the consistency of the user's input to the model.

Option (ii) relates to substitution of sulphur containing fuels for a different *type* of energy source with low or no sulphur content. Although this is often an

---

[38]The distinction between option (ii) and (iii) is that the former is the substitition between different fuel types, for example using gas instead of coal, while the latter is the use of a low sulphur fuel within one fuel type, for example the use of low sulphur coal.

effective way of reducing emissions, it may be limited for historical, political, institutional or technical reasons. This limitation makes it difficult to derive cost estimates. Moreover, the costs of fuel switching may have a large effect on the economy, a factor which is not included in RAINS. For these reasons the option of fuel substitution is excluded (Amann, 1990). It should be noted, however, that it is possible to define scenarios implying fuel substitution, for example by replacing brown coal in power plants by hard coal, but RAINS does not calculate the costs of this option.

The main control options for sulphur dioxide are the use of low sulphur fuels and the desulphurization of flue gases. If aiming at international consistency and comparability, a basic assumption in calculating control costs is the existence of free trade and exchange of emission abatement technology. This means that, in principle, technologies for the same installations are available for all countries at equal investment costs. In addition, the costs of each abatement technology are influenced by national circumstances through which abatement costs differ for every country. The computation of the costs of using low sulphur fuels is rather simple since the use of low sulphur fuels requires no investments. The costs of using low sulphur hard coal are derived from an analysis of the long term price differences on the world coal market. Substitution of brown coal by imports with low sulphur fuel content is excluded because of high transportation costs which make substitution unlikely.

There are several techniques that allow for desulphurization during and after combustion. These techniques require investment at the plant site and therefore abatement costs are more difficult to estimate. Costs of desulphurization are mainly based on data from the former West Germany which had extensive experience on desulphurization. The investment costs depend on the type of technology applied, the fuel type and the average boiler size. To convert the investment costs to costs per removed ton of SO_2, the country specific real interest rate and the average lifetime of plants are used. The capacity utilization (annual operating hours) and the sulphur removal efficiency relate those annualized costs to the actual amount of removed sulphur (Amann and Kornai, 1987). Because of a lack of accurate data, costs for removing process emissions have been estimated very roughly. It is assumed that every country can remove its process emissions up to 80% of the uncontrolled emission rates at stepwise increasing costs.

Given the available pollution control options and their costs, the options are ranked according to their cost effectiveness. For a selected energy pathway a compilation of the lowest cost solutions results in a national cost curve that describes the cost optimal combination of measures needed to achieve specified levels of national emission reductions. Figure 4.2 shows the national SO_2 abatement cost function for the Netherlands, based on the official energy pathway for the year 2000.

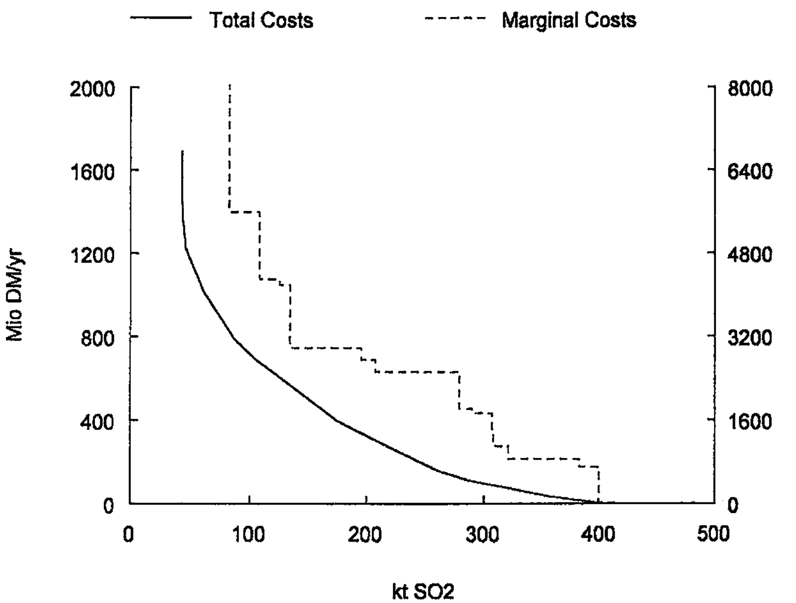

Figure 4.2 *SO_2 abatement cost function for the Netherlands, 2000, OEP (Source: RAINS 6.0)*

Nitrogen Removal

The RAINS model includes only direct NO_x control measures, applicable to existing energy infrastructures. Emission reductions as a result of structural changes of the energy system were excluded since such changes have macro economic impacts, which are not covered in RAINS. Actually, two options for nitrogen removal are included: (i) combustion modification and (ii) flue gas cleaning. The routine for calculating the abatement costs of these options is first to calculate the total costs related to an emission reduction option and next to derive the annual costs from these total costs (Amann, 1989).

A distinction is made in stationary sources between investment related costs and operating costs, as well as investment costs. For a control technology, all costs are related to one unit of fuel input, with fuel type and the economic sector being taken into account. Next, unit costs are derived, taking into account the capacity utilization factor. Finally, the cost efficiency of different control options is evaluated by relating the abatement costs to the amount of NO_x removed. The technology specific data, describing economic and technical properties of control technologies are assumed to be equal for all European countries. In addition, use is made of country-specific data like average capacity utilization of plants, average boiler size and interest rates, that describe national conditions under which abatement technologies are applied. Technology specific data are mainly European data. Country-specific data are based on international statistics, where possible (Amann, 1989). The cost evaluation for mobile sources basically follows the same approach as for stationary sources. The amount of abated NO_x emissions is calculated on the basis of the emission factor for unabated emissions, the removal efficiency of the control option, and the average annual fuel consumption of a vehicle and its lifetime. The calculations assume that the emission factors are equal for all countries. In reality however, it is more likely that they will differ between countries, but no data for this are available.

In summary, two sources of variations in costs for applying the same control technology over Europe can be distinguished. One is the national circumstances, like the interest rate. The second source of variations is related to the structural differences of national energy systems which determine the potential for application of individual control options. Given all available control options and the related costs, national cost curves can be derived that define the lowest cost for achieving varying reduction levels. Because the overall potential of reducing emissions from a specific source has to be consistent, special provisions are taken for control options excluding each other and having different costs and removal rates. Examples are the uncontrolled catalyst and the controlled three-way system for mobile sources. In ranking these options, the marginal costs for the more expensive but also more efficient control option is related to the incremental emission reductions compared to the cheaper option (Amann, 1989).

Ammonia Removal

Estimates of abatement costs for ammonia are based on specific data for country, animal, and technology. Lack of experience means that the cost estimates for ammonia reduction are more uncertain compared to those for controlling sulphur and nitrogen emissions. There are three options for controlling ammonia originating from livestock farming: (i) changes in the nitrogen content of the fodder, (ii) adaptations during stable and storage of manure, and (iii) low ammonia application of manure. Several techniques are available within these three options. Manure processing is assumed to take place only in the Netherlands. Since the costs of manure processing are much higher than other techniques, it cannot be limited to controlling ammonia emissions only, but is used for controlling manure (mineral) surpluses. Ammonia emissions from industrial processes can be reduced through the application of stripping and absorption techniques. The costs of controlling these emissions are a fixed amount per ton NH_3 removed, which implies that there are constant marginal abatement costs.

Estimation of control costs for ammonia originating from livestock farming is more complex. For all available techniques, unit costs are calculated per animal per year. In order to obtain the cost per unit of NH_3 removed, the costs are related to the amount of emissions reduced. In doing so it has to be remembered that abatement options may simultaneously reduce emissions during stable and storage, application and in the meadow. Another aspect that is taken into account is that the options for reducing ammonia emissions for each animal category can be applied in combination. But removal efficiencies of combinations are less than or equal to the sum of removal efficiencies of separate options. For example, low nitrogen feed for dairy cows reduces emissions by 20% while manure injection reduces emission by 90%. However, in combination both options reduce emissions only by 92%. Removal efficiencies of combinations are calculated using nitrogen balances for each animal type.

The routine for computing national cost curves for ammonia abatement differs slightly from the routine for computing national cost curves of sulphur and nitrogen abatement. First, control options within each animal category are ranked according to their marginal costs and their individual potential for removal. The option with the lowest cost is selected first. The marginal costs for the remaining options are calculated in relation to this first option, and the option with the lowest marginal costs is then considered the second best alternative. If further alternatives remain, the marginal costs are then computed in relation to the second best alternative. This procedure is repeated for each animal category as well as for industrial control. Finally, all options that are shown to be efficient are ranked according to marginal costs. This is expressed in the national cost function. Moreover, because of structural differences between agricultural systems, especially the structure of the livestock population and the intensity and type of fertilizer use, the potential of control opportunities is not always fully applicable in each country. Therefore the potential application of techniques is reduced to less than 100% for some countries.

According to the national cost functions, the maximum feasible overall reduction in ammonia emissions for Europe that can be achieved by technical means is 30 to 40% over the 1980 level (Klaassen, 1994).

Discussion on control technology and costs

The aim of international consistency and comparability of national control costs limits the level of detail that can be maintained. For example, national average boiler size and average capacity utilization are used for the calculation of national SO_2 and NO_x abatement costs. Using these averages affects the national abatement costs since the distribution of capacity utilization and the distribution of boiler sizes has a significant influence on the abatement costs in a country. Another issue of discussion relates to the fact that the abatement costs for SO_2 and NO_x date from the late eighties and are now out of date. This is especially true for eastern European countries where the transition of centrally planned economies to market economies, which started in 1989, causes changes in the energy sector, and therefore also in control costs.[39]

A difficult problem arises when control technologies simultaneously reduce emissions of other air pollutants (e.g. reduction of NO_x by a three way catalytic converter also reduces CO and VOC). If the control of NO_x is selected as a single target, such control options may turn out to be less cost efficient than other techniques (Amann, 1989). Because the RAINS model refers solely to acidification, other linked environmental problems are ignored, and from a broader point of view this results in inefficient solutions. If efficiency is not only related to acidification but to other linked environmental problems as well, abatement options which are inefficient in reducing acidification only may turn out to be efficient. However, since work is in progress to include other pollutants in the RAINS model, this disadvantage will vanish in the near future. A point indicated in previous sections is that the macro economic effects of emission reductions are not taken into account. A study concerning the macro economic impact of the implementation of the EC directive to control air pollution indicates, however, that this impact is small (Klaassen and Nentjes, 1991).

The EFOM-ENV model, which has already been mentioned, can be used to identify cost optimal combinations of technologies that satisfy the exogenously determined final energy demand and other user-determined constraints. Examples are a restriction of total emissions, a selection of techniques or a restriction of fuel types. Cost estimates available by using this model include energy conservation, an option that is excluded in the RAINS model. The EFOM-ENV model analyses the temporal development of the energy system over a long term planning horizon taking into account the age structure of the existing plants, the dynamics of market penetration of innovative technologies and the time structure of emission control regulations. Therefore the cost functions provided by this model are based on a dynamic approach. Compare this to the RAINS model which uses a static approach to the estimation of cost functions since dynamic effects such as uneven age structures are not taken into account (Amann et al., 1992a). The EFOM-ENV model is currently in use in all member states of the European Community. Work

---

[39]An energy-efficiency scenario for Eastern Europe is described in Amann et al., 1992.

is in progress to apply EFOM-ENV to Central and Eastern European countries as well (Rentz et al., 1993).

The model results from EFOM-ENV show that the relevance of energy conversion measures greatly depends on the energy scenario definition. A policy aiming at SO_2 emission reduction does not usually lead to a significant increase in energy savings, because the costs of additional energy conservation measures are generally higher than the costs of conventional SO_2 emission reduction technologies. If the policy aim is to reduce emissions of CO_2, SO_2 and NO_x emissions simultaneously, energy saving becomes a cost-efficient abatement option (Rentz et al., 1994).

4.2.4 Atmospheric transport

The atmospheric transport of acid pollutants in the RAINS model is based on the atmospheric model developed within the Co-operative Program for Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe (E-MEP) (Eliassen and Saltbones, 1983; Iversen et al., 1991). The main inputs of this model are emissions on a grid scale of 150 x 150 km and meteorological data. The model simulates the transport of emissions within 96 hours. Based on this model, EMEP provides transport matrices for SO_2, NO_x and NH_3, which relate national emissions to deposition at each grid-point. Background sources are also taken into account. These include natural sources, very long transport of emissions from North-America and anthropogenic emissions in Europe that have spent more than 96 hours in the atmosphere. Since part of the background deposition is caused by anthropogenic emissions in Europe, it is assumed that two-thirds of the background deposition will decrease if emissions decrease. The relationship between a country's emissions and its contribution to deposition at any grid-point is assumed to be linear. It is generally accepted that over long time and space scales the assumption of linearity between SO_2 emissions and deposition is appropriate (Alcamo et al., 1990). For NO_x and NH_3 this assumption has to be modified: the assumption of linearity only holds in part.

A main assumption in the atmospheric model is that pollutants are homogeneously mixed in the mixing layer. This layer is the layer above the earth's surface in which acidifying pollutants diffuse; it varies geographically and with time of day and season. During the day this layer extends up to an elevation of about one or two kilometres, whereas at night it drops to within a few hundred metres. Pollutants from high chimney stacks may then be mixed into the unstable air above the mixing layer and transported over long distances. The variability of deposition at any location depends on the number of emission sources taken into account and on the average period of deposition. If both factors are large enough, the EMEP model predicts the deposition satisfactorily. Starting from an estimate of annual European emissions, the EMEP model calculates the annual deposition with low variability. Decreasing the spatial and time scales increases the uncertainty of the model calculations significantly. The EMEP model uses input of emission data on a grid scale of 150 x 150 km.

Estimating future atmospheric transport requires assumptions about future meteorological conditions. The choice is between using the meteorology of a recent year or using average meteorological data. Climate patterns of individual years vary significantly from one another and from the long term average (Iversen et al., 1991). Since 1994 an eight-year average (1985-1992) is used in RAINS. Figures 4.3 and 4.4 illustrate the atmospheric transport of emissions. In these figures the export, import and the domestic contribution of sulphur emissions is illustrated for two arbitrarily chosen countries. Both figures are based on the emissions according to the Official Energy Pathway (OEP) for the year 2000 and the average transport matrix used in the RAINS model.

Another source of uncertainty about atmospheric transport is that of climatic change caused by global warming. Global warming may affect the precipitation and wind patterns in Europe and consequently alter the spatial patterns of deposition. Uncertainty of transfer coefficients does not seem to be a function of the distance between sources and receptors. There is no relationship between relative uncertainty and distance (Alcamo et al., 1990).

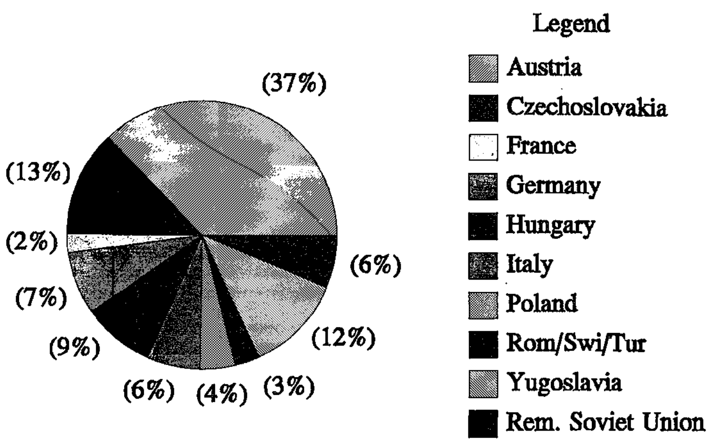

Figure 4.3 *Distribution of sulphur originating from Austria (Source: RAINS 6.0)*

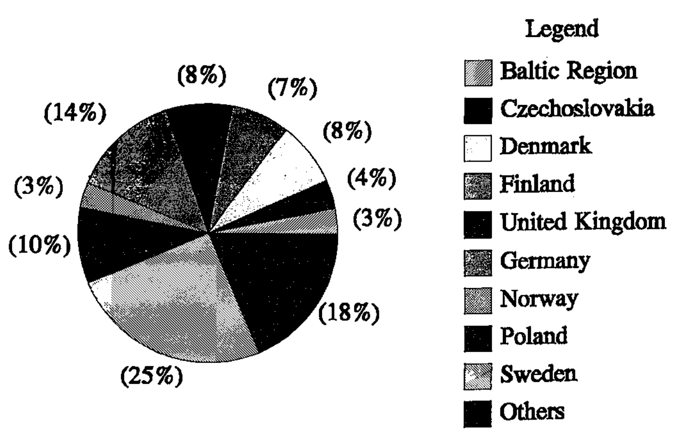

Figure 4.4 *Origin of sulphur deposition in Sweden (Source: RAINS 6.0)*

4.2.5 Environmental Impacts

Having examined emissions, control options and control costs as well as the atmospheric transport of acidifying emissions, the final issue for discussion is environmental impacts of acid deposition. To analyse the environmental impacts of acid deposition, the RAINS model currently uses the critical loads concept. This means that deposition patterns resulting from various emission allocations can be compared with these critical loads.[40] The exceedance of critical loads indicates the environmental damage of an emission allocation.

Critical loads for an ecosystem are defined as 'the quantitative estimate of exposure to one or more pollutants below which harmful effects which are judged to be significant on specified elements of the environment do not occur according to present knowledge' (Nilsson and Grennfelt, 1988). A map showing critical loads of acidity, for sulphur and for nitrogen was developed by the Coordination Center for Effects (CCE). The map shows the cumulative distribution of critical loads for a mixture of forest soil combinations and surface waters for each cell of the EMEP-grid system (150x150 km) (Hettelingh et al., 1991; Downing et al., 1993). Critical loads are given in percentiles. The 1- and 5-percentile maps of acidity, sulphur and nitrogen were produced by the CCE. The 1- (or 5-) percentile map of critical loads reflects the upper limit of the range of critical loads in each EMEP grid which covers 1% (or 5%) of the area in an EMEP grid cell. Consequently, this critical load protects 99% (or 95%) of the entire EMEP grid cell area.

Critical loads are computed by means of the steady state mass balance method. This method assumes a time independent equilibrium between the production and consumption of acidic compounds in soils. Dynamic processes are assumed to be unimportant for the assessment of long-term critical loads. The steady-state mass-balance method computes the critical load as the difference between the dissolution of base cations and the critical alkalinity leaching during weathering. The relative importance of sulphur and nitrogen depositions is expressed in the critical load of sulphur and of nitrogen respectively. The critical load for sulphur is derived from the computed critical load of actual acidity by using a sulphur fraction. The sulphur fraction is the ratio of sulphur deposition to the net deposition of sulphur and nitrogen. In calculating the sulphur fraction, uptake and immobilization of nitrogen is taken into account. The critical load of nitrogen reflects the sensitivity of ecosystems to both eutrophication and acidity. Nitrogen uptake for managed forests is taken as the maximum allowable eutrophication

---

[40]In an earlier version of the model soil acidity, lake acidity and forest impacts of acidification were treated in separate submodels.

limit. Therefore the critical load of nitrogen is the share of the critical load of acidity attributed to nitrogen. For forests this critical load is corrected for the nitrogen uptake (Hettelingh et al., 1991).

Discussion on environmental impacts

In the RAINS model deposition patterns emerging from scenarios can be compared with critical loads. Damage resulting from acidifying deposition is therefore expressed in physical terms, i.e. the exceedance of critical loads. The RAINS model makes no attempt to assign a monetary value to the damage of critical load exceedance. As a result, it is not possible to apply a cost-benefit analysis to acidification within the model. The model can only compute a cost- effective emission allocation if given a certain deposition target. In Chapter 2, I called this approach the 'ecological approach'. A disadvantage of the approach is that, in a neo-classical optimal pollution-control context, it is implicitly assumed that the damage above the deposition target is infinitely large. Exceedance of critical loads are assumed to have an equal weight, which is an unrealistic assumption.

A drawback to the critical loads as used in the RAINS model is that these loads are based only on the sensitivity of soils and surface waters. This means that other damage related to acidification, like direct damage to vegetation and buildings, and health damage is not taken into account. Another problem is that the critical loads are annual average loads. Peak concentrations, which may cause substantial health damage for example, are not taken into account (Chadwick and Kuylenstierna, 1992). Finally, the methodology used for subdividing the critical load for acidification into a critical load for sulphur and a critical load for nitrogen has to be criticized since this methodology is strongly debatable. A main disadvantage of the current formulation of these critical loads for sulphur and nitrogen is that the loads do not depend only on ecosystem properties, but also on the current deposition ratio of sulphur and nitrogen. However, the ratio between sulphur and nitrogen deposition is not a fixed number but may change over time. This suggests that the critical loads should be recalculated whenever the deposition composition of sulphur and nitrogen changes (Downing et al., 1993).

4.3 The ASAM model

The Abatement Strategies Assessment Model (ASAM) was developed against the background of the international negotiations concerning the abatement of SO_2 and NO_x. To date the model only contains the emissions of SO_2. The aim of the model is to serve as a computer tool to assist in the development of progressive reductions of acid deposition for achieving environmental goals (ApSimon et al., 1994).

As I indicated in the introduction, the structure of the ASAM is rather similar to the structure of the RAINS model. The EMEP transfer matrix is used to describe the atmospheric transport of SO_2, critical loads being used to assess the environmental damage of a certain emission allocation. The model differs with respect to the spatial scale on which emissions are considered and, as a result, abatement costs are no longer national cost curves. A second major difference is the approach used to derive effective emission reductions. The following subsections explain these two differences in more detail.

4.3.1 Emissions and costs

In the ASAM, the SO_2 emissions are not taken at national level but at the grid level of 150 x 150 km. One advantage of this approach is that major emission sources can be treated explicitly. This approach compensates for the drawback of the assumption of a constant spatial distribution within a country, as made in the RAINS model. Unfortunately, the EMEP emission inventory does not distinguish different emission sources, like power plants and industrial and domestic energy uses. The ASAM, however, is able to use official national emission data that do distinguish these source categories. In this model the spatial distribution of future emissions is kept constant over time and is based on the current emission pattern (ApSimon et al., 1994). Available literature does not make it clear which official emission data are ultimately used in the ASAM.

As emissions are considered on a 150 x 150 km grid scale, ASAM has provision for individual specification of the costs of emission reduction for each individual grid cell. However, since in practice national emission data are used, costs of abatement are estimated accordingly. The abatement costs for sulphur dioxide are based on the national cost functions of the RAINS model. These cost functions are scaled for each source grid square in a country according to the proportion of the country's emissions in that square. Because cost curves can involve some large reductions at constant costs, the ASAM defines a maximum reduction step equivalent to emissions from a single power plant, or limited by a ceiling on expenditure per step (ApSimon et al., 1994). Because the RAINS abatement cost functions are used, the criticism that the option of energy conservation is not included in these functions holds for the ASAM as well. Although the ASAM claims to take into account the emissions from individual sources and their related costs, the emission and cost data are not available for this purpose. Therefore the intended accurate spatial distribution of emissions and costs is actually only a rough estimate.

The grid square to grid square approach might only cause changes for the largest countries in Europe with significant internal differences in emission densities. Model results show that there appears to be no significant influence from the use of grid square to grid square atmospheric transport (IIASA, 1991).

4.3.2 Effective emission reductions

The ASAM can be used to derive economically and environmentally effective emission reductions. This is known as the Best Economic Environmental Pathway (BEEP). It is based on the assumption that environmental improvement will be achieved through a series of successive steps during a specified time period. At each step the ratio between the benefit, in terms of reduction of deposition towards specified target loads, and the associated costs should be maximized. Accordingly, ASAM produces a sequence of emission reductions at selected emitters, with deposition converging towards target levels as a function of the cumulative costs (ApSimon et al., 1994). Deposition targets can be critical loads or other specified target loads. At each step the model selects for each emitter the cheapest available emission reduction option not yet implemented and the associated benefit of the corresponding reduction. The benefit of a reduction for a source j consists of the change in deposition at any receptor caused by that emission reduction and of the contribution of this emission reduction towards the reduction of the difference between current deposition and the target deposition at any receptor. If the target is met at a receptor, this latter contribution is reduced to zero. Weighting functions built into the model can be used to put more emphasis on sensitive areas, or to weigh susceptible areas where the exceedance is particularly large. Successive steps are implemented until the target loads are attained or the maximum expenditure allowed is exhausted (ApSimon et al., 1994).

The BEEP approach has the advantage that it clearly shows how the benefit to cost ratio changes with increasing expenditure, and how well environmental goals are being reached given an overall amount of money to spend on emission reduction. However, this is only a quasi-advantage. Similar information can be provided by the RAINS model, for example by comparing cost effective emission allocations with different deposition targets. Another benefit of the sequential procedure applied in the ASAM is that it is not subject to discontinuities in the pattern of emitters at which reductions are selected. For example, in the case of tightening targets, flue gas desulphurization once fitted to a power plant will not be switched to another. A third advantage is that ASAM still produces useful results even if the target loads specified cannot be completely attained. More sophisticated linear optimization techniques concentrate on obtaining just a single "best solution" strategy to meet the specified target deposition (ApSimon and Wilson, 1991). This is only a small advantage since LP techniques also provide information about which target loads may not be achieved. LP solutions also provide so called shadow prices which indicate the relative difficulty of attaining targets at the receptors.

In conclusion, the advantages of using the BEEP approach instead of linear programming to derive the cost effective emission allocation are not very great. Moreover, in the available literature on the ASAM, it is not clear exactly which optimization procedure is followed to determine the Best Economic Environmental Pathway. Conceptually, the ASAM allows for more accurate calculations by using emissions at grid cell level. However, lack of data means that an approximation of the associated control costs for each grid cell is required and that no change in the future spatial distribution of emissions can be assumed. Thus accuracy is cancelled out.

4.4 The CASM model

The structure of the Co-ordinated Abatement Strategy Model (CASM) is similar to the structure of the RAINS model. CASM is built around a linear programming package and was developed to help develop acid rain control strategies. The model includes SO_2 and NO_x emissions. All data required for use in CASM is prepared by the Stockholm Environmental Institute (SEI), with the exception of EMEP transfer matrices (Gough et al., 1993).

The CASM offers a choice of optimization criteria. Like the RAINS model, CASM can perform cost minimization, subject to constraints. Other additional optimization criteria that are available are exceedance, damage and area minimizations, given a budget constraint of total allowable abatement expenditure (Rosemarin, 1991). Exceedance minimization minimizes the number of tonnes of sulphur exceeding critical loads. Damage minimization minimizes a weighted exceedance of critical loads counted up over all receptors. In this alternative, receptors have different influence depending on the extent of exceedance. In area minimization, the exceedance of critical loads is restricted to the smallest possible area. The extent of exceedance is of no importance in this optimization alternative. In the following subsections I discuss emissions, cost curves and critical loads of the CASM.

4.4.1 Emissions

As in the RAINS model, national emissions in the CASM are based on energy use data. The energy demand in OECD countries is derived from official energy projections. The energy demand in non-OECD countries is based on energy projections prepared for the Stockholm Environmental Institute. Demand for energy is divided into five sectors, and seven fuel types are distinguished. In order to calculate national emissions, the energy demand for every sector is combined with fuel sulphur content estimates and fuel conversion retention factors. Where appropriate, emissions from industrial processes are also included. The way emissions are calculated in the ASAM is very similar to the approach used in the RAINS model, the difference being that other energy data are used.

4.4.2 Costs

In the CASM, control costs functions are on national basis. As in the RAINS model, these national cost functions are based on the application of several control technologies. Also as in the RAINS model, energy conservation is not taken into account. The methodology for establishing these cost functions is as follows. Total annual costs and the sulphur removed are listed for every technology. The technologies are sorted by the total annual costs, and uneconomic technologies are rejected. Criteria for rejection are not explicit in the available publications on CASM. Next, the marginal costs and the marginal amount of sulphur removed are calculated. This gives the marginal costs per ton of sulphur removed. Finally the cost curve is smoothed (Rosemarin, 1991).

The sources for which the costs of abatement technologies are specified are power stations, industrial boilers and refineries. To obtain the cost functions, the energy demand of a sector and the sulphur content of fuel are taken into account. The technical abatement options for sulphur removal differ per sector and fuel type used. These include coal washing, flue gas desulphurization and fluidized bed combustion. A complete list of abatement options is given in Rosemarin (1991). Technology cost functions are a combination of a capital cost function and an operating and maintenance cost function. Cost factors are construction costs, labour costs, electricity costs and by-product disposal costs.

Similarities between cost functions in the CASM and the RAINS model differ from country to country. For instance, the marginal costs for Poland for the year 2000 in the CASM are very similar to the costs in the RAINS model. However, for France, the CASM estimates show substantial higher marginal abatement costs compared to the abatement costs in the RAINS model (Rosemarin, 1991). Accurate information that explains this difference is not available. To obtain the abatement cost functions of NO_x, a distinction is made between stationary and mobile sources. Abatement options for stationary sources are combustion modifications and selective catalytic reductions. The abatement options for mobile sources depend upon the type of engine. A main abatement option for petrol engines is the use of catalytic converters. Measures for diesel engines that meet certain emission standards are specified. As in the RAINS model, the technical information on abatement measures in the CASM is combined with cost estimates for each source category. Finally, these cost estimates are combined in a national cost curve (Gough et al., 1993).

Model results show that for most emission reduction scenarios the cost estimates of the CASM are in broad agreement with the RAINS model. The differences in costs have a range of 20% (IIASA, 1991).

4.4.3 Critical Loads

For analysing the environmental effects of acid deposition, the Stockholm Environmental Institute (SEI) developed a method of estimating the differential relative sensitivity of surface areas of Europe to acidic deposition. In this section this method is explained and differences from the critical loads as developed by the Coordination Center for Effects (CCE) are indicated.

The SEI aims at the determination of the relative sensitivity of ecosystems for showing the relative size of the effect that a unit of acid deposition will have on living organisms of different aquatic or terrestrial ecosystems (Chadwick and Kuylenstierna, 1992). The effect of acid deposition depends on the direct and indirect effects. Indirect effects occur because of acid deposition on the soil. The most important site factors influencing the sensitivity of ecosystems to the indirect effects are rock type, soil type, land use and annual rainfall. A small number of categories of these factors which have different effects on sensitivity are distinguished. The effects of these categories are expressed in weights. Next, weights are applied to the four site factors in proportion to the known or assumed degree to which they affect sensitivity. When combined, a range of sensitivity classes results. From this range, five relative sensitivity classes are defined. These five classes are mapped for areas in Europe. Since it does not seem justifiable to give absolute values to a relative scale, targets are assigned to the classes of relative sensitivity which are based on critical load values (Chadwick and Kuylenstierna, 1992). The target of Class 4 is twice as large as the target of Class 5, which is the most sensitive and the target of Class 3 is twice as large as that of Class 4 and so on.

A main advantage of the method to assign critical loads to sensitive areas as developed by the SEI is the use of several factors to specify acidic sensitivity of the environment. In contrast to the critical loads developed by the CCE, the sensitivity classes developed by the SEI are not only based on soil type, but other relevant factors are also taken into account. However, like the critical loads developed by the CCE, the sensitivity classes developed by the SEI ignore acidic sensitivity of cultural properties or health implications caused by acidic pollution. A drawback of the method used by the SEI is that finally only five sensitivity classes are distinguished. The map developed by the SEI was compared with the critical loads map of the CCE. In general the SEI map has lower critical loads; in other words the amount of deposition not causing damage is lower. The largest similarities are in Northern Europe and in the USSR. Differences occur in Southern, Western and South-Eastern Europe. Maps showing the differences are given in Hettelingh et al. (1991).

The effect of using the different critical load maps was analysed by a CASM run. For this purpose an exceedance minimization subject to a cost constraint was calculated. This cost constraint consisted of the cost of a uniform 50% emission reduction on 1980 figures in all countries. Calculation results show that compared to SEI critical loads, the total amount of sulphur abated is similar, but if CCE critical loads are used, a slightly greater degree of abatement takes place in Northern and Central Europe and the USSR and consequently less in Southern, Western and South-Eastern Europe. In keeping with the pattern of abatement, the greater number of emission reduction in Northern and Central Europe, using CCE critical load maps results in lower critical load exceedance in this region. Likewise, a greater exceedance of critical loads occurs in Southeastern, Southern and Western Europe if the CCE map rather than the SEI map is used (Hettelingh, et al., 1991). However, the use of this single example does not allow a general conclusion to be drawn on the effect of the use of different critical load maps. Hetteling et al. (1991) concluded that as the overall differences in SEI and CCE European critical load maps are small and as they result in broadly similar abatement strategies, the confidence in the use of critical load maps is promoted.

4.5 Non European acid rain models

In previous sections of this chapter I limited discussion to European acid rain models. However, acid rain is not a typically European environmental problem only. To give a more complete overview of acid rain models, in this section non-

European acid rain models are reviewed: first an Asian model and then United States models.

In July 1992, a research project was begun on acid rain and emission reduction in Asia, the model being based on the framework of the RAINS model (Foell et al., 1995). RAINS-Asia is an integrated assessment model for 23 countries in Asia and focuses on SO_2 emissions. It consists of three sub modules: (i) the energy and emissions module, (ii) the acid deposition module, (iii) the ecosystems impact module. The energy and emission module calculates the emission for a range of energy scenarios and the abatement costs. The core of the module is a sectoral and fuel specific end-use energy model. Sulphur emissions are calculated from the energy scenarios taking fuel characteristics, combustion technology and emission control assumptions into account. The module provides a gridded inventory of SO_2 emissions. The deposition module calculates the ambient levels and the deposition loading throughout Asia that result from a certain emission allocation. Next, in the ecosystem impact module, deposition levels can be compared with critical loads. By assessing the consequences of emission strategies, RAINS-Asia will be a useful tool for policy makers on acid rain at both national and regional levels.

In the US several types of acid rain models have been developed. As well as an integrated assessment model, there is also a model which takes uncertainty into account and another which takes probability into account. In 1984, a framework for an integrated assessment model was designed within the US National Acid Precipitation Assessment Program (NAPAP). However, the model has never been operational. It was designed to link models or data bases in such a manner that policy decisions could be made on a cost-benefit analysis (Streets, 1989). The model has 6 modules: (i) emissions, (ii) atmospheric processes, (iii) receptor systems, (iv) economic valuation, (v) control and mitigation and (vi) policy evaluation. Modules (i) to (iii) are more or less similar to the modules in the RAINS and RAINS-Asia models. The economic valuation module calculates the direct and indirect economic damage (or benefit) of changes in deposition. Module (v) calculates the costs of emission control. In the RAINS model this module is integrated with the emissions module. The policy analysis module (vi) integrates all of the information generated in the rest of the model in a form useful to the policy-maker. All that the policy analysis module can expect to achieve is to provide the best available tools to weigh the merits of alternative courses of action. In fact, the policy evaluation aims at providing a cost benefit analysis.

The models discussed so far have not dealt with uncertainty. I now want to describe two models that include treatment of uncertainty. The first is the ADEPT (Acid DEPosition decision Tree program) which takes uncertainty into account in the analysis of acid rain (North et al., 1985). This model focuses on the balancing of costs of SO_2 emission abatement with the ecological effects. In the absence of perfect foresight, the model aims at providing the best decision based on the information available today. It is based on decision analysis, which provides a formal theory for choosing among alternatives whose consequences are uncertain. There are three stages in the ADEPT model. First, there is the effect of control strategies on emissions. Secondly, the emissions are related to changes in deposition and finally changes in deposition must be related to changes in impacts.

To describe a set of scenarios at different points in time, use is made of a decision tree. A decision tree can include a large number of scenarios defined by different combinations of decisions and outcomes at each stage. Once the scenarios have been described in terms of probabilities and values, the decision tree can be used to compare the decision alternatives. Each scenario shows the consequences of a particular strategy. The uncertainty in a scenario is represented by the extent to which emission reduction reduces acid deposition and on the relationship between deposition and long term ecological impacts. The judgement on uncertainties, however, is a subjective matter. The use of the decision framework relies heavily on the assessment of judgemental probabilities. The ADEPT model has been used for illustrative calculations of the costs and benefits to individual states for acid deposition control policies (North et al., 1985).

A probabilistic model was developed by Rubin et al. (1988). This model aims at calculating cost-effective emission controls for coal-fired power plants. The model is called the Integrated Environmental Control (IEC) Model since it takes both NO_x and SO_2 emissions into account. A unique feature of the IEC model is its ability to analyse uncertainty. A range of output results is achieved which describes statistically the combined effects of uncertainties in many different parameters that vary simultaneously. A cumulative frequency distribution then characterizes the probability that some result (e.g. cost) will exceed a specific value or lie within some specified range. This type of result is believed to be more meaningful than the single deterministic answers generally produced by studies. However, the IEC Model only focuses on coal-fired power plants, whereas in decision-making on cost effective acid rain abatement other emission sources should be taken into account.

There are two types of non-European acid rain models: integrated assessment models and models including uncertainty. Taking uncertainty into account in acid rain models seems an attractive alternative. However, as the judgement on uncertainties is a subjective matter this will influence model results.

4.6 Conclusions

In this chapter I have discussed acid rain models paying most attention to integrated assessment models. These models are policy oriented and provide information on the costs and environmental effects of different emission abatement strategies. Environmental damage is not expressed in monetary terms in the integrated assessment models, because accurate monetarization of environmental damage is not possible in practise. Instead, the environmental damage is treated in physical terms. Physical environmental targets are often used in environmental policy. Given these targets, a cost-effective emission abatement policy can be stipulated. The use of physical environmental targets is very usefull for policy oriented models .

Among the European integrated assessment models the RAINS model is the most complete and extensive model. Although the RAINS model, the ASAM and the CASM are very similar, they diverge on various points as well. One difference is the number of acidifying compounds taken into account. Another difference concerns the optimization options that are implemented. A third point of difference is the emission accuracy. Finally, different data sources are used. The main benefit of the ASAM model is its accuracy on emission locations and the resulting atmospheric transport. However, because data for this purpose is unavailable, the intended accuracy is not entirely reliable. Model results show no significant differences caused by the grid to grid approach used. A serious drawback of this model is the lack of clarity of the procedure to derive the so called Best Economic Environmental Pathway.

The CASM is attractive because of its large number of optimization criteria. However, since policy aims at reaching certain deposition targets, optimization criteria to minimize environmental damage given a budget constraint are actually not very useful for policy purposes. Comparison of the CASM and the RAINS model shows that the model results are broadly in line with each other. The current status of acceptance of the RAINS model might be explained by two reasons. First, the model is the most completely integrated assessment model for acidification. The alternative integrated assessment models, the CASM and the ASAM, do not substantially improve on the RAINS model. Second, RAINS was developed by scientists of various disciplines at an international "East-West" institute. This aspect's main role was the acceptance of the RAINS model in international policy-making (Hordijk, 1991, 1995).

Despite the completeness of the integrated assessment models, they can still be improved. A main point for improvement is the abatement cost functions. At the moment these are not very accurate. One difficulty is that the model should predict the optimal emissions for a future year. This requires assumptions about future energy use which among others depends upon things like fuel prices and economic development; factors that are difficult to estimate. In this context, models including uncertainty may be a valuable complement to deterministic models. Obviously, the judgement on uncertainty remains a subjective matter. Another point for improvement is the valuation of environmental damage. As indicated in this chapter, the critical loads developed by CCE and by the SEI do not take all damage factors into account. Estimation of environmental damage requires factors like the presence of (historical) buildings and agricultural crops to be taken into account. Despite the need for improvement, the acid rain models have substantially increased knowledge about the acid rain problem and provide useful information to assist and improve policy-making.

In my research I used the RAINS model for calculations. Since I needed to calculate cost-effective emission allocations and to analyse deposition levels resulting from emission allocations, the use of an integrated assessment model was the most appropriate. I chose the RAINS model for two reasons: first, it is the most complete model for acidification; second, it is broadly accepted, since it was actually used in the international negotiations on the Second Sulphur Protocol.

5 PERMIT TRADING FOR NON-UNIFORMLY MIXING POLLUTANTS

5.1 Introduction

As I showed in Chapter 2, trading emission permits that is subject to rules might be a promising instrument for achieving a cost-effective emission abatement allocation, while taking deposition targets into account. In Chapter 3 I discussed the theory on tradeable permits more thoroughly. This present chapter elaborates on permit trading, on the assumption that trading is a bilateral and sequential process. I analyse whether cost effective emission abatement can be achieved by bilateral permit trading, and how such a bilateral trading system would look. In particular, the following topics will be examined: (i) the cost effective allocation of emission abatement, (ii) the need for a bilateral trading process versus a simultaneous trading process, and (iii) conditions needed for the bilateral trading process to achieve the cost effective allocation of emission abatement.

Permit trading as a bilateral and sequential process was first simulated by Atkinson and Tietenberg (1991), by applying a non-degradation offset system. According to this system, every trade transaction must not exceed deposition targets, not may total emissions increase. The simulation results of Atkinson and Tietenberg show that under these specific settings bilateral trade may improve efficiency, but is not able to approach the cost-effective allocation of emission abatement. Applying the deposition constraint to every bilateral trade transaction in the way Atkinson and Tietenberg (1991) did, is a very restrictive rule that prevents the achievement of the cost-effective solution.

In this chapter I introduce an alternative bilateral trading system which is the able to achieve the cost-effective emission abatement. According to this system not every bilateral trade transaction has to meet the deposition targets, but deposition targets should be met in the final stage, after the completion of a sequence of trade transactions. Because of this liberalization of restrictions, bilateral trade might coincide with the cost-effective solution.[41] To meet the deposition targets after a sequence of trade transactions, the amounts traded have to be controlled by a trade institution. The maximum number of traded permits between two sources allowed by the trade institution can be derived from the cost-effective solution for each transaction. Such an alternative bilateral trading scheme is indicated as 'guided bilateral trade'.

The aim of this chapter is to develop the methodology and the principles for guided bilateral trade by means of theoretical and hypothetical examples. I show how and under which conditions guided bilateral trade may lead to the cost-effective solution. I use a small model to illustrate the system of guided bilateral trade. Profitable trade opportunities appear to depend on a combination of the source-receptor matrix and the abatement-cost functions. Because guided bilateral trade is formulated as an optimization problem with a large number of equations and inequalities, it has not been possible to provide an analytical derivation of the optimum conditions for profitable trades. This means that it is not possible to derive conditions for the source-receptor matrix and for the abatement-cost functions that guarantee profitable trade opportunities.

This introduction concludes with a reminder of the classification of trade systems in Table 5.1. Two main criteria are cost effectiveness and violation of deposition targets. There are two explanations for cost-effective emission abatement: (I) an emission abatement allocation which abates a certain amount of emission at minimum abatement costs or (II)an emission abatement allocation which meets certain deposition targets at minimum abatement costs. This distinction is relevant since in the acid rain problem not only are the number of emissions important but the deposition resulting from an emission allocation also. In this study Iam mainly interested in cost effectiveness of type II, as nowadays this approach is very relevant for policy-making.

---

[41]Assuming a fixed number of emission permits, bilateral trade will only result in cost-effective emission abatement if the total number of issued emission permits coincides with the sum of emissions in the cost-effective abatement allocation.

Table 5.1 Overview of different trade systems

Trade system:		Offset rate:	Description:	Cost effective: emission (I)	Cost effective: deposition (II)	Non-violation of deposition targets	Discussed in section:
simultaneous	unconstrained			yes	no	no	5.3
	constrained	offset rate = 1	non-degradation offset	no	no	yes	5.3
		offset rate ≠ 1	pollution offset	no	yes	yes	5.3 (5.4.2)
			modified pollution offset	no	yes	yes	5.3 (5.4.2)
bilateral	unconstrained			yes	no	no	5.4.1
	constrained	offset rate = 1	non-degradation offset	no	no	yes	5.4.1
		offset rate ≠ 1	pollution offset	no	?	yes	5.4.2
			modified pollution offset	no	?	yes	5.4.2
		offset rate = 1	guided bilateral trade	no	yes	yes*	5.5

* only if all trade transactions defined by the trade vector are profitable (see section 5.5)

A main distinction in trade systems is that between *simultaneous* permit trading and *bilateral* permit trading. This distinction is important because, subject to the deposition targets, simultaneous permit trading can capitalize instantaneously on all increases and decreases among sources whereas bilateral trade cannot (Atkinson and Tietenberg, 1991). The second distinction is that between *unconstrained* and *constrained* permit trading. To reach cost-effective emission abatement of the latter, permit trading has to be restricted in some way so as to take account of deposition targets. I shall argue that constrained permit trading is only possible if permit trading is regarded as a bilateral process. However, in the literature, constrained permit trading is also regarded as a simultaneous process. Therefore Table 5.1 includes constrained simultaneous permit trading is included. A final distinction concerning trade systems that should be mentioned concerns the offset rate. Offset rate is defined as the amount of emissions that a source has to reduce if another source increases its emission by one unit. The most natural way to think of permit trading is on a one-to-one basis, which implies an offset rate of 1. However, if deposition targets are taken into account, it might not be necessary for the amount of emission reduction to equal the amount of emission increase. In such cases the offset rate might be unequal to one.

The organization of this chapter is as follows. First, in section 5.2, an example is introduced and the cost-effective abatement allocation is derived. This example is used throughout the chapter to illustrate the effect of different trade systems. Next, in section 5.3, simultaneous permit trading is reviewed. Section 5.4 discusses bilateral permit trading. The requirement that every trade has to meet given deposition targets is explored and the offset rates for different trade-rules are derived. In section 5.5 the idea of 'guided bilateral' trade is elaborated and illustrated. Section 5.6 is conclusions.

5.2 Cost-effective emission abatement

Cost-effective emission allocation that does not violate deposition targets follows from the optimization problem in which deposition targets (d^*_i) at every receptor i are met at minimum aggregate abatement costs (C_j) of the n sources. The level of abatement (r_j) depends on the difference between initial emissions (E_j) and actual emissions (e_j). The deposition at a receptor depends upon the emission of the sources affecting that receptor, multiplied by the transfer coefficients (a_{ij}). In the remainder of this chapter, cost-effective emission abatement refers to achieving deposition targets at minimum abatement costs. Mathematically the problem is defined as follows:

$$Min\sum_{j=1}^{n} C_j(r_j) \tag{5.1}$$

$$s.t.\sum_{j=1}^{n} a_{ij}e_j \leq d_i^*, \textit{ for every } i \tag{5.2}$$

$$r_j = E_j - e_j \tag{5.3}$$

$$r_j \geq 0,\ r_j \leq E_j \tag{5.4}$$

To illustrate the cost effective solution, a numerical example is developed for 5 emission sources (n=5) and 5 receptor sites (i=5). In the base situation it is assumed that the initial emission (E_j) of source j = 1,...,5 amounts, of 275, 200, 175, 400, 350 respectively. The transfer of emissions is given by the transfer matrix A. An element a_{ij} represents the share of source j's emission deposits at receptor i.

	0.5	0.1	0	0	0
	0.25	0.4	0.05	0.05	0
A:	0	0.05	0.45	0.1	0.20
	0	0.15	0.05	0.5	0.15
	0	0	0.05	0.35	0.45

The deposition at receptor i = 1,...,5 resulting from the initial emission levels amounts of 158, 178, 199, 291 and 306 respectively. It is assumed that the deposition target (d^*_i) are 100 for each receptor. This requires substantial emission abatement.

For reason of simplicity in this initial example, abatement-cost functions for source j = 1,...,5 are assumed to be affine (r_j is the amount of emission abatement by source j):[42]

$$C_1 = 10 + 100r_1$$
$$C_2 = 50 + 200r_2$$
$$C_3 = 75 + 300r_3$$
$$C_4 = 100 + 50r_4$$
$$C_5 = 25 + 175r_5$$

It is assumed that no emission abatement takes place in the base situation. The solution to the optimization problem as formulated by equation *(5.1)- (5.4)* shows that the minimum costs needed to meet the deposition targets are 87,010.[43] Total emissions are 575. The average emission reduction for the base situation is 59%. Table 5.2 gives an overview of abatement, costs, emissions and deposition in the cost-effective emission abatement allocation.

Table 5.2 EXAMPLE I (59%): Overview of emissions, abatement, costs and deposition in the cost effective allocation.

Source/ receptor	E_j	e_j	r_j	C_j	d_i
1	275	59	216	21610	60
2	200	200	0	0	100
3	175	106	69	20775	100
4	400	0	400	20100	67
5	350	210	140	24525	100
sum	1400	575	825	87010	

In the cost-effective abatement allocation, source 4 abates all emission. This can be explained by the low marginal abatement costs of source 4. Moreover, the

[42]This assumption is released in section 5.5.3.

[43]This solution follows from solving the optimization problem with the linear programming package 'pcprog'.

column sum in the transfer matrix of source 4 equals 1, indicating that all emissions from source 4 are deposited in the region under consideration. All other sources have a column sum smaller than 1, which means that part of the emissions are deposited outside the region. Source 3 reduces 40% of its initial emissions, in spite of high marginal abatement costs. This might be explained by the relatively large transfer coefficient of source 3 for receptor 3, which is one of the binding receptors. Receptors 1 and 4 are non-binding in the cost-effective abatement allocation. Consequently, relaxing the deposition target for receptor 1 and/or receptor 4 will not influence total abatement costs. Receptor 2, 3 and 5 are binding. A graphical illustration of the cost-effective abatement allocation is given in Figure 5.1.[44] The cost-effective allocation of emissions is at point CE. The initial emissions are represented by point E.

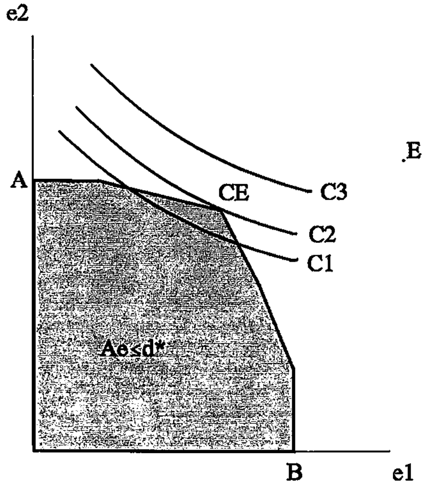

Figure 5.1 *Cost-effective allocation of emission*

Figure 5.1 considers the emissions of two sources. Emissions of source 1 *(e1)* are measured along the x-axis and emissions of source 2 *(e2)* along the y-axis. The deposition target can be met by different combinations of e1 and e2. The

[44]In contrast to the numerical illustration, in the graphical illustration, in agreement with common practice, abatement-cost functions are assumed to be non-linear.

polygon AB defines the feasible area; i.e. AB represents combinations of emissions of both sources which do not violate the deposition targets.

Different emission allocations imply different abatement costs. Total abatement costs are represented by iso-cost curves C1, C2, and C3. A low emission level implies high abatement costs. Consequently, moving to the origin, iso-cost curves represent a higher level of abatement costs. Therefore C3 represents the lowest level of abatement costs. However, the emission combinations corresponding to C3 violate the deposition targets. The deposition target meets the lowest cost at CE. Given the initial emission allocation E, both sources have to reduce emissions though not to the same extent, to reach point CE.

5.3 Simultaneous permit trading

Having presented a benchmark in the previous section, in this section I want to introduce permit trading. Simultaneous permit trading assumes that all sources trade multilaterally. Assuming profit-maximizing behaviour, competitive markets, the availability of full information and lack of restrictions, simultaneous permit trading will equalize marginal abatement costs.[45] As a result, the emission abatement is cost effective but will not meet deposition targets.[46] Figure 5.2 illustrates simultaneous trade.

As in Figure 5.1, Figure 5.2 shows the emissions of two sources, e1 and e2. Point CE represents the cost-effective emission allocation which meets the deposition targets at minimum costs. If the number of initially issued permits equals the number of emissions in the cost-effective allocation (CE), then the initial permit allocation has to be somewhere on the dotted line ΣEP_j.[47] Simultaneous permit trading will result in point S, since at that point the allocation of permits is tangent to the highest iso-cost curve, which implies that, given the total amount of emission permits, abatement costs are at a minimum. There is no rationale for sources to stop trading permits at point CE because moving to point S means

---

[45]This has been explained in Chapter 2.

[46]In section 4.1 this is indicated as cost effectiveness of type I.

[47]The choice of an initial permit allocation is extensively discussed in section 4.5.

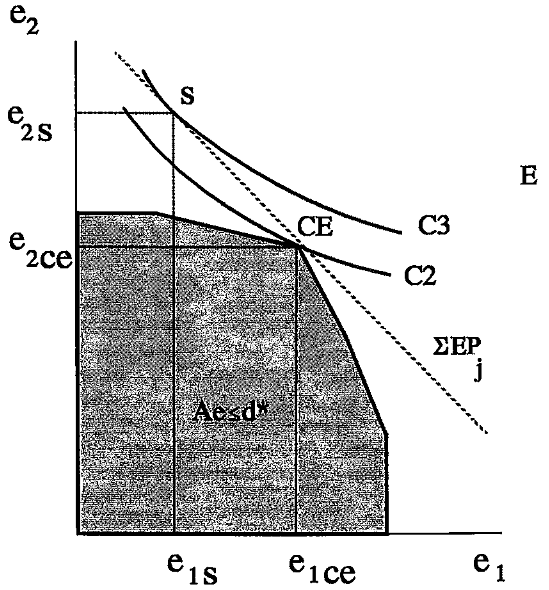

Figure 5.2 *Simultaneous Trading in Permits*

additional cost savings. However, the simultaneous trade result violates the deposition targets. This illustrates that permit trading has to be constrained in some way to take deposition targets into account.

Constrained simultaneous trade by means of trading subject to rules has already been discussed by several authors (see Chapter 3, section 3.4). However, the concept of simultaneous permit trading being constrained is vague. No answer is given in the literature to the question of how to implement a system of constrained simultaneous trade (see e.g. Krupnick et al., 1983). It is difficult to think of a simple mechanism that mimics simultaneous trade subject to rules. In the literature (e.g. Krupnick et al., 1983) it is mistakenly said that simultaneous trade will automatically result in the cost-effective solution. This is unrealistic because there is no indication of how constraints should be implemented in practice. Therefore, the idea that a cost-effective abatement allocation satisfying the deposition targets (point CE in Figure 5.2) can be reached by constrained simultaneous trade is purely theoretical. A more realistic assumption is to see permit trading as a bilateral sequential process (Atkinson and Tietenberg, 1991).

5.4 Bilateral Permit Trading

5.4.1 One-to-one emission trading

A natural way to think of permit trading in practice is of trading being a bilateral process. This section looks at bilateral trade between a seller (S) and a buyer (B). Assuming that permit trading takes place bilaterally, a sequence of trade transactions takes place in which different sources can act as seller and buyer.

When S and B are allowed to exchange emission permits that do not exceed deposition targets, they do so under cost minimization. The trade result can be derived from the following optimization problem:[48]

$$Min(C_S(r_S)+C_B(r_B)) \tag{5.5}$$

$$s.t. \sum_{j \neq S,\, B} e_j^0 a_{ij} + e_S^1 a_{iS} + e_B^1 a_{iB} \leq d_i^*, \text{ for every } i \tag{5.6}$$

$$e_S^1 = e_S^0 - \sigma \tag{5.7}$$

$$e_j^1 = e_j^0,\ j \neq S,\ j \neq B \tag{5.9}$$

$$r_j = E_j - e_j^1,\ j = S,B \tag{5.10}$$

$$e_S^1 + e_B^1 \leq e_S^0 + e_B^0 \tag{5.11}$$

$$\sigma, \beta, e_S^1, e_B^1 \geq 0 \tag{5.12}$$

where:

$C_j(r_j)$: abatement-cost function of source j

e^0_j: pre-trade emission of source j

e^1_j: post-trade emission of source j

E_j: initial emission of source j

[48]In this analysis it is assumed that banking of permits does not take place. The effect of banking was analysed by Cronshaw and Kruse (1994).

σ: number of emission permits sold
β: number of emission permits bought
r_j: emission abatement of source j
a_{ij}: transfer coefficient of source j to receptor i
d^*_i: deposition target at receptor i;

In this optimization problem, source S is selling emission permits to source B. Consequently, source S has to increase emission reduction whereas source B will reduce less than originally required. The change in emission reduction results from the difference between initial emissions and pre-trade emissions and initial emissions and post-trade emissions ($\Delta r_j = e^1_j - e^0_j$). The emission allocation after the trade transaction has to meet the deposition target at every receptor. First, the pre-trade emissions follow from the number of permits a source is initially equipped with. The systems of issuing emission permits were discussed in Chapter 3, section 3.6. As a result of trading, pre-trade emissions change for a subsequent trade.

This section analyses bilateral trade according to Atkinson and Tietenberg (1991), assuming permit trading takes place on a one-to-one basis (i.e. σ=β). It illustrates that the cost-effective solution cannot be achieved by means of this trading system.

Trade opportunities depend largely on the transfer coefficients. If $a_{iB} \leq a_{iS}$, trading emission permits are always allowed with respect to receptor i, because in this case the decrease in deposition caused by source S is equal to, or exceeds the increase in deposition caused by source B. If $a_{iB} > a_{iS}$, trade is only possible where receptor i is non-binding. The maximum traded number of emission permits for non-binding receptors (i.e. $d^0_i < d^*_i$; d^0_i represents the pre-trade deposition level) equals $(d^*_i - d^0_i)/(a_{iB} - a_{iS})$.[49] Where a receptor is binding, trading is only allowed if $a_{iB} \leq a_{iS}$. Although one-to-one trading is attractive because of its simplicity, trade opportunities are limited if trading permits take place on a one-to-one basis. This is first illustrated graphically and then numerically, using the model introduced in section 5.2.

---

[49] This only holds for $a_{iB} > a_{iS}$.

Before bilateral permit trading can be simulated, an initial emission permit allocation has to be specified. The rule that every trade transaction has to meet deposition targets also requires the pre-trade allocation to meet the deposition targets. In this example it is assumed that every source is provided with a number of emission permits based upon a uniform emission-reduction percentage.[50] The number of issued permits is established so that the deposition caused by the initial permit allocation meets the targets (i.e. $d^0_i \leq d^*_i$). In Figures 5.3 and 5.4 the initial emission-permit allocation and the constrained and unconstrained bilateral trading result are represented, assuming an offset rate of one.

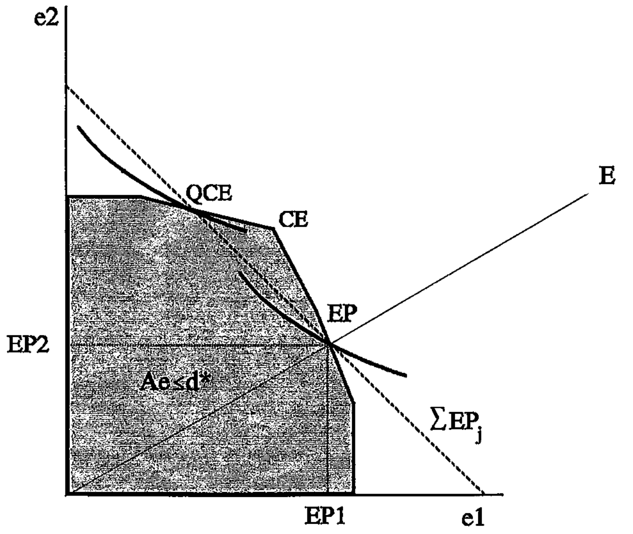

Figure 5.3 *Constrained trading in permits on a one-to-one basis*

As in Figure 5.1, emission combinations which do not harm the deposition target are represented by the shaded polygon in both figures. The initial emissions are at point E. Basing initial emission permits on a uniform emission reduction, the initial permits are represented by point EP. Because permit trading is assumed

[50]This is also known as the sovereignty criterion (Rose and Stevens, 1993). Other criteria for the issuing of permits have been discussed in Chapter 3, section 3.6.

to take place with an offset rate of 1, the total number of emission permits remains constant. In the figures this is represented by the dotted 45-degree line ΣEP_j through EP. Trading permits imply a movement along line ΣEP_j.

Compared to the initial permit allocation (EP), Figure 5.3 shows that, if the sum of emissions e1 and e2 remains constant, deposition targets could be met at lower costs in the so called Quasi Cost Effective allocation (QCE). This is because point QCE is on a higher iso-cost curve, and represents lower total abatement costs than at point EP.

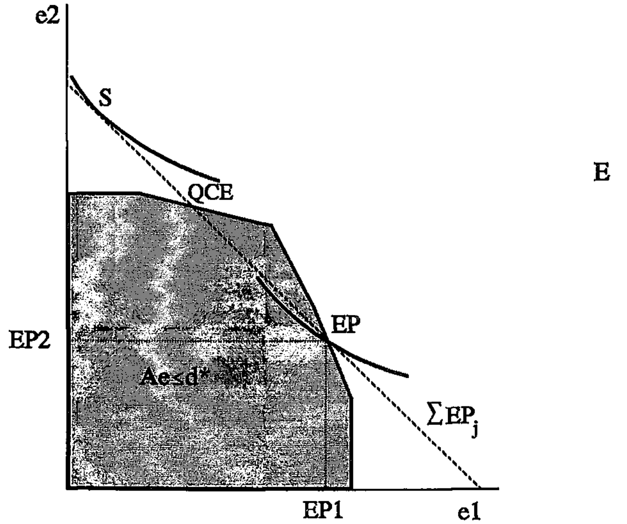

Figure 5.4 *Unconstrained bilateral trading in permits*

Figure 5.4 shows the need for constrained permit trading. If trading permits are unconstrained, permit trading will result in point S, equal to the simultaneous trade result. Given the sum of emission permits, the allocation is at lowest cost in point S. However, point S is outside the feasible area. Only if trading permits are allowed on condition that deposition targets are not violated, will permit trading result in point QCE. The quasi cost-effective emission allocation is the emission allocation which minimizes abatement costs so that deposition targets are met, given the total number of available emission permits. The quasi cost-effective allocation can only be reached if permit trading is limited. As shown in section 5.4, trade constraints can only be defined if the trading process is bilateral.

Figure 5.1 indicated that the cost-effective emission allocation is at point CE. Figure 5.5 illustrates that, given the initial emission permit allocation based upon uniform emission reduction and assuming that deposition targets must be met while permit trading takes place at a one-to-one basis, then the cost effective emission allocation will never be reached. If the aim is to reach the cost-effective emission allocation by permit trading on a one-to-one basis, the initial permit allocation has to be chosen differently.

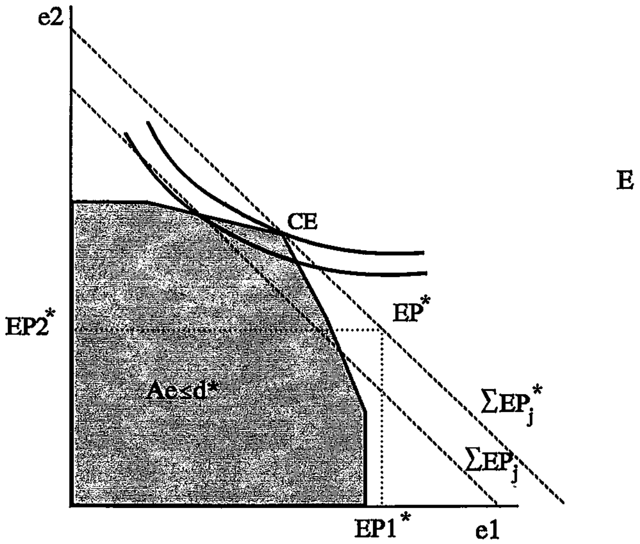

Figure 5.5 *Cost-effective trading in permits on a one-to-one basis*

In order to reach the cost-effective emission allocation (CE) by one-to-one constrained permit trading, the total number of issued emission permits has to equal the total emissions in point CE. Therefore the initial permit allocation has to be on line ΣEP_j^*, which intersects point CE. If the initial permit allocation is based on a uniform emission reduction, for example, the initial allocation is at EP^*. Note that to reach the cost-effective solution, the initial permit allocation does not necessarily has to meet the deposition targets. Permit trading, restricted by a deposition constraint will now result in allocation CE.[51] In CE the highest possible iso-cost curve representing the least abatement costs is reached and deposition targets are met. Permit trading still has to be constrained in some way to take care of deposition targets.

The bilateral trading process, assuming an offset rate of 1, can be illustrated numerically using the model introduced in section 5.2. Following Atkinson and Tietenberg (1991) it is assumed that every trade transaction has to meet the deposition targets. Consequently, the initial emission permit allocation based upon an equal reduction percentage is such that deposition targets are met. Figure 5.3 illustrated that this system of permit trading will only result in a quasi cost-effective abatement allocation and not in the cost-effective abatement allocation.

For the initial permit allocation EP not to violate deposition targets a total reduction percentage of 67.5% is required.[52] In the cost-effective allocation, the average reduction percentage only amounts to 59%. If the initial permit allocation is based on a 59% reduction, point EP^* from Figure 5.5 will be the initial allocation that violates deposition targets.

Table 5.3 gives an overview of initial emissions (E_j), initial permit allocation (EP_j), the pre-trade costs (C_j^0) and depositions (d_i^0) as well as the quasi cost-effective solution. The difference between the initial emission permit allocation and the emission allocation which meets deposition targets at the lowest cost, is indicated as a trade vector.[53] In Figure 5.3 the trade vector is represented by EP-QCE. In Figure 5.5, the trade vector is represented by EP^*-CE. In Table 5.3, column five represents the trade vector.

[51]In practice, the rule that permit trading is only allowed as long as deposition targets are not violated is no longer usable since the initial permit allocation EP^* is outside the feasible area.

[52]In PCPROG this is calculated by solving the equations that describe the relation between emission reduction and deposition, taking into account a uniform emission reduction percentage.

[53]Whether this allocation is the quasi cost-effective allocation or the cost effective allocation depends upon the total amount of issued emission permits.

Table 5.3 EXAMPLE II (67.5%): Overview of pre-trade data and the quasi cost-effective solution.

Source j/ receptor i	E_j	EP_j	C_j^0	d_i^0	Buy (+) Sell (-)	e_j^I	C_j^I	d_i^I
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
1	275	90	18510	57	- 65	25	25010	50
2	200	65	27050	58	+ 135	200	50	95
3	175	57	35475	65	+ 118	175	75	100
4	400	130	13600	95	- 130	0	20100	47
5	350	114	41325	100	- 58	56	51475	34
Sum	1400	456	135960		0	456	96710	

The cost-saving of permit trading, given the initial number of emission permits, amounts to 39250. This is the difference between abatement costs resulting from the initial permit allocation and the abatement costs in the quasi cost-effective emission allocation. In this example, bilateral and sequential trade in emission permits, requiring every trade not to exceed deposition targets captures all cost savings. That is buyers and sellers can trade all their permits at a profit. However, bilateral permit trading does not necessarily capture all cost savings. Whether all savings are captured depends on the source-receptor matrix and the abatement costs. Nevertheless, deposition targets can be met by lower abatement costs. A potential cost saving of 9700 (i.e. the difference between the costs in the quasi cost-effective allocation and the costs in the cost effective allocation) cannot be captured by means of the specified trade system. Table 5.4 shows the bilateral trade result stepwise. The sources that trade in emission permits are in bold and are underlined.

Table 5.4 EXAMPLE II (67,5%): Emissions, deposition and abatement cost after every bilateral trade transaction.[54]

				BILATERAL TRANSACTIONS:						
Source	pre-trade	I	II	III	IV	V	VI	VII	VIII	IX
				EMISSIONS:						
1	90	90	90	**54**	54	**36**	**28**	**27**	27	**25**
2	65	65	**95**	**131**	**196**	196	196	**197**	**200**	200
3	57	**157**	157	157	157	**175**	175	175	175	175
4	130	**30**	**0**	0	0	0	0	0	0	0
5	114	114	114	114	**49**	49	**57**	57	**54**	**56**
				DEPOSITION:						
1	57	67	70	56	62	55	51	51	51	50
2	58	58	68	74	100	96	94	94	96	95
3	65	100	98	100	90	98	100	100	100	100
4	95	50	39	45	45	46	47	47	47	47
5	100	70	59	59	30	31	34	34	33	34
				ABATEMENT COSTS (x 1000):						
1	18.5	18.5	18.5	**22.1**	22.1	**23.9**	**24.7**	**24.8**	24.8	**25**
2	27	27	**21**	**13.8**	**0.8**	0.8	0.8	**0.6**	**0**	0
3	35.5	**5.5**	5.5	5.5	5.5	**0**	0	0	0	0
4	13.6	**18.6**	**20.1**	20.1	20.1	20.1	20.1	20.1	20.1	20.1
5	41.3	41.3	41.3	41.3	**52.7**	52.7	**51.3**	51.3	**51.8**	**51.5**
Sum	136	111	106	102.8	101.2	97.5	96.9	96.8	96.7	96.6
Saving		25	5	3.2	1.6	3.7	0.6	0.1	0.1	0.1

In this example 9 bilateral trade transactions take place. After these transactions have taken place, permit trading is no longer possible without violating deposition

[54]Small differences in costs compared to those in table 5.3 are caused by rounding off the figures.

targets. It is assumed that permits are traded in whole units. The trade sequence is defined as follows. Given the pre-trade emission allocation, the trade transaction generating the largest cost saving is assumed to take place first. Emission allocation alters after every trade transaction, consequently the opportunities for subsequent trade and the potential for cost savings in a subsequent trade alter. This explains why the cost savings differ in every trade transaction and do not follow a specific pattern. The quasi cost-effective allocation is achieved after the completion of all trade transactions. Notice that source 5 has to sell and buy permits successively in transaction VIII and IX. This is rather unrealistic for dynamics. Selling permits implies implementing additional reduction measures while buying permits allows these measures to be cancelled.[55]

5.4.2 Offset Rates

An alternative to permit trading on a one-to-one basis is permit trading with offset rates. An offset rate is defined as the emission amount a source S has to decrease if another source B increases emissions by one unit in order to meet the deposition constraint. The idea behind using offset rates is that so long as deposition targets are not violated, the amount of emission reduction does not have to equal the amount of emission increase. For instance, so long as deposition targets are met, two sources S and B are allowed to trade emission permits with an offset rate of 0.5 which means that the increase of emissions from source B is twice as much as the decrease of emission from source S. Consequently, total emissions will increase. This is only possible if source B is able to obtain the remaining number of emission permits from the authorities.[56]

The concept of offset rates unequal to one is illustrated graphically in Figure 5.6. This figure considers emissions from two sources, e1 and e2. The shaded area represents emission combinations which do not harm the deposition targets. The initial emission permit allocation, that meets the deposition target is assumed to be at EP. Since permit trading does not take place on a one-to-one base, Figure 5.6 shows that it is possible to move from the initial permit allocation (EP) to the

---

[55]See also section 5.5.4.

[56]In that case a trading agency functions as a "third' party in the bilateral trade transaction.

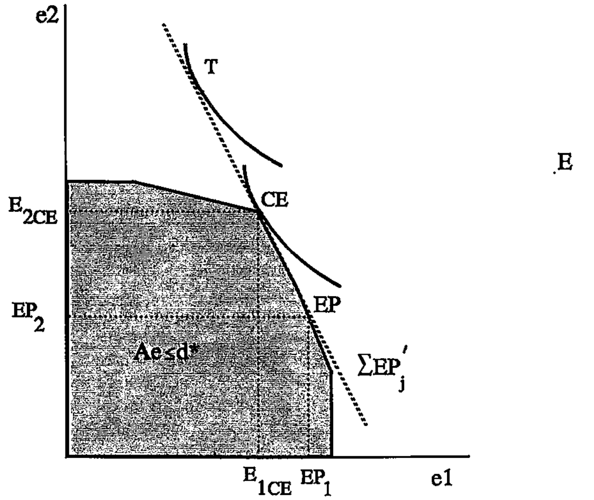

Figure 5.6 *Permit trading with an offset rate*

cost-effective allocation (CE). Applying an offset rate unequal to one allows permit trading to move along line $\Sigma EP_j'$, which is not a 45° line. Because of trading along this line, in this illustration the total amount of permits increases. The need to define constraints for permit trading still holds. If trading permits were not constrained, trading would result in allocation T violating deposition targets.

When deriving the offset rate for a trade between two sources S and B, distinction has to be made between binding and non-binding receptors. If a receptor is binding no increase in deposition at that receptor is allowed. Consequently, the deposition at receptor i originating from source B has to be smaller than, or equal to the deposition originating from source S:

$$e_B^1 a_{iB} \leq e_S^1 a_{iS} \qquad (5.13)$$

From equation *(5.13)* the offset rate follows: $V = \Delta e_S / \Delta e_B \geq a_{iB}/a_{iS}$. This offset rate only holds for one binding receptor. In general the offset rate has to be set so as not to violate the deposition target at any binding receptor. Therefore the offset rate equals (for i corresponding to a binding receptor):

$$V \geq Max\ \{a_{iB}/a_{iS}\},\ for\ i=1,..,I \tag{5.14}$$

Consequently, opportunities for permit trading applying an offset rate based upon transfer coefficients are larger compared to one-to-one permit trading. If a_{iB} equals zero, trade is always allowed as far as receptor i is concerned. In this situation source B is allowed to increase emission without any compensation from another source. If a_{iB} is not equal to zero and a_{iS} equals zero, trading in emission permits is never possible without exceeding the deposition target at receptor i. In this situation source S is not able to compensate for the additional deposition caused by receptor B.

Trade opportunities that apply a pollution offset depend largely on the initial permit allocation. If it implies a deposition lower than the deposition targets, trade options are increased. After trade transactions have taken place, some receptors will become binding, thus restricting opportunities for trade. Trading in permits according to the modified pollution offset offers fewer trade opportunities. According to this rule, trading in permits is only allowed so long as neither deposition targets nor pre-trade deposition levels are violated. Consequently, all receptors are binding, thus restricting trade opportunities.

5.4.3 Optimum condition for bilateral trade

Finally, in this section the optimum condition for bilateral permit trading is derived, applying an offset rate. Assuming both trading sources minimize costs, the optimal traded amount of emission permits applying an offset rate *(V)* can be derived from the following optimization problem (Klaassen and Amann, 1992):

$$Min.\ C_S(r_S) + C_B(r_B) \tag{5.15}$$

$$s.t.\ \ e_S^1 = e_S^0 - \sigma \tag{5.16}$$

$$e_B^1 = e_B^0 + \beta \tag{5.17}$$

$$\beta = V\sigma \tag{5.18}$$

$$r_j = E_j - e_j \qquad j = S,B \tag{5.19}$$

After reformulation, the condition for a cost minimum solution can be derived by using the Lagrange function (Klaassen and Amann, 1992):

$$(e_S^1 \ x \ V) + e_B^1 = (e_S^0 \ x \ V) + e_B^0 \tag{5.20}$$

$$C_S^{\prime}(r_S) = V \ x \ C_B^{\prime}(r_B) \tag{5.21}$$

The optimal amount of traded emission permits is the amount for which the marginal abatement costs of the source that is selling emission permits equals the marginal abatement costs of the source buying emission permits, weighted by the offset ratio. Marginal costs are equalized in one-to-one permit trading because in this case the offset rate equals one. From condition *(5.21)* it follows that the buying of emission permits is only profitable for source B if in the pre-trade situation $C'_S(r_S) \leq V \ x \ C'_B(r_B)$, as buying emission permits decreases the marginal abatement costs of source B while selling emission permits increases the marginal abatement costs of source S. Although the use of offset rates unequal to unity enlarges trade opportunities, it has also drawbacks. Offset rates are not simple, fixed rates but depend on the pre-trade emissions and deposition targets (Klaassen and Amann, 1992). As a result, the trade systems become rather complex. The following section develops a system of guided permit trading to meet the drawbacks of trading with an offset rate.

5.5 Guided Bilateral Trade

5.5.1 Introduction

In the previous section I assumed that every bilateral trade has to meet the deposition targets. I also showed that this is a rather restrictive condition. If sources affect different receptors, a source selling permits is not able to compensate all receptors for an increasing deposition caused by the increase in emission from the source that is buying permits. Moreover, the initial permit allocation has to meet the deposition target. It is therefore necessary to find other rules for trading. One way to do this is to introduce the concept of 'guided bilateral trade'.

The central principle in the concept of guided bilateral trade is that not every bilateral trade transaction has to meet the deposition targets. This is because a subsequent bilateral trade transaction might compensate for the excess deposition at a receptor brought about by a previous trade transaction. A sufficient condition in guided bilateral trade is that deposition at all receptor points has to meet the targets only after all bilateral trade transactions have been completed. To ensure that deposition targets are eventually met, the number of permits traded in a bilateral transaction has to be controlled by a trade institution. The allowed number of traded emission permits is derived from the cost-effective solution. The question raised in this section is how the cost-effective allocation of emission abatement can be reached by guided bilateral trade. The concept of guided bilateral trade is explained by using the simple model with 5 emission sources and 5 receptor sites, which was introduced in section 5.3. For reasons of simplicity the abatement-cost functions are (for the time being) linear.

A bilateral trade transaction will only take place if the two trading parties benefit from the trade. In guided bilateral trade, the decision by the trade institution whether trade is allowed or not is based on the cost-effective allocation of emission abatement. Only after all allowed bilateral trade transactions have been completed, will the deposition targets be met at all receptors. Cost-effective abatement implicitly assumes full cooperation. However, as I discussed in Chapter 2, it is not in every country's interest to participate in a full cooperation agreement since some countries may be disadvantaged by full cooperation. Correspondingly, there might be a discrepancy between what is rational for a certain country and what is rational for Europe as a whole. Given the initial emission permit allocation, three situations can be distinguished.

(I) A situation in which every bilateral trade transaction is rational for each trading partner and contributes to the cost-effective solution, i.e every seller might trade with every buyer at a profit.

(II) A situation in which a bilateral trade transaction is necessary to reach the cost-effective abatement allocation, but which is not profitable for some individual countries.

(III) A situation in which a bilateral trade transaction is necessary if the cost-effective abatement is to be reached, but which is only profitable if trade takes place involuntary, i.e. with a certain trading partner.

These three situations are successively be illustrated in the next section.

5.5.2 Deriving guided bilateral trade

Emission trading on a one-to-one basis might only reach the cost-effective solution as derived in section 5.5.1, if the number of issued permits equals the total emissions in the cost-effective emission abatement allocation. In this section the emission permits (EP) initially issued are based on the average emission reduction percentage in the cost-effective abatement allocation.[57] The average reduction percentage amounts to 59% (derived in section 5.3). Consequently, it is assumed that the initially assigned emission permits equal 41% of the initial emission (E_j) of every source. Since deposition not only depends on the total amount of emissions but also on the allocation of emissions among the sources, this permit allocation does not meet the deposition targets in all receptor areas. Trading in emission permits is necessary if deposition targets are ultimately to be met. An overview of the cost-effective emission abatement allocation to be reached by bilateral trade, was given in Table 5.2.

Now all information is available that is needed to analyse guided bilateral permit trading. Table 5.5 gives an overview of the pre-trade data and the cost-effective emission allocation. The first column shows the unrestricted emissions, the second column gives the emission allocation according to the initially allocated emission permits, the third column shows the abatement costs resulting from this initial permit allocation and the fourth column presents the deposition in the pre-trade situation. The cost-effective emission allocation is presented in the sixth column and the associated costs are presented in the seventh column. The last column shows the deposition in the cost-effective allocation. The fifth column shows the difference between the initial permit allocation and the cost-effective allocation. This difference is indicated as a trade vector. When countries trade the

[57] This is in agreement with uniform emission reduction as applied in the SO_2 Protocol. Uniform emission reduction was criticized in Chapter 2.

number of permits as indicated by the trade vector, the result is the cost-effective allocation. The total potential savings from emission trading amount to 31936 (the difference between $\sum C_j^1$ and $\sum C_j^0$).

Table 5.5 EXAMPLE I: Overview of pre-trade data and cost-effective allocation

Source j/ receptor i	E_j	EP_j	C_j^0	d_i^0	Buy (+) Sell (-)	e_j^1	C_j^1	d_i^1
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)
1	275	113	16210	72	-54	59	21640	60
2	200	82	23650	73	+118	200	50	100
3	175	72	30975	108	+34	106	20627	100
4	400	164	11900	131	-164	0	20100	67
5	350	144	36075	126	+66	210	24457	100
Sum	1400	575	118810		0	575	86874	

The trade vector can be used for a scheme of bilateral trade to indicate which sources might be potentially trading partners and to show the maximum number of emission permits that sources are allowed to trade (see Table 5.6). Table 5.6 has to be read as follows. Sources selling emission permits are represented vertically. The trade vector shows that these are source numbers 1 and 4. Sources buying emission permits are represented horizontally, i.e. numbers 2, 3 and 5. The marginal abatement costs (MC) of every source are given in italics. The number of permits sold (σ) and bought (β) according to the trade vector are also given.

To derive the trade scheme an assumption has to be made about the sequence of bilateral trades. An obvious assumption is that a trade transaction generating the largest cost saving will take place first.[58] Table 5.6 has been filled by applying this rule. Consequently, the sequence of trades is as follows. First, source 4 sells

[58] Transaction costs have not been taken into account.

118 permits to source 2. Next, source 4 sells 34 permits to source 3. Source 1 then sells 54 permits to source 5. Finally, source 4 sells 12 permits to source 5.

Alternatively, the sequence of trades might be based upon the ratio in marginal costs of two trading sources. If so, the sequence of trades changes. First source 4 sells permits to source 3, then to source 2 and finally to source 5. Next, source 1 sells permits to source 5.

Table 5.6 EXAMPLE I: Trade scheme for bilateral trade.

			MC:	*200*	*300*	*175*
			source:	2	3	5
			β:	118	34	66
MC:	source:	σ:				
100	1	54				54
50	4	164		118	34	12

The model used for simulation is specified by equation *(5.22) - (5.31)*:

$$Min \sum_S \sum_t C_{S,t}(r_{S,t}) + \sum_B \sum_t C_{B,t}(r_{B,t}) \tag{5.22}$$

$$s.t.\ e_{S,t} = e_{S,t-1} - \sum_B G_{S,B,t}\,,\quad for\ every\ S\ and\ t \tag{5.23}$$

$$e_{B,t} = e_{B,t-1} + \sum_S G_{S,B,t}\,,\quad for\ every\ B\ and\ t \tag{5.24}$$

$$\sum_t G_{S,B,t} = S_S\,,\quad for\ every\ S \tag{5.25}$$

$$\sum_t G_{S,B,t} = B_B\,,\quad for\ every\ B \tag{5.26}$$

$$r_{S,t} = E_S - e_{S,t}\,,\quad for\ every\ S\ and\ t \tag{5.27}$$

$$r_{B,t} = E_B - e_{B,t}\,,\quad for\ every\ B\ and\ t \tag{5.28}$$

$$G_{S,B,t} \leq Min\ \{S_S;\ B_B\}\ x\ H_{S,B,t}\,,\quad for\ every\ S,\ B,\ t \tag{5.29}$$

$$\sum_S \sum_B H_{S,B,t} \leq 1,\quad for\ every\ t \tag{5.30}$$

where: $e_{S,t}$, $e_{B,t}$, $r_{S,t}$, $r_{B,t}$, $G_{S,B,t} \geq 0$ *(5.31)*

$C_{S,t}(r_{S,t})$ = abatement-cost function of a seller S

$C_{B,t}(r_{B,t})$ = abatement-cost function of a buyer B

Variables:

$r_{S,t}$ = emission reduction by seller S at stage t

$r_{B,t}$ = emission reduction by buyer B at stage t

$e_{S,t}$ = emissions of seller S at stage t

$e_{B,t}$ = emissions of buyer B at stage t

$G_{S,B,t}$ = traded permits between seller S and buyer B at stage t

$H_{S,B,t}$ = binary variable for every buyer-seller combination at stage t

Data:

E_S = initial emissions of country S

E_B = initial emissions of country B

S_S = maximum permits seller S is allowed to sell

B_B = maximum permits buyer B is allowed to buy

The objective is to minimize total abatement costs. By minimizing total abatement costs over all stages t, the trade between sources S and B that generates the largest cost saving will take place first. A trade transaction causes the emission of a source selling permits to decrease, and the emission of a source that is buying to increase. The emissions of sources not involved in trade remain constant. The number of permits sold equals the number of permits bought in every trade transaction between two sources. The total number of permits that sources are allowed to buy or sell is limited by the trade vector (elements S_S and B_B). To clarify the sequence of bilateral trade transactions equation *(5.29)* defines that only one transaction takes place in any stage. Variable $H_{S,B,t}$ is a binary variable defining the trade between a seller and a buyer at stage t.

Simulating bilateral trade results finally in the same emission allocation as the cost-effective emission abatement allocation. In this example, every bilateral trade generates a profit because the marginal abatement costs of the sources that are buying permits always exceed the marginal abatement costs of the sources selling emission permits. Consequently, trading in permits is always profitable. This illustrates situation (I), as described in section 5.6.1. Table 5.7 gives an overview of abatement costs and savings caused by bilateral permit trading.

Table 5.7 EXAMPLE I: Overview of abatement costs after every bilateral trade transaction.

	t0	t1	t2	t3	t4
C1	16210	16210	16210	**21648**	21648
C2	23650	**50**	50	50	50
C3	30975	30975	**20506**	20506	20506
C4	11900	**17800**	**19545**	19545	**20100**
C5	36075	36075	36075	**26559**	**24615**
Sum	118810	101110	92386	88307	86919
Profit		17700	8724	4082	1388

The pre-trade situation is represented by t0. The abatement costs of trading sources are given in bold and are underlined. According to the specified sequence of trades the first trade transaction between source 2 and source 4 is the most profitable and the last trade transaction between source 3 and source 4 generates the least profit. The profit in every transaction depends upon the difference in marginal abatement costs between the two trading sources. In this example the marginal abatement costs differ widely from 50 to 300. Consequently, profits in every transaction differ widely. The total profit generated by emission trading amounts to 31894. All potential cost savings are fully reached. Every bilateral trade transaction is rational for each trading partner. In this situation individual rationality coincides with full cooperation. Assuming that the emission sources aim at cost minimizing, all trade transactions will take place and the final trade result will coincide with the cost effective allocation of emission abatement.

However, as I indicated in section 5.5.1, rationality does not necessarily have to coincide with full cooperation. To illustrate situation (II) I have changed some of the transfer coefficients in the source-receptor matrix. Consider a new matrix A (the altered transfer coefficients are underlined):

0.5	0.1	0.1	0	0
0.25	0.4	0.05	0.05	0
0	0.15	0.6	0.1	0.25
0	0.25	0.1	0.5	0.15
0	0	0.05	0.35	0.45

Altered transfer coefficients affect the cost-effective solution. The solution to the optimization problem as formulated by equation *(5.1) - (5.4)* is solved by applying the new transport matrix. Table 5.8 gives an overview of the emissions, abatement and costs in the cost-effective solution.

Table 5.8 EXAMPLE II: Overview of emissions, abatement, costs and deposition in the cost effective abatement allocation.

Source j/ receptor i	E_j	e_j	r_j	C_j	d_i
1	275	163	112	11172	100
2	200	143	57	11478	100
3	175	40	135	40508	100
4	400	0	400	20100	73
5	350	218	132	23168	100
Sum	1400	564	836	106426	

The distribution of emission among the sources caused by the new source-receptor matrix has changed considerably in the cost-effective solution. The average abatement percentage in the cost-effective solution equals 60%. The initially issued emission permits are again based on the average abatement percentage in the cost-effective solution. Table 5.9 gives an overview of the pre-trade data and the cost-effective emission allocation.

Table 5.9 EXAMPLE II: Overview of pre trade data and the cost effective allocation

Source j/ receptor i	E_j	EP_j	C_j^0	d_i^0	Buy (+) Sale (-)	e_j^1	C_j^1	d_i^1
1	275	111	16410	71	+51	163	11214	100
2	200	81	23850	72	+62	143	11423	100
3	175	70	31575	106	-29	40	40530	100
4	400	161	12050	129	-161	0	20100	73
5	350	141	36600	123	+77	218	23167	100
Sum	1400	564	120485		0	564	106434	

The potential savings from emission trading amount to 14051 (the difference between ΣC_j^1 and ΣC_j^0). If the trade vector (i.e. the number of permits sources have to buy or to sell) is known, then a scheme for bilateral trade can be filled in. It is still assumed that sources generating the largest cost saving will trade first. The trade scheme for bilateral trade is represented in Table 5.10.

Table 5.10 EXAMPLE II: Trade scheme for bilateral trade.

			MC:	*100*	*200*	*175*
			source:	1	2	5
			β:	52	62	77
MC:	source:	σ:				
300	3	30		30		
50	4	161		22	62	77

The sequence of trades is as follows. First source 4 sells permits to source 5. Next, source 4 sells permits to source 2. Then source 4 sells permits to source 1. Finally, source 3 sells permits to source 1. Simulating bilateral trade according to scheme 1 results finally in the same emission allocation as the cost effective allocation of emission abatement. Table 5.11 gives an overview of abatement costs after every bilateral trade transaction.

Table 5.11 EXAMPLE II: Overview of abatement costs after every bilateral trade transactions

	t0	t1	t2	t3	t4
C1	16410	16410	16410	**14210**	**11210**
C2	23850	23850	**11450**	11450	11450
C3	31575	31575	31575	31575	**40575**
C4	12050	**15900**	**19000**	**20100**	20100
C5	36600	**23125**	23125	23125	23125
Sum	120485	110860	101560	100460	106460
Profit		9625	9300	1100	-6000

According to the selected trading sequence, the most profitable trade takes place first (t1). The last trade taking place (t4) has a negative profit. This is caused by the fact that the marginal abatement costs of source 3 exceed the marginal abatement costs of source 1. As a result of permit trading, the increase in abatement cost of source 3 exceeds the decrease in abatement cost of source 1. However, this trade transaction is necessary for meeting the deposition targets.

From an individual rational point of view, source 3 will never sell emission permits to source 1 voluntarily unless both sources are compensated for this trade by a side payment. From a common point of view, however, this trade is necessary for reaching the cost-effective solution, given the deposition targets. The total profit resulting from the permit trading is sufficient to pay a side payment to sources that are disadvantaged by permit trading. To be able to pay a side payment to sources 3 and 1, funds have to be created through, for example, the trade institution taxing the profit from trading.

Because of the high marginal abatement costs of source 3, the selling of permits by that source will always result in increasing total abatement costs, regardless of the trading partner. The disadvantage of permit trading is reduced if source 3 sells emission permits to source 2, as the difference in marginal between these sources is the smallest. The loss would now only amount to 3000. However, the assumption of a trade sequence based on the profitability of trades excludes a transaction between sources 2 and 3. The selling of permits by source 3 to source 5 also results in negative profit. It should be noted that the sequence of trade does not affect the total profit of emission trading, but does affect the distribution of the profit.

Simulated permit trading is affected by the specified transfer matrix. The row-sum of receptor 3 equals 1.1 and, in addition, the contribution of source 3 to receptor 3 is substantial. This explains why source 3 has to sell emission permits in spite of its high marginal abatement costs. By means of this example I have illustrated that trade might not be rational for both trading partners, even though the final trade is necessary for reaching a cost-effective solution.

Finally a situation might be distinguished in which, for all trade transactions to be profitable, the choice of trading partners is of overriding importance. This is an illustration of situation (III) as described in section 5.5.1. To create such a situation, the contribution of source 2 to receptor 2 is raised. To 'compensate' this increase, two other coefficients are changed. The changes in this matrix are based on trial and error. The altered coefficients (compared to the previous transfer matrix) are underlined. Consider the following transfer matrix A:

0.5	0.1	0.1	0	0
0.25	<u>0.65</u>	<u>0.1</u>	0.05	0
0	0.05	0.45	0.1	0.20
0	<u>0.1</u>	0.05	0.5	0.15
0	0	0.05	0.35	0.45

Analogous to the previous simulations, a cost-effective abatement allocation can be calculated using this new transfer matrix A. The initial emission permit allocation is defined according to the average abatement percentage. Table 5.12 gives an overview of pre-trade data and the cost-effective emission abatement allocation.

Table 5.12 EXAMPLE III: Overview of pre trade data and the cost effective allocation.

Source j/ receptor i	E_j	EP_j	C_j^0	d_i^0	Buy (+) Sale (-)	e_j^1	C_j^1	d_i^1
1	275	111	16410	71	+49	161	11410	100
2	200	81	23850	96	-7	74	25250	100
3	175	71	31275	80	+51	122	15975	100
4	400	161	12050	113	-161	0	20100	45
5	350	141	36600	123	+68	208	24875	100
Sum	1400	565	120185		0	565	97610	

Knowing the number of permits that sources buy and sell, a trade scheme for bilateral trade can be filled in by applying the specified trade sequence that trade with the highest potential cost savings is assumed to take place first. Regarding the marginal abatement costs of sources buying and selling emission permits, it is noticeable that to generate a profitable trade, source 2 can only trade with source 3 since for a profitable transaction the marginal abatement costs of the buyer should exceed those of the seller. In this example, source 3 is the only buyer whose marginal abatement costs exceed the marginal abatement costs of source 2. This is clarified in Table 5.13. If source 2 were to trade emission permits with source 1 or with source 5, the trade would generate a loss. Consequently, from an individual point of view source 2 would never trade with source 1 or with source 5.

As source 2 has to trade with source 3, the sequence of trades is as follows. First, source 4 sells emission permits to source 3. Source 4 then sells emission permits to source 5. Then, source 4 sells emission permits to source 1. Finally, source 2 sells emission permits to source 3. In this example the sequence as such is not affected by the fact that source 2 has to trade with source 3. However, because source 2 has to trade 7 permits with source 3, the maximum number of traded emission permits between source 4 and source 3 is only 44.

Table 5.13 EXAMPLE III: Trade scheme for bilateral trade.

			MC:	*100*	*300*	*175*
			source:	1	3	5
			β:	49	51	68
MC:	source:	σ:				
200	2	7			7	
50	4	161		49	44	68

Simulating bilateral emission permit trading according to Table 5.13 means that the trade result coincides with the cost-effective emission abatement allocation. Table 5.14 gives an overview of abatement costs after every bilateral trade.

Table 5.14 EXAMPLE III: Overview of abatement costs after every bilateral trade transactions

	t0	t1	t2	t3	t4
C1	16410	16410	16410	**11510**	11510
C2	23850	23850	23850	23850	**25250**
C3	31275	**18075**	18075	18075	**15975**
C4	12050	**14250**	**17650**	**20100**	20100
C5	36600	36600	**24700**	24700	24700
Sum	120185	109185	100685	98235	97535
Profit		11000	8500	2450	700

Because the number of emission permits equals the total emission in the cost-effective solution, bilateral permit trading results in cost-effective emission abatement. Since the number of permits source 4 and source 3 are allowed to trade is restricted, and since source 2 necessarily has to trade with source 3, all bilateral trades generate a profit.

The three distinguished situations in the introduction have been illustrated by changing the transfer matrix. However, the three situations could also be illustrated by changing the cost functions of the sources, because trade opportunities depend upon a combination of the source-receptor matrix and the abatement cost functions.

Obviously, in a situation in which the marginal abatement costs of all sources selling emission permits exceeds the marginal abatement costs of sources buying emission permits, guided bilateral permit trading will result in a cost-effective emission-abatement allocation. Whether sources have to sell permits or are allowed to buy permits depends upon the pre-trade emissions, the deposition target, the transfer matrix and the marginal abatement costs. In general, a source has to sell emission permits if it has low marginal abatement costs and a high transfer coefficient for a receptor i, which deposition has to be reduced. Nevertheless, sources with high marginal abatement costs and a high transfer coefficient for a receptor i, which deposition is allowed to increase will buy emission permits.

5.5.3 Extension: non-linear abatement cost functions

In practice, abatement cost functions for SO_2 emissions do not have a linear shape. A more likely assumption is that abatement costs increase exponentially the more emission is abated. The introduction of non-linear abatement cost functions does not alter the method of guided bilateral trade. This is looked at in this section. Assume the following abatement cost functions:

$C_1 = 20r_1^2$

$C_2 = 30r_2^{1.8}$

$C_3 = 35r_3^{1.65}$

$C_4 = 25r_4^{1.75}$

$C_5 = 5r_5^{1.85}$

Cost-effective emission abatement allocation can be derived by using the data on initial emissions, the transfer matrix and the deposition targets from section 5.3. Table 5.15 gives an overview of the cost-effective emission-abatement allocation.

Table 5.15 EXAMPLE NONLIN: Overview of emissions, abatement, costs and deposition in the cost-effective emission-abatement allocation.

Source j/ receptor i	E_j	e_j	r_j	C_j	d_i	MC_j
1	275	181	94	175321	100	3760
2	200	93	107	134590	88	2269
3	175	0	175	135809	58	1658
4	400	115	285	494732	100	3035
5	375	129	221	108967	98	910
Sum	1400	518	882	1049419		

Applying the new cost functions, the minimum costs in needed to meet the deposition targets amount to 1,049,419. Total emissions in the cost-effective solution amount to 518. This is a reduction of 63% compared to the base situation. Next, the overall emission-reduction percentage is used for the initial distribution of emission permits. The initial emission permits amount to 37% of the base emission *(E_j)* for every source.

Given the initial permit allocation, the cost-effective emission-abatement allocation can now be derived. The pre-trade data and the cost-effective emission allocation are represented in Table 5.16.

Table 5.16 EXAMPLE NONLIN: Overview of pre-trade data and the cost-effective allocation

Source j/ receptor i	E_j	ER_j	C_j^0	d_i^0	Buy (+) Sell (-)	e_j^1	C_j^1	d_i^1
1	275	102	598580	65	+79	181	175321	100
2	200	74	181046	66	+19	93	134591	88
3	175	65	64609	97	-65	0	135809	58
4	400	148	398466	118	-33	115	494718	100
5	350	129	108667	113	0	129	108980	98
Sum	1400	518	1351368		0	518	1049419	

In this example, the potential savings from permit trading amount to 301,949. The fifth column in Table 5.16 contains the trade vector, indicating how many permits sources have to buy or to sell in order to reach the cost-effective allocation. This trade vector is used for a scheme of bilateral trade. The scheme of bilateral trade is given in Table 5.17. According to the trade vector, source 5 does not trade any permits. Consequently, source 5 is not included in the trade scheme. Source 3 and source 4 sell emission permits and source 1 and source 2 buy emission permits. Because of the introduction of exponential abatement-cost functions, marginal abatement costs are no longer constant but depend on the level of abatement. Therefore it is not clear at first sight which sources have to trade with one another to generate the greatest profit.

Table 5.17 EXAMPLE NONLIN: Trade scheme for bilateral trade

		source:	1	2
		β:	79	19
source:	σ:			
3	65		65	
4	33		14	19

The sequence of trade that allows the first trade transaction to generate the greatest profit and the last trade transaction to generate the smallest profit is as follows. First, source 3 sells permits to source 1. Source 4 then sells permits to source 1 and finally source 4 sells permits to source 2. Table 5.18 gives an overview of the abatement costs after every bilateral trade transaction. The sources involved in a trade transaction are given in bold an are underlined. In addition the marginal abatement costs after the completion of all bilateral trade transactions are in the last column of Table 5.18.

Table 5.18 EXAMPLE NONLIN: Overview of abatement costs after every bilateral trade transaction.

	t0	t1	t2	t3	MC[1]
C1	598580	**233280**	**176720**	176720	3760
C2	181046	181046	181046	**134900**	2269
C3	64609	**135809**	135809	135809	1658
C4	398466	398466	**438009**	**482143**	3035
C5	108667	108667	108667	108667	910
Sum	1351368	1057268	1040251	1038239	
Profit		294100	17017	2012	

Table 5.18 shows that the cost-effective emission abatement allocation results (deviation in total abatement costs due to round off) after all trade transactions have been completed. By introducing exponential abatement-cost functions, the sequence of trades might become more complex. Permit trading is not only limited by the maximum number of allowed traded permits resulting from the trade vector, but if the marginal abatement costs of the sources intersect, then the number of traded permits at maximum profit also depend on the level of abatement.

5.5.4 Guided bilateral trade and administrative procedures

As shown in the previous section, guided bilateral trade requires a trade institution to guide the process of bilateral trade. This institution has to consider whether or not trade transactions are allowed. To be able to control trade transactions, all transactions have to be announced in advance. Next, the institution decides whether to allow the transaction. This decision will be based upon the trade vector. Consequently, trade transactions which are rational for both trading parties but do not contribute to the cost-effective solution are excluded.

If a trade transaction is necessary to reach the cost-effective solution, that is not rational from an individual point of view (situation II), the institution has to play an active role in stimulating the transaction by providing a side payment. Funds for side payments can be created for example by taxing profitable trade transactions. Using a progressive tax system might equalize the benefits of emission trading between the trade parties. In some situations the trade institution can also take care of the choice of trading partners. If a source has to trade with a certain other source to be able to generate a profit from trade (situation III), the institution can guide the bilateral trades in such a way that the sources are able to trade with one another.

As stated in section 5.4.1, to reach the-cost effective solution by bilateral permit trading on a one-to-one basis, the number of issued permits has to equal the total emissions in the cost-effective solution. Where the number of issued permits does not equal the total emissions in the cost-effective solution, the trade institution can play a part in distributing additional permits so that the cost-effective solution can be reached.[59] For this reason it might be necessary for the trade institution to function as a "third" trade partner by buying permits from one source and selling additional permits to another source. A final remark concerns the periods relating to a bilateral trade sequence. From a practical point of view it is unrealistic to assume that sources will adapt to their new allowed number of emissions immediately after every bilateral trade transaction. Sources may trade more than once. It is conceivable that after the issue of permits, a certain time-period should be allowed for trades to take place.

[59]see for example variant 2 in Chapter 6.

5.6 Conclusions

In this chapter I have discussed several systems of permit trading in relation to cost-effective emission abatement. I have shown that for non-uniformly mixing pollutants, permit trading has to be constrained if it is to reach a cost-effective emission allocation. This allocation is one that minimizes the total abatement costs while at the same time does not violate deposition targets. If permit trading is constrained it has been argued that permit trading should be regarded as a bilateral process.

A system of bilateral permit trading which defines that every single transaction has to meet deposition targets is restricted by the transfer coefficients and by the ratio of marginal abatement costs of both trading sources. If trade takes place on a one-to-one basis, trade opportunities are limited and permit trading will not reach the cost-effective abatement allocation. Applying an offset rate unequal to 1, while enlarging trade opportunities, does make trading very complex. Therefore, in this chapter I have developed a new system for bilateral permit trading, indicated as guided bilateral trade. In this system the maximum number of permits that sources are allowed to sell or buy is based upon the cost-effective abatement allocation. It is not necessary for every trade transaction to meet the deposition targets, because these targets are taken care of by means of the maximum numbers that sources are allowed to buy and sell. The term 'guided bilateral trade' indicates that the permit market as such is not able to generate the desired outcome. The trade process has to be guided by a trade institution that takes care of the number of traded permits. However, any trading system that takes deposition targets into account will need some kind of control.

To guide trade transactions to a cost-effective emission allocation, the allocation has to be known. Therefore the trade institution needs full information to be able to establish the trade vector. Unfortunately, this cancels one of the characteristics of tradeable permits namely that full information is not needed. In the system of guided bilateral trade, the trade coordinating institute has to derive the trade vector. However, situations may occur in which reaching a cost-effective allocation at once is politically not possible.[60] In that case a system of guided bilateral

[60]As in the Second Sulphur Protocol.

trade is able to achieve further cost savings. After all, given a specified emission allocation, this system clearly indicates which transactions can take place in which both seller and buyer benefit. In other words, a system of guided bilateral trade provides a recipe for reaching further costs savings.

A drawback to guided bilateral permit trading is that trade transactions do not necessarily have to be profitable. Non-profitable trade transactions need some sort of substitution if deposition targets are to be met. Whether this occurs depends on the source-receptor matrix, the abatement-cost functions, the deposition targets and the initial permit distribution. The main advantage of the guided bilateral trade system is that the cost-effective emission abatement allocation can be achieved if all trade transactions are profitable or if non-profitable transactions are subsidized. In the past, achieving the cost-effective allocation was thought to be possible by means of a simultaneous trade process. However, as I showed in section 5.3, the practical implication of such a trade system is a black box. By means of the system of guided bilateral trade this black box is opened.

To analyse the consequences of guided permit trading for SO_2 emissions in Europe, guided bilateral permit trading needs to be simulated. The simulation results will shed light on the question as to whether the cost-effective emission allocation can actually be reached by guided bilateral permit trading.

6 SIMULATING GUIDED BILATERAL TRADE FOR SO_2 REDUCTIONS IN EUROPE

6.1 Introduction

In this chapter, I apply the concept of guided bilateral trade that has been developed in Chapter 5 to sulphur dioxide (SO_2) emission trading between European countries. Having discussed the concept of guided bilateral trade theoretically in Chapter 5, in this chapter, I raise the question of to what extent guided bilateral trade results empirically in the aimed emission allocation.

Recapitulating briefly, the guided bilateral trade system aims at reaching a cost-effective abatement allocation, taking into account deposition targets. In order to be able to reach the cost-effective allocation by permit trading, the total number of emission permits is based upon the total emissions in the cost-effective allocation. In addition the trading of emission permits is subject to deposition targets. For this purpose a trade coordinating institute specifies a trade vector that indicates how many permits countries are allowed to buy or sell.

To calculate the cost-effective abatement allocation for SO_2 emissions in Europe the RAINS model was used. Assumptions about the deposition targets were based upon the 1994 SO_2 Protocol, which implies a gap closure of 60% between the actual deposition and the critical loads. Emission data and cost curves were taken from the RAINS model too. Having both the cost-effective allocation and a rule for initially distributing the emission permits, then the trade vector was derived. Having this trade vector and the national abatement-cost functions guided bilateral trade between European countries for SO_2 emissions was simulated. The simulation result was analysed for both the environmental consequences and abatement costs.

As well as examining the Base Case in this chapter, I also look at three variants of guided bilateral permit trading. First, I will formulate a Base Case based on a gap closure of 60%. Next, in variant 1, I formulate a modification on the Base Case. This modification examines the consequences of ignoring the current emission reduction plans of the countries. Finally, in variant 2, I simulate guided bilateral permit trading, using the Second SO_2 Protocol as a guideline for the initial distribution of emission permits. In this variant, countries not signing the protocol are excluded from permit trading. Variant 2 is split into 2 subvariants: variant 2A assumes that the Trade Coordinating Institute (TCI) sells permits but not at a fixed price, whereas variant 2B assumes that the TCI sells permits at a fixed price. The main assumption of the variants are summarized in Table 6.1.

Table 6.1 Overview of the main assumptions of the variants discussed for guided bilateral permit trading.

VARIANT	Deposition Target	Initial Permit Allocation	Current Reduction Plans	Other
Base Case	60% gap closure	historical emissions	yes	-
Variant 1	60% gap closure	historical emissions	no	-
Variant 2A	modified 60% gap closure	second SO_2 protocol	only for non-signing countries	TCI permit price not fixed
Variant 2B	modified 60% gap closure	second SO_2 protocol	only for non-signing countries	TCI permit price fixed

I compare the Base Case with two alternative abatement strategies. Firstly, it is compared with non-guided permit trading; this is permit trading without any restriction. Secondly, it is compared with a uniform emission reduction strategy that meets the deposition targets.

The organization of this chapter is as follows. Section 6.2 describes the model used for simulation. Section 6.3 formulates the Base Case. Given the assumptions of the Base Case guided bilateral permit trading is simulated and compared with alternative abatement strategies. Section 6.4 presents the modification on the Base Case. Section 6.5 is a discussion of guided bilateral permit trading, starting from the SO_2 Protocol. Finally in section 6.6 conclusions are drawn.

6.2 The model

6.2.1 General description

In order to simulate a system of guided bilateral trade among European countries four steps have to be taken. These four steps are represented schematically in Figure 6.1. First, a cost-effective emission-abatement allocation is calculated, given the national abatement cost functions, the emissions, the source-receptor matrix and the deposition targets, by using the RAINS model. In this allocation, deposition targets are met at minimum abatement costs. The second step is to define an initial distribution of emission permits.[61] To reach the cost-effective abatement allocation by guided bilateral permit trading, the total emissions corresponding to the number of initially distributed emission permits must equal the total emissions in the cost-effective abatement allocation. Since it is assumed that permits are traded on a one-to-one basis, which implies that the total number of emissions permits is fixed, this requirement is essential.

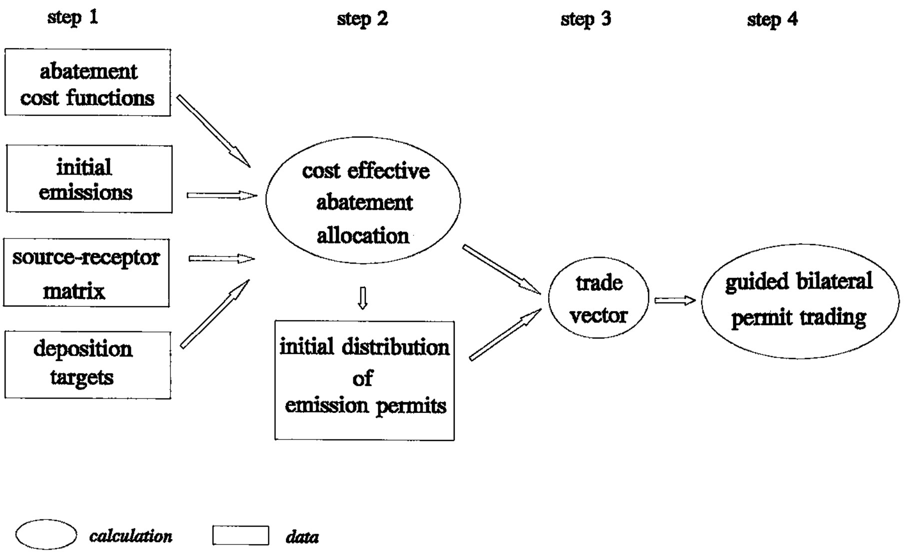

Figure 6.1 *Schematic overview of steps to be taken to simulate guided bilateral permit trading.*

The third step is to derive the trade vector. In Chapter 5, I stipulated that the trade vector indicates the number of permits countries are allowed to buy or sell. The trade vector is calculated as the difference between the cost-effective

[61]The issue of the initial distribution of emission permits was discussed in detail in section 3.6.

abatement allocation and the initially distributed emission-permit allocation. A positive number in the trade vector is the maximum number of emission permits a country is allowed to buy, whereas a negative number in the trade vector is the maximum number of emission permits a country is allowed to sell. Obviously, the initial distribution of emission permits largely influences the trade vector.

Finally, having obtained the trade vector, the fourth step is to simulate guided bilateral trade for SO_2 emission permits, assuming that a trade-coordinating institute restricts the amounts that countries are allowed to buy and sell, in accordance with the maximum quantities indicated by the trade vector. To do this I developed an optimization model in OMP.[62] The aim of simulating guided bilateral trade is to clarify the sequence and the volume of trade transactions between European countries, under the assumption that countries generating the largest cost savings trade first. The model equations of this optimization model are given by equations *(5.22) - (5.31)* in section 5.6.2. In Appendix I the model is represented as formulated in the OMP-modelling language.

To ensure the trade that generates the largest cost saving will take place first, the objective function maximizes the profit caused by permit trading in period t. The profit of trade is defined as the total reduction in abatement costs that result from the modified abatement allocation as a result of trading. To be able ultimately to reach the cost effective allocation, a constraint has been added to the optimization model so that for every source, the permits to be sold or bought, summed over all periods t are restricted according to the trade vector. For every country, the degree of emission abatement in a period t equals the initial emissions minus the number of initially issued permits and minus the number of permits bought, or plus the number of permits sold before period t. Whether a source is a buyer or a seller is defined by the trade vector.

6.2.2 Abatement cost functions

Generally, abatement costs increase as more emissions are abated. These costs generally increase sharply beyond a certain emission reduction. This is rather obvious since the more emission is abated the more expensive are the techniques

[62] OMP is a linear programming modelling package.

that are needed. A convex shaped abatement cost function can be expressed by the functional form: $C(r) = a_x r^b$ in which r represents the amount of emission reduction and a and b are cost coefficients.[63] A linear programming model, however, requires linear abatement functions.[64] Therefore the convex cost functions may be approached by piece-wise linear functions. Figure 6.2 is a graph showing a piece-wise approximation of a convex function.

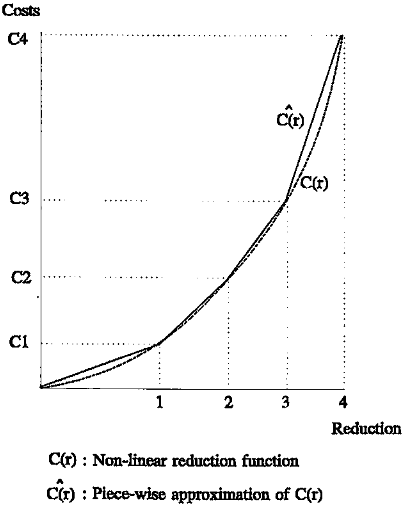

Figure 6.2 *Approximation of a non-linear abatement-cost function by using a piece-wise linear cost function.*

The abatement cost functions used in the optimization model to simulate guided bilateral trade are taken from the RAINS model. In this model, the abatement-cost functions are already piece-wise linear. As I explained in Chapter 4, these functions are technologically based and depend upon the energy scenario chosen. The abatement cost functions used in the optimization model are based upon the Official Energy Pathway for the year 2000. Each step in the RAINS abatement-cost function implies another abatement technique. The step size differs for every

[63]Cronshaw and Kruse (1993) indicated that because of the discrete choice of abatement technologies abatement cost-functions are unlikely in practice to be strictly convex.

[64]Because I used binary variables to model the trade sequence, as explained in section 5.2.1, linear programming was necessary. When an optimization problem includes binary variables, the problem can only be solved if the objective function and the constraints are linear in decision variables.

technique, because the ranges of application of techniques differ. Moreover, the step sizes and the number of steps vary for every country because the installation capacities, the fuels used and other characteristics differ for every country. The abatement functions also define the maximum possible emission reduction. Broadly, the maximum abatement percentage ranges from 80 to 90% of the projected emissions for the year 2000.[65]

6.3 The Base Case

6.3.1 Basic assumptions

Deposition target

The deposition target for the Base Case was chosen in accordance with the negotiations that led to a new SO_2 protocol in June 1994. The deposition target currently aimed at is based on a pan European reduction of the difference (the 'gap') between the actual deposition level and the 5-percentile critical loads for sulphur. The 5-percentile critical load protects 95% of the ecosystem area within a grid cell of 150 x 150 km against the damaging effect of sulphur deposition.

A 'gap closure' scenario would bring all European countries closer to fully achieving the 5-percentile critical loads with an equal percentage. This scenario takes into account regional differences in the environmental sensitivities, the dispersion of air pollutants, and differences between the countries' economic and energy situations and some notion of economic effort required to achieve these targets (Amann et al., 1992a).

In the negotiations for the 1994 protocol, a gap-closure of 60% has been specified as the deposition target (Amann and Schöpp, 1993). In this scenario, the difference between the 1990 deposition level and the 5-percentile critical load is reduced by 60%. For grids where the 1990 deposition already meets the 5-percentile critical loads, the deposition target equals these critical loads. This deposition target is applied to 483 land based receptors of 150 x 150 km each.

---

[65] A main exception is Switzerland where only 40% of the emissions can be abated because of the large extent of industrial emissions.

The gap-closure concept seems to be a fair target because it aims at bringing all countries closer to critical loads with an equal percentage, with the exception for those receptors where the deposition already meets the critical loads. However, it should be noticed that reaching this target at the binding receptors implies a gap-closure of more than 60% at non-binding receptors. Moreover, since the deposition target is expressed as a percentage, countries that were confronted with high deposition levels in 1990, will suffer more from environmental damage than countries confronted with low deposition levels in 1990. Although the relative improvement is equal for all countries, the absolute differences may differ substantially.

Cost-effective emission allocation

The aim of guided bilateral permit trading is to reach the cost-effective emission allocation. This allocation meets the deposition target, in this case the 60% gap closure target, at the lowest possible abatement costs. Apart from the necessary abatement efforts required to reach the specified deposition targets, some countries have already committed themselves to achieving a certain emission reduction. The target years for these plans vary but, for reasons of comparison have been assumed to be the year 2000 across Europe. The cost-effective emission allocation used in the Base Case takes into account these so-called Current Reduction Plans (CRP). The CRP are taken into account by adding a constraint in the optimization problem for calculating the cost-effective emission allocation. This stipulates that the emission reduction should be greater or equal to the CRP reduction. As a result, the cost-effective allocation does not represent the lowest possible costs, but the lowest possible costs given the CRP.

Further, it is assumed that ship emissions will also be reduced. As well as distinguishing between emissions from countries, the RAINS model also distinguishes betweeen ship emissions.[66] However, it is not assumed that these will participate in permit trading. The cost-effective emission allocation is calculated for the year 2000. Table 6.2 gives an overview of several emission data.

[66]Ship emissions are emissions in the Baltic Sea, the North Sea and the Atlantic Ocean.

Table 6.2 Overview of SO_2 emission data and projections (kt)

Country	Emissions 1990	Unabated Emissions 2000	CRP-Emissions 2000
Albania	140	137	138
Austria	98	352	78
Belgium	414	721	430
Bulgaria	1266	815	520
Czechoslovakia	2443	2346	2170
Denmark	184	369	176
Finland	260	603	116
France	1260	1309	1101
West Germany	940	2271	520
East Germany	4800	2363	230
Greece	740	905	595
Hungary	1010	1075	1094
Ireland	168	234	240
Italy	2180	3000	1976
Luxembourg	16	15	10
Netherlands	207	424	106
Norway	54	144	70
Poland	3210	3738	2600
Portugal	204	389	304
Rumania	1800	2592	2592
Spain	2316	2793	2143
Sweden	169	412	100
Switzerland	62	74	60
Turkey	1459	2887	2889
United Kingdom	3774	3333	2552
Yugoslavia	1480	1576	1576
Baltic Sea	73	73	73
North Sea	173	173	173
Atlantic Ocean	316	316	316
Kola Karelia	701	728	396
St. Petersburg	409	334	307
Baltic Region	653	629	435
Belorussia	596	567	456
Ukraine	2782	5066	1696
Moldavia	313	378	231
RSU*	3350	6347	3725
Europe	40020	49488	32194

* Remaining Soviet Union

The first column shows the emissions in 1990, the second column shows the projected unabated emissions in 2000 based on the Official Energy Pathway and the third column shows emissions according to the CRP. The emissions are presented for all countries (or regions where applicable) that are distinguished in the RAINS model. This classification is not up to date. Political changes of the past years have not been taken into account. This is revealed by the inclusion of Czechoslovakia, both West and East Germany, and Yugoslavia in Table 6.2. Emissions in 1990 are used to determine the initial distribution and to calculate the deposition in 1990, so as to derive the gap closure deposition target. The projected emissions for the year 2000 are the initial (unabated) emissions in Figure 6.1 (step 1) that are used to calculate the cost-effective abatement allocation. The CRP-emissions are used as a restriction for calculating the cost-effective abatement allocation in the Base Case.

The cost-effective emission allocation taking into account the CRP for the year 2000 is represented in Table 6.3. The first column represents the emissions in the cost-effective allocation, the second column shows the abatement required to reach these emissions and the third column shows the abatement costs.

Table 6.3 Overview of the cost effective emission allocation for 60% gap closure for the year 2000 taking into account CRP (emissions and abatement in kt SO_2. Abatement costs in million DM.).

Country	Cost-Effective Emissions	Emission Abatement	Abatement Costs
Albania	132	5	0
Austria	78	274	537
Belgium	200	521	712
Bulgaria	520	295	251
Czechoslovakia	1792	554	423
Denmark	51	318	790
Finland	116	487	916
France	675	634	461
West Germany	520	1751	3914
East Germany	230	2133	2437
Greece	595	310	212
Hungary	521	554	290
Ireland	53	181	234
Italy	1016	1984	1384
Luxembourg	10	5	27
Netherlands	106	318	716
Norway	34	110	379
Poland	1717	2021	1782
Portugal	294	95	69
Rumania	1036	1556	844
Spain	1486	1307	496
Sweden	88	324	979
Switzerland	60	14	16
Turkey	2887	0	0
United Kingdom	949	2384	2533
Yugoslavia	983	593	819
Baltic Sea	18	55	101
North Sea	42	131	238
Atlantic Ocean	317	0	0
Kola Karelia	157	571	463
St. Petersburg	239	95	61
Baltic Region	266	363	317
Belorussia	456	111	72
Ukraine	1696	3370	1988
Moldavia	231	147	89
RSU	3725	2622	1320
Europe	23296	26193	25870

Emission abatement (Table 6.3) is the difference between unabated emissions (Table 6.2) and the cost-effective emissions (Table 6.2). Comparing the CRP-emissions (Table 6.2) with the cost-effective emissions shows that for the majority of countries cost-effective emissions are lower than CRP-emissions. Since it is assumed that countries will at least carry out their current reduction plans in calculating cost-effective emission allocation, the cost-effective emissions are always lower or equal to CRP-emissions.

Trade vector

As a first step in trading, countries have to receive a number of emission permits. Since guided bilateral trade aims at reaching the cost-effective emission allocation, the sum of all permits over all countries has to correspond to the total European emission in the cost-effective allocation. Therefore, in the Base Case, the total number of emission permits is equal to 23296 kt SO_2.[67]

The distribution of the total number of permits between the countries in the Base Case is based upon historical emissions. The year chosen for reference is 1990. The initial distribution of permits is equal to the emission that would result for each country if all countries reduced emissions by a uniform percentage. In the Base Case this percentage is 42%, since it is equivalent to a European emission level of 23296 kt SO_2. It should be noted that the emission allocation resulting from this initial distribution of emission permits does not meet the deposition targets. The trade vector follows from the difference between the initial distribution of permits and the cost-effective emission allocation. The initial distribution of emission permits and the trade vector are represented in Table 6.4. For comparison, the cost-effective emission allocation is repeated from Table 6.3. The countries in Table 6.4 are ordered according to the trade vector. First the buyers are represented and next the sellers are represented.

[67]Throughout this chapter it is assumed that 1 permit allows the emission of 1 kt SO_2.

Table 6.4 Base Case: calculation of the trade vector (kt SO_2).

Country	Cost-Effective Emissions	Initial Distribution	Trade Vector (buyers +; sellers -)
Albania	132	81	51
Austria	78	57	21
Czechoslovakia	1792	1419	373
Greece	595	430	165
Luxembourg	10	9	1
Norway	34	33*	1*
Portugal	294	118	176
Spain	1486	1345	141
Switzerland	60	43*	17*
Turkey	2887	847	2039
Yugoslavia	983	860	123
St. Petersburg	239	238	1
Belorussia	456	346	110
Ukraine	1696	1616	80
Moldavia	231	182	49
RSU	3725	1946	1779
Belgium	200	240	-40
Bulgaria	520	735	-215
Denmark	51	107	-56
Finland	116	151	-35
France	675	732	-57
West Germany	520	546	-26
East Germany	230	2779*	-2549*
Hungary	521	587	-66
Ireland	53	98	-45
Italy	1016	1266	-250
Netherlands	106	120	-14
Poland	1717	1865	-148
Rumania	1036	1046	-10
Sweden	88	98	-10
United Kingdom	949	2192	-1243
Kola Karelia	157	407	-250
Baltic Region	266	379	-113
Baltic Sea	18	18	-
North Sea	42	42	-
Atlantic Ocean	317	317	-
Europe	23296	23296	0

* adapted data

Unfortunately the specified initial distribution requires Switzerland and Norway to exceed their maximum possible abatement.[68] Since this is not possible, the initial distribution has been adapted such that both countries can satisfy their abatement requirements.[69]

6.3.2 Results of guided bilateral trade

Having described and motivated the basic assumptions for guided bilateral trade, all elements for trading are now available. In terms of Figure 6.1, I have completed steps 1 to 3. Now step 4 has to be taken. From section 5.6.1, we know that bilateral transactions are not by definition all profitable. To obtain insight into the profitability of trading, guided bilateral permit trading is simulated.

After the initial distribution of permits, countries can start trading them. The maximum number of permits countries are allowed to buy or sell is defined by the trade vector. Assuming that the trade transaction with the greatest profit will take place first, the sequence of profitable bilateral permit trades is represented in Table 6.5.[70] The first two columns show the seller and the buyer in a trade transaction. The third column shows the number of permits traded in a trade transaction. The fourth column shows the profit generated by the transaction. The profit depends on both the quantity of permits traded and the differences in marginal abatement costs. The fifth column shows the market price of the traded permits in a transaction. This price equals the marginal abatement costs of the seller. However, since I assume that permit trading is a bilateral process, it is reasonable also to assume that both trade partners will negotiate a permit price so that both have the same advantage with permit trading. The last column in Table 6.5 shows

---

[68]This maximum is according to the abatement functions in the RAINS model. If all abatement technologies are applied in Switzerland and Norway, their minimum emissions equal 43 and 33 kt SO_2 respectively. An overview of maximum abatement for all European countries is given in Appendix II.

[69]Therefore Switzerland and Norway are additionally allocated emission permits for 7 and 2 kt SO_2 respectively. Since the total number of emission permits cannot be changed, the permits initially distributed to former East Germany have been reduced by 9. East Germany was chosen because this is the only country that is able to reduce 9 kt at zero costs because the number of initially distributed emission permits to East Germany exceeds the amount of unabated emissions in the year 2000. In table 6.3 the adapted initial distribution and trade vector are presented.

[70]Abbreviations of names of countries are given in Appendix III.

this implicit permit price. This price is calculated on the assumption that both seller and buyer receive an equal profit share from permit trading.

Table 6.5 Base Case: sequence of profitable trade transactions

Seller	Buyer	Traded Amount (kt. SO_2)	Profit (Mio.DM)	Market Permit Price (Mio. DM/ kt SO_2)	Implicit Permit Price (Mio. DM/kt SO_2)
(1)	(2)	(3)	(4)	(5)	(6)
GDR	TUR	1077	2700	0	1.63
GDR	RSU	965	176	0.71	0.80
HUN	AUS	21	155	0.53	4.17
KOL	SWI	17	124	0.70	4.35
GDR	POR	176	122	0.71	1.06
UK	YUG	123	119	0.72	1.22
UK	BYE	110	47	0.73	0.94
HUN	LUX	1	18	0.53	10
KOL	GRE	124	17	0.70	0.90
HUN	NOR	1	10	0.53	6
UK	GRE	41	9	0.73	0.84
GDR	ALB	26	4	0.71	0.81
UK	ALB	19	3	0.73	0.82
HUN	CZE	10	2	0.53	0.80
ITA	CZE	12	1	0.73	0.79
UK	S-PET	1	0.15	0.73	1.08

The benefit of permit trading by the buyer equals the decrease in abatement costs *(ΔC_b)* minus the permit price *(P)* that has to be paid, multiplied by the number of permits bought *(Q)*. The benefit of permit trading by the seller equals the permit price *(P)* that is received, multiplied by the number of permits *(Q)* minus the increase in abatement costs *(ΔC_s)*. The implicit permit price can be calculated either from a seller's perspective or from a buyer's perspective. The following is the calculation from a seller's point of view. If both the seller and the buyer receive half the profit of trading, the following equation holds:

$$P * Q - \Delta C_s = 0.5 * Profit \qquad (6.1)$$

The total profit equals the decrease in the buyer's abatement costs minus the increase in the seller's abatement costs:

$$Profit = \Delta C_b - \Delta C_s \qquad (6.2)$$

Substitution of *(6.2)* in *(6.1)* and rewriting gives:

$$P = (\Delta C_b + \Delta C_s) \;/\; 2\,Q \qquad (6.3)$$

Equation *(6.3)* defines that the implicit permit price *(P)* equals the decrease in the buyer's abatement costs plus the increase in the seller's abatement costs, divided by twice the traded quantity of permits *(Q)*. Analogously, *(6.3)* can be derived from the buyer's perspective.

Consider for example the first trade transaction. The decrease in abatement costs (ΔC_b) for Turkey amounts to 3106 million DM and the increase in abatement costs (ΔC_s) for former East Germany amounts to 406. The traded quantity of permits in the first trade transaction amounts to 1077. If we substitute these numbers in equation *(6.3)* it follows that the permit price equals 1.63. This first trade transaction is between East Germany and Turkey, generating a profit of 2.7 billion DM. Since East Germany is able to sell a number of permits at zero costs, the profit from the first transaction is observably large. Turkey is allowed a great increase in emissions in 2000 because they only affect minor environmentally sensitive areas. As the initial allocation of emission permits is based on the emission level of 1990, the trade vector allows Turkey to purchase a large number of permits. The subsequent trade transaction between East Germany and the Remaining Soviet Union is about the same size in terms of amount of emission, but the profit is considerably smaller.

The calculations show that countries do not always trade the maximum number of permits allowed by the trade vector. This can be explained by the intersection of marginal abatement-cost curves. As long as marginal abatement-cost curves do not intersect, countries will trade their maximum allowed number of permits. Otherwise it is more profitable to trade only until the point of intersection. The variation in implicit permit prices is considerable. Since the trade sequence is based solely on the profitability of a trade transaction, these prices do not follow a pattern. From equation *(6.3)* it follows that there is no clear relationship between the profit and the implicit permit price. After all profitable trade transactions have been completed the total profit generated by permit trading is 3507.15 million DM., but the cost-effective emission allocation has still not been reached.

To analyse guided bilateral permit trading, Figure 6.3 and Figure 6.4 illustrate the initial trade vector and the corresponding marginal abatement costs in graph form. Each bar indicates a country. The y-axis in Figure 6.3 represents the number of permits a country is allowed to buy (positive number) or sell (negative number). The y-axis in Figure 6.4 represents the marginal abatement-costs in the pre- trade allocation. In both figures, buyers are represented first and seller second.

Trade is only profitable if the marginal abatement cost of a buyer exceeds the marginal abatement cost of the seller. From Figure 6.4 it is clear that this condition does not hold for each buyer-seller combination. The most striking example is Sweden. The marginal abatement costs of Sweden exceed the marginal abatement cost of any buyer, therefore Sweden will not sell any permits. Other countries with relative high marginal costs not selling permits are West Germany, Finland, Denmark and the Netherlands.

From Table 6.5 and from Figures 6.3 and 6.4 it can also be concluded that the buyer with the highest marginal abatement cost (Luxembourg) does not buy first. Since the trade order is defined by the total generated profit, the number of permits a country is allowed to buy is also of importance. Therefore Turkey and the Remaining Soviet Union are the first two buyers. Luxembourg, the buyer with the highest marginal costs is only the eighth buyer, since it is allowed to buy only one emission permit.

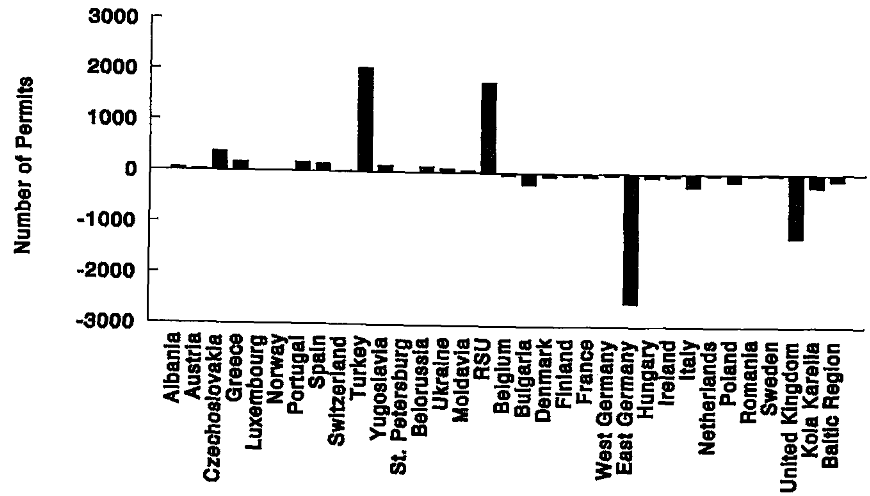

Figure 6.3 *Base Case: initial trade vector (buyers +; sellers -).*

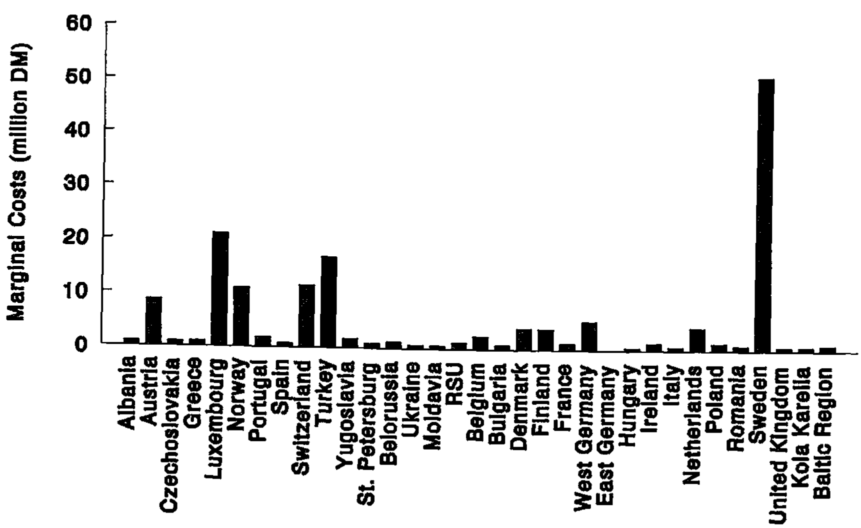

Figure 6.4 *Base Case: marginal abatement costs in the pre-trade allocation.*

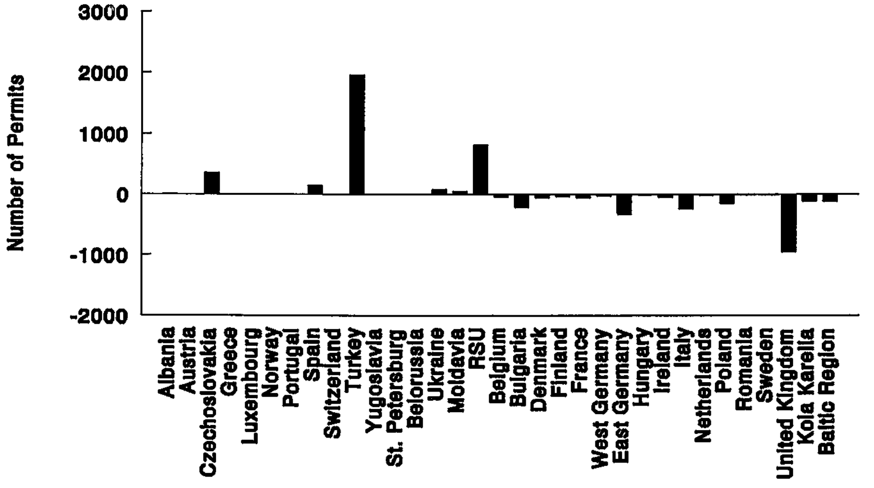

Figure 6.5 *Base Case: trade vector after all profitable transactions have taken place (buyers+; sellers -).*

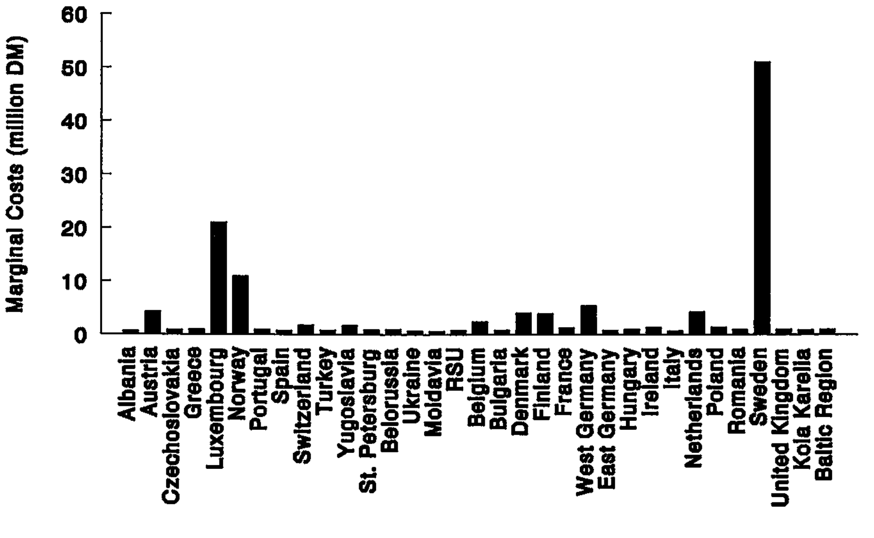

Figure 6.6 *Base Case: marginal abatement costs after all profitable trade transactions have taken place.*

Finally, Figures 6.5 and 6.6 illustrate the resulting trade vector and the corresponding marginal abatement-costs after all profitable trade transactions have taken place. No more profitable permit trading can take place because the marginal costs of potential sellers exceed the marginal costs of potential buyers. For example, Czechoslovakia, a potential buyer, cannot generate a profitable trade with any of the potential sellers, because the marginal costs of all potential sellers are too high compared to the marginal costs of Czechoslovakia.

The trade allocation does not meet the deposition targets at all receptors. Figure 6.7 shows that the deposition target is exceeded at only 46 of the 483 receptors and at most of those it is only slightly exceeded. At one receptor the relative exceedance is 35%.

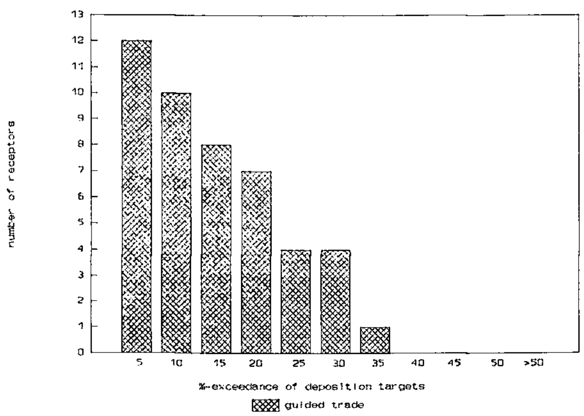

Figure 6.7 ***Relative exceedance of deposition targets for the guided trade allocation.***

Figure 6.8 shows the location of the 46 receptors where the deposition target is exceeded. These receptors are shaded dark. A disproportionately large number of these receptors are located in Norway, the Netherlands, Switzerland and Italy. In section 6.3.4 deposition levels are compared in more detail.

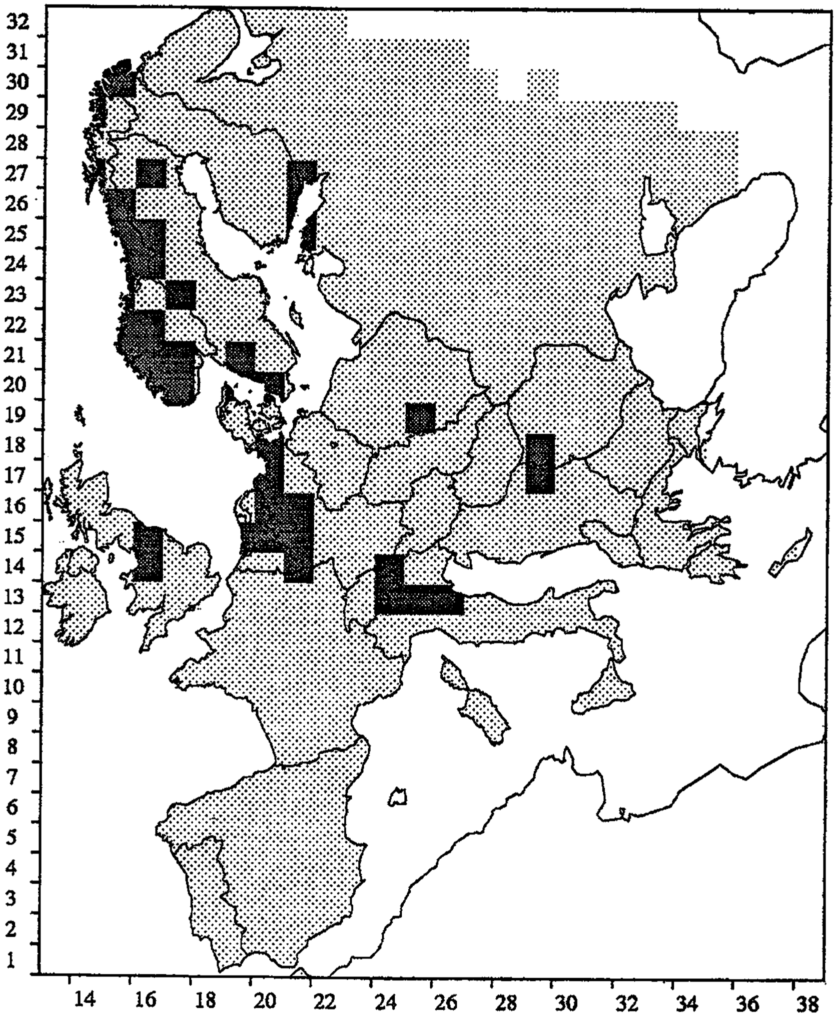

Figure 6.8 *Location of receptors in Europe exceeding the deposition targets in the guided trade allocation (if only profitable transactions take place).*

If the cost-effective emission allocation is to be reached a number of loss-making transactions has to take place. These transactions will result in an increase in abatement costs because such an increase by the sellers can no longer be compensated by the decrease in abatement costs of the buyers. This phenomenon has already been pointed out in Chapter 5. To stimulate further emission trading, a side payment (e.g. a subsidy) has to be introduced. Since the cost difference between the pre-trade allocation and the cost-effective allocation amounts to 135 million DM, there is in principle an opportunity for side payments. The sequence of trades after introducing side payments strongly depends upon the amount of the side payments and upon the way such payments are introduced.

For reason of completeness and to give some insight into the volume of loss-making trade transactions, these transactions are simulated, assuming that a subsidy per permit is paid.[71] This subsidy is introduced after all profitable trades have taken place. First, a subsidy of 5 million DM per permit is paid (i.e. 5000 DM for 1 tonne SO_2). Next, the subsidy is raised in a stepwise manner by 5 million DM per permit to generate trade. Table 6.6 gives an overview of the sequence of loss-making trade transactions. The first two columns gives the seller and the buyer of a trade transaction. The third column gives the number of permits traded. The fourth column represents the profit from trade, including the subsidy. Correcting the profit for this subsidy results in an increase in abatement costs (that is a loss) caused by permit trading, as represented in the fifth column. This loss has been calculated as the difference between the profit and the total subsidy. For example, in the first transaction the subsidy equals 4715 (943/5), whereas the profit equals only 3813. The loss (or increase in abatement costs) equals 902. Table 6.6 illustrates that the introduction of a subsidy results in very unequally distributed profits. Starting with a subsidy of 5 million DM per permit results in extremely high profits for the first countries that trade.

[71]It is not suggested that this is a realistic policy alternative.

Table 6.6 Base Case: sequence of trade transactions involving losses

Seller	Buyer	Traded Amount (kt SO_2)	Profit (million DM)	Increase in total Abatement Costs (Mio. DM)
UK	TUR	943	3813[1]	902
ITA	CZE	238	1111[1]	79
BUL	RSU	215	1022[1]	53
GDR	RSU	274	939[1]	431
POL	RSU	147.5	641[1]	97
BAL	SPA	113	529[1]	36
KOL	CZE	107	491[1]	44
FRA	UKR	57	245[1]	40
IRE	RSU	44.5	167[1]	55
HUN	RSU	33	156[1]	9
BEL	RSU	40	89[1]	111
DEN	RSU	38	63[1]	127
ROM	SPA	9.5	46[1]	2
NET	SPA	14	13[1]	57
FIN	UKR	15	8[1]	67
KOL	SPA	2	6[1]	4
ROM	SPA	0.5	2.3[1]	0.2
POL	SPA	0.5	2.2[1]	0.3
IRE	SPA	0.5	1[1]	1.5
FRG	UKR	2	0.5[1]	9.5
FRG	MOL	24	100[2]	140
FIN	RSU	20	68[2]	132
GDR	MOL	17	24[2]	146
DEN	MOL	8	11[2]	69
DEN	UKR	6	7[2]	53
DEN	ALB	4	4[2]	36
SWE	TUR	10	22[3]	228
GDR	RSU	2	1[4]	59
GDR	ALB	2	1[4]	59
GDR	SPA	1	0.6[4]	29.4
GDR	TUR	9	43[5]	272

[1] Including a subsidy of 5 million DM/permit
[2] Including a subsidy of 10 million DM/permit
[3] Including a subsidy of 25 million DM/permit
[4] Including a subsidy of 30 million DM/permit
[5] Including a subsidy of 35 million DM/permit

Assuming still that the most profitable trade takes place first, the profit slowly decreases the more transactions take place. This pattern is disrupted after 20 trade transactions by the need to raise the subsidy to generate further trade. As a result the profit abruptly increases at this point. The total number of subsidies paid is extremely high: 2294 permits are traded with a subsidy of 5 million DM per permit, 79 permits are traded with a subsidy of 10 million DM per permit, 10 permits are traded with a subsidy of 25 million DM per permit, 5 permits are traded with a subsidy of 30 million DM per permit and finally 9 permits are traded with a subsidy of 35 million DM per permit. The total amount paid in subsidies equals 12975 million DM.

The stepwise increase of subsidies seems to be very unfair for countries who trade first. If countries behaved rationally they would postpone trading until after the highest subsidy is paid. In fact, the introduction of subsidies to stimulate permit trading is likely to be politically infeasible. Therefore permit trading will only be regarded as far as it is profitable, taking into account that deposition targets will not be fully reached. Therefore, in further analyses only the profitable trades shown in Table 6.5 are taken.

6.3.3 Alternative abatement strategies

Non-guided permit trading

An interesting analysis is to look at what advantages guided SO_2 permit trade has above non-guided SO_2 permit trading in Europe. Therefore non-guided permit trading was simulated, based upon the initial permit distribution used for guided permit trading. Non-guided permit trading has no restrictions on buying and selling permits. It is also not clear beforehand which sources are buyers and which are sellers.

After permits have been distributed, sources can start trading. A source will buy permits so long as the marginal abatement costs exceed the permit price. A source will sell emission permits so long as the permit price exceeds the marginal abatement costs. Assuming that countries behave rationally non-guided permit trading will result in the lowest cost allocation of emissions, given the total emission permits available and not taking deposition targets into account.

Table 6.7 Overview of the non-guided trade allocation (emissions in kt SO_2, abatement costs in million DM).

Country	Initial Distribution	Trade Allocation	Abatement Costs in the Trade Allocation	Traded Permits (bought +; sold -)
Albania	81	65	60	-16
Austria	57	292	37	235
Belgium	240	559	97	319
Bulgaria	735	554	212	-181
Czechoslovakia	1419	965	1126	-454
Denmark	107	279	69	172
Finland	151	383	163	232
France	732	736	377	4
West Germany	546	1904	292	1358
East Germany	2779	428	1338	-2351
Greece	430	191	607	-239
Hungary	587	514	297	-73
Ireland	98	173	50	75
Italy	1266	1183	1167	-83
Luxembourg	9	15	0	6
Netherlands	120	358	36	238
Norway	33	106	31	73
Poland	1865	2578	764	713
Portugal	118	256	106	138
Rumania	1046	915	960	-131
Spain	1345	1193	750	-152
Sweden	98	327	67	229
Switzerland	43	67	5	24
Turkey	847	1583	884	736
United Kingdom	2192	1827	1109	-365
Yugoslavia	860	1452	70	592
Baltic Sea	18	18	-	-
North Sea	42	42	-	-
Atlantic Ocean	317	317	-	-
Kola Karelia	407	266	324	-141
St. Petersburg	238	125	166	-113
Baltic Region	379	304	265	-75
Belorussia	346	433	95	87
Ukraine	1616	788	2765	-828
Moldavia	182	62	222	-120
RSU	1946	2037	2630	91
Europe	23296	23296	17141	0

Table 6.7 gives an overview of the emission allocation after trade. The initial distribution of emission permits is equal to the distribution in the guided bilateral trade situation. The second column represents the emission allocation after all trades have taken place. The abatement costs resulting from this allocation are represented in the third column. The last column represents how many permits are bought and sold by the countries. Compared to guided permit trading, many countries turn out sellers rather than buyers and vice versa. This change occurs because deposition targets are not taken into account in non-guided permit trading.

Since deposition targets are not taken into account in non-guided permit trading the total abatement costs after trade strongly decrease, compared to the cost-effective allocation that meets the deposition targets. However, as I will show in detail in section 6.3.4, these cost savings have an environmental drawback because deposition targets are not reached.

Uniform emission reduction

A second alternative emission reduction strategy that is presented here is the so-called uniform emission reduction strategy. This strategy aims at reaching the deposition target of 60% gap closure at all receptors by requiring an equal reduction percentage for all countries. The base year for emission reduction is 1990. The uniform reduction percentage to meet the 60% gap closure deposition target at all receptors equals 66%. Table 6.8 gives an overview of the emission allocation and abatement costs of this emission reduction strategy. It should be noted that not all countries are able to reduce 66% of their 1990 emissions. For these countries (marked with an asterisk) Table 6.8 shows the maximum feasible abatement.

It is common knowledge that a uniform emission reduction strategy generally results in excessive abatement costs. Compared to the cost-effective emission allocation, the abatement costs increase by 16394 million DM., an increase in abatement costs of 63%.

Table 6.8 Uniform emission reduction: 66% (emission and abatement in kt SO_2; abatement costs in million DM.)

Country	Emissions	Abatement	Abatement Costs
Albania	48	89	85
Austria	33	319	920
Belgium	141	580	999
Bulgaria	430	385	354
Czechoslovakia	831	1515	1309
Denmark	63	306	671
Finland	88	515	1147
France	428	881	938
West Germany	320	1951	5659
East Germany	1632	731	457
Greece	252	653	543
Hungary*	395	680	800
Ireland	57	177	219
Italy	741	2259	1828
Luxembourg	5	10	121
Netherlands	70	354	992
Norway*	33	111	390
Poland	1091	2647	2869
Portugal	69	320	456
Rumania	612	1980	1331
Spain	787	2006	1263
Sweden*	84	328	1070
Switzerland*	43	31	152
Turkey*	813	2074	5509
United Kingdom	1283	2050	1869
Yugoslavia	503	1073	1632
Baltic Sea	25	48	88
North Sea	59	114	207
Atlantic Ocean	107	209	378
Kola Karelia	238	490	355
St. Petersburg	139	195	153
Baltic Region	222	407	396
Belorussia	203	364	371
Ukraine	946	4120	2598
Moldavia	106	272	180
RSU	1139	5208	3955
Europe	14036	35452	42264

* Maximum abatement

6.3.4 Comparison of deposition levels

The allocation reached by profitable guided permit trading and by non-guided permit trading can be compared with the pre-trade allocation in terms of environmental quality.[72] In the pre-trade allocation, the deposition target is exceeded at 88 receptors (of a total of 483 receptors). In the non-guided trade allocation, the deposition target is exceeded at 109 receptors, whereas in the guided trade allocation, the deposition target is only exceeded at 46 receptors (see Figure 6.8, section 6.3.2).

A point of interest concerns the relative exceedance of deposition targets. In general, the smaller the relative exceedance the less environmental damage occurs. Figure 6.9 illustrates the relative exceedance of deposition targets in the pre-trade allocation, the post non-guided trade allocation and in the post-guided trade allocation. In this figure, the relative exceedance is expressed on the x-axis and the number of permits is measured along the y-axis. For example, the first bar in Figure 6.9 indicates that in the pre-trade allocation the deposition target is exceeded by 5% at 17 receptors. The second bar indicates that in the non-guided trade allocation, the deposition target is exceeded by 5% at 10 receptors and the third bar indicates that in the guided trade allocation, this exceedance of 5% occurs at 12 receptors.

Figure 6.9 indicates that the guided trade allocation causes the lowest environmental damage. The guided trade allocation clearly shows an environmental improvement compared to the pre-trade allocation. The environmental damage caused by non-guided permit trading is the largest. The non-guided trade allocation results in relative exceedances of more than 50% at 35 receptors. The guided trade allocation only results in relative exceedances of maximum 35%.

6.3.5 Distribution of abatement costs

A final issue for comparison concerns the distribution of abatement costs. Generally, distribution issues are most important to policy makers. In Table 6.9 an overview of abatement costs is presented for several abatement allocations. First

---

[72]Both the 'cost-effective' allocation and the '66% uniform emission reduction' allocation are excluded in this comparison since deposition targets are fully met by these allocations.

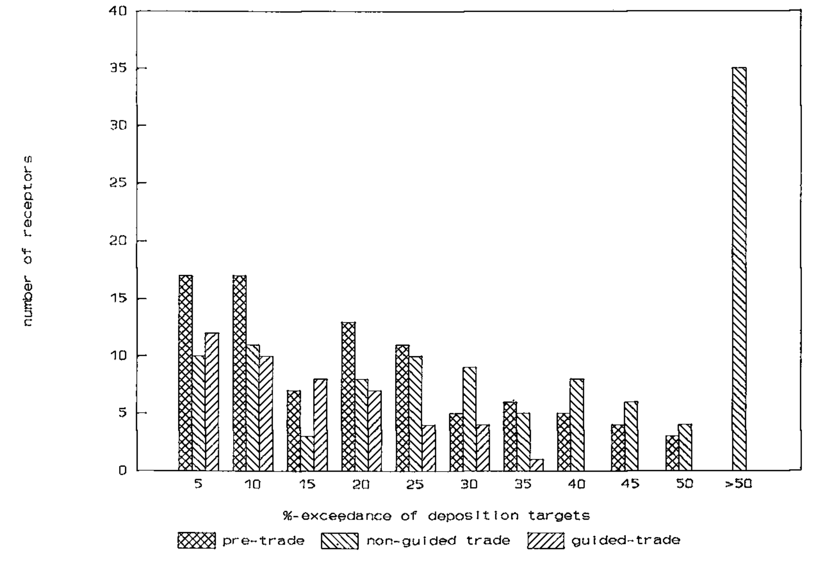

Figure 6.9 *Relative exceedance of deposition targets for the pre-trade, non-guided trade and the guided trade allocation.*

the abatement costs in the cost-effective allocation are represented. Second, the abatement costs of a 66% uniform emission allocation are presented. Third, the abatement costs of the initial permit allocation are represented. The fourth column of Table 6.9 represents the abatement costs after profitable guided bilateral permit trading. Permit trading not only affects abatement costs, but sources are also faced with costs or benefits of buying c.q. selling permits. These distributional effects are represented in the last column, indicated with total net costs after trade. For example, the abatement costs for Luxembourg after trade amount to 27 million DM. However, Luxembourg bought 1 permit at a price of 10 million DM. Therefore the total costs for Luxembourg, as represented in the fifth column amount to 37 million DM. Obviously, for sellers of emission permits, the total costs decrease. For example, the United Kingdom sells permits to Greece, Albania and St. Petersburg, causing an increase in abatement costs from 825 to 1040 million DM. However, the proceeds raised by selling permits exceeds this increase in abatement costs. As a result, the total costs for the UK decrease from 825 to 735 million DM. The total European net costs after trade equal the total costs in the trade allocation because the aggregate benefits for selling sources equal the aggregate costs for buying sources.

Generally, the 66% uniform reduction allocation results in the highest abatement costs. However, for some countries the cost-effective allocation corresponds to higher abatement costs. This is so for Denmark, former East Germany, Ireland and the Baltic Sea. In the cost-effective allocation, these countries have to reduce their emissions by more than 66%. Since the uniform emission reduction allocation is based on a reduction of 66%, it is obvious that the initial permit allocation, which only requires a uniform reduction of 42%, results in fewer abatement costs. Since countries will only trade permits if this is profitable, the total cost trade allocation corresponds to lower (or equal) costs compared to the initial permit allocation.

It is an ambiguous task to define who benefits from permit trading. The benefits strongly depend on the point of departure. With the exception of former East Germany, the trade allocation results in lower abatement costs compared to the 66% uniform reduction allocation for all countries. Compared to the cost-effective allocation, the trade allocation results in higher abatement costs for Albania, Czechoslovakia, Spain, Turkey, Ukraine, Moldavia and the Remaining Soviet Union. Considering the total cost of the trade allocation (in this case including the revenues or costs of selling or as the case may be buying permits) it follows that Albania, Austria, Czechoslovakia, Greece, Luxembourg, Norway, Portugal, Spain, Switzerland, Turkey, former Yugoslavia, Belorussia and the Remaining Soviet Union suffer from higher costs than in the cost-effective allocation. Notably, all these countries are buyers of emission permits.

Table 6.9 Overview of abatement costs (million DM.) for four emission allocations and total net costs after trade.

Country	Abatement Costs				
	Cost-Effective Allocation	66% Uniform Reduction Allocation	Initial Permit Allocation	Post Trade Allocation	Total Net Costs after Trade*
Albania	0	85	44	4	40
Austria	537	920	701	537	625
Belgium	712	999	575	575	575
Bulgaria	251	354	48	48	48
Czechoslovakia	423	1309	736	717	734
Denmark	790	671	467	467	467
Finland	916	1147	692	692	692
France	461	938	382	382	382
West Germany	3914	5659	3747	3747	3747
East Germany	2437	457	0	1239	-1496
Greece	212	543	371	212	358
Hungary	290	800	240	259	147
Ireland	234	219	147	147	147
Italy	1384	1828	1096	1105	1096
Luxembourg	27	121	46	27	37
Netherlands	716	992	649	649	649
Norway	379	390	390	379	385
Poland	1782	2869	1585	1585	1585
Portugal	69	456	317	69	257
Rumania	844	1331	834	834	834
Spain	496	1263	607	607	607
Sweden	979	1070	725	725	725
Switzerland	16	152	152	16	90
Turkey	0	5509	3692	587	2343
United Kingdom	2533	1869	825	1040	735
Yugoslavia	819	1632	1027	819	969
Baltic Sea	101	88	101	101	101
North Sea	238	207	238	238	238
Atlantic Ocean	0	378	0	0	0
Kola Karelia	463	355	209	324	138
St. Petersburg	61	153	62	61	62
Baltic Region	317	396	192	192	192
Belorussia	72	371	199	72	175
Ukraine	1988	2598	2044	2044	2044
Moldavia	89	180	122	122	122
RSU	1320	3955	2743	1879	2651
Europe	25870	42264	26005	22501	22501

* after including revenues from permit selling or expenditures for permit buying

6.4 Variant 1: excluding current reduction plans

In this section I examine the possibility of achieving the emission allocation with absolute lowest abatement costs. In contrast to the Base Case, current reduction plans are not taken into account in calculating the cost-effective abatement allocation. Thus, the constraint that countries should reduce a minimum amount of emission according their CRP is released in calculating the cost-effective emission allocation. The deposition targets used for this variant are those resulting from the 60% gap closure. The cost-effective emission allocation is calculated for the year 2000, taking into account the official energy pathway when calculating the unabated emissions and the abatement-cost functions. The resulting abatement allocation reflects the cheapest way of reaching the deposition target of 60% gap closure. It is assumed that the ship emissions are also reduced according to this cost-effective emission allocation, although ship emissions are excluded from permit trading.

Given this cost-effective emission allocation which has to be reached by means of permit trading, an appropriate number of emission permits has to be first distributed. As in the Base Case, the initial permit distribution is based on a uniform reduction of the emission level of 1990. The reduction percentage amounts to 34% percent compared to 42% in the Base Case. Next, the trade vector follows from the difference between the initial distribution of permits and the cost-effective emission allocation. Table 6.10 gives an overview of the cost effective emissions, the initial permit distribution and the trade vector.[73]

The initial distribution derived in this way implies lower total abatement costs than the abatement costs in the cost-effective allocation (see Table 6.11). This cost difference is explained by the fact that the initial distribution of emission permits is based on a uniform reduction percentage, and does not take the emission location into account. As a result deposition targets are violated.

[73] The initial distribution of permits again involves a problem. According to the initial distribution of permits, Switzerland has to abate 35 kt SO_2, but the maximum technically feasible abatement of Switzerland only amounts to 31 kt SO_2. For Switzerland it is assumed that 4 permits can be obtained (bought) from the seller with lowest costs (East Germany), before the trade process starts.

Table 6.10 Variant 1. Overview of SO_2 emission data: cost-effective emissions, initial distribution and the trade vector (kt SO_2).

Country	Cost-Effective Emissions	Initial Distribution	Trade Vector (buyer +; seller -)
Albania	132	92	40
Austria	219	64	155
West Germany	670	618	52
Greece	862	487	375
Luxembourg	14	11	3
Portugal	294	134	160
Switzerland	64	43	21
Turkey	2752	959	1793
Yugoslavia	990	973	17
Belorussia	546	392	154
Ukraine	3122	1829	1293
Moldavia	334	206	128
Russia	5456	2202	3254
Belgium	132	272	-140
Bulgaria	794	832	-38
Czechoslovakia	853	1606	-753
Denmark	58	121	-63
Finland	125	171	-46
France	675	828	-153
East Germany	391	3155	-2764
Hungary	521	664	-143
Ireland	53	110	-57
Italy	1019	1433	-414
Netherlands	78	136	-58
Norway	33	36	-3
Poland	1823	2110	-287
Rumania	946	1183	-237
Spain	1486	1523	-37
Sweden	88	111	-23
United Kingdom	1000	2481	-1481
Kola Karelia	157	461	-304
St. Petersburg	105	296	-164
Baltic Region	149	429	-280
Baltic Sea	18	18	-
North Sea	42	42	-
Atlantic Ocean	317	317	-
Europe	26317	26317	-

Table 6.11 Variant 1. Overview of abatement costs in the pre-trade allocation and in the cost-effective allocation (million DM/yr).

Country	Pre-Trade	Cost-Effective Allocation
Albania	35	0
Austria	639	147
Belgium	496	1079
Bulgaria	0	0
Czechoslovakia	579	1268
Denmark	411	722
Finland	613	850
France	301	461
West Germany	3373	3187
East Germany	0	1400
Greece	316	0
Hungary	199	290
Ireland	131	234
Italy	974	1380
Luxembourg	8	2
Netherlands	581	921
Norway	357	390
Poland	1298	1641
Portugal	288	69
Rumania	709	929
Spain	467	496
Sweden	568	971
Switzerland	152	9
Turkey	2542	3
United Kingdom	613	2428
Yugoslavia	835	807
Kola Karelia	171	463
St. Petersburg	37	196
Baltic Region	143	532
Belorussia	144	1
Ukraine	1901	1097
Moldavia	106	21
Russia	2488	338
Europe	21474	22332

The cost-effective allocation, however, requires some countries to reduce much more emission. This results in a considerable increase in abatement costs. The most apparent example is Czechoslovakia. According to the initial distribution, Czechoslovakia is allocated with 1606 kt SO_2. The cost-effective allocation, however, allows Czechoslovakia to emit only 853 kt SO_2. By way of comparison, in the Base Case Czechoslovakia's cost-effective emissions amount to 1792 kt SO_2.

Table 6.11 gives an overview of the abatement costs corresponding to the initial permit distribution and the abatement costs in the cost-effective emission allocation. Although the total post-trade costs are higher than the total pre-trade costs, a number of profitable trade transactions will take place, if countries behave rationally. Table 6.12 gives an overview of the sequence of profitable trades, assuming that the trade with the greatest profit will take place first.

Table 6.12 Variant 1. Sequence of profitable trades

Seller	Buyer	Traded Quantity (kt. SO_2)	Profit (million DM)	Market Permit Price (million DM/kt SO_2)	Implicit Permit Price (million DM/kt SO_2)
GDR	TUR	1171	1895	0	0.98
GDR	AUS	155	383	0.71	1.95
HUN	FRG	52	159	0.53	2.05
BUL	SWI	21	143	0	3.40
GDR	POR	160	104	0.71	1.04
GDR	RSU	709	103	0.71	0.79
GDR	GRE	375	49	0.71	0.78
KOL	BYE	102	35	0.70	0.88
BUL	YUG	17	29	0	0.85
HUN	ALB	40	13	0.53	0.71
HUN	LUX	3	4	0.53	1.27
HUN	BYE	7	1.2	0.53	0.66
ROM	BYE	5	0.1	0.70	0.69

As in the Base Case, the first seller of emission permits is the former East Germany. Since the number of initially distributed emission permits exceeds the level of unabated emissions in the year 2000, former East Germany can sell the first permits at zero costs. Turkey on the other hand, planned an increase of emissions in 2000 whereas the number of permits is based on 1990-emissions.

Consequently Turkey has to buy a large amount of permits. The profit generated by the first trade transaction is observably large. The quantity of permits traded in the first transaction is also remarkably large compared to the other trade transactions. The total profit generated by permit trading amounts to 2.9 billion DM and is about the same size as the generated total profit in the Base Case. The number of permits traded in this variant is also about the same size as the number of permits traded in the Base Case.

After all profitable trades have taken place, many permits still have to be traded to reach the cost-effective emission allocation. Since these transactions are no longer profitable, a kind of side payment has to be introduced. However, since the total abatement costs in the cost-effective allocation exceed the total abatement costs in the pre-trade allocation, there are no funds for paying side payments.

The allocation reached by profitable permit trading (post-trade allocation) can be compared with the pre-trade allocation with respect to environmental quality. In the pre-trade allocation, the deposition target is exceeded at 119 receptors. In the post-trade allocation, the deposition target is exceeded at 84 receptors. Figure 6.10 illustrates the relative exceedance of deposition targets in the pre-trade and post-trade allocation. Figure 6.10 illustrates that both the number of receptors where the deposition target is exceeded and the relative exceedance of deposition targets are reduced by permit trading. However, if we compare this variant with the Base Case, both the number of receptors and the relative exceedance of deposition targets has increased. An obvious explanation for this lower environmental benefit is that there is an increase in the total emissions in the cost-effective emission allocation not taking into account CRP.

6.5 Variant 2. The Second SO_2-Protocol

This variant examines whether guided bilateral permit trading, starting from the actual SO_2 Protocol can improve the cost-effectiveness of emission abatement. The difficulty of getting concerning political agreement on the initial distribution of permits is generally regarded as a major drawback to a system of permit trading. By using the emission distribution of the actual SO_2 Protocol as a starting point for permit trading, this drawback is met.

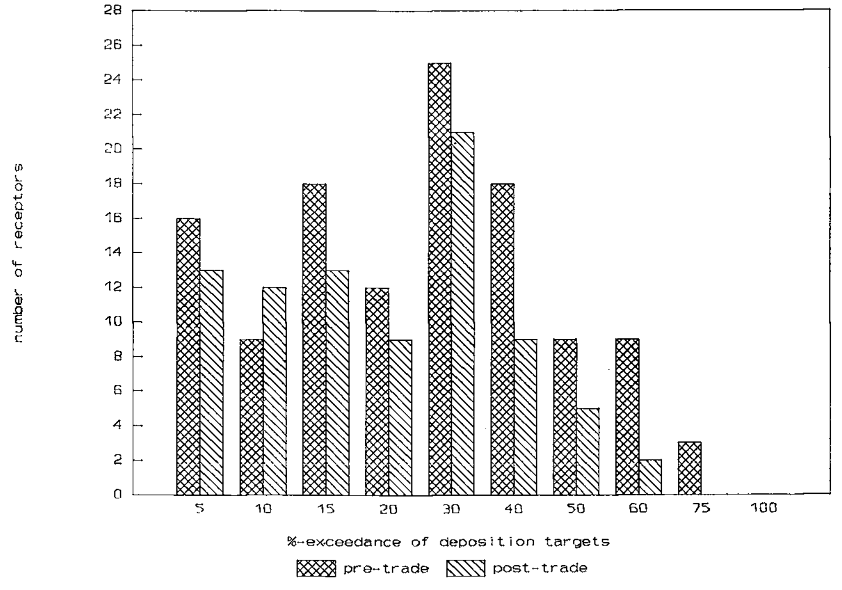

Figure 6.10 *Variant 1. Relative exceedance of deposition targets in the pre-trade and post-trade allocation.*

It should be noted that not all European countries signed the 1994 SO_2 Protocol. Therefore I have assumed here that only those countries who have signed the Protocol will participate in a system of guided bilateral permit trading. Table 6.13 gives an overview of the signatory countries, their committed emission ceilings for the year 2000 and the accompanying abatement costs according to the RAINS model.[74,75] The main countries who did not sign the Protocol were Albania, Rumania, Turkey and the Baltic Region.

Although the SO_2 Protocol in principle aimed at the deposition target corresponding to 60% gap closure, the actual deposition resulting from the protocol will not reach this deposition target at all receptors. There are two reasons for this.

---

[74]Canada, not included in Table 6.12, signed the Protocol as well.

[75]The Protocol data have been adapted to fit the country classification of the RAINS model.

Table 6.13. Sulphur emissions according to the 1994 SO_2 Protocol and the accompanying abatement costs.

Country	Emissions (kt SO_2)	Abatement Costs (million DM)
Austria	78	537
Belgium	248	555
Bulgaria	1374	0
Czechoslovakia	1465[a]	697
Denmark	90	536
Finland	116	866
France	868	269
West Germany	1019[b]	1940
East Germany	281	1768
Greece	595	212
Hungary	898	76
Ireland	155	72
Italy	1330	1049
Luxembourg	10	27
Netherlands	106	716
Norway	34	377
Poland	2583	759
Portugal	304	59
Spain	2143	163
Sweden	100	688
Switzerland	60	16
United Kingdom	2449	637
Yugoslavia	1378[c]	167
Belorussia	456	72
Ukraine	2310	1602
RSU	4440	836
Total	24890	14696

[a] Czech Republic: 1128 kt SO_2 and Slovakia: 337 kt SO_2

[b] Germany committed itself to emit 1300 kt SO_2. However, the RAINS model distinguishes West Germany from East Germany. The emission distribution has been chosen so that total abatement costs for both West and East Germany are minimal.

[c] Only Croatia (133 kt SO_2) and Slovenia (130 kt SO_2) signed the Protocol. It is assumed that the Republic of Yugoslavia emits according to CRP.

Firstly, some countries did not sign the protocol[76] and secondly, some countries did not commit themselves to a reduction of as many emissions as were necessary to achieve this deposition target. Therefore, a new deposition target was formulated to simulate guided bilateral permit trading based upon the SO_2 Protocol. This new deposition target is as follows. First, the deposition resulting from the SO_2 Protocol is calculated, assuming that the non-signatory countries act according to their CRP. Next, the deposition at every receptor is compared to the 60% gap closure deposition target. For receptors where this deposition target is exceeded, the deposition target is adapted to the deposition resulting from the SO_2 Protocol. For receptors where the deposition is below or equal to the 60% gap closure deposition target, this target is maintained. Moreover, deposition targets are only formulated for receptors in the countries that signed the protocol.

Next, given this new deposition target, a cost-effective emission allocation can be calculated. This cost-effective emission allocation meets the adapted deposition targets at minimum abatement costs for the countries signing the SO_2 Protocol, given the CRP emissions of the non-signatory countries. Table 6.14 gives an overview of emissions and abatement costs in the cost effective emission allocation. Table 6.14 indicates that the deposition target based upon the SO_2 Protocol can be achieved at lower abatement costs compared to the abatement costs resulting from the SO_2 Protocol. The difference amounts to 1211 million DM.

Once the initial allocation of emission permits and the cost-effective allocation of emissions have been established, the trade vector can be derived. The problem is that the sum of emissions in the cost-effective allocation exceeds the initial amount of allocated emissions. In other words, the total number of permits for all countries that may be bought exceeds the total number of permits for all countries that may be sold. To deal with this problem I have introduced an additional seller, the Trade Coordinating Institute which can sell an additional amount of permits. Table 6.15 shows the resulting trade vector.

---

[76] I have assumed that those countries have acted according to their own CRP.

Table 6.14 Variant 2. Emissions and abatement costs in the cost-effective emission allocation for countries signing the SO_2 Protocol.

Country	Emissions (kt SO_2)	Abatement Costs (million DM)
Austria	106	418
Belgium	250	550
Bulgaria	794	0
Czechoslovakia	1346	797
Denmark	98	504
Finland	116	863
France	810	315
West Germany	1025	1918
East Germany	391	1400
Greece	894	0
Hungary	521	290
Ireland	169	55
Italy	1244	1114
Luxembourg	14	2
Netherlands	123	636
Norway	41	275
Poland	2541	805
Portugal	304	59
Spain	2148	162
Sweden	110	578
Switzerland	66	6
United Kingdom	2235	793
Yugoslavia	1398	141
Belorussia	453	75
Ukraine	2538	1460
RSU	5607	269
Total	25342	13485

According to the trade vector, Finland and Portugal are not allowed to trade. The emission according to the SO_2 Protocol equals the cost-effective emission for these countries. Countries that should sell permits are Bulgaria, Czechoslovakia, Hungary, Italy, Poland, the United Kingdom and Belorussia. That is, these countries should further reduce their emissions. The Trade Coordinating Institute is allowed to sell 452 emission permits. The remaining 16 countries are allowed to buy permits.

Table 6.15 Variant 2. Overview of the cost-effective emission allocation, the initial emission distribution according to the SO_2 Protocol and the trade vector (kt SO_2).

Country	Cost-Effective Emissions	Initial Emission Distribution	Trade Vector (buyers +; sellers -)
Austria	106	78	28
Belgium	250	248	2
Denmark	98	90	8
West Germany	1025	1019	6
East Germany	391	281	110
Greece	894	595	299
Ireland	169	155	14
Luxembourg	14	10	4
Netherlands	123	106	17
Norway	41	34	7
Spain	2148	2143	5
Sweden	110	100	10
Switzerland	66	60	6
Yugoslavia	1398	1378	20
Ukraine	2538	2310	228
RSU	5607	4440	1167
Bulgaria	794	1374	-580
Czechoslovakia	1346	1465	-119
France	810	868	-58
Hungary	521	898	-377
Italy	1244	1330	-86
Poland	2541	2583	-42
United Kingdom	2235	2449	-241
Belorussia	453	456	-3
Trade Coordinating Institute	-	452	-452
Finland	116	116	0
Portugal	304	304	0
Total	25342	25342	-

Once the initial distribution of emission permits and the trade vector are established, guided bilateral permit trading can be simulated. As in the Base Case and in the modification on the Base Case, the sequence of trade transactions is based upon the profitability of the transactions.

The effect on trade of the selling of permits by the Trade Coordinating Institute depends upon the price of these permits. First I will assume that the TCI does not set a price but will receives half of the reduction in abatement costs of a country purchasing permits form the TCI. This reduction in abatement costs equals the profit generated by permit trading. The sequence of profitable trade transactions resulting from guided bilateral permit trading is shown in Table 6.16. For each transaction the seller, the buyer, the traded quantity, the profit and the implicit permit price is represented.

Table 6.16 Variant 2A. Sequence of profitable trade transactions if the TCI receives half of the generated profit.

Seller	Buyer	Traded Quantity (kt. SO_2)	Profit (million DM)	Revenue TCI (million DM)	Implicit Permit Price (million DM/kt SO_2)
BUL	GDR	110	368	-	1.67
BUL	RSU	470	296	-	0.32
TCI	GRE	299	212	106	0.35
TCI	AUS	28	118	59	2.12
TCI	SWE	10	110	55	5.46
TCI	NET	17	80	40	2.36
TCI	NOR	7	76	38	5.45
TCI	UKR	91	56	28	0.31
HUN	DEN	8	28	-	2.27
HUN	LUX	4	23	-	3.39
HUN	FRG	6	18	-	2.05
HUN	YUG	20	16	-	0.94
HUN	UKR	137	13	-	0.57
HUN	IRE	14	10	-	0.88
HUN	SWI	6	7	-	1.02
HUN	BEL	2	4	-	1.64

Simulation results indicate that 16 profitable trade transactions will take place. In the first two, Bulgaria is involved. According to the cost functions of the RAINS model, Bulgaria is able to reduce emissions to 794 kt SO_2 at zero costs. Therefore Bulgaria is able to sell 580 permits at zero costs. Notably, the Trade Coordinating Institute sells all 452 permits. The total profit generated by permit trading amounts to 1.4 billion DM. The TCI receives 0.326 billion DM by selling permits. After all profitable trade transactions have taken place, the cost effective emission allocation is not fully reached.

Figures 6.11 and 6.12 illustrate the trade vector and the pre-trade marginal abatement costs as a graph. For trade to be profitable, the marginal abatement costs of the buyers have to exceed the marginal abatement costs of the sellers. The first buyer of emission permits, the former East Germany does not have the highest marginal abatement costs. But, since it is allowed to buy a substantial number of permits, the largest profit is generated. The buyer with the highest marginal costs, Luxembourg is only allowed to buy 4 emission permits and is therefore only the tenth buyer. Explaining the order of sellers is obvious. First the sellers with zero marginal costs sell permits. Then comes Hungary, the seller with the lowest marginal abatement costs. Since the marginal abatement costs of most buyers exceed the marginal abatement costs of the sellers, most of the trade vector is traded. After all profitable trades have taken place, only Spain and the Remaining Soviet Union should buy 5 and 697 emission permits respectively in order to reach the cost-effective solution.

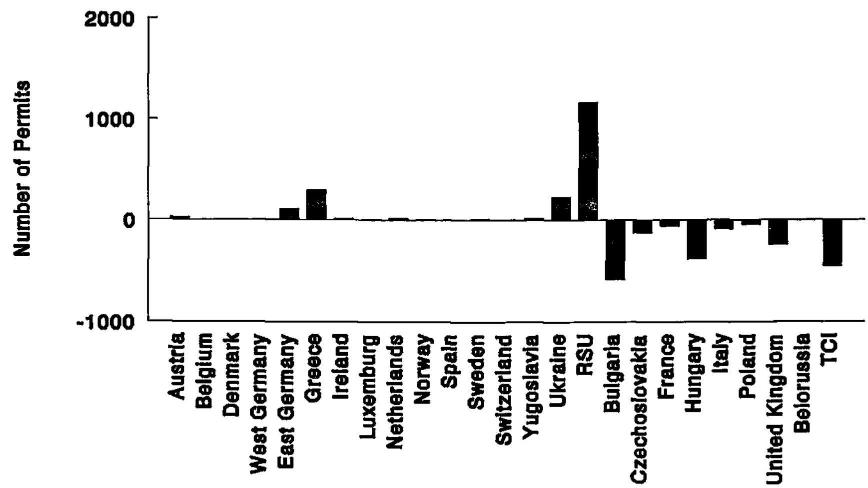

Figure 6.11 *Variant 2A. Initial trade vector (buyers +; sellers -).*

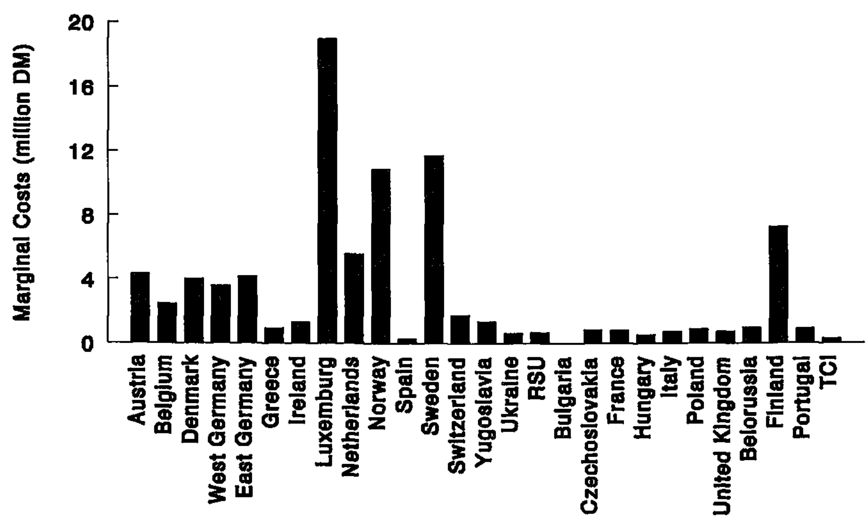

Figure 6.12 *Variant 2A. Marginal abatement costs in the pre-trade allocation.*

The implicit price per permit the TCI receives varies considerably. A more reasonable assumption is that the TCI will sell permits to every buyer at the same price. It is also reasonable to assume that the TCI wants to sell all permits since this minimizes the total abatement costs. The permit price that involves the sale of all permits depends upon the marginal abatement costs. If the TCI wants to be sure of selling all permits, the permit price should equal the lowest marginal cost in the cost-effective allocation. This marginal cost amounts to 0.29 million DM/kt SO_2. Table 6.17 shows the sequence of profitable trade transactions assuming this permit price.

Table 6.17 Variant 2B. Sequence of profitable trade transactions if the price of the permits sold by the TCI amounts to 0.29 million DM/kt SO_2.

Seller	Buyer	Traded Quantity (kt. SO_2)	Profit (million DM)	Revenue TCI (million DM)	Implicit Permit Price (million DM/kt SO_2)
BUL	GDR	110	368	-	1.67
BUL	RSU	470	245	-	0.26
TCI	GRE	267	135	77.43	0.29*
TCI	AUS	28	110	8.12	0.29*
TCI	SWE	10	107	2.9	0.29*
TCI	NET	17	75	4.93	0.29*
TCI	NOR	7	74	2.03	0.29*
TCI	UKR	123	41	35.67	0.29*
HUN	DEN	8	28	-	2.27
HUN	LUX	4	23	-	10.58
HUN	FRG	6	18	-	2.04
HUN	YUG	20	16	-	0.94
HUN	UKR	105	10	-	0.57
HUN	IRE	14	10	-	0.89
HUN	SWI	6	7	-	1.07
HUN	BEL	2	4	-	1.49

* fixed price

By assuming that the TCI sells permits at a price of 0.29 million DM, the total profit generated by trade decreases and amounts to 1.27 billion DM. The TCI sells all 452 permits. The revenue of the TCI decreases to 131 million DM. The number of permits that is traded has only decreased by 32.

An interesting question that arose was to what extent the generated profit depends upon the assumed trade sequence. To answer this question the model was adjusted to simulate permit trading with an arbitrary trade sequence. This new model was then used to run 2500 simulations in which the trade sequence was randomly chosen. The simulation results indicate that the average profit generated amounts to 1.19 billion DM. The smallest profit amounts to 0.97 billion and the largest profit amounts to 1.26 billion DM. Figure 6.13 shows the distribution of the generated profits for these 2500 simulations. The generated profit shows a fair stability. Only in one trade sequence does the profit generated amount to less than 1 billion DM. Most trade transactions generate a profit of between 1.15 and 1.25 billion DM, which indicates that the generated profit is not heavily sensitive to the trade sequence.

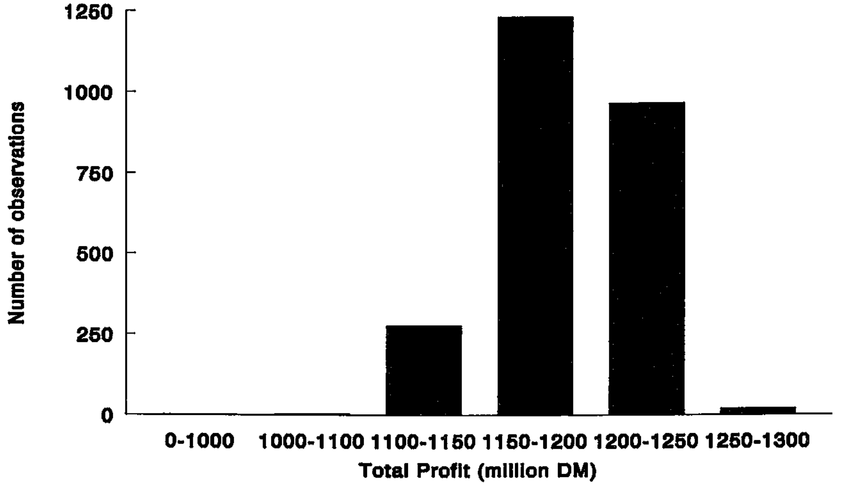

Figure 6.13 *Overview of generated profits caused by guided bilateral trade for 2500 different trade sequences.*

6.6 Conclusions

In this chapter I have discussed simulated guided bilateral permit trading among European countries for SO_2 emissions. Since SO_2 is a non-uniformly mixing pollutant, permit trading has to be guided to take care of deposition targets. Obviously, this guiding limits trade opportunities and makes the system more complex than that of free trade in emission permits. The system of guided bilateral trade aims at reaching the cost effective emission allocation. For this purpose, the number of initially issued permits has to correspond to the emissions in the cost-effective abatement allocation. To be able to reach the cost effective emission allocation by permit trading, trade has to be restricted according to the trade vector that indicates the number of permits countries are allowed to buy or sell.

Guided bilateral permit trading was simulated for three variants. In the policy-oriented Base Case, permit trading aimed at reaching the cost-effective emission allocation, given the Current Reduction Plans of the European countries. In Variant 1, the restriction of Current Reduction Plans was released. Variant 2 examined guided bilateral permit trading starting from the SO_2 Protocol. The simulation results indicate that permit trading steers the trade process towards the cost-effective allocation of emission abatement, but the cost-effective allocation of emission abatement will not be fully reached by profitable trade transactions. In all three variants, the profit generated by guided bilateral permit trading is substantial. In the Base Case and in Variant 1, the cost saving of permit trading amounts to 13.5% of the total abatement costs. By excluding the Current Reduction Plans in Variant 1 the total abatement costs decreased but, since guided bilateral trade allows more initial emissions, this decrease in abatement costs occurred initially and the profit by permit trading did not increase.

The simulation result of Variant 2 indicates that guided bilateral permit trading starting from the emission allocation in the SO_2 Protocol results is a cost saving of 7.8 % of total abatement costs. Given this result, I conclude that the SO_2 Protocol does not result in highly excessive abatement costs, when given the deposition target, but that some cost-effective improvement is possible by guided bilateral permit trading.

Finally, the simulation results indicate that the trade sequence influences the generated profit. However, the simulations also show that the impact of the trade sequences on the profit level is small. As a result the trade coordinating institute only has to control whether countries trade according to the trade vector. No influencing is necessary to generate a certain trade sequence, since this has a minor impact on total profits.

Having illustrated the consequences of guided bilateral permit trading, the question arises whether this system should be introduced in practice. This question can be answered by using the criteria to judge on policy instruments as introduced in Chapter 2. As far as economic efficiency and environmental effectiveness is concerned, guided bilateral trade is a suitable instrument because deposition targets are hardly exceeded and emission reduction takes place at minimum abatement costs. Other arguments in favour of guided bilateral permit trading are that the use of permits provides a continuous incentive to search for technology innovation and provides flexibility the way polluters want to comply with the environmental targets. Moreover, the system of guided bilateral permit trading can be introduced in accordance with the Second SO_2 Protocol, since this protocol provides the opportunity for two or more parties to implement the emission reduction obligations together. This opens up the opportunity for one party to decrease its emission reductions if another party increases it reductions.

This thesis only takes the costs of emission reduction into account. It is difficult to estimate the cost of implementation of a guided bilateral permit trading system. There is no indication that the costs of this system will be substantially higher than the costs of other policy instruments, when non-violation of deposition targets is taken into account. Since the trade restrictions of guided bilateral permit trading are clear and rather simple to the polluters, the mode of operation of this system seems not very complex. The remaining criteria, integration in sectoral policies, distributional effects and political acceptability are difficult to judge without specific research on these criteria. In conclusion, on the basis of the applied criteria, guided bilateral permit trading seems to offer opportunities for cost savings and practical implementation.

7 SUMMARY AND CONCLUSIONS

7.1 Introduction

This study has examined the question of whether a system of tradeable emission permits can contribute to a cost-effective reduction of SO_2 emissions in Europe, taking into account prespecified deposition targets and what sort of system it should be. To deal with this question three research topics have been examined.

(I) What are the main economic aspects of a European acid rain policy?

(II) What are the advantages and the disadvantages of using a system of tradeable permits to implement a European acid rain policy?

(III) What should a system of tradeable emission permits for a non-uniformly mixing pollutant look like in order to take deposition targets into account?

This chapter summarizes and presents the conclusions of this research. First I present the conclusion on the efficacy of the different integrated assessment models in analysing the effects of emission reduction. I then summarize the conclusions on guided bilateral permit trading. Finally, I make several suggestions for further research.

7.2 Economic theory of pollution control

In this research I have followed the ecological approach to pollution control. This approach does not aim at optimal pollution control levels where marginal abatement costs equal marginal benefits. Marginal benefits are usually too difficult to determine. Instead it aims at cost-effective abatement of pollutants to reach prespecified environmental targets. Since acid rain is an international environmental problem, cost-effective abatement needs to be considered in an international context. To achieve cost-effective international abatement it is necessary to have cooperation among countries. Although full cooperation results in minimized total costs for the cooperating countries, individual countries may, however, not act accordingly. This has been explained by the existence of the prisoner's dilemma. However, the assumptions underlying the prisoner's dilemma do not always hold in reality, and in the real world, countries do have some incentives to cooperate. Moreover, side payments can be introduced to stimulate countries to participate in cooperative abatement strategies.

Another issue I discussed is which policy instruments can be used to implement pollution control policy. A main distinction can be made between economic and non-economic instruments (regulation). A main advantage of economic instruments is their cost-effective character because, unlike regulation, economic instruments result in minimum control costs. Moreover, they allow more flexibility to the polluters and provide a continuous incentive for developing and adapting new control technologies. This thesis has elaborated on the use of tradeable emission permits.

7.3 Tradeable permits for non-uniformly mixing pollutants

Environmental degradation by non-uniformly mixing pollutants depends not only on the amount of emission, but also on the location of that emission. Accordingly, the environmental aim I followed in this study was not to reach an emission target, but rather to reach deposition targets. This has main consequences for using a system of tradeable emission permits as this system, in principle, only regulates emissions. It does not take the resulting depositions into account. The implications of permit trading for non-uniformly mixing pollutants have been illustrated and the alternatives for taking emission location into account in permit trading, that were put forward in the literature were discussed in Chapter 3.

It cannot be concluded in general that emission permits are a suitable instrument for controlling non-uniformly mixing pollutants. The use of emission permits results in a cost-minimum reduction of emissions, but the environmental implications of this emission reduction are uncertain, as deposition may be unacceptably large at some locations. To what extent the resulting cost-effective emission allocation deviates from the cost-effective deposition allocation depends on the source-receptor relationships and the marginal control costs of controlling the sources. In the USA, however, tradeable emission permits are actually used for the non-uniformly mixing pollutant SO_2. Nevertheless, one should be aware that this emission trading does not focus on reaching a deposition target, but is only emission oriented. Environmental targets are taken care of by State Implementation Plans in the USA.

Deposition permits are an obvious alternative to emission permits for taking emission location into account. Theoretically, the use of tradeable deposition permits results in a cost-effective allocation that meets deposition targets. According to this system, sources have to own the appropriate number of deposition permits for each receptor affected. In this way authorities can control deposition targets. However, a disadvantage of this system is that it may result in an increase in emissions, which is politically delicate. Another argument against this system is the trading complexity that occurs because sources have to operate simultaneously in many different markets.

Since the use of both emission and deposition permits is not considered to be a very suitable instrument, several alternatives permit systems for non-uniformly mixing pollutants have been proposed in the literature. Relatively simple alternatives are permit trading within zones and the 'worst case' approach, but unfortunately, these systems do not guarantee the non-violation of deposition targets.

A promising permit trading system that does not violate deposition targets seems to be a more advanced one where the trading is subject to rules. Three variants are known: the pollution offset, the non-degradation offset and the modified pollution offset. All systems guarantee non-violation of deposition targets. The constraints on emissions and on deposition targets, however, differ between these systems. The pollution offset offers the largest trade opportunities, but allows for an increase in emissions. The non-degradation offset and the modified pollution offset are more restricting and consequently result in higher abatement costs. Which of the two has the lower cost cannot be shown on theoretical grounds.

In judging permit trading systems, special attention has to be paid to the nature of the trading process. Although many studies implicitly assume a simultaneous trading process, it seems more reasonable to consider permit trading as a bilateral and sequential process. Combining permit trading subject to rules with the bilateral and sequential trading concept sheds new light on the cost-effectiveness of these permit trading systems. Empirical studies indicate that bilateral permit trading applying offset rules do not result in cost-effective abatement. However, bilateral and sequential trading has only been recently applied to studies on permit trading, and has been examined further in this study. A main conclusion is that permit trading for non-uniformly mixing pollutants has to be restricted in some way if deposition targets are to be taken into account. What is needed are restrictions that result in non-violation of deposition targets, and at the same time, do not hamper the attainment of the cost-effective allocation.

7.4 Integrated assessment models

Integrated assessment models can be used for quantifying the consequences of abatement allocations. These models provide a scientific basis for evaluating alternative abatement allocations and are therefore a useful tool in policy making. Three integrated assessment models are available for Europe : (i) the Regional Acidification INformation and Simulation model (RAINS); (ii) the Abatement Strategies Assessment Model (ASAM) and (iii) the Coordinated Abatement Strategy Model (CASM). Integrated assessment models provide information on emissions, abatement costs, the atmospheric transport of emissions and the environmental effects of alternative abatement strategies. Since accurate monetarization of environmental damage is very difficult and complex in practice, the environmental damage is expressed in physical terms. For acidification, the environmental damage is expressed in exceedance of critical loads. Given these targets, a cost-effective emission abatement policy can be found. This makes these models very useful for policy analysis.

Among the European integrated assessment models, the RAINS model is the most complete and extensive. Although the RAINS model, the ASAM and the CASM are roughly similar, they diverge on various points as well. One difference is the number of acidifying compounds they take into account. Other differences are the optimization options that are implemented and the accuracy of the emission location. Finally, the data sources used differ in some submodels. The main benefit of the ASAM is its accuracy on emission locations and the resulting atmospheric transport. However, because of lack of data for this purpose, the intended accuracy is only partial. Model results show no significant differences caused by the grid to grid approach used. A serious drawback of this model is the lack of clarity of the procedure needed to derive the so called Best Economic

Environmental Pathway. The CASM is attractive because of its large number of optimization criteria. However, since policy aims at reaching certain deposition targets, optimization criteria to minimize environmental damage, given a budget constraint are not actually very useful for policy purposes. Comparison of the CASM and the RAINS model shows that the model results are broadly in line with each other.

There are two reasons for the current status of acceptance of the RAINS model. First, the model is the most completely integrated assessment model for acidification. The alternative integrated assessment models, the CASM and the ASAM do not substantially improve the RAINS model. Second, RAINS was developed by scientists of various disciplines at an international "East-West" institute. This aspect mainly played a role in the acceptance of RAINS in international policy making (Hordijk, 1991, 1995). Acid rain models have substantially increased knowledge of the acid rain problem and have provided useful information for assisting and improving policy making. In this research I used the RAINS model for calculations. The reasons for this choice were those given above: (i) the model is the most complete one for acidification and (ii) the model is widely accepted and was used in the international negotiations on the Second Sulphur Protocol.

7.5 Guided bilateral trade

7.5.1 Theory

Research has shown that in case of non-uniformly mixing pollutants, permit trading has to be constrained if a cost-effective emission allocation is to be attained. It has been argued that if permit trading is constrained, it is necessary to consider permit trading as a bilateral process. In Chapter 5 I developed a new system for bilateral permit trading describing it as guided bilateral trade. The aim of this system is to provide large trade opportunities while at the same time preventing the violation of deposition targets. However, it is not necessary to require that every trade transaction meets the deposition targets. A subsequent trade transaction might compensate for the deposition exceedance at a receptor brought about by a previous trade transaction. To be able to take care of the deposition targets, the guided bilateral trade system restricts trading by a trade vector which indicates the number of permits sources are allowed to sell or buy.

Guided bilateral permit trading achieves a cost-effective allocation if all trade transactions that are in accordance with the trade vector are profitable. A drawback of guided bilateral permit trading is that trade transactions are not necessarily profitable. Whether this is so depends on the source-receptor relationships, the abatement cost functions, the deposition targets and the initial permit distribution. To reach the deposition targets ultimately, non-profitable trade transactions need to be compensated in some way. Before trade transactions can be guided to the cost- effective emission allocation, this allocation has to be known. Therefore full information is required by the trade institution in order to establish the trade vector. Unfortunately, this cancels out one of the characteristics of tradeable permits, namely that full information is not needed. Knowledge on the cost-effective emission allocation by policy makers might cause the guided bilateral permit system to become unnecessary. However, knowing this allocation does not by definition imply that it will be agreed upon at once, as can be seen from the Second Sulphur Protocol. In that case, a suitable instrument for achieving further cost savings is a system of guided bilateral trade.

The term guided bilateral trade indicates that the permit market as such is not able to generate the desired outcome. The trade process has to be guided by a trade institution that takes care of the number of traded permits. However, any trading system that takes deposition targets into account will need some kind of control. The main advantage of the guided bilateral trade system is that the cost-effective emission abatement allocation can be achieved if all trade transactions are profitable, or, if not, when non profitable transactions are subsidized. In the past, achieving the cost-effective allocation was thought to be possible by means of a simultaneous trade process. However, as I indicated in section 5.3, the practical functioning of such a trade system is a black box. This box can be opened by means of the system of guided bilateral trade.

7.5.2 Simulation results

To examine to what extent the guided bilateral trade system results in a cost-effective reduction of SO_2 emissions in Europe, taking into account deposition targets, I used the RAINS model to simulate guided bilateral permit trading among European countries for SO_2 emissions. In calculating the cost-effective emission allocation, emission is measured at national level. Accordingly, guided bilateral permit trading only allocates emission reduction between countries. The way in which countries meet their emission reduction is a next step. For the sake of completeness it should be noted that the European abatement allocation is only cost-effective if all individual countries also reduce their national emissions in a cost minimum way. I simulated guided bilateral permit trading for three cases. In the policy oriented Base Case, permit trading aimed at the cost-effective emission allocation, given the Current Reduction Plans of the European countries. In variant 1 the restriction of Current Reduction Plans was released. Variant 2 examined guided bilateral permit trading, starting from the SO_2 Protocol.

The simulation results indicated that permit trading steers the trade process towards the cost-effective allocation of emission abatement, but the cost-effective allocation of emission abatement will not be fully reached by profitable trade transactions. In all three cases, the profit generated by guided bilateral permit trading was substantial. In the Base Case and in variant 1, the cost saving of permit trading amounted to 13.5% of the total abatement costs. From a policy point of view, variant 2 is very interesting since it draws heavily on the actual situation in Europe. The simulation result of Variant 2 indicated that guided bilateral permit trading, starting from the emission allocation in the Second SO_2 Protocol, results is a cost saving of 7.8% of the total abatement costs. Given this result, we may conclude that, compared to the SO_2 Protocol, some cost-effective improvement is possible by guided bilateral permit trading. However, it should be noted that the simulation results excluded transaction costs. Therefore, the indicated cost savings can be regarded as the upper limit of possible savings.

The simulation results indicated that the trade sequence influenced the generated profit. However, the simulations showed that the impact of trade sequences on the profit level was small. This suggests that a trade coordinating institute does not have to play an active role in the matching of trade partners, but only has to control whether countries trade according to the trade vector.

Having discussed the main aspects of this study I now want to turn to the third research question. By developing the guided bilateral permit trading system I have shown how a permit trading system for non-uniformly mixing pollutants could be formulated that takes deposition targets into account. The main drawback of this system is that full information on the cost-effective allocation is needed by the environmental authorities if such a trading system is to be successfully implemented. But, it should also be realized that in the current practice this information has already been used: in the negotiations on the Second Sulphur Protocol the information on the cost-effective emission allocation provided by the RAINS model served as a guideline for the agreed emissions reduction. Since countries agreed upon the emission reductions in the Second Sulphur Protocol, they might agree on a permit trading system that is based on the same information.

7.6 Guided bilateral trade revisited

In this study I have been extensively illustrated that aiming at deposition targets for acidification while minimizing abatement costs is a complex matter. It depends on three factors: the source receptor relationships, the deposition targets and the differences in marginal abatement costs. Obviously, a policy instrument that is designed to generate a cost-effective abatement allocation needs to deal with this complexity. After having analysed the system of guided bilateral trade for SO_2 emissions in Europe thoroughly, the final question that has to be looked at is whether this system would be suitable for implementation in practice and how it could be introduced. This question is discussed in this section, paying attention to cost effectiveness, innovation, international agreements, implementation costs, distributional effects and practical implementation.

The system of guided bilateral permit trading succeeds in reaching the *cost-effective* abatement allocation if all allowed trade transactions are profitable. If not, the cost-effective allocation will only be fully reached if the trade transactions needed to reach this cost-effective allocation are subsidized. This will be difficult since it is necessary to prune away profits from profitable trades to generate funds for subsidizing. However, the simulation results of guided bilateral permit trading indicate that the violation of deposition targets is only very moderate if the cost-effective allocation is not fully reached by guided bilateral permit trading. Therefore it can be stated that guided bilateral permit trading is sufficient for the cost-effectiveness criterion. Arguments generally favouring the use of tradeable emission permits are that the use of permits provides a continuous incentive to search for technology *innovation* and provides flexibility to polluters in the way they want to comply with the environmental targets. Guided bilateral trade provides this incentive and flexibility as well.

An important aspect in judging the suitability of guided bilateral permit trading is the conformity with *international agreements*. In this thesis I have shown that given the Second Sulphur Protocol, guided bilateral permit trading can improve the cost-effectiveness of emission abatement. Whether a system of guided bilateral permit trading will really be successful will largely depend on the willingness of the European countries to accept the specified trade rules and the initial distribution of permits. A disadvantage for negotiating countries may be that emission permits, by definition, put a price on pollution that was previously free. To make permits acceptable, countries first have to receive permits for free. In this thesis it was suggested that the agreed emission levels from the Second Sulphur Protocol could serve as a guideline for the initial distribution of permits, since countries have already committed themselves to these amounts. In fact, given the current sulphur protocol, the guided bilateral permit trading system provides a 'recipe' for reaching the cost-effective sulphur dioxide allocation in Europe. Since the sulphur protocol is an agreement among national governments, permit trading among these governments is an obvious succession. However, alternatively, a system could be implemented in which permit trading could take place among firms (see section 7.7).

The *costs of implementation* of a guided bilateral permit trading system have not been studied in this research and it is therefore difficult to judge the system on this criterion. However, compared to the current international practice of negotiations, there seems to be no indication so far that the costs of this trading system will be substantially higher. Neither the *distributional aspects* of guided bilateral permit trading have been dealt with extensively. In general, no country will be worse off with permit trading, since a country will only trade permits if this results in a profit, resulting in a Pareto improvement. A country selling permits will only sell permits if the permit price exceeds the marginal abatement costs, and a country will only buy permits if this is cheaper than emission abatement. So, in so far as the abatement costs of countries change as a result of permit trading, this change is always compensated by the buying or selling of permits. Therefore, guided bilateral permit trading will only result in very moderate distributional effects.

The most obvious way for *practical implementation* of guided bilateral permit trading is to extend the Second Sulphur Protocol with a guided bilateral permit trading scheme in an additional annex, in accordance with annex II in which the current committed emission reductions are stated. Currently, countries are obliged to report the annual levels of sulphur emissions to the Executive Body of the Convention each year. Likewise, countries could be obliged to report their trade transactions annually. I suggest that it is better to include a trade scheme in the protocol rather than the trade vector, since a trade scheme as presented in Table 6.15 or 6.16 results in the largest cost savings. Furthermore, by presenting a trade scheme, countries do not have to search for their trading partner. This form of implementation does not require the establishment of a new agency since it fits into the current international negotiation arrangement on acidification, through the Convention under the Economic Commission for Europe of the United Nations.

In this section I have indicated that guided bilateral permit trading principally offers a suitable supplement to the sulphur protocol for reaching a cost-effective emission allocation. I have also indicated how such a trading system could be implemented. Remembering that the simulation results showed that guided bilateral permit trading may generate a cost saving of 143 million DM, this system can be taken as a suitable policy instrument for reducing sulphur dioxide emissions. Whether it can really be successfully implemented largely depends on the political willingness of countries to accept such a trading scheme.

7.7 Suggestions for further research

Many aspects on tradeable emission permits have been examined in the literature. In this thesis I have emphasized the application of such permits to non-uniformly mixing pollutants for which a bilateral and sequential permit trading has been developed. This research could be extended to include topics such as banking of permits or market power. Additional suggestions for further research include (i) the aggregation in permit trading, (ii) the enlargement of permit trading to total acidification and its relation to other environmental problems, (iii) permit trading in relation to other environmental problems ensuing from common sources and (iv) the implementation of guided bilateral permit trading where there is no protocol.

(i) In this study I have assumed permit trading takes place among national governments. One justification for this is that a permit trading system has a fair chance of being accepted if it is close to the current multiparty negotiations that assign targets to nations, as in the sulphur protocols. An alternative would be that emission permits be distributed to and traded by firms. Arguments in favour of this are that national governments lack experience and lack knowledge about operations and compliance options at the firm level. The question is what guided bilateral permit trading would look like if trading took place among firms. Although the general methodology of guided bilateral permit trading needs no modification, if trading is taking place among firms rather than among countries, its implementation requires additional research. To simulate permit trading among firms, additional data are needed. The source receptor matrix has to be adapted since it is the location of firms that matters now. The source receptor matrix has to link the firm's emissions to deposition. Moreover, abatement costs have to be specified at the firm level. The current RAINS-Europe model does not contain these necessary data and refinements. However, the RAINS-Asia model, a policy-oriented model which provides a framework for integrated assessment of acid deposition in Asia, does already provide an analysis for large point sources. In this model, the emissions of 355 large point sources are linked to deposition at 1°x 1° grid cells (Foell et al., 1995).

Alternatively, permit trading between firms could be analysed by a 'two-stage method'. This method, which needs no modification of the source receptor matrix, consists of first selecting two trading countries and next of selecting the firms with the lowest and the highest marginal control costs within these countries. Obviously, the firm with the lowest marginal costs is selected for the country selling permits and the firm with the highest marginal costs for the country buying permits. The next trade would be between the firms with the second highest and the second lowest costs, and so on.

(ii) Acidification is not only caused by sulphur dioxide emission, but also by nitrogen oxides and ammonia. Instead of analysing trade in of sulphur permits, as

I have done in this thesis, it would be interesting to study the opportunities for the use of tradeable permits for total acidification. This multipollutant approach is in accordance with the ongoing negotiations on a following protocol on acidification. However, it requires additional research, including the following topics. Firstly, the three pollutants contribute in a different way to acidity. This would lead to possible substitution of emission reduction. Further, spatial patterns of emission and deposition differ, and abatement-cost functions show differences. These differences imply that permit trading for total acidification will be complex, but considering all acidifying emissions simultaneously may provide additional cost savings. An interesting topic to analyse in this context is how to allocate the emission reductions of sulphur dioxide and nitrogen oxides as the latter contribute both to acidification and to tropospheric ozone. The role of nitrogen oxides in acidification has been discussed in Chapter 1. Tropospheric ozone is formed by complex chemical reactions of volatile hydrocarbons (VOC) and nitrogen oxides. The ratio of VOC and NO_x, and not merely their total amounts, is important for ozone formation. This implies that when the ratio of VOC to NO_x is very small, reduction of NO_x can actually increase ozone formation. In general, the abatement of nitrogen oxide is more expensive than the abatement of sulphur dioxide. This implies that a shift from nitrogen oxides to sulphur dioxide emission reduction would result in cost savings. However, as nitrogen oxides contribute to tropospheric ozone, preventing high ozone concentrations might, in some cases, prohibit the reduction of nitrogen oxides. An additional restriction on NO_x emissions would be needed to take this into account in a permit trading system .

(iii) A third extension of guided bilateral permit trading for sulphur dioxide emission would be to link sulphur dioxide emission to climatic change. The rationale for this is that both problems have a common source, that is, energy use. In the RAINS model for Europe which was used for this study, only end-of-pipe technologies are available for reducing emissions. However, changes in energy use may be of benefit to both acidification and climatic change. To analyse the effect of changes in energy use for sulphur dioxide emissions the RAINS model should be extended with an accurate energy scenario module. The RAINS-Asia model is already equipped to generate energy scenarios. The so called energy scenario generator in this model estimates energy consumption contributing to SO_2 emissions (Foell et al., 1995).

(iv) A final topic for further research is how to implement a system of guided bilateral permit trading if a protocol had not yet been signed. Given an agreed initial permit allocation and given the restriction of the trade vector, it would be obvious to provide the opportunity to trade emission permits. Countries will have a strong incentive to trading since this is beneficial in terms of costs. If it appeared that countries would not trade, this could be an indication that transaction costs are too high. If transactions costs exceeded the indicated cost savings, permit trading would not take place, and countries would comply with the emission reduction without trading.

It has become clear that research on tradeable emission permits for sulphur dioxide emissions is an interesting topic that needs further investigation. Extending the research by the suggested options may provide new insights into how to achieve emission reductions at minimum abatement costs. Especially growing economies, such as those in Asia and Latin America, that will suffer from sulphur dioxide emissions and deposition in the near future, can benefit from a better insight into tradeable emission permits for reaching cost-effective reductions.

APPENDIX I

This appendix presents the simulation model for guided bilateral permit trading as formulated in the OMP modelling language.

```
*Model voor het berekenen van de volgorde van handel in SO2 emissie rechten
*
Scenario = basecase
*
*
MAX profit
*
SET = BUYER:      ALB AUS CZE GRE LUX NOR POR SPA SWI TUR YUG
                  S_PET BYE UKR MOL RSU
SET = SELLER:     BEL BUL DEN FIN FRA FRG_W FRG_E HUN IRE ITA
                  NET POL ROM SWE UK KOL BAL
SET = N: 1 TO 21
SET = TIME: T1 T2
*
REL = STEPS, SET = TAU.SELLER(&).N(&), DATA = TRAJECTS
REL = STEPB, SET = TAU.BUYER(&).N(&) , DATA = TRAJECTB
*
X=COST.BUYER(&)=C
X=COST.SELLER(&)=C
X=TAU.BUYER(&).N(&)=C
X=TAU.SELLER(&).N(&)=C
X=ABATEMENT.BUYER(&) = C >-INF
X=ABATEMENT.SELLER(&) = C  >-INF
X=TRADE.SELLER(&).BUYER(&).TIME(&) = C
X=HELP.SELLER(&).BUYER(&).TIME(1) = B
X=PROFIT=C >-INF $
X=SUMCOST = C
*
C= P = PROFIT = 22188.5 - COST.BUYER(S&) - COST.SELLER(S&)
```

```
C=SUMC = SUMCOST = COST.SELLER(S&) + COST.BUYER(S&)

C=AB.SELLER(&)= ABATEMENT.SELLER(&)=
      + /HS/ * TAU.SELLER(&).N(S&)

C=AB.BUYER(&) = ABATEMENT.BUYER(&) =
      + /HB/ * TAU.BUYER(&).N(S&)

C=C.SELLER(&) = COST.SELLER(&) =
      + /ZS/ * TAU.SELLER(&).N(S&)

C=C.BUYER(&) = COST.BUYER(&) =
      + /ZB/ * TAU.BUYER(&).N(S&)

C=HULP1.SELLER(&) = TAU.SELLER(&).N(S&) = 1

C=HULP2.BUYER(&) = TAU.BUYER(&).N(S&) = 1

C = A.SELLER(&) = ABATEMENT.SELLER(&) = /IN_EMS/ - /IN_EPS/
      + TRADE.SELLER(&).BUYER(S&).TIME(1)

C = A.BUYER(&) = ABATEMENT.BUYER(&) = /IN_EMB/ - /IN_EPB/
      - TRADE.SELLER(S&).BUYER(&).TIME(1)

C = MAX.BUYER(&) = TRADE.SELLER(S&).BUYER(&).TIME(S&) > /B/

C = MAX.SELLER(&) = TRADE.SELLER(&).BUYER(S&).TIME(S&) < /S/

C = HVW1.SELLER(&).BUYER(&).TIME(1) =
TRADE.SELLER(&).BUYER(&).TIME(1)
            < /MBS/ * HELP.SELLER(&).BUYER(&).TIME(1)

C = HVW2 = HELP.SELLER(S&).BUYER(S&).TIME(1) =1
```

```
*
DATA=IN_EMS,      FILE=INEMS0.WK1,   L=SELLER(&), C=EMISSIE
DATA=IN_EMB,      FILE=INEMB0.WK1,   L=BUYER(&),  C=EMISSIE
DATA=IN_EPS,      FILE=INEPS-V.WK1,  L=SELLER(&), C=RECHT
DATA=IN_EPB,      FILE=INEPB-V.WK1,  L=BUYER(&),  C=RECHT
DATA=S,           FILE=MVERK0.WK1,   L=SELLER(&), C=MVERK
DATA=B,           FILE=MKOOP0.WK1,   L=BUYER(&),  C=MKOOP
DATA=MBS,         FILE=MINBS0.WK1,   L=SELLER(&), C=BUYER(&)
DATA=HS,          FILE=HS0.WK1,      L=SELLER(&), C=N(&)
DATA=HB,          FILE=HB0.WK1,      L=BUYER(&),  C=N(&)
DATA=ZS,          FILE=ZS0.WK1,      L=SELLER(&), C=N(&)
DATA=ZB,          FILE=ZB0.WK1,      L=BUYER(&),  C=N(&)
DATA=TRAJECTS,    FILE=SELLER0.WK1,  L=SELLER(&), C=N(&)
DATA=TRAJECTB,    FILE=BUYER0.WK1,   L=BUYER(&),  C=N(&)
*
```

APPENDIX II

This appendix gives an overview of the maximum (no emission reduction) and minimum (maximum emission reduction) SO_2 emissions and the maximum reduction percentage according to the RAINS abatement cost functions based on the Official Energy Pathway 2000.

Country	Maximum Emissions (kton SO_2)	Minimum Emissions (kton SO_2)	Percentage Maximum Emission Reduction
Albania	138	32	77
Austria	352	43	88
Belgium	722	53	93
Bulgaria	816	71	91
Czechoslovakia	2346	631	73
Denmark	370	37	90
Finland	604	68	89
France	1309	117	91
Germany-West	2272	224	90
Germany-East	2363	226	90
Greece	906	73	92
Hungary	1076	369	66
Ireland	234	19	92
Italy	3001	171	94
Luxembourg	16	1	94
Netherlands	425	43	90
Norway	144	33	77
Poland	3739	499	87
Portugal	389	21	95
Romania	2595	338	87
Spain	2793	161	94
Sweden	412	84	80
Switserland	75	44	41
Turkey	2887	814	72
United Kingdom	3334	387	88
Yugoslavia	1577	160	90
Baltic Sea	74	18	76
North Sea	174	42	76
Atlantic Ocean	317	76	76
Kola-Karelia	728	103	86
St. Petersburg	334	34	90
Baltic Region	629	67	89
Belarus	567	48	92
Ukraine	5066	604	88
Moldavia	379	40	89
RSU	6348	703	89

APPENDIX III

This appendix gives an overview of the abbreviatons for country names as used in this study.

Country	Abbreviation
Albania	ALB
Austria	AUS
Belgium	BEL
Bulgaria	BUL
Czechoslovakia	CZE
Denmark	DEN
Finland	FIN
France	FRA
West Germany	FRG
East Germany	GDR
Greece	GRE
Hungary	HUN
Ireland	IRE
Italy	ITA
Luxembourg	LUX
Netherlands	NET
Norway	NOR
Poland	POL
Portugal	POR
Romania	ROM
Spain	SPA
Sweden	SWE
Switzerland	SWI
Turkey	TUR
United Kingdom	UK
Yugoslavia	YUG
Baltic Sea	BAS
North Sea	NOS
Atlantic Ocean	ATL
Kola Karelia	KOL
St. Petersburg	S-PET
Baltic Region	BAL
Belorussia	BYE
Ukraine	UKR
Moldavia	MOL
Remaining Soviet Union	RSU

REFERENCES

Alcamo, J., R.W. Shaw and L. Hordijk (eds.), 1990, *The RAINS model of acidification: science and strategies in Europe*, Kluwer Academic Publishers, Dordrecht.

Amann, M., 1989, *Potential and costs for control of NO_x emissions in Europe*, Status Report 89-1, IIASA, Laxenburg, Austria.

Amann, M., 1990, *Energy use, emissions and abatement costs.* in: Alcamo, J., R.W. Shaw and L. Hordijk (eds.), 1990.

Amann, M., 1990a, Recent and future development of emissions of nitrogen oxides in Europe, *Atmospheric Environment*, 24A (11), pp. 2759-2765.

Amann, M., I. Bertok, J. Cofala, G. Klaassen and W. Schöpp, 1992a, *Strategies for reducing sulphur dioxode emissions in Europe*, SR-92-08, IIASA, Laxenburg, Austria.

Amann, M., L. Hordijk, G. Klaassen, W. Schöpp and L. Sorensen, 1992, Economic restructuring in Eastern Europe and acid rain abatement strategies, *Energy Policy*, 20, pp. 1186-1197.

Amann, M., L. Hordijk, G. Klaassen, W. Schöpp, 1994, *Reduction of sulphur emissions in Europe: a combination of the 'best available technology' concept and the critical loads approach.* In: E.C. van Ierland (ed.), 1994.

Amann, M., G. Klaassen and W. Schöpp, 1993, *Closing the gap between the 1990 deposition and the critical sulphur deposition values*, Background paper prepared for the UN/ECE Task Force on Integrated Assessment Modelling, June 7-8, 1993, IIASA, Laxenburg, Austria.

Amann, M. and G. Kornai, 1987, *Cost functions for controlling SO_2 emissions in Europe*, Working Paper 87-065, IIASA, Laxenburg, Austria.

Amann, M. and W. Schöpp, 1993, *Reducing excess sulphur deposition in Europe by 60 percent. Further analysis of an international reduction scheme suggested as a basis for a new sulphur protocol in Europe.* Background paper prepared for the UN/ECE Working Group on Abatement Strategies, August 30 - September 2, 1993, IIASA, Laxenburg, Austria.

ApSimon, H.M., R.F. Warren and J.J.N. Wilson, 1994, The abatement strategies assessment model - ASAM: applications to reductions of sulphur dioxide emissions across Europe, *Atmospheric Environment*, 28 (4), pp. 649-664.

ApSimon, H.M. and J.N.N. Wilson, 1991, *Calculations with the ASAM-II model for the Task Force on Integrated Assessment Modelling*, Geneva 3-6 December 1991, Air Pollution Group, Imperial College, London.

Asman, W.A.H., 1990, *Ammonia emissions in Europe: updated emission and seasonal emission variation*, Report DMU Luft - A132, National Environmental Research Institute, Roskilde.

Atkinson, S.E. and T.H. Tietenberg, 1982, The empirical properties of two classes of designs for transferable discharge permit markets, *Journal of Environmental Economics and Management*, 9, pp. 101-121.

Atkinson, S.E. and T.H. Tietenberg, 1984, Approaches for reaching ambient standards in non-attainment areas, *Land Economics*, 60, pp. 148-159.

Atkinson S.E. and T.H. Tietenberg, 1987, Economic implications of emissions trading rules for local and regional pollutants, *Canadian Journal of Economics*, 2, pp. 370-386.

Atkinson S.E. and T.H. Tietenberg, 1991, Market failure in incentive based regulation: the case of emissions trading, *Journal of Environmental Economics and Management*, 21, pp. 17-31.

Barde, J-P., 1995, *Environmental policy and policy instruments.* In: Folmer, H., H. Landis Gabel and J.B. Opschoor (eds.), 1995.

Barrett, S., 1990, The problem of global environmental protection, *Oxford Review of Economic Policy*, 6 (1), pp. 68-79.

Barrett, S., 1991, *The paradox of international environmental agreements*, mimeo, London Business School.

Baumol, W.J. and W.E. Oates, 1988, *The theory of environmental policy*, second edition, Cambridge University Press, Cambridge.

Berdén, M., S.I. Nilsson, K. Rosén and G. Tyler, 1987, *Soil acidification; extent, causes and consequences*, report 3292, National Swedish Environmental Protection Board, Solna.

Bergman, L., H. Cesar and G. Klaassen, 1992, *Efficiency in transboundary pollution abatement: a scheme for sharing the costs of reducing sulphur emissions in Europe*. In: Krabbe, J.J. and W. Heijman (eds.), 1992.

Boadway R.W. and N. Bruce, 1984, *Welfare economics*, Basil Blackwell Ltd, Oxford.

Bohm, P. and B. Larsen, 1994, Fairness in a tradeable permit treaty for carbon emissions reductions in Europe and the former Soviet Union, *Environmental and Resource Economics*, 4, pp. 219 - 239.

Bohm, P. and C.S. Russell, 1985, *Comparative analysis of alternative policy instruments*. In: Kneese, A.V. and J.L. Sweeney (eds.), 1985.

Bolo, L., J. Alcamo, J. Bartnicki, K. Olendrzynski, 1991, *Heavy metals contamination in eastern europe: background load from the atmosphere*, WP-91-46, IIASA, Laxenburg, Austria.

Bovenberg, A.L., M.E.T. van de Broek, R.J. Mulder, 1991, *Instrumentkeuze in het milieubeleid*, Discussienota 9102, Ministerie van Economische Zaken, The Hague.

Bromley, D.W., 1991, *Environment and economy. Property rights and public policy*, Blackwell, Oxford UK and Cambridge USA.

Buijsman, E., H. Maas and W. Asman, 1985, *Ammonia Emissions in Europe*, Institute for Oceanography, State University Utrecht, Utrecht.

Burtraw, D., 1994, *Agency in international permit trading*. In: Klaassen, G. and F. R. Førsund (eds.), 1994.

Burtraw, D., 1996, Trading emissions to clean the air: exchanges few but savings many, *Resources*, 122, pp. 3-6.

Chadwick, M.J. and J.C.L. Kuylenstierna, 1992, Effects of acidic depositions on living organisms, *International Journal of Environment and Pollution*, 2 (1/2), pp. 23-42.

Chicago Board of Trade, 1995, *1995 EPA allowance auction results*, Chicago, Illinois.

Coase, R.H., 1960, The problem of social cost, *Journal of Law and Economics*, 3, pp. 1-44.

Collins, A., 1995, International pollution control: a review of marketable permits - a comment, *Journal of Environmental Management*, 43, pp. 185-188.

Cronshaw, M.B. and J.B. Kruse, 1994, *Regulated firms in polluting permit markets with banking*, discussion paper, University of Colorado, Boulder.

Dales, J.H., 1968, *Pollution, property and prices; an essay in policy making and economics*, University of Toronto Press.

Downing, R.J., J-P. Hettelingh and P.A.M. de Smet, 1993, *Calculation and mapping of critical loads in Europe: status report 1993*, RIVM report no. 25910-1003, Coordination Center for Effects, National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands.

ECE, 1985, *Air pollution across boundaries*, Economic Commission for Europe, United Nations, Geneva.

Eerden, L.J. van der, A.E.G. Tonneijck, J.H.M. Wijnands, 1987, *Economische schade door luchtverontreiniging aan de gewasteelt in Nederland*, Ministry of Public Housing, Physical Planning and Environmental Management, the Hague.

Eliassen, A., J. Saltbones, 1983, Modelling of long-range transport of sulphur over Europe: a two-year model run and some model experiments, *Atmospheric Environment*, 17, pp. 1457-1473.

Environmental Resources Limited, 1983, *Acid rain: a review of the phenomenon in the EEC & Europe*, Graham & Trotman, United Kingdom.

Erisman, J.W. and G.J. Heij, 1991, *Concentration and deposition of acidifying compounds*. In: Heij, G.J. and Schneider, T. (eds.), 1991, Final report second phase dutch priority programme on acidification, report no. 200-09, National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands.

Ermoliev, Y., G. Klaassen and A. Nentjes, 1996, Cost-effective ambient charges under incomplete information, *Journal of Environmental Economics and Management*, 31, pp. 37-48.

Foell, W., C. Green, M. Amann, S. Bhattacharya, G. Carmichael, M. Chadwick, T. Haugland, J.-P. Hettelingh, L. Hordijk, J. Shah, R. Shrestha, D. Streets, D. Zhao, 1995, Energy use, emissions, and air pollution reduction strategies in Asia, *Water, Air and Soil Pollution*, 85, pp. 2277 - 2282.

Folmer, H. and E.C. van Ierland (eds.), 1989, *Valuation methods and policy making in environmental economics*, Studies in Environmental Science 36, Elsevier, Amsterdam.

Folmer, H. and I. Musu, 1992, Transboundary pollution problems, environmental policy and international cooperation: an introduction, *Environmental and Resource Economics*, 2 (2), pp. 107-116.

Folmer, H., P. van Mouche and S. Ragland, 1993, Interconnected games and international environmental problems, *Environmental and Resource Economics*, 3 (4), pp. 313-335.

Folmer, H., H. Landis Gabel and J.B. Opschoor (eds.), 1995, *Principles of environmental and resource economics*, Edward Elgar, Brookfield, U.S.

Gough, C., P. Bailey and M.J. Chadwick, 1993, *Results from the Stockholm Environment Institute's Co-ordinated Abatement Strategy Model (CASM)*, report to the UN-ECE Working Group on Strategies, 24-28 May 1993, Geneva, The Stockholm Environment Institute, York.

Groot, R.S. de, 1992, *Functions of nature: evaluation of nature in environmental planning management and decision making*, Wolters-Noordhoff, Groningen.

Hahn R.W., 1984, Market power and transferable property rights, *Quarterly Journal of Economics*, 4, pp. 753-765.

Helm, D. and D.W. Pearce, 1990, Assessment: economic policy towards the environment, *Oxford Review of Economic Policy*, 6 (1), pp. 1-16.

Hettelingh, J-P., R.J. Downing and P.A.M. de Smet, 1991, *Mapping critical loads for Europe*, RIVM report no. 259109009, Coordination Center for Effects, National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Hoel, M., 1992, *How should international greenhouse gas agreements be designed?* In: P. Dasgupta, K-G. Mäler and A. Vercelli (eds.), 1992, The economics of transnational commons, Oxford University Press, Oxford.

Hordijk, L., 1991, Use of the RAINS model in acid rain negotiations in Europe, *Environment, Science & Technology*, 25 (4), pp. 569-602.

Hordijk, L., 1995, Integrated assessment models as a basis for air pollution negotiations, *Water, Air and Soil Pollution*, 85, pp. 249-260.

Howe, C.W., 1994, Taxes versus tradeable discharge permits: a review in the light of the US and European experience, *Environmental and Resource Economics*, 4, pp. 151-169.

Ierland, E.C. van, 1993, *Macroeconomic analysis of environmental policy*, Elsevier Scienctific Publishers, Amsterdam.

Ierland, E.C. van (ed.), 1994, *International environmental economics; theories, models and applications to climate change, international trade and the environment*, Developments in environmental economics - 4, Elsevier Science Publishers, Amsterdam.

Ierland, E.C. van (ed.), 1994a, Proceedings of the advanced research workshop *"The economics of atmospheric pollution: theories, applied models and implications for international policy making"*. Wageningen International Congres Centre, Wageningen, The Netherlands, November 16-18, 1994.

IMP, 1984, *Indicatief meerjarenprogramma lucht 1985 - 1989*, Ministerie van Volkshuisvesting Ruimtelijke Ordening en Milieubeheer, Leidschendam, The Netherlands.

IIASA, 1991, *Comparison of European integrated assessment models for evaluation of acid rain control policies*, final report to the Commission of the European Communities, Directorate-General for Environment, Nuclear Safety and Civil Protection, Laxenburg, Austria.

Iversen, T., N.E. Halvorsen, S. Mylona, H. Sandnes, 1991, *Calculated budgets for airborne acidifying compounds in Europe, 1985, 1987, 1988, 1989 and 1990*, The Norwegian Meteorological Institute, Oslo.

Kete, N., 1992, *The U.S. acid rain allowance trading system*. In: OECD, 1992, Climate change: designing a tradeable permit system, Organization for Economic Co-operation and Development, Paris.

Kete, N., 1994, *Air pollution control in the United States: a mixed portfolio approach*. In: Klaassen, G. and F.R. Førsund, 1994.

Klaassen, G., 1991, *Past and future emissions of ammonia in Europe*, Status Report 91-01, IIASA, Laxenburg, Austria.

Klaassen, G., 1994, Options and costs of controlling ammonia emissions in Europe, *European Review of Agricultural Economics*, 21 (2), pp. 219-240.

Klaassen, G., 1994a, *Trading sulphur emission permits in Europe using an exchange rate*. In: Ierland, E.C. van (ed.), 1994.

Klaassen, G., 1995, Trade-offs in sulfur emission trading in Europe, *Environmental and Resource Economics*, 5, pp. 191 - 219.

Klaassen, G., 1995a, *Trading sulphur emission reduction commitments in Europe: a theoretical and empirical analyses*, Ph.D. thesis, Free University, Amsterdam.

Klaassen, G. and M. Amann, 1992, *Trading of emission reduction commitments for sulphur dioxide in Europe*, Status Report 92-03, IIASA, Laxenburg.

Klaassen, G. and F.R. Førsund (eds.), 1994, *Economic Instruments for Air Pollution Control*, Economy and Environment, 9, IIASA, Kluwer Academic Publishers, Dordrecht.

Klaassen, G. and H.M.A. Jansen, 1989, *Economic principles for allocating the costs of reducing sulfur emissions in Europe*, Institute for Environmental Studies, Free University, Amsterdam.

Klaassen, G. and A. Nentjes, 1991, Macroeconomic impacts of an EEC policy to control air pollution, *Journal of Policy Modeling*, 13 (1), pp. 347-366.

Kneese, A.V. and J.L Sweeney (eds.), 1985, *Handbook of natural resource and energy economics*, vol. I, Elsevier Scientific Publishers, Amsterdam.

Komen, M.H.C. and H. Folmer, 1995, Vluchten bedrijven voor milieubeleid?, *Economisch Statistische Berichten*, 3997, pp. 148-152.

Krabbe, J.J. and W. Heijman (eds.), 1992, *National income and nature: externalities, growth and steady state*, Economy and Environment, 5, Kluwer Academic Publishers, Dordrecht.

Krupnick, A.J., W.E. Oates and E. de Verg, 1983, On marketable air pollution permits: the case for a system of pollution offsets, *Journal of Environmental Economics and Management*, 10, pp. 233-247.

Lübkert, B., 1987, *Development of a nitrogen oxide emission model at IIASA*, paper for presentation at the EURASAP workshop on methodologies for air pollutant emission inventories, Paris, June 29 - July 2, 1987.

Mäler, K-G., 1989, *The acid rain game.* in: Folmer, H. and E.C. van Ierland (eds.), 1989.

Mäler, K-G., 1990, International environmental problems, *Oxford Review of Economic Policy*, 6 (1), pp. 80-108.

Mäler, K-G., 1991, *Incentives in international environmental problems.* In: Siebert, H. (ed.), 1991.

McGartland, A.M., 1988, A comparison of two marketable discharge permits systems, *Journal of Environmental Economics and Management*, 15, pp. 35-44.

McGartland, A.M. and W.E. Oates, 1985, Marketable permits for the prevention of environmental deterioration, *Journal of Environmental Economics and Management*, 12, pp. 207-228.

Meister, A.D., 1990, *Environmental regulation and use of economic instruments for environmental planning and management: an overview.* Discussion paper in natural resource economics no.15, Massey University, Palmerstone North, New Zealand.

Montgomery, W.D., 1972, Markets in licenses and efficient pollution control programs, *Journal of Economic Theory*, 5, pp. 395-418.

Munro, A., N. Hanley, R. Faichney and J.S. Shortle, 1995, *Impediments to trade in markets for pollution permits*, paper presented to the European Association of Environmental and Resource Economists Conference, Umeå, 1995.

Musgrave, R.A and P.B. Musgrave, 1984, *Public finance in theory and practice*, 4th edition, McGraw-Hill, New York.

NAPAP, 1991, *The U.S. national acid precipitation assessment program: 1990 integrated assessment report*, Washington D.C.

Netusil, N.R. and J.B. Braden, 1995, *Efficiency gains in transferable permit markets: evidence on discrete, bilateral, sequential trades of sediment allowances*, research paper, Oregon/Illinois.

Nillson, J. and P. Grennfelt, 1988, *Critical loads for sulphur and nitrogen*, Miljørapport 1988: 15, Nordic Council of Ministers, Copenhagen, Denmark.

North, D.W., W.E. Balson and D.W. Boyd, 1985, *Acid deposition: a decision framework that includes uncertainty.* In: D.D. Adams and W.P. Page (eds.), 1985, Acid deposition. Environmental, economic and policy issues, Plenum Press, New York.

Opschoor, J.B, and J.B. Vos, 1989, *Economic instruments for environmental protection*, OECD, Paris.

Peeters, M., 1992, *Marktconform milieurecht? Een rechtsvergelijkende studie naar de verhandelbaarheid van vervuilingsrechten*, W.E.J. Tjeenk Willink, Zwolle, the Netherlands.

Posch, M., P.A.M. de Smet, J-P. Hettelingh, R.J. Downing (eds.), 1995, *Calculating and mapping of critical tresholds in Europe*, RIVM report no. 2591010-04, Coordination Centre for Effects, National Institute of Public Health and the Environment, Bilthoven, the Netherlands.

Rentz, O., H.D. Haasis, A. Jattke, P. Ruβ, M. Wietschel and M. Amann, 1992, *Impacts of energy supply structure on national emission reduction potentials and emission reduction costs*, Institute for Industrial Production, University of Karlsruhe and International Institute of Applied Systems Analysis, Laxenburg, Austria.

Rentz, O., H.D. Haasis, A. Jattke, C. Oder, M. Wietschel and M. Amann, 1993, *Strategies for reducing emissions and deposition in central and eastern European countries*, interim report, Institute for Industrial Production, University of Karlsruhe and International Institute of Applied Systems Analysis, Laxenburg, Austria.

Rentz, O., H.D. Haasis, A. Jattke, P. Ruβ, M. Wietschel and M. Amann, 1994, Influence of energy supply structure on emission reduction costs, *Energy*, 19 (6), pp. 641-675.

RIVM, 1989, *Concern for tomorrow*, National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands.

RIVM, 1991, *National environmental outlook 2: 1990-2010*, National Institute of Public Health and Environmental Protection, Bilthoven, the Netherlands.

Rose, A. and B. Stevens, 1993, The efficiency and equity of marketable permits for CO_2 emissions, *Resource and Energy Economics*, 15 (1), pp. 117-146.

Rosemarin, A., 1991, *An outline of the Stockholm Environment Institute's Coordinated Abatement Strategy Model*, Stockholm Environment Institute, Sweden.

Rubin, E.S., J.S. Salmento and H.C. Frey, 1988, Cost-effective emission controls for coal-fired power plants, *Chem. Eng. Comm.*, 74, pp. 155-167.

Siebert, H., 1987, *Economics of the environment, theory and policy*, second revised and enlarged edition, Springer Verlag, Berlin.

Siebert, H. (ed.), 1991, *Environmental scarcity: the international dimension.* Institüt für Weltwirtschaft an der Universität Kiel, Germany.

Streets, D.G., 1989, Integrated assessment: missing link in the acid rain debate?, *Environmental Management*, 13 (4), pp. 393-399.

Thomas, R., W.G. van Arkel, H-P. Baars, E.C. van Ierland, K.F. de Boer, E. Buijsman, T.J.H.M. Hutten, R.J. Swart, 1988, *Emissions of SO_2, NO_x, VOC and NH_3 in the Netherlands and Europe in the period 1950 - 2030*, RIVM, report nr. 758472002, Bilthoven, the Netherlands.

Tietenberg, T.H., 1978, Spatially differentiated air pollutant emission charges: an economic and legal analysis, *The American economic review land economics*, 54, pp. 265-277.

Tietenberg, T.H., 1985, *Emissions trading: an exercise in reforming pollution policy*, Resources for the Future, Washington D.C.

Tietenberg, T.H., 1995, Tradeable permits for pollution control when emission location matters: what have we learned?, *Environmental and Resource Economics*, 5, pp. 95-113.

Tschirart, J.T., 1984, *Transferable discharge permits and profit maximizing behaviour*, In: T.D. Crocker (ed.), 1984, Economic perspectives on acid deposition control, Butterworth Publishers, Boston MA.

Tuovinen, J-H., K. Barrett, K. and H. Styve, 1994, *Transboundary acidifying pollution in Europe: calculated fields and budgets 1985 -93*, EMEP/MSC-W report 1/94. Norwegian Meteorological Institute, Oslo.

UN, 1973, *Report of the UN conference on the human environment*, Stockholm, Sweden, 5-16 June 1972, United Nations, New York.

UN/ECE, 1994, *Protocol to the 1979 Convention on Long-range Transboundary Air Pollution on further reduction of sulphur emissions*, United Nations, Geneva.

Wetstone, G.S. and A. Rosencranz, 1983, *Acid rain in Europe and North America: national responses to an international problem.* Environmental Law Institute, Washington D.C.

Wüster, H., 1994, *Towards efficiency in international agreements: the new sulphur protocol.* In: E.C. van Ierland (ed.), 1994.

Young, J.W.S. and R.W. Shaw, 1986, A proposed strategy for reducing sulphate deposition in North America - I. Methodology for minimizing sulphur removal, *Atmospheric Environment*, 20, pp. 189 - 199.

Zhang, Z. and H. Folmer, 1995, The choice of policy instruments for the control of carbon dioxide emissions, *Intereconomics*, 30 (3), pp. 133-142.

Zylicz, T., 1993, *Improving environment through permit trading: the limits to market approach*, Beijer Discussion Paper Series No.23, Beijer International Institute of Ecological Economics, Stockholm, Sweden.

SAMENVATTING

Inleiding

Verzuring is een grensoverschrijdend milieuprobleem dat veroorzaakt wordt door de emissie van zwaveldioxide (SO_2), stikstofoxiden (NO_x) en ammoniak (NH_3). In deze studie beperken we ons tot SO_2. Voor een effectieve reductie van SO_2 is een internationaal reductiebeleid nodig. In Europa zijn reeds twee protocollen afgesloten waarin landen zich verplichten tot het terugdringen van SO_2 emissie. Het eerste protocol gaat uit van een uniforme emissiereductie; het tweede protocol is gericht op het bereiken van bepaalde depositiedoelstellingen. Dit huidige protocol impliceert echter geen kosteneffectieve emissiebestrijding.

Deze studie handelt over de vraag of een systeem van verhandelbare emissierechten kan bijdragen aan een kosteneffectieve bestrijding van zwaveldioxide emissie in Europa, gegeven bepaalde depositiedoelstellingen, en hoe zo'n systeem er uit zou moeten zien. Deze centrale vraag wordt behandeld aan de hand van drie onderzoeksonderwerpen:

(I) Wat zijn de belangrijkste economische aspecten van een Europees verzuringsbeleid?

(II) Wat zijn de voor- en nadelen van het gebruik van verhandelbare emissierechten om het Europese verzuringsbeleid gestalte te geven?

(III) Hoe zou een systeem van verhandelbare emissierechten voor niet uniform verspreide vervuiling er uit moeten zien, indien rekening wordt gehouden met gegeven depositiedoelstellingen?

Deze onderzoeksonderwerpen komen achtereenvolgens aan de orde.

Economische aspecten van vervuilingsbestrijding

Idealiter is het streven gericht op een optimale emissiebestrijding waarbij de marginale kosten van emissiebestrijding gelijk zijn aan de marginale baten daarvan. Immers deze optimale bestrijdingsomvang impliceert nutsmaximalisatie. Echter, omdat de marginale baten van emissiebestrijding in praktijk moeilijk zijn vast te stellen, wordt in deze studie uitgegaan van kosteneffectieve emissiebestrijding gegeven bepaalde depositiedoelstellingen. De kosteneffectieve bestrijdingsallocatie is gedefinieerd als die allocatie van emissiebestrijding waarbij gegeven milieudoelstellingen tegen de laagst mogelijke kosten worden gerealiseerd. Omdat verzuring een internationaal milieuprobleem is, moet kosteneffectieve emissiebestrijding ook in een internationaal kader worden beschouwd.

Om een internationale kosteneffectieve emissiereductie te bewerkstelligen, is samenwerking tussen landen onontbeerlijk. Hoewel volledige samenwerking tot een internationale kosteneffectieve emissiereductie kan leiden, handelen individuele landen echter niet altijd conform een coöperatieve bestrijdingsstrategie. Dit verschijnsel kan worden verklaard door het bestaan van 'prisoners dilemma' situaties. In de praktijk zijn de veronderstellingen die ten grondslag liggen aan het bestaan van het prisoners dilemma echter niet altijd aanwezig en bestaan er wel degelijk motieven voor landen om samen te werken. Bovendien kan gebruik worden gemaakt van side-payments om landen te stimuleren deel te nemen aan coöperatieve bestrijdingsstragieën.

Voorts wordt ingegaan op de vraag welke beleidsinstrumenten ingezet zouden kunnen worden om emissiereductie te realiseren. Een belangrijk onderscheid kan gemaakt worden tussen economische en niet-economische instrumenten (regulering). Het grootste voordeel van economische instrumenten is dat ze in theorie tot een kosteneffectieve bestrijding leiden. In tegenstelling tot regulering leidt het gebruik van economische instrumenten tot een minimalisatie van bestrijdingskosten. Bovendien bieden economische instrumenten meer flexibiliteit aan vervuilers en zorgen ze voor een voortdurende prikkel voor de ontwikkeling en toepassing van nieuwe bestrijdinstechnologieën. In deze studie wordt het gebruik van één van de economische instrumenten, namelijk verhandelbare emissierechten, voor de emissie van zwaveldioxide in Europa nader geanalyseerd.

Verhandelbare emissierechten voor niet uniform verspreide vervuiling

De aantasting van het milieu door niet uniform verspreide vervuiling, zoals zwaveldioxide, hangt niet alleen af van de *hoeveelheid* geëmitteerde vervuiling maar ook van de *locatie* van emissie. In overeenstemming hiermee is de milieudoelstelling die in deze studie gehanteerd wordt niet uitgedrukt in een emissiedoelstelling maar in een *depositiedoelstelling.* Dit heeft belangrijke gevolgen voor het gebruik van een systeem van verhandelbare emissierechten omdat een systeem van vrij verhandelbare emissierechten alleen de totale emissie van verontreinigende stoffen reguleert. De depositiedoelstellingen worden daarbij niet in beschouwing genomen. De implicaties van het gebruik van verhandelbare emissierechten voor niet uniform verspreide vervuiling en de alternatieven die zijn aangedragen in de literatuur om rekening te houden met depositiedoelstellingen, worden uitgebreid behandeld.

In het algemeen kan niet worden geconcludeerd dat emissierechten een geschikt instrument zijn voor het bestrijden van niet uniform verspreide vervuiling. Het gebruik van volledig vrij verhandelbare emissierechten leidt tot een emissiereductie tegen minimale bestrijdingskosten, maar de gevolgen voor de aantasting van het milieu zijn onzeker omdat de depositie op bepaalde locaties onaanvaardbaar hoog kan worden.

Om rekening te houden met depositiedoelstellingen zijn *depositierechten* een mogelijk alternatief voor emissierechten. In theorie leidt het gebruik van verhandelbare depositierechten tot een kosteneffectieve allocatie waarbij aan de depositiedoelstellingen wordt voldaan. Bij een systeem van verhandelbare depositierechten moeten emissiebronnen over de juiste hoeveelheid depositierechten beschikken voor iedere receptor die ze beïnvloeden. Op deze manier kan de overheid bewerkstelligen dat de depositiedoelstellingen worden bereikt. Een nadeel van verhandelbare depositierechten is dat de totale emissies in principe kunnen toenemen en dit is een politiek gevoelig aspect. Een ander nadeel dat in de literatuur naar voren wordt gehaald is dat handel in depositierechten complex is omdat bronnen simultaan in een groot aantal verschillende markten moeten opereren.

Omdat het gebruik van zowel emissie- als depositierechten op bezwaren stuit, zijn in de literatuur verschillende alternatieve systemen van verhandelbare rechten ontwikkeld. Het gemeenschappelijke kenmerk van deze systemen is dat de handel in emissierechten niet meer volledig vrij is. Relatief eenvoudige systemen zijn de "worst case" benadering en de situatie waarin de handel in emissierechten slechts in een bepaalde zones mag plaats vinden. Echter, deze systemen garanderen niet dat depositiedoelstellingen gehaald worden.

Complexere systemen van verhandelbare emissierechten, waarbij handel gebonden is aan regels, leiden niet tot een overschrijding van depositiedoelstellingen. Er bestaan drie varianten: (i) pollution offset, (ii) non degradation offset en (iii) modified pollution offset. De emissie- en depositierestricties in deze systemen variëren, maar al deze varianten garanderen het realiseren van depositiedoelstellingen. Het pollution offset systeem biedt de meeste handelsmogelijkheden doordat de totale emissie mag toenemen, mits aan de depositiedoelstellingen wordt voldaan. De handelsmogelijkheden in de non degradation en de modified pollution offset systemen zijn beperkter en deze systemen leiden daardoor tot hogere bestrijdingskosten.

Bij het beoordelen van systemen van verhandelbare emissierechten is het van belang om rekening te houden met de aard van het handelsproces. Veel studies veronderstellen impliciet dat handel in emissierechten een simultaan en multilateraal proces is. Echter, in afwezigheid van een veiling is het aannemelijker om te veronderstellen dat handel bilateraal en sequentieel plaats vindt. De resultaten van empirische studies geven aan dat bilaterale en sequentiële handel, waarbij handel aan regels gebonden is, niet tot kosteneffectieve emissiebestrijding leidt.

Een belangrijke conclusie is dat een systeem van verhandelbare emissierechten voor niet uniform verspreide emissie op één of andere manier gerestricteerd dient te worden om rekening te houden met depositiedoelstellingen. Deze restricties moeten depositiedoelstellinge garanderen maar tegelijkertijd het bereiken van de kosteneffectieve emissie allocatie niet in de weg staan.

Guided bilateral trade: theorie

In deze studie is een nieuw systeem van verhandelbare emissierechten ontwikkeld, genaamd "guided bilateral trade" dat ruime handelsmogelijkheden biedt waarbij depositiedoelstellingen niet overschreden worden. Een centrale gedachte in het systeem van guided bilateral trade is dat niet iedere bilaterale handelstransactie hoeft te voldoen aan de depositiedoelstellingen, maar dat moet worden voldaan aan de depositiedoelstellingen nadat alle bilaterale transacties hebben plaats gevonden. Immers, overschrijding op een willekeurige receptor kan door een volgende handelstransactie ongedaan worden gemaakt. Om rekening te houden met depositiedoelstellingen is de handel volgens het guided bilateral trade systeem gerestricteerd door een handelsvector die aangeeft hoeveel emissierechten emissiebronnen mogen aankopen dan wel mogen verkopen. Deze handelsvector is berekend als het verschil tussen de initieel aan landen toegekende rechten (emissie) en de hoeveelheid emissie in de kosteneffectieve emissieallocatie.

Guided bilateral trade leidt tot de kosteneffectieve allocatie indien alle handelstransacties die voortvloeien uit de handelsvector winstgevend zijn. Een nadeel van guided bilateral trade is dat deze handelstransacties niet per definitie winstgevend zijn. Of hiervan sprake is hangt af van de verspreidingsmatrix, de bestrijdingskosten functies, de depositiedoelstellingen en de initiële verdeling van emissierechten. Om de kosteneffectieve allocatie te bereiken, zouden in sommige gevallen verliesgevende handelstransacties op één of ander manier moeten worden gesubsidieerd.

Om de emissies naar de kosteneffectieve allocatie te leiden, moet deze allocatie bekend zijn. Daarom moet het instituut dat de handel coördineert over volledige informatie beschikken teneinde de handelsvector te kunnen vaststellen. De term "guided bilateral trade" geeft aan dat de markt als zodanig niet in staat is het gewenste resultaat te bewerkstelligen. Het handelsproces moet geleid worden door een instituut dat toeziet op de verhandelde hoeveelheid emissierechten.

Guided bilateral trade: simulatie

Om na te gaan in hoeverre guided bilateral trade tot een kosteneffectieve allocatie van SO_2 reductie in Europa leidt, is guided bilateral trade voor SO_2 emissies tussen Europese landen gesimuleerd. Voor het bepalen van de kosteneffectieve allocatie is gebruik gemaakt van het RAINS model. Om handel in emissierechten te simuleren is een simulatiemodel ontwikkeld.

Guided bilateral trade is gesimuleerd voor drie situaties. In de Base Case is de kosteneffectieve allocatie berekend rekening houdend met de huidige reductievoornemens van de Europese landen. De initiële verdeling van emissierechten is gebaseerd op historische emissies. In variant 1 is de restrictie betreffende de huidige reductievoornemens losgelaten. Variant 2 simuleert guided bilateral trade startende vanuit het tweede zwavelprotocol.

De simulatieresultaten tonen aan dat het handelsproces de emissieallocatie in de richting van de kosteneffectieve oplossing stuurt. De kosteneffectieve oplossing wordt echter niet volledig bereikt indien alleen de winstgevende handelstransacties plaats vinden.

In alle varianten wordt een substantiële kostenbesparing gerealiseerd. In de Base Case en in variant 1 bedraagt de kostenbesparing 13,5 % van de totale bestrijdingskosten. Vanuit beleidsoogpunt is variant 2 bijzonder interessant omdat deze variant de huidige afspraken omtrent emissiereducties in Europa als uitgangspunt neemt voor handel in emissierechten. De simulatieresultaten van variant 2 geven aan dat guided bilateral permit trading, startende vanuit de emissie allocatie van het tweede zwavelprotocol tot een kostenbesparing leidt van 7,8 % van de totale bestrijdingskosten. Gegeven deze uitkomst mogen we concluderen dat, vergeleken met het tweede zwavelprotocol, guided bilateral trade tot kosteneffectiviteitsverbetering leidt. Overigens moet worden opgemerkt dat transactiekosten in deze studie buiten beschouwing zijn gelaten.

Voor wat betreft de handelsvolgorde geven de simulatieresultaten aan dat de volgorde waarin de bilaterale handelstransacties plaats vinden invloed heeft op de gerealiseerde kostenbesparingen. Uit de simulaties blijkt echter dat dit effect gering is.

Het systeem van guided bilateral trade laat zien hoe een systeem van verhandelbare emissierechten voor niet uniform verspreide vervuiling, rekening houdend met depositiedoelstellingen, vorm kan worden gegeven. Een nadeel van het systeem is dat de coördinerende instantie volledige informatie nodig heeft voor het vaststellen van de kosteneffectieve emissie allocatie teneinde een systeem van guided bilateral trade succesvol te kunnen invoeren. Men moet zich echter realiseren dat bij de tot standkoming van het huidige zwavelprotocol, de kosteneffectieve allocatie, berekend met het RAINS model, ook al als een richtlijn heeft gediend.

Hoewel een systeem van 'guided bilateral trade' enige praktische complicaties kent, zou het grotere flexibiliteit opleveren en substantiële kostenbesparing mogelijk maken. Of een dergelijk systeem daadwerkelijk succesvol geïntroduceerd zou kunnen worden hangt in belangrijke mate af van de politieke bereidheid van landen om een dergelijk handelssysteem te accepteren.

De toepassing van verhandelbare emissierechten voor zwaveldioxide vormt een complex en breed aandachtsgebied. Het onderzoek hiernaar is met deze studie niet ten einde. Perspectieven voor verder onderzoek liggen onder andere op het terrein van het analyseren van verhandelbare emissierechten tussen bedrijven in plaats van tussen landen, het analyseren van verhandelbare emissierechten voor totale verzuring, de interactie van verhandelbare emissierechten voor zwaveldioxide met andere milieuproblemen zoals het broeikaseffect en het implementeren van guided bilateral trade in afwezigheid van een bestaande protocol.

CURRICULUM VITAE

Sonja Kruitwagen werd op 6 juli 1967 geboren te Heerlen. In 1985 behaalde zij aan het Eijkhagencollege te Schaesberg het diploma ongedeeld VWO. In september 1985 werd een start gemaakt met de studie Agrarische Economie aan de Landbouwuniversiteit Wageningen. In augustus 1990 sloot zij deze studie af. Het afstudeervak algemene en agrarische economie werd in 1990 onderscheiden met een scriptieprijs van het Nederlands Meststoffen Instituut.

Aansluitend aan haar afstuderen was zij tot december 1993 werkzaam als Onderzoeker in Opleiding. Deze aanstelling vond plaats in het kader van het NWO-project 'Bestrijding van grensoverschrijdende luchtverontreiniging als internationaal allocatievraagstuk'. De Landbouwuniversiteit heeft een vervolg aanstelling als assistent in opleiding gefinancierd. In de zomer van 1992 werd 3 maanden aan dit proefschrift gewerkt bij het International Institute of Applied System Analysis (IIASA) in Oostenrijk in het kader van het Young Scientist's Summer Program.

Sinds augustus 1994 is zij werkzaam als universitair docent bij de vakgroep Staathuishoudkunde op het gebied van de milieu-economie.