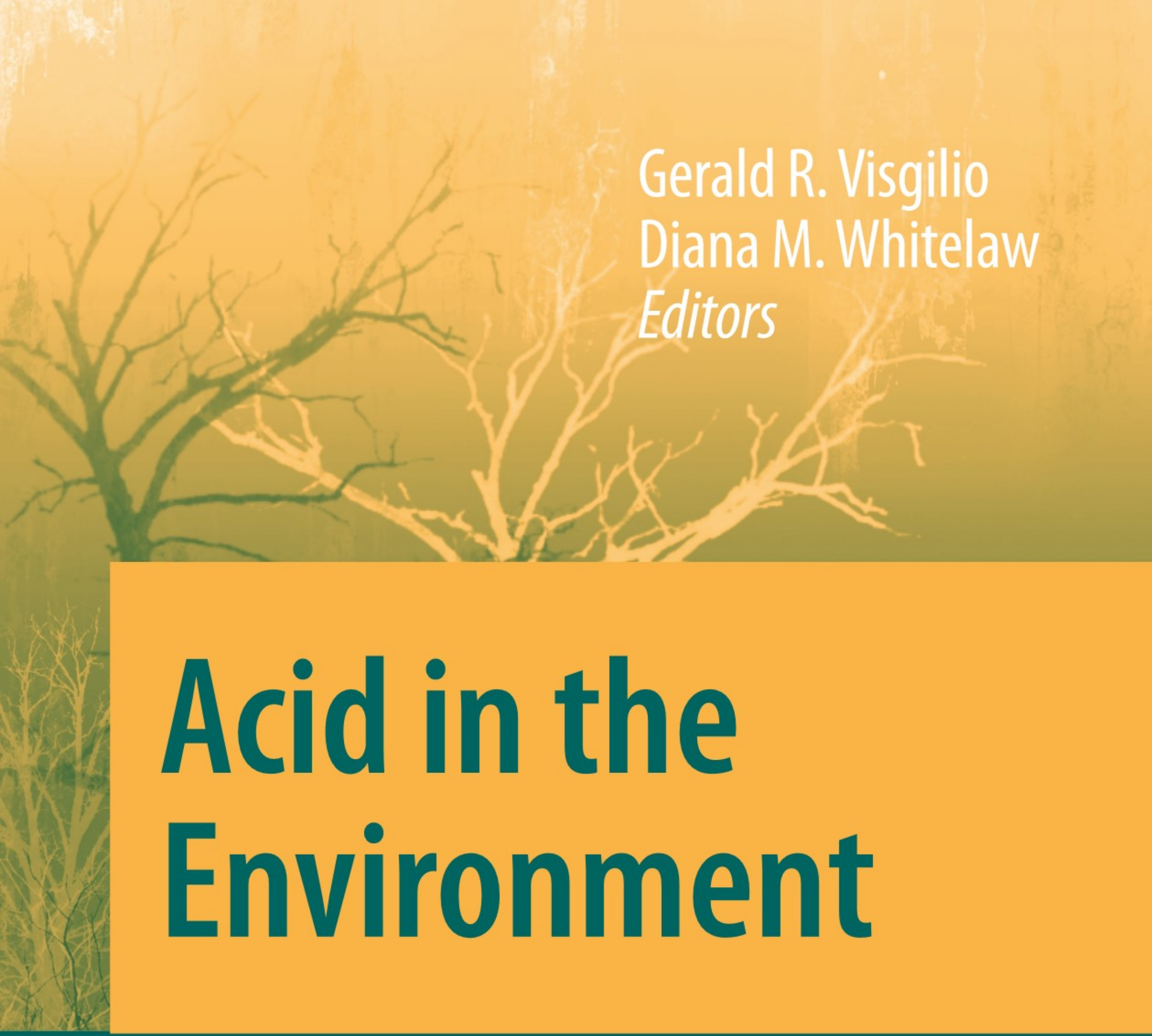

Gerald R. Visgilio
Diana M. Whitelaw
Editors

Acid in the Environment

Lessons Learned and Future Prospects

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Edited by

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and

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Preface

The Goodwin-Niering Center for Conservation Biology and Environmental Studies at Connecticut College is a comprehensive, interdisciplinary program that builds on one of the nation's leading undergraduate environmental studies programs. The Center fosters research, education and curriculum development aimed at understanding contemporary ecological challenges. One of the major goals of the Goodwin-Niering Center involves enhancing the understanding of both the College community and the general public with respect to ecological, political, social, and economic factors that affect natural resource use. To this end, the Center has offered five conferences at which academicians, representatives of federal and state government, and individuals from non-government environmental organizations are brought together for an in-depth, interdisciplinary evaluation of important environmental issues. On April 1 and 2, 2005, the Center presented the Elizabeth Babbott Conant interdisciplinary conference on *Acid in the Environment: Lessons Learned and Future Prospects.* The Connecticut Institute of Water Resources at the University of Connecticut, the Connecticut Chapter of The Nature Conservancy and the Connecticut Sea Grant College Program joined the Center as conference sponsors.

During the past twenty five years acid rain, formally referred to as acid deposition, has been the focus of much political debate and scholarly research. Acid deposition occurs when important precursor pollutants, such as sulfur dioxide (SO_2) and nitrogen oxides (NO_x), mix with water vapor and oxidants in the atmosphere and fall back to earth in either wet or dry form. Research has shown that acid deposition adversely affects fresh water lakes and streams, coastal habitats, agricultural production, building materials, forests, soils, and human health. Acid deposition is an environmental problem that crosses state and national boundaries, and is closely linked to energy policy since much of it originates as emissions from fossil-fuel power plants. A landmark in the evolution of international cooperation on the environment, the Convention on Long-Range Transboundary Air Pollution (LRTAP) came into force in 1983. In the United States, Title IV of the 1990 Clean Air Act Amendments established a "cap-and-trade" program to reduce SO_2 emissions to approximately half of their 1980 level. By allocating tradable SO_2 emission allowances to electric utilities, Title IV is designed to provide a cost effective approach to reducing SO_2 emissions.

The overall goals of this interdisciplinary conference were to summarize scientific and policy lessons learned from the attempt to mitigate acid deposition, and to discuss the future of transboundary pollutants and market-based emission control systems. Anthony Janetos, Vice President of the Heinz Center for Science, Economics and the Environment gave the keynote address providing the conferees with an historical perspective on lessons learned from the acid deposition research experience. In the evening address, Paul Portney, the former President and Senior Fellow of Resources for the Future discussed economical benefits and costs of air pollution control in the United States. The conference provided an opportunity for experts in the field to discuss important ecological impacts of acid deposition, the transboundary nature of pollutants that cause acid deposition, and domestic and international policies that are designed to reduce the emission of these pollutants. The audience included concerned citizens, NGO representatives and policymakers, and students and faculty from Connecticut College and other universities. This book, *Acid in the Environment: Lessons Learned and Future Prospects,* is based on the papers presented at the conference.

Acknowledgements

We greatly appreciate the financial support provided for the conference by the Connecticut Institute of Water Resources at the University of Connecticut, the Connecticut Chapter of The Nature Conservancy; the Connecticut Sea Grant College Program; the Marjorie Dilley Fund; the Beaver Brook Fund; the Connecticut College departments of Anthropology, Arboretum, Biology, Botany, Economics, Government, Sociology; the Environmental Studies Program; and the Offices of the Dean of Faculty and the President. We thank the following faculty, staff and students of Connecticut College for their assistance in a number of ways including planning and carrying out the conference and writing, reviewing, editing and proofing chapters for this book: Robert A. Askins, Professor of Biology; Jane Dawson, Professor of Government; Arlan Mantz, Professor of Physics; Yong Jin Park, Professor of Economics; Peter Siver, Professor of Botany; Glenn D. Dreyer, Arboretum Director; Nancy Lewandowski and Melissa Mylchreest, administrative assistants; and Betsy Ginn '05 and Sarah Lumnah '05, seniors. Finally, we are most grateful to all the contributing authors for their patience, understanding and professionalism during the long process of responding to comments and recommendations received during the review and editing phases of this book.

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1. Acid in the Environment: An Overview

Gerald R. Visgilio, Jane Dawson, Peter A. Siver and Diana M. Whitelaw

During the past twenty five years acid rain, formally referred to as acid deposition, has been the focus of much political debate and scholarly research. Acid deposition occurs when important precursor pollutants, such as sulfur dioxide (SO_2) and nitrogen oxides (NO_X), chemically mix with water vapor and oxidants in the atmosphere and fall back to earth in wet or dry form. Wet deposition comes in the form of dew, fog, snow or rain, while dry deposition occurs as either gasses or dry particulates. Research has shown that acid deposition adversely affects freshwater lakes and streams, coastal habitats, agricultural production, forests, soils, human health and building materials. Fossil-fuel power plants, refineries, and paper and pulp mills are the major sources of SO_2 emissions, while automobiles and other vehicles are the primary emitters of NO_X.

Acid deposition is an environmental problem that crosses state and national boundaries, and is closely linked to energy policy since much of it originates as emissions from fossil-fuel power stations. The Scandinavians first identified transboundary acid deposition as a serious environmental issue in the late 1960s, when they found themselves subjected to downwind acid-producing emissions flowing from Europe, particularly from the United Kingdom and Germany. Similarly, the United States has struggled to address inequities caused by the flow of emissions across domestic state boundaries, and has gradually come to recognize the need to work with Canada to find an equitable bilateral solution to transmission across their international boundary. In 1979, an agreement focusing primarily on Europe, but also including the United States and Canada, was reached on a comprehensive, multilateral treaty to restrict the transboundary flow of acid-causing emissions. A landmark in the evolution of international cooperation on the environment, the Convention on Long-Range Transboundary Air Pollution (LRTAP) came into force in 1983, and has been greatly strengthened over the past two decades by a series of protocols further restricting these emissions. While generally considered quite successful in addressing this transboundary issue in Western Europe, implementation of the treaty in the Central and Eastern European countries (CEE) was not seriously undertaken until after the collapse of the Soviet Bloc. With the integration of the CEE countries in a wider Europe, however, has come the challenge of dealing with the scores of antiquated, coal-fired power plants that dot the post-communist landscape and bringing these new and candidate members of the EU into full compliance with LRTAP standards.

A major source of pollutants that cause acid deposition in areas of eastern Canada and northeastern United States are sulfur dioxide and nitrogen oxide emissions from electric-generating facilities located in the northeastern and midwestern regions of the United States. The Acid Rain Program in the United States, or more formally Title IV of the 1990 Clean Air Act Amendments (CAAA), is an important policy initiative with respect to the control of sulfur dioxide emissions. This program, as a major environmental policy that is rooted in a market-based system, represents a shift in U.S. emission control policy away from the command-and control approach of previous years. Title IV, which has been called the "grant policy experiment,"[1] instituted a cap-and-trade program to regulate SO_2 emissions. From this perspective, Title IV combines the benefit of a "solid environmental goal" with the "flexibility to trade or bank" emission discharge permits.[2]

In an attempt to achieve a 10 million ton reduction in the discharge of SO_2 from its 1980 level, Title IV established a national cap of 8.95 million tons of annual SO_2 emissions. The achievement of this national cap is separated into two phases. Phase I commenced in 1995 and pertains to the largest and most polluting electric-generating facilities, while Phase II began in 2000 and extends coverage to smaller and less polluting facilities. Title IV sets up an emission trading market in which each electric utility receives an annual allocation of SO_2 emission discharge permits or, as they are called in the law, allowances. Each allowance held by an individual utility entitles it to discharge one ton of SO_2 into the atmosphere and the utility may sell unused allowances or bank them for use in subsequent years. A decentralized allowance trading market seeks to minimize the cost of abating SO_2 emissions by encouraging utilities with high marginal abatement costs to buy allowances from those with low marginal abatement costs. Trading markets also should provide utilities with an incentive to seek new and innovate ways to reduce SO_2 emissions. In short, the success of the U.S. Acid Rain Program is predicated on its ability to achieve its emission cap in a cost-effective manner.

Acid in the Environment: Lessons Learned and Future Prospects provides an overview of the important science and policy issues pertaining to acid deposition. *Acid in the Environment* uses an interdisciplinary approach that focuses on important ecological impacts of acid deposition, the transboundary nature of the pollutants that cause acid rain, and domestic and international policies designed to reduce the emission of these pollutants. By emphasizing issues such as the scientific lessons learned from acid deposition and the future prospects for market-based emission control policies, our book presents a broad approach to the study of acid deposition. In this context, *Acid in the Environment* blends the research findings and the policy analyses of individuals from different academic disciplines with the positions advanced by representatives of various nongovernmental organizations (NGOs). Because *Acid in the Environment* deals with many aspects of the acid rain issue, it should be of interest to a diverse audience that includes researchers, students, concerned citizens, policy analysts, and members of NGOs and government agencies. *Acid in the Environment* also may serve as a book of readings in introductory courses pertaining to ecology, environmental policy and environmental economics.

We see our book as a springboard for a more enlightened discussion pertaining to the science and policy of acid deposition. We offer the following questions as examples of issues that may facilitate dialogue between such diverse groups as teachers and students, concerned citizens and legislators, or scientists and policy makers. Has research informed the policy debate on the issue of acid rain? What are the long-term effects of acid deposition on forest soils and plants? Have sulfate and nitrate deposition rates decreased in eastern North America during the past decade? Is the atmospheric deposition of nitrogen onto the landscape an important non-point source of nitrogen emissions in coastal ecosystems? What are the sources of the transboundary acid rain issues between the United States and Canada? Has bilateral cooperation on the transboundary issue of acid rain between the United States and Canada been effective in addressing the problem? What lessons have the European Union (EU) and the United States learned from each other with respect to the control of emissions that cause acid rain? How will pan-European efforts to control acid rain be affected by the inclusion of Central and Eastern European countries in the European Union? As an environmental policy tool, are market-based approaches more effective than more traditional command-and-control regulatory approaches? Does Title IV raise significant environmental justice concerns in creating hotspots in less affluent regions? Do minorities and the poor receive a disproportionately smaller share of the net benefits from the U.S. SO_2 emission-trading program? Does the cap-and-trade program provide a model for controlling other types of emissions, including mercury and climate change gases? What roles should and do conservation organizations play in reducing the threat of atmospheric deposition?

Did science contribute to the formation of acid rain policy? In Chapter 2, "Lessons Learned From the Acid Deposition Research Experience: An Historical Perspective," Anthony Janetos uses his experiences as a participant in the National Acid Precipitation Program (NAPAP) to review important lessons that may be learned from acid rain research. In this respect, Janetos focuses much of his discussion on the interplay between research and policy. Although research has influenced the public policy debate about acid rain, it may not have always focused on issues pertaining to decision making in a real world context. Janetos notes that the availability of good time series data on rainfall pH and surface water acidity enhanced our understanding of the process by which acid rain impacts water, soil, and forest ecosystems. However, he also identifies a disconnect between science and policy in the early years of acid deposition research. From his perspective the scientific community initially focused its efforts on understanding the "process by which acid deposition affected ecosystems," while the policy and regulatory community within the EPA sought data pertaining to the "extent, magnitude, and (future) ecological consequences" of acid rain. Although Janetos recognizes Title IV of the 1990 Clean Air Act Amendments as being cost effective with respect to reducing SO_2 emission, he argues that judging the success or failure of our acid rain policy is complex and perhaps premature. An evaluation system, according to Janetos, should at the very least combine "periodic assessments of...physical and ecological outcomes" with "measures of economic and regulatory effectiveness."

Part I. Ecological Impacts of Acid Deposition

Acid deposition delivers acidifying compounds to the surface of the Earth which, in turn, can produce a cascade of negative ecological effects. In Chapter 3, "Acidic Deposition: Sources and Ecological Effects," Charles Driscoll, Kathy Fallon Lambert and Limin Chen describe the composition of acid deposition, changes in the amounts over recent decades and key effects to ecosystems. They note that acid deposition has altered forest soils by accelerating the leaching of available base cations, enhancing the accumulation of sulfur and nitrogen, and increasing the concentration of dissolved inorganic aluminum in soil water. Driscoll and his colleagues argue that, because of these changes in the soil, the structure of many forest ecosystems have changed and they have become more sensitive to the effects of additional acidic deposition. They also point out that acid deposition has impaired surface water quality by lowering pH, decreasing acid-neutralizing capacity, and increasing concentrations of dissolved inorganic aluminum. In affected waterbodies, such changes have resulted in a reduction in both species diversity and the abundance of aquatic life. Finally, they contend that long-term research indicates that additional reduction in the emission of sulfur dioxide and nitrogen oxides will be necessary to accelerate the recovery of affected ecosystems.

Does climate play a role in the recovery of surface water from acid deposition? In Chapter 4, "Long-Term Changes in Boreal Lake and Stream Chemistry: Recovery From Acid Deposition and the Role of Climate," Peter Dillon, Shaun Watmough, Catherine Eimers and Julian Aherne discuss the results of their research on sulfate deposition. In their work, which covers approximately three decades of observations, Dillon and his colleagues evaluate trends in elemental budgets for 8 lakes and 20 sub-catchments located in south-central Ontario, Canada. Their research indicates a link between drought conditions and increased sulfate concentrations in surface waters, which is most likely a result of the mobilization of stored sulfur from wetlands. As a result, despite an almost 50% decrease in atmospheric deposition of sulfate to the region, the concentration of sulfate in the surface waters has not declined to levels as initially predicted, and subsequent improvements in buffering capacity and pH levels have not been fully realized. Based on their long-term records, drought years appear to be highly correlated with El Niño events.

Have the increased rates of nitrogen deposition associated with acid deposition impaired terrestrial ecosystems? In Chapter 5, "Atmospheric Nitrogen Deposition: Implications for Terrestrial Ecosystem Structure and Functioning," Knute Nadelhoffer indicates that, unlike sulfur, rates of nitrogen deposition in eastern North America are not decreasing. In his chapter, Nadelhoffer describes the effects of nitrogen deposition, including both nitrate and ammonium, on plant community composition, net primary production, carbon gains and losses, and patterns of nutrient cycling in terrestrial ecosystems. He not only summarizes the results of research pertaining to the mechanisms by which terrestrial ecosystems retain or release nitrogen inputs, but he also predicts the long-term effects of nitrogen deposits on the composition and functioning of these ecosystems.

Non-point source emissions account for a substantial amount of nitrogen pollution in coastal waters. In Chapter 6, "Atmospheric Deposition and Nitrogen Pollution in Coastal Marine Ecosystems," Robert Howarth recognizes atmospheric deposition as an important source of coastal nutrient pollution. However, since the onset of public concern in the early 1970s over acid rain, attention has centered primarily on the effects of the acid component of the precipitation on terrestrial and freshwater ecosystems. Marine systems were largely ignored because they were strongly buffered and considered immune from the falling acids. What was initially overlooked, but now a growing concern, were the anion components of the acid compounds, in particular the nitrogen oxides. As in many terrestrial ecosystems, nitrogen is often the limiting nutrient in marine ecosystems and recent findings indicate that the contribution of nitrogen from atmospheric deposition to coastal waters in the United States is indeed significant. In his contribution, Howarth argues that acidic deposition accounts for up to 40% of the nitrogen budget either directly from the atmosphere, or indirectly through transport from terrestrial landscapes via surface waters to coastal ecosystems along the United States. Howarth clearly articulates current research needs and outlines nitrogen loading estimates that are especially subject to uncertainty.

Part II. Acid Emissions Energy and Policy

Are there different approaches used in the EU and the United States to control emissions that cause acid rain? In Chapter 7, "The Politics of Acid Rain in Europe," Miranda Schreurs provides a historical overview of the evolving recognition of the transboundary nature of the acid rain problem, first called to world attention by the Scandinavians in the late 1960s, and the emergence of multilateral cooperative solutions, with the earliest and most far-reaching being the steps taken in Europe in the 1970s and beyond. While the Convention on Long-Range Transboundary Air Pollution (LRTAP) which went into force in 1983 included the United States and Canada, the North American members of the Convention never ratified the series of protocols adopted by the European members after 1983, thus setting the United States and Canada on a quite different path in the solution of transboundary acid deposition problems than the Europeans. With over 40 participating European states, LRTAP and the concurrent development of European Union standards on acid-producing emissions have presented a much more challenging arena for treaty negotiation and cooperative solutions than the North American case. Schreurs traces the role of expert communities, nongovernmental organizations, technological innovations in the power and industry sector, and the expansion of the EU's jurisdiction to include environmental issues, in facilitating successful cooperation on transboundary air pollution in Europe over the past three decades. Schreurs finds that the European approach, which emphasizes targets and regulations rather than the market-based approach adopted in the United States, has been very successful in decreasing SO_2 emissions across many of the wealthiest countries of Europe, but has been less successful in the less affluent countries of Southern Europe and has had very limited success in addressing the more difficult problem of NO_x emissions from non-stationary sources. Given the relative success of the European approach to addressing this transboundary issue among a diverse group of states encompassing both rich and poor, emitters and receivers, LRTAP has been put forth as a more appropriate model than the bilateral Canada-United States agreement for building cooperative air quality regimes in other regions of closely packed countries, particularly South and East Asia.

Because of the prevailing atmospheric and geographic patterns, the emissions of air pollutants in Central and Eastern European (CEE) countries contribute significantly to the transboundary acidification problem in Europe. In Chapter 8, "Acid Rain in a Wider Europe: The Post-Communist Transition and the Future European Acid Rain Policies," Liliana Andonova examines the development of acid deposition policies in ten CEE countries undergoing post-communist transition and European integration in the 1990s, and finds a surprising level of success in decreasing power plant emissions across the region. She argues that the success of these policies is due largely to the role of international actors and institutions—particularly the lure and support of the EU—but that the level of success varies across countries. By looking more closely at the cases of the Czech Republic, Poland, and Bulgaria, whose adherence to LRTAP and EU standards range from over-compliance in the Czech case to negligible action in Bulgaria, Andonova also draws out the important role played by domestic institutions and actors in determining the extent to which CEE governments have been willing and able to bring their power plants into compliance with tough European standards. The Czech Republic's well-developed democratic institutions, strong public opinion supporting quick action on air quality issues, and a power sector eager to be integrated into the West European electricity grid all came together to promote dramatic measures and the over-compliance with LRTAP as now observed. Many observers have suggested that bringing the ten CEE states into the EU will erode the high air quality standards upheld by both LRTAP and EU policies and continued movement toward even tougher standards will be halted by the votes of these new members. Andonova, however, does not support this viewpoint. Although cautious in her optimism, she argues that a widening of the EU has been accompanied by the imposition of very high emissions standards which will shape the economic restructuring of the CEE countries for the next decade and set them on the road to continued improvement on this front in the future. Thus European expansion should not be seen as watering down the successes of Western Europe, but rather fostering genuine "pan European efforts" to address the transboundary acidification problem.

SO_2 and NO_x emissions, which are often dispersed by prevailing winds from the United States to Canada, create a contentious transboundary pollution problem between the two countries. In Chapter 9, "Acid Rain Politics in North America: Conflict to Cooperation to Collusion," Don Munton provides an "overview of the politics" of transboundary acid deposition in the United States and Canada. He identifies the years from the late 1970s to 1990 as a period of conflict in which Canada acted as the "demandeur" and the United States as the recalcitrant participant in their negotiations on acid rain, with little achieved in addressing the transboundary complaints continuously voiced by the Canadian government. The passage of the 1990 CAAA in the United States and the negotiation of the 1991 Canada-United States Air Quality Agreement brought a period of "significant bilateral cooperation" between the two countries in the implementation of emission reduction programs – though cooperation that came about largely as a result of U.S. domestic public concerns about acid rain and the need to address it at home, with the U.S.-Canadian cooperation primarily a by-product of U.S. domestic policy. While the Bilateral Air Quality Agreement has largely been celebrated as a great success for both countries, Munton's research reveals a certain hollowness to these claims; Munton argues that in recent years cooperation has given way to "blatant collusion" between the two governments as each country studiously avoids mentioning or recognizing the serious inadequacies in compliance with the bilateral Air Quality Agreement, which have become ever more common since the late 1990s.

In Chapter 10, "Air Quality and Power Production in the United States: Emissions Trading and State-Level Initiatives in the Control of Acid-Producing Emissions, Mercury, and Carbon Dioxide," Daniel Sosland argues that the federal government has recently subordinated efforts to reduce the emission of air pollutants in favor of policies that are intended to promote greater energy production. Rather than focusing solely on acid-producing emissions, Sosland expands his discussion to include other pollutants emitted by coal-fired power plants, including mercury, which is a potent toxin, and carbon dioxide, the primary culprit in the climate change debates. He thus looks at the broader issue of the air quality impact of our national energy strategy and reliance on fossil-fuel based power plants, and considers whether the market-based approach, used so successfully to control emissions of acid-producing pollutants in the United States since 1990, offers opportunities for addressing ongoing challenges in curtailing both mercury and carbon dioxide emissions from power stations. While he finds the lessons of the cap-and-trade approach to SO_2 quite appropriate to future policies to limit carbon dioxide emissions, he also argues that it is entirely the wrong approach to addressing mercury emissions. In looking ahead toward the formulation of policy to address climate change and carbon dioxide emissions, Sosland not only looks to the cap-and-trade successes of the Acid Rain Program in the United States, but also earlier successes in building a clean air regime starting from the local, state, and regional levels. In this forward looking chapter, Sosland reviews the encouraging steps being taken by states and regions in the United States to adopt largely market-based policies toward the control of CO_2 emissions and predicts that this momentum will eventually catapult the issue up to the federal level.

Part III. Sulfur Dioxide and the Market

Will market-based policies improve the level of environmental quality? In Chapter 11, "Market-Based Approaches to Environmental Policy: A 'Refresher' Course," Paul Portney argues that, by charging prices for air and water resources, emission sources have an incentive to economize on the use of these resources. Portney notes, however, that for many years command-and-control regulations with technology based effluent standards have been a major part of U.S. environmental policy. These regulations, however, limited the flexibility of emission sources to be internally cost effective with respect to emission abatement. Portney also discusses market-based policies, such as emission taxes and cap-and-trade systems, as providing emitters with the flexibility to select least cost abatement strategies. He sees Title IV of the 1990 CAAA as the first successful "large scale application of cap-and-trade" in the United States. Although not seen as a panacea for all of our environmental problems, Portney maintains that market-based approaches are "the default option in much of modern environmental policy."

When evaluating environmental policies, it is important to review their performance in a "real world" context. In Chapter 12, "Economic Incentives Versus Command and Control: What's the Best Approach for Solving Environmental Problems?" Winston Harrington and Richard Morgenstern use several case studies to provide an ex post evaluation of command-and-control versus economic incentive policies. Although there is no "one-size-fits-all" answer to the question of how to judge the best environmental policy, Harrington and Morgenstern focus on the effectiveness of each type of policy in achieving its environmental goals at lowest costs. In this context, they compared the actual performance of environmental policies in the United States with those of various Western European countries with respect to six environmental problems. From the results of their case studies, Harrington and Morgenstern argue that environmental policies bring about desired environmental results and that the economic incentive policies provide "cost savings in pollution abatement" as well as a strong incentive for emitters to reduce "overall costs" through technological innovations.

The environmental justice movement in the United States deals with the issue of disadvantaged communities bearing a disproportionate burden of society's environmental risks and receiving a smaller share of the benefits from the implementation of environmental policies. In Chapter 13, "Benefits and Costs From Sulfur Dioxide Trading: A Distributional Analysis," Ronald Shadbegian, Wayne Gray, and Cynthia Morgan look at the spatial distribution of costs and benefits resulting from air quality improvements under Title IV of the 1990 CAAA. Their work reveals substantial net benefits from the reduction of SO_2 emissions and a high concentration of these costs and benefits in four EPA regions. They also found no indication of environmental injustices among minority communities, with African-American and Hispanic communities enjoying a "substantially greater share" of the benefits relative to the costs from Title IV SO_2 emission reduction. Their work, however, shows some evidence of an environmental inequity in the distribution of costs and benefits to the poor.

Has the U.S. SO_2 emission-trading program influenced the design of domestic and international climate change policies? In Chapter 14, "From Sulfur Dioxide to Greenhouse Gases: Trends and Events Shaping Future Emissions Trading Programs in the United States," Joseph Kruger describes the U.S. SO_2 trading program as "a model" for future cap-and-trade programs and emissions trading, in general, as fundamental to framing an international policy for climate change. Although he recognizes our SO_2 trading program as providing important lessons for controlling greenhouse gas emissions, Kruger discusses how differences in "sources, science, mitigation options, and economics" may affect the design of climate change policies. He concludes with an assessment of likely impacts of alternative climate change initiatives such as voluntary targets, regional state government agreements, and the EU trading program on the development of a national greenhouse gas trading program in the United States.

Part IV. Lessons Learned and Future Prospects

It is widely recognized that the interplay among science, technology, and policy is important in the control of atmospheric deposition. In Chapter 15, "Atmospheric Deposition and Conservation: What is the Role for Conservation Organizations?" Timothy Tear explores the role of conservation organizations in reducing the ecological impacts of atmospheric deposition. He argues that conservation organizations must play an active part along with science, technology, and policy in addressing the threat to ecosystems from atmospheric deposition. Because of global to local links, parcels of land and bodies of water protected by conservation organizations are being adversely impacted by acid deposition. In this regard, Tear notes that The Nature Conservancy is currently taking a proactive approach in dealing with atmospheric deposition. Not only has the Conservancy expanded its scientific capacity to evaluate the adverse effects of atmospheric deposition on ecosystems in the northeastern United States, but it also plans to help formulate public policy to mitigate the impact of these effects. Tear argues that conservation organizations need to raise public awareness about the impacts of atmospheric deposition on ecosystems, engage in long-term monitoring of ecosystems, and support scientific research.

Where do we go from here? Are there important lessons to be learned from the study of acid deposition? In Chapter 16, "Achieving a Solution to Acid Deposition and Other International Environmental Problems," Robert Askins summarizes major themes presented in our book. In his discussion of these themes, he insightfully weaves together positions advanced by the contributors to this volume with respect to the science and policy of acid deposition. To further the chemical and biological recovery of ecosystems, Askins notes a consensus among several of our authors for more stringent control of major acid deposition sources. In addition to regulating the discharge of sulfur dioxide from electric utilities, they call for controlling the emissions of nitrous oxides from the transportation sector and of ammonia from the agricultural sector of the economy. They also argue for the need to understand and monitor the ecological impacts of acid deposition on a wide array of ecosystems. Another general theme advanced by some of our contributors is the need to pursue sound environmental policies even when, as Askins indicates, there is little, if any, consensus among those involved in the problem, as is often the case with transboundary pollutants. Finally, there is the theme that current acid deposition policies may serve as a model for other emission control policies. Here, Askins notes that the lessons learned from the control of sulfur dioxide might be applied to the control of greenhouse gases.

Acid deposition is an on-going, long-term environmental problem with "a greater environmental impact than previously projected."[3] Since many ecosystems are now "more sensitive to the input of additional acids," their recovery from the adverse affects of acid rain will most "likely be delayed."[4] In *Acid in the Environment,* we provide the perspectives of various authors with respect to the lessons learned and future prospects associated with the issue of acid deposition. We use an interdisciplinary approach that combines a discussion of important ecological issues associated with acid deposition with an analysis of domestic and international policies to control the emission of pollutants that cause acid rain. In this context, *Acid in the Environ-*

ment exposes students with a science background to significant policy issues while it also exposes those with a policy orientation to important ecological impacts. We also see our book serving as a solid platform for class discussions on issues such as the process of ecosystem recovery or direction of climate change policy in the United States. Finally, we refer the interested reader to the footnotes, references, or readings presented at the end of each chapter for additional coverage of the many topics that are discussed in this volume.

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2. Lessons Learned From the Acid Deposition Research Experience: An Historical Perspective

Anthony C. Janetos[1]

I began a fifteen-year career as a federal science program manager in the middle of the 1980s, when I took a position in the U.S. Environmental Protection Agency's (EPA) Office of Research and Development on the Acid Deposition Research Staff. EPA was emerging from an unprecedented trough in public perception and official performance. William Ruckelshaus had returned as Administrator, in part to repair the damage done by the previous Administrator, and among the many vexing environmental issues that needed to be addressed, the challenge of acid deposition was among the greatest. In this chapter, I reflect not so much on the science of acid deposition per se, but on my personal experiences as a participant in an important federal science and assessment program on a very visible public environmental issue, and what lessons can be drawn from them.

For purposes of clarity, I will address four categories of experiences:

- Scientific lessons, focusing especially on the design of science and assessment programs;
- Policy and political lessons, focusing on whether the science really affected policy decisions and what it finally took to get policy action;
- Institutional lessons, examining the challenges to coordination and collaboration in large, interagency programs and implications for today's issues; and
- Career lessons, examining the incentives and disincentives for managers to participate in such programs.

1. Scientific Lessons

1.1. Background

The basis of the acid deposition issue in the United States in the 1980s had been forming for over a decade. The observations of acidified surface

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[1] The views presented in this chapter are my thoughts about my own learning experiences in the National Acid Precipitation Assessment Program and beyond.

waters, mostly in lakes in the Northeast, often accompanied by dramatic reductions in fish and other biological populations, mirrored phenomena that had also been documented in Europe. Canadian lakes in the eastern provinces showed identical phenomena. Europe, Canada and the eastern United States also exhibited acidity in rainfall that far surpassed the expected natural background, accompanied with high levels of deposition of sulfate, both in solution in rainfall and in particulate form. Nitrate deposition was also much higher than expected from natural processes (see below). By the mid-1980s, there was relatively strong scientific consensus that in many places, the deposition of strong mineral acids in rain and snow had in fact led to acidification of surface waters over time. The chemical processes in soils were relatively less well understood, and there was a controversy over whether there would be additional delayed consequences of continued deposition. There was very little debate over where the sources of the excess sulfate were: they were understood to be the big stationary sources of coal-fired power plants, mostly although not exclusively in the midwestern states of Ohio, Indiana and Kentucky, but also including many large sources throughout the Northeast. It was also clear by that time that U.S. sources of sulfate influenced Canadian resources, and vice versa.

There was also beginning to be concern over acid deposition's potential effects on forests. Coniferous forests in Scandinavia, Germany, and Eastern Europe were experiencing a dieback phenomenon that was characterized by foliar damage, reductions in growth rates, a surprising variety of leaf pathologies, and eventual tree death. In part because there were few other candidates, air pollution was strongly suspected to be a cause of the dieback. In the northeastern United States, at high elevations in the Adirondacks and even further south in the Appalachians, a similar, but not identical dieback was beginning to be noticed. Red spruce was the species most affected, and symptoms included obvious damage to its needles, reduced growth rates, and increased mortality in affected stands. Air pollution was also a strong candidate as a cause, in part because there were no other obvious candidates for this pathology, and in part because the early stages of the phenomenon were limited to high elevation, high deposition regions. The growing visibility of this phenomenon, and the possibility of its link to acid deposition and air pollution, only served to increase the public's attention to acid deposition and as a spur to increased federal science funding to quantify and understand these phenomena.

The policy scene was substantially less clear. It was known that reductions in sulfur emissions were going to be needed to reduce the levels of lake acidification, but the amounts of reductions, the costs of doing so, the regulatory mechanisms to be used, and the fact that the environmental damage of concern was in many ways separate from where the emissions originated all served to make policy and political solutions difficult to negotiate. Although debate was vigorous, and the political rhetoric heated, there was no policy in place for dealing with the issue under the Clean Air Act.

1.2. The Importance of Time Series

The consensus in scientific understanding at the time was largely due to having a few locations, such as the Hubbard Brook Experimental Forest, described in detail in subsequent chapters, where there were high quality time series of both rainfall pH and surface water acidity, and where very careful experimentation had been done to understand the processes involved. Such high quality time series were of enormous value in understanding the processes by which acid deposition was affecting surface waters, soils, and ultimately forest ecosystems.

But the forest dieback phenomenon lacked such carefully investigated, long time series. As a result, this newer phenomenon was relatively poorly understood, and there were simultaneously arguments over its very existence as a significant environmental issue, as well as the mechanisms by which acid deposition might or might not be involved. Added to the complexity was the realization that the phenomena observed in forests were not immediately diagnosable as being due specifically to an air pollution stress. This was quite unlike the situation with surface waters, where the acidification and loss of aquatic life was clearly due to the continued, long-term deposition of strong mineral acids; the only real questions were the particular soil chemical processes involved, how much buffering capacity there was, and how quickly the systems might respond to changes in deposition. The situation for forests was far less clear.

1.3. The Importance of Extent and Magnitude

There was a significant difference between the desire for knowledge on the part of the policy and regulatory community within EPA (and more broadly, the federal agencies), and those of the scientific community. For the most part, the scientists involved were interested in understanding the processes by which acid deposition affected ecosystems. The policy community respected and valued this mechanistic knowledge base, of course, because it was fundamentally important to establishing the cause-and-effect linkage that was itself fundamentally important to establishing levels of emissions reductions that were going to be necessary. But the policy community also needed to know the overall extent and magnitude of the (then) current effects that could be attributed to acid deposition. It also needed to know the degree to which the current effects might constitute all there were going to be, or whether substantial additional effects might be in the offing, or what the ecosystem's sensitivities might be, and whether there were thresholds in effects such that very rapid or large additional changes might be coming.

These concerns went well beyond the traditional academic interests in understanding mechanisms and processes. They quickly entered the realms of environmental assessment (how much damage and where) and predictability (what might the future hold under different policy/environmental scenarios). The scientific community could address these concerns but at least in the beginning, the research programs in place were ill-equipped to do so. They had been reasonably designed to understand processes and mechanisms, but not designed to quantify the extent and magnitude, nor designed to investigate alternative futures.

It was in the mid-to-late 1980s that the EPA and federal research programs were dramatically re-oriented to provide quantitative information on the extent and magnitude of acidification effects on surface waters to respond to this need of the policy community. EPA's survey of the extent of acidified surface waters in lakes, and later, in streams, provided substantial information on the degree to which the environment had already been altered by acid deposition. New mechanistic research on forests, initiated around the same time, provided much better information on the processes leading to the observed forest dieback phenomena, and the U. S. Forest Service developed visual survey methodologies that began to be incorporated into its routine forest survey methodologies to estimate the extent of forest dieback.

1.4. Working in Parallel

A natural and logical way for scientists and laypeople alike to understand the genesis of any environmental phenomenon is to work from its origins, to its consequences, to remedies. For acid deposition this meant following the path from sulfur and nitrogen emissions, to atmospheric chemistry and transport, to deposition processes, to ecological consequences, and then to treatment (e.g., liming of lakes to reduce acidification) or to reduction of emissions. This template outlined the major delineations of the EPA and federal research programs.

In terms of program design, this logical structure becomes problematic. There was a tendency in both the scientific community and in the policy communities to view acid deposition as primarily an atmospheric issue. This was not surprising, but it was limiting. Programs that are designed to treat environmental problems in essentially chronological order presuppose that there are questions about whether the atmospheric stresses actually lead to ecological consequences, i.e., whether there really are cause and effect mechanisms. In the case of acid deposition and the deposition of surface waters, nearly a decade of research was spent on the mechanisms of acid formation and transport in the atmosphere, when what had driven the public's interest and concern were observed consequences that by the mid-80s had already been linked to deposition. The real question from the standpoint of consequences was how extensive the damage was, and whether it was going to get worse.

When the forest issues began to be raised, those questions were magnified. It was unrealistic to expect that the research programs initiated in the mid-80s were going to yield quantitative information on processes and extent of the phenomena in a short period of time. But the consequence of structuring the overall program in the same way as it was most easily understood was that the bulk of the funding had already been put into the atmospheric components of the problem. So there were extensive field experiments on model validation, and indeed, extensive model development. But the major questions that were being asked by the policy community were not so much about the atmospheric science issues as they were about the extent, magnitude and potential futures of ecological consequences. Unfortunately, significant research funding on ecosystem effects was precluded because of the original formulation of the problem as being primarily an atmospheric issue. Had the ecological research been undertaken when the phenomena were first observed, in parallel with the atmospheric research, there would have been substantially more and better information about potential target loadings and whether thresholds for damage existed. In addition, the fact that the ecological research required had intrinsically longer time scales, i.e., simply took longer to do in the field, than the atmospheric research, meant that it was very difficult to respond to the degree of urgency that had already entered the public, policy, and political debate.

Even though one can argue that there should have been more work on ecological effects funded in parallel with the atmospheric science, one cannot argue that the atmospheric science was unimportant. Obviously, it was the understanding of the atmosphere's behavior that made specific policy recommendations about targets, timetables, technologies and compliance costs, and ultimately emissions caps possible to have in a substantive way.

In my view, the program would have been more responsive to the actual needs of the policy makers had it been structured and funded in a more "Bayesian" fashion. It could then have asked and addressed fundamental questions about the ecological consequences of different deposition levels at the same time as it sponsored research on the atmospheric and engineering challenges involved in controlling emissions and deposition. It is certainly possible that this would have meant less overall investment in the atmospheric sciences in the research program. I can only note, however, that the technical details of the vast majority of atmospheric sciences supported in the acid deposition program did not enter the policy discussions about cap and trade legislation in any significant way.

1.5. Holes in the Strategy

At the time, it was reasonably well understood among the ecological scientists involved in the acid deposition program that there were potentially significant gaps in the overall research strategy. For instance, the deposition of nitrates also has an acidifying effect on most of the soils in the eastern United States, as had already been documented in parts of Europe. Nitrogen deposition also came into sharp focus as an important contributor to the forest dieback problems experienced by red spruce, especially at high elevations in the Adirondacks and Appalachians. There was also considerable concern among the scientists working on forests, in particular, that there were synergistic effects of other air pollutants, especially ozone, and acid deposition that were important stressors on forests in the eastern United States. The acid deposition program simply could not deal with all of these issues at once, and the policy/regulatory apparatus of EPA certainly could not deal with these issues of multiple environmental stresses. The end result was that there was policy action that was quite effective at reducing the emissions and deposition of sulfur, but nitrogen and ozone issues were left for another day, regardless of the fact that it was already well-understood that they had become important stressors on the ecosystems.

The final known gap in the overall strategy was ensuring that adequate monitoring of both deposition and effects would continue. The research staffs in both EPA and other agencies were acutely aware of this need. Indeed, some of the deposition chemistry monitoring has continued to this day, in spite of substantial pressure on the funding of such monitoring networks. But the situation for effects has been less impressive. A few of the long-term research sites, such as Hubbard Brook, have been able to maintain their research support over the years. But several of the research networks that were originally implemented by the EPA focused on understanding the links between deposition and forest dieback, were quickly cannibalized for funds for new research interests by the early 1990s. Subsequent surveys of lake and stream chemistry optimized for change detection were not done, with the result that if this problem were to arise anew today, we would almost be limited to the same set of long-term research sites as we had in 1984 to find high quality time series of environmental data.

2. Policy and Political Lessons

Did the science matter? The National Acid Precipitation Assessment Program (NAPAP) was, at the time, the biggest interagency research and assessment program that the federal government had organized to understand an environmental problem. Was that investment worth it?

In my view, the answer is both yes and no. Yes, because the research clearly provided important information in terms of developing understanding about the extent, rates, and magnitudes of the consequences of acid deposition, and about the deposition itself. No, because at least some of the research was not focused well on real decision-making. For example, some of the dose-response research on materials damage was successful in identifying damage functions, but it was not focused on the factors that people actually used in real decisions about replacement, repainting, and recovery. The end result was that the cost numbers for potential damages were largely irrelevant to any sensible emissions reduction program.

Another way in which popular perception of policy interests was at odds with expressed policy interests was in the realm of health effects. It is a tenet of many people concerned with environmental issues that human health effects are in some way the *gold standard*. In the acid deposition program, research on potential health effects focused mainly on the potential health effects of acidified ground and surface water leaching metals out of municipal water systems. This turned out, however, to be largely a hypothetical concern. Interestingly, integrated assessment models done at the time identified potential health consequences as a major concern in policy formulation, although their probability of occurrence was understood to be quite low. In large part, this was because such models could quantify in economic terms the health impacts in terms of treatment costs and lost income, but could not quantify at all the consequences of losses in ecosystems and their services in economic terms. Thus, the very low probability of health consequences trumped the known large ecosystem consequences because the latter were calculated as having almost no economic value.

On the other hand, the public really did care about ecological outcomes, even when clear and large economic consequences were not identified. The sense of place that many people have turned out to be enormously important, as reflected in their acceptance of bearing regulatory costs in order to protect a region's ecological heritage. This was true in acid deposition, and remains true today in the climate change debate, even as we begin to understand more about the role that ecosystems play in human well-being by providing services.

The final aspect of policy and political lessons to be addressed here is the notion of whether the reduction in sulfur emissions through the cap-and-trade program in the Clean Air Act amendments has been successful. From one perspective, the answer is obviously yes. An enormous amount of sulfur that would have been emitted into the atmosphere has not been, and the overall cost of compliance with the targets is lower than they would have been under standard command and control regulations. However, as other chapters in this volume point out, there are still substantial numbers of acidified lakes and damaged forests in the northeastern United States. The recovery of systems has been much slower than was originally thought, and the gaps in the regulatory system, in particular the relative inattention to nitrogen sources from transportation sources, means that there is still substantial acid input to those systems.

This argues that judgments about the success or failure of the regulatory regime for acid deposition are both more difficult than anticipated and possibly premature. We must move towards an evaluation system that at the very least incorporates periodic environmental assessment of the state of the physical and ecological outcomes of concern as well as measures of economic and regulatory effectiveness. Such a system should be able to maintain financial support of the necessary ecological monitoring as well as monitoring of the atmospheric stressors, and include measures of economic costs and benefits, and those other measures of costs and benefits that are difficult to price and therefore trade in markets.

3. Institutional Lessons

At the time, NAPAP was the biggest coordinated interagency research and assessment program that was dedicated to studying the processes, impacts, and potential solutions of a national environmental problem. At its height, the research budget exceeded $60M annually, and more than a dozen agencies participated, about half of them actively. The program was the province of the Research and Development arms of the various agencies, and their policy equivalents served on an oversight board of agency executives. EPA's Office of Research and Development, where I worked, was the largest single player in terms of budget and people, and sponsored research both in its own laboratories and in the external university community.

But an interesting feature then, as now, of large interagency research programs in the federal government was the diversity of philosophies that the agencies had about research and the diversity of relationships with the external scientific community. Many agencies had in-house scientific expertise, and those scientists had the normal relationships with their peers in universities, think tanks, and consulting companies that one would expect to see anywhere. But those agencies often had relatively little experience in funding outside scientists, especially using competitively awarded, peer-reviewed grants. Others had primarily in-house science management expertise, and were oriented primarily towards the award of grants to outside scientists. Still others, including EPA at the time, had a mixture of each, but had not often made an institutional commitment to one mode of operation or another. The result of this last mode of relating to the broader scientific community was a continual tension between the external community and the internal community of agency scientists over scarce resources.

It was also the case that the agencies in NAPAP had very different management philosophies, which also led to some tensions. A few, such as EPA, had traditions of reasonably strong central management. Others were extensively decentralized. Understandably, this difference in agencies' approaches to their own management led to tensions within NAPAP, especially when decisions needed to be made quickly, or when longer-term strategic decisions required different lengths of time to be vetted in the participating agencies.

However, there was one common feature to all of the science agencies involved in NAPAP. All experienced at least some degree of difficulty in communicating with their internal policy counterparts. There often was some degree of distrust, often driven by what the science managers perceived as a desire for too-rapid decision making by their policy colleagues. But in my view, there was primarily a lack of a common understanding between policy analysts/managers and scientist/managers about what science would in fact be helpful and useful in making policy recommendations and decisions.

Interestingly, NAPAP was quite effective at galvanizing participation in research with the private sector. The Electric Power Research Institute (EPRI) and the National Council for Air and Stream Improvement (NCASI) each brought substantial intellectual and financial resources to the table on ecological research, for example. They became important partners in both field and modeling research, and many of the collaborations that began in NAPAP have lasted to the present day, focusing on different topics now, but building on a strong relationship of trust that began in NAPAP.

Finally, the creation and use of the interagency program office was, I believe, extraordinarily important for NAPAP's functioning. Many of the agencies preferred to keep the central office weak; doing so enhanced their own ability to influence NAPAP's agenda. But the central office provided a neutral forum for the agency participants to meet and negotiate, and indeed many of the programs that NAPAP sponsored went far beyond simply coordinating the efforts of individual agencies. They required real negotiation over activities, organizational structures, schedules, and resources, and would have been far more difficult to achieve without that central focal point.

Where the central office of NAPAP finally proved its worth, however, was in the assessment process. After a very visible, public false start on assessment, where an Executive Director of NAPAP put his personal political conclusions on the results of the science without benefit of peer review, damaging NAPAP's credibility nearly beyond repair, the central office played a crucial role in restoring that credibility. The new Executive Director, James Mahoney, very visibly took steps to restore transparency to NAPAP's assessment processes, ensured that a set of high-quality scientific assessments were done, reviewed, and revised, and generally brought a high level of professionalism back to the program's operations. These steps could not have been taken without a central office that was strong enough and visible enough to ensure that there was some management consistency across a multitude of efforts.

4. Career Lessons

Finally, I will touch on some career lessons that stem from my experience with NAPAP. One of the interesting features of many of the people involved with NAPAP was how relatively inexperienced so many of them were. Of course, there were senior agency managers involved in each agency. But there was also a plethora of junior, early and mid-career employees in each agency, many of whom had been entrusted with a great deal of authority and responsibility. NAPAP was for many of them a seminal feature of their professional development, as it was for me.

A large cadre of these participants has gone on to have extremely interesting and quite varied careers in government, academia, industry, environmental groups, and think tanks. I am convinced that this cadre of people learned early on that it was indeed possible to have meaningful collaboration among different government partners, among the government and industry and the NGOs, and that they have used this knowledge to their advantage in their own careers. Indeed, it is now relatively common for serious environmental problems to generate an interagency scientific response, and the experience that the NAPAP participants had themselves, and the institutional experience of their agencies suggests that current interagency partnerships have sometimes been made easier and more productive as a result.

It is also true, I believe, that a large part of the learning experience of the NAPAP participants was a far better understanding of how science supported by the federal government and private sector does or does not interact with policy and decision making. Acid deposition was and remains a case with large environmental stakes, and potentially large compliance costs. The knowledge we gained in NAPAP will continue to benefit us as we deal with the current challenges of acid in the environment.

Part I

Ecological Impacts of Acid Deposition

3. Acidic Deposition: Sources and Ecological Effects

Charles T. Driscoll, Kathy Fallon Lambert and Limin Chen

1. Acidic Deposition

Acidic atmospheric deposition, popularly referred to as acid rain, is the transfer of strong acids and acid forming substances from the atmosphere to the Earth's surface. Acidic deposition is comprised of sulfuric and nitric acids, and ammonium derived from atmospheric emissions of sulfur dioxide, nitrogen oxides, and ammonia respectively. These compounds are emitted by the burning of fossil fuels and by agricultural activities. Once such compounds enter an ecosystem, they can acidify soil and surface waters and bring about a series of ecological changes. The term acidic deposition encompasses all forms in which these compounds are deposited to the Earth, including gases, particles, rain, snow, clouds, and fog (see Box 3.1). Acidic deposition was first reported in the United Kingdom in the later half of the 19th Century (Gorham 1992). Ecological effects were first documented in Scandinavia in the 1960s with the link between acidic deposition, surface water acidification and loss of fisheries (Gorham 1992). Atmospheric deposition of sulfate, nitrate and ammonium are elevated in eastern North America, Europe and large portions of Asia (Rodhe et al. 1995).

Electric utilities account for the greatest proportion of anthropogenic sulfur dioxide emissions in Europe and North America. For example, in 2002 the major sources of sulfur dioxide emissions in the United States were electric utilities (67%), industrial combustion (15%), and industrial processes (9%; United States Environmental Protection Agency 2004). Transportation sources—including cars, trucks, and non-road vehicles (i.e., construction equipment)—accounted for more than 50% of anthropogenic nitrogen oxide emissions in the United States. Other major sources of nitrogen oxides include electric utilities (22%) and industrial combustion (14%). Ammonia emissions are derived largely from livestock waste and fertilized soils (83% of total ammonia; Driscoll et al. 2003). Motor vehicles and industrial processes also contribute to ammonia emissions.

An airshed or source area is an area where "significant portions of emissions result in deposition of air pollutants to a region" (www.epa.gov). In North America, emissions of sulfur dioxide are highest in the mid-western

United States (hereafter the Midwest), with seven states in the Ohio River Valley accounting for 51% of total U.S. emissions in 2002 (Figure 3.1a). Five of these states are also among the highest emitters of nitrogen oxides (Figure 3.1b). Moreover, the Midwest is a significant source of atmospheric ammonia. In addition to regional pollution sources, local emissions of sulfur dioxide and nitrogen oxides from electric utilities and motor vehicles have significant impacts on local air quality. Analysis of continental air currents shows that a multi-state region, including the Midwest, comprises the source area for sulfur dioxide, nitrogen oxide and ammonium emissions that are transported downwind to acid-sensitive areas of eastern North America (Butler et al. 2001).

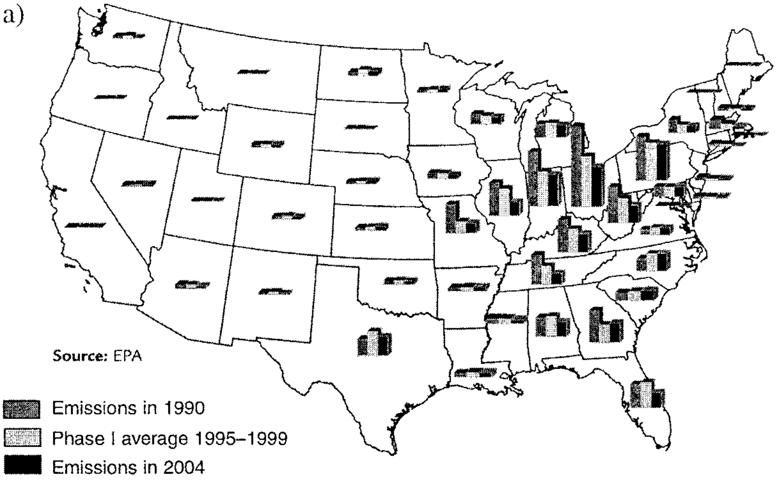
There have been significant efforts to reduce emissions of acidic and acidifying substances in North America and Europe over the past three decades. Although regulatory controls have decreased emissions, levels remain high compared to background conditions. Importantly, emissions and deposition of base cations (i.e., elements such as calcium and magnesium that help counteract acidic deposition) have declined substantially since the early 1960s with the enactment of pollution controls to reduce fine particulate matter (Hedin et al. 1994).

Total sulfur dioxide emissions in the United States peaked in 1973 at approximately 29 million metric tons annually. The 1970 and 1990 Amend-

Box 3.1. How is acidic deposition monitored?

Acidic deposition occurs in three forms: wet deposition, which falls as rain, snow, sleet, and hail; dry deposition, which includes particles, gases, and vapor; and cloud or fog deposition which occurs at high altitudes and coastal areas. In the United States wet deposition has been monitored at more than 200 sites, by both independent researchers and the inter-agency National Atmospheric Deposition Program/National Trends Network (http://nadp.sws.uiuc.edu/). Dry deposition is monitored at 70 sites in the United States by the U.S. Environmental Protection Agency Clean Air Status and Trends Network (http://www.epa.gov/castnet/) and at 13 other sites by the National Oceanic and Atmospheric Administration AIRMON-dry Network. Cloud and fog deposition has been monitored for limited periods at selected high-elevation sites, largely by independent researchers. Dry and cloud deposition patterns are extremely variable over space and time, making it difficult to characterize patterns. Therefore, even though cloud and dry deposition comprise a significant proportion of total deposition, this report primarily presents general patterns and trends of wet deposition. Some researchers also measure bulk deposition, which is collected in an open collector. Bulk deposition is greater than wet deposition because it includes some dry deposition.

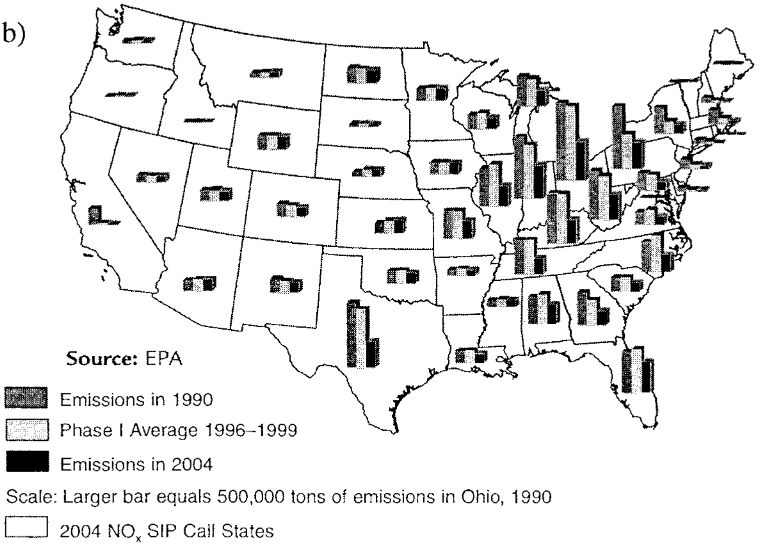
a)


State-by-State SO_2 Emissions Levels, 1990–2004

Source: EPA

Scale: Largest bar equals 2.2 million tons of emissions in Ohio, 1990

b)


State-by-State NO_x Emission Levels for All Acid Rain Program Sources 1990 to 2004

Source: EPA

Scale: Larger bar equals 500,000 tons of emissions in Ohio, 1990

Figure 3.1. State by state emissions of sulfur dioxide (a) and nitrogen oxides (b) in the United States. Values are shown for three periods: 1990, after Phase I (1996–1999) and after Phase II (2004) of control of utility emissions in response to 1990 Amendments of the Clean Air Act (after U.S. EPA 2005). Note the bars are scales to 1990 emissions for Ohio (2 million metric tons for sulfur dioxide and 454,000 tons for nitrogen oxides)

ments of the Clean Air Act (CAAA) led to a 52% decrease in sulfur dioxide emissions nationwide, to approximately 13.9 million metric tons in 2002. The multi-state source area for eastern North America has shown substantial decreases in sulfur dioxide emissions over this period (Figure 3.2). The 1990 CAAA set a cap of 14 million metric tons of total annual sulfur dioxide emissions to be achieved by 2010. The cap on electric utilities is set at 8.9 million metric tons and the cap on industrial sources is 5.6 million metric tons to be reached by 2010.

Nitrogen oxide emissions in the United States have increased over the past decades, peaking at nearly 22.7 million metric tons in 1990. From 1990 to 2002 nitrogen oxide emissions have decreased 12%. The 1990 CAAA calls for an additional reduction that will result in the emission of 1.8 million fewer tons of nitrogen oxide than the level that would have occurred without the CAAA. However, no cap on total annual emissions of nitrogen oxides was set. Nevertheless, it is expected that nitrogen emissions will decrease gradually in the future due to a variety of federal and state emission control programs. In contrast to sulfur dioxide, the multi-state source area for eastern North America has shown little change in nitrogen dioxide emissions since the early 1970s although some decrease has been evident in recent years (Figure 3.2).

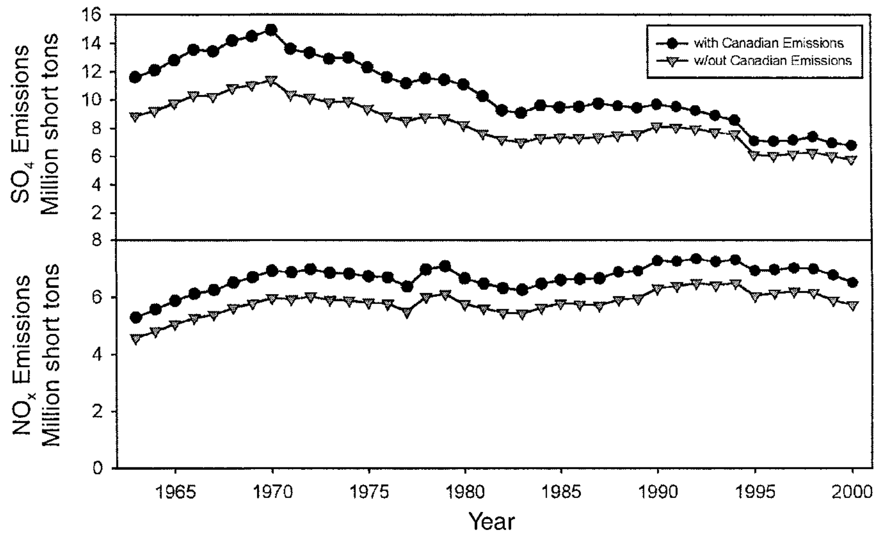


Figure 3.2. Annual emissions of sulfur dioxide and nitrogen oxides for the source area of the Hubbard Brook Experimental Forest. The source area was determined by 24-hour back trajectory analysis. Shown are emissions from both U.S. and Canadian sources

Ammonia emissions play an important role in the acidification of soil and surface waters. Deposition of ammonium accounts for approximately 30% of the total nitrogen deposition measured in eastern North America and has not changed appreciably over the past 30 years. Trends in U.S. ammonia emissions are consistent with this pattern and have shown little change over the past 10 years.

European efforts to reduce emissions of air pollutants have been brought together in a series of protocols under the United Nations Economic Council for Europe (UN/ECE) Convention on Long Range Transboundary Air Pollution (LRTAP; Sundqvist et al. 2002; Ferrier 2001). The first binding protocol was the 1985 Protocol on the Reduction of Sulfur Emissions, which was intended to reduce sulfur dioxide emissions at least 30% by 1993 compared to 1980 levels. This was expanded in 1994 with the Protocol on Further Reductions of Sulfur with the objective of decreasing sulfur dioxide emissions 80% by 2010 from 1980 values. In 1988 the Protocol Concerning Control of Emissions of Nitrogen Oxides stabilized nitrogen oxide emissions. Finally, the 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone established national caps for multiple air pollutants, including sulfur dioxide, nitrogen oxides and ammonia, with attainment expected by 2010 (Kurz et al. 2001). These actions have resulted in marked reductions in emissions of sulfur dioxide (65%) and nitrogen oxides (30%) from 1990 to 2002. Ammonia emissions have remained essentially constant over the same period. Once implemented, sulfur dioxide emissions will decrease by at least 63%, nitrogen oxide emissions by 41%, and ammonia emissions 17% relative to 1990 values.

An important framework for emission reductions and ecosystem recovery is "critical loads." Critical loads are based on the idea that emission control strategies should be effects-driven. A critical load is total deposition of a pollutant to an ecosystem below which significant harmful ecological effects are not known to occur (Nilsson and Grennfelt 1988). Following the UN/ECE Convention on LRTAP, maps of critical loads were developed for Europe (Posch et al. 1995). These maps have been revised (Posch et al. 2001) and the resulting critical loads were used to regulate emissions through the 1999 Protocol. Critical loads have only been applied to a limited extent in the United States. In contrast to North America and Europe, emissions in Asia have been increasing and are expected to continue to increase in the coming decades (Galloway 1995; Klimont et al. 2001).

Trends in acidic deposition mirror emission trends in the source area. For example, over the past 30 years in eastern North America, sulfate deposition has declined but nitrogen and ammonium deposition have remained relatively stable (Figure 3.3). Decreases in precipitation sulfate have coincided

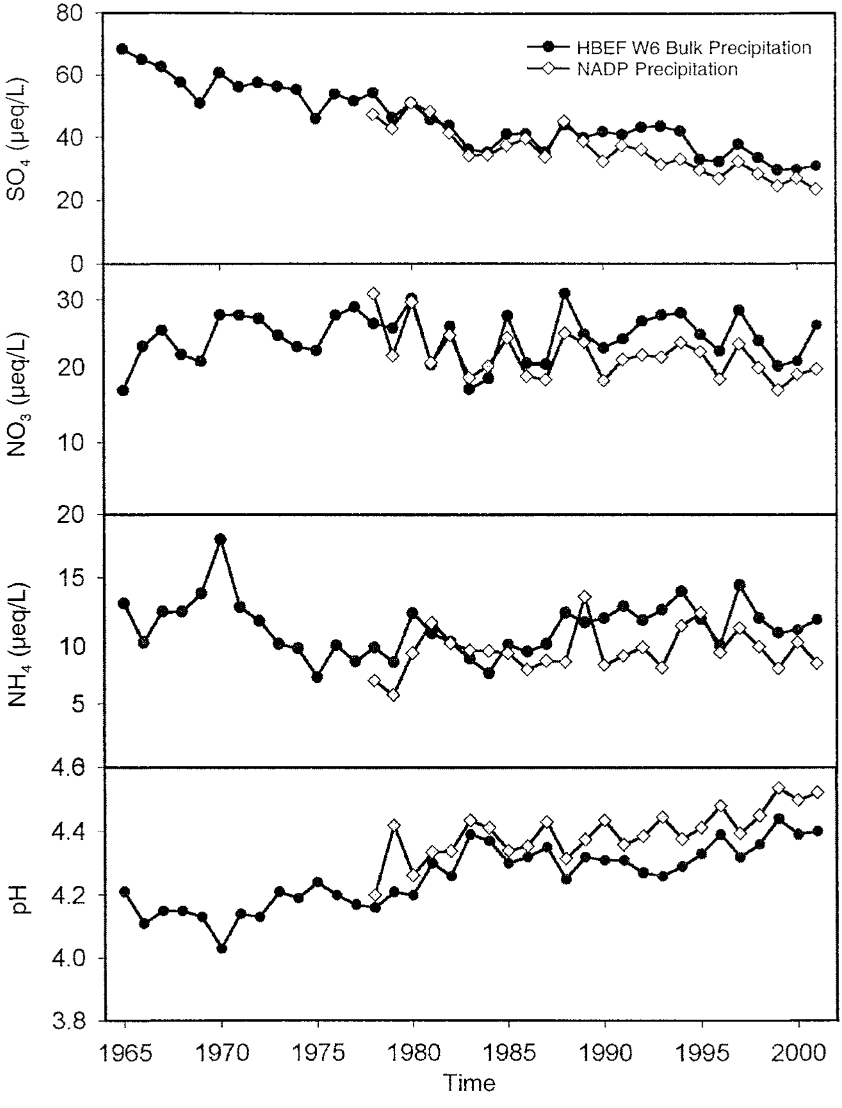


Figure 3.3. Annual volume-weighted sulfate, nitrate, and ammonium concentrations and pH in bulk and wet deposition at the Hubbard Brook Experimental Forest, New Hampshire 1963-2000

with increases in pH. The Hubbard Brook Experimental Forest in New Hampshire has one of the longest continuous records of precipitation chemistry (see Box 3.2). Long-term data from Hubbard Brook show declining concentrations of sulfate in bulk deposition since the mid-1960s and wet deposition since the late 1970s (see Figure 3.3). Based on these long-term data, there is a strong positive correlation between sulfur dioxide emissions in the source area and sulfate concentrations in precipitation at Hubbard Brook (Figure 3.4). It is now expected that the sulfate concentration of atmospheric deposition will decrease in a direct linear response to decreases of sulfur dioxide emissions in the source area.

The relationship between sulfur dioxide emissions and wet sulfate deposition extends throughout the eastern United States. The portion of the east-

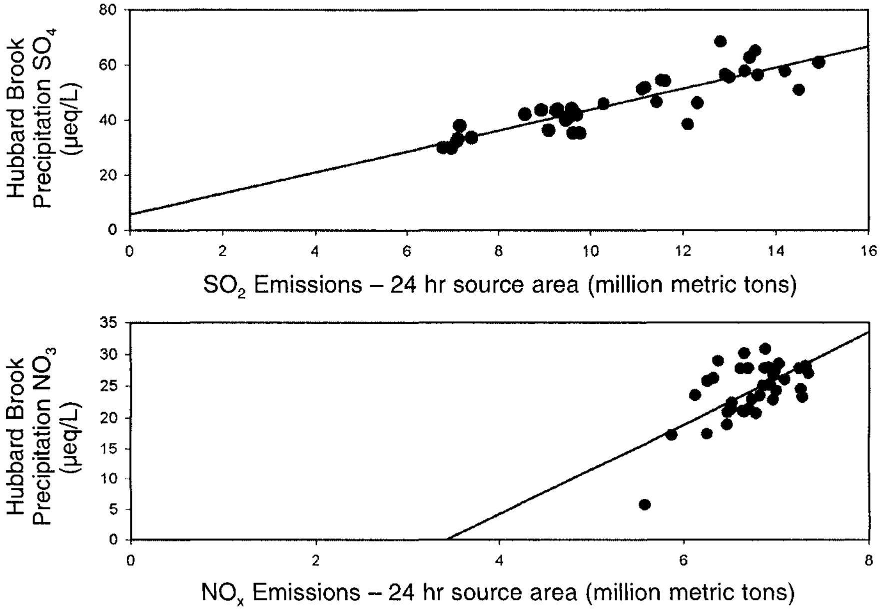


Figure 3.4. Relationships between sulfur dioxide and nitrogen oxide emissions for the source area of the Hubbard Brook Experimental Forest (see Figure 3.2) and annual volume-weighted concentrations of sulfate and nitrate in bulk deposition

Box 3.2. The Hubbard Brook Experimental Forest

The Hubbard Brook Experimental Forest is a long-term ecological research site established by the U.S. Department of Agriculture Forest Service in the White Mountains of New Hampshire to investigate the structure and function of forest and aquatic ecosystems, and their response to disturbance (Likens and Bormann 1995; Groffman et al. 2004). Hubbard Brook was the site where acidic deposition was first reported in North America (Likens et al. 1972). Hubbard Brook receives elevated inputs of acidic deposition and the forest ecosystem is very sensitive to these inputs. There have been long-term measurements and studies of acidic deposition and its effects on forests and streams at Hubbard Brook (Likens et al. 1996; Driscoll et al. 2001).

ern United States with high wet deposition of sulfate decreased markedly over the interval 1984–1986 to 2002–2004 (see Figure 3.5). These reductions in wet sulfate deposition are consistent with the emissions reductions called for in the 1990 CAAA.

In contrast to sulfate trends in wet deposition, concentrations of nitrate or ammonium at Hubbard Brook have not shown large changes since 1963 (see Figure 3.3). There is a relationship between nitrate concentrations in bulk deposition at Hubbard Brook and nitrogen oxide emissions in the source area (Butler et al. 2003), but the relationship is much weaker than observed for sulfate (Figure 3.4). This weak relationship is due to the fact that nitrogen oxide emissions and nitrate in bulk deposition have not changed much since measurements were initiated in 1963. Patterns of wet deposition of nitrogen at Hubbard Brook are consistent with the pattern across the entire eastern United States, which shows limited change over the last several years (see Figure 3.6).

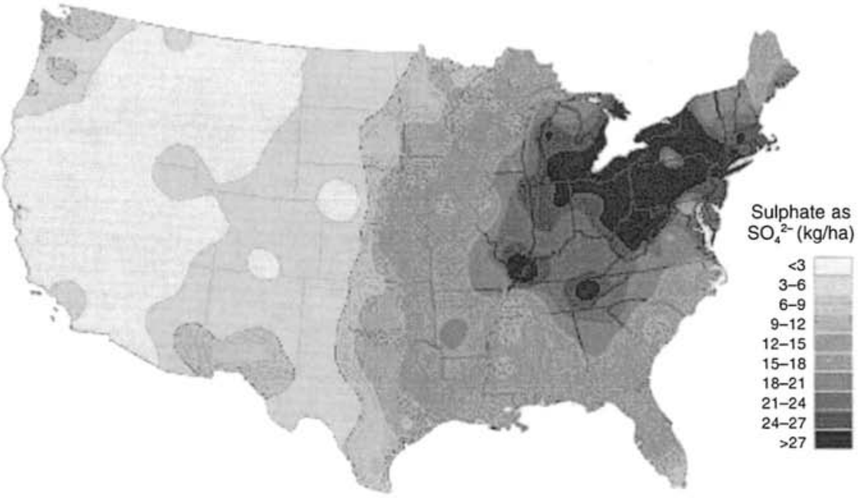
2. Effects of Acidic Deposition on Ecosystems

Acidic deposition alters soils, stresses forest vegetation, acidifies lakes and streams, and harms fish and other aquatic life. These effects can alter important ecosystem services such as forest productivity and water quality. Decades of acidic deposition have also made many ecosystems more sensitive to continuing pollution. Moreover, the same pollutants that cause acidic deposition contribute to a wide array of other important environmental issues at local, regional, and global scales (see Table 3.1).

2.1. Effects of Acidic Deposition on Forest Ecosystems

Until recently, understanding of the effects of acidic deposition on soils was limited. However, current research has shown that acidic deposition has chemically altered forest soils with serious consequences for acid-sensitive ecosystems. Soils compromised by acidic deposition lose their ability to neutralize continuing inputs of strong acids, provide poorer growing conditions for plants, and extend the time needed for ecosystems to recover from acidic deposition. Acidic deposition has altered and continues to alter base-poor forest soils in three important ways. Acidic deposition depletes available calcium and other nutrient cations (e.g., magnesium, potassium) from soil; facilitates the mobilization of dissolved inorganic aluminum into soil water; and increases the accumulation of sulfur and nitrogen in soil.

Sulphate Ion Wet Deposition 1984–1986



Sulphate Ion Wet Deposition 2002–2004

Figure 3.5. Annual sulfate in wet deposition in the eastern United States for 1984–1986 and 2002–2004. (data were obtained from the National Atmospheric Deposition Program). Note that there have been marked reductions in wet sulfate deposition in response to emission controls

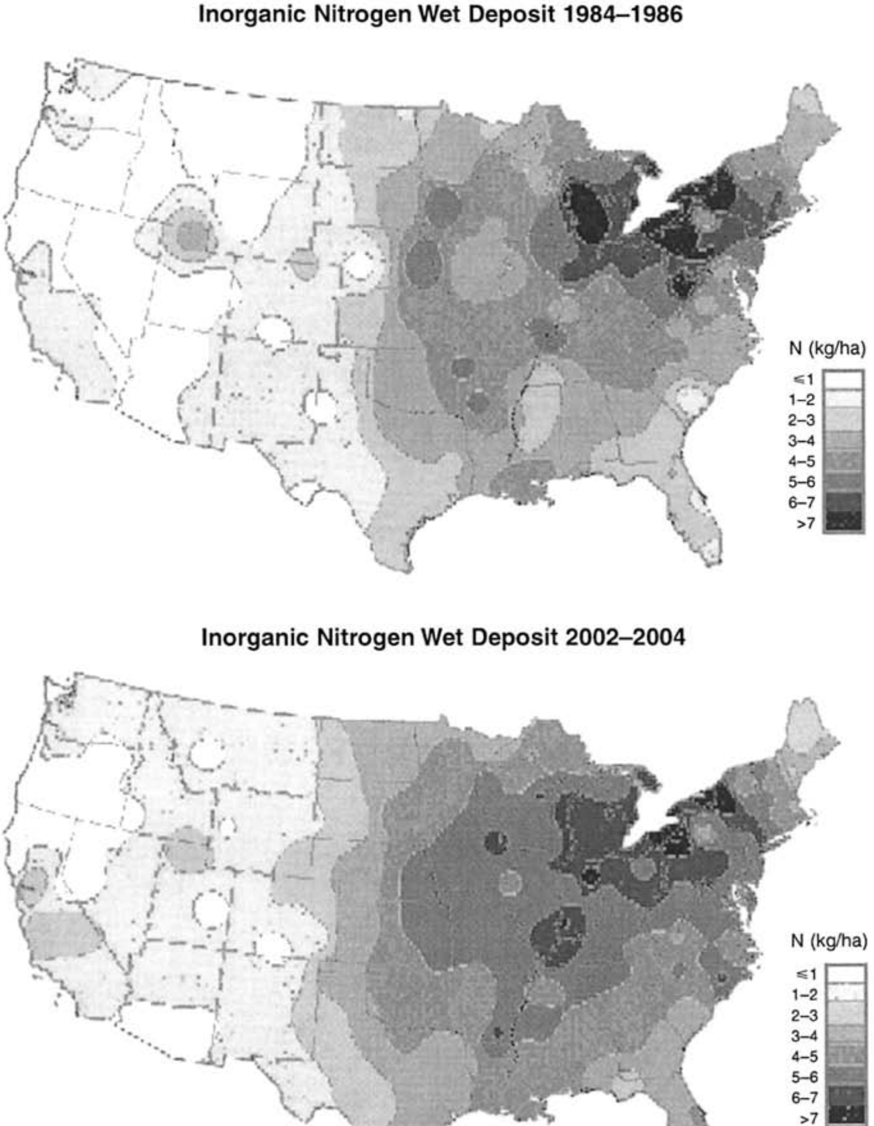


Figure 3.6. Annual inorganic nitrogen (ammonium plus nitrate) deposited in wet deposition in the eastern United States for 1984–1986 and 2002–2004 (data were obtained from the National Atmospheric Deposition Program). Note that there have been limited changes in wet deposition of inorganic nitrogen

Table 3.1. The links between sulfur dioxide and nitrogen oxide emissions, acidic deposition, and a range of environmental issues

Problem	Linkage to Acid Deposition	Reference
Coastal eutrophication	Atmospheric deposition adds nitrogen to coastal waters	Paerl et al. 2002, Howarth, this volume
Mercury	Deposition of sulfate enhances methylation of mercury. Surface water acidification increases mercury accumulation in fish	Branfireun et al. 1999, Driscoll et al. 1994
Visibility	Sulfate aerosols diminish visibility and views	Malm et al. 1994
Climate Change	Sulfate, nitrate and ammonium aerosols may offset global warming in the short-term, but nitrous oxide is a potent greenhouse gas	Moore et al. 1997 Dillon et al. this volume
Tropospheric ozone	Emissions of nitrogen oxides contribute to the formation of ozone	NAPAP 1998
Airborne particulate matter	Emissions of sulfur dioxide, nitrogen oxides and ammonia contribute to airborne particulate matter and associated health effects	http://www.epa.gov/air/urbanair/pm/index.html
Corrosion and damage to structures and monuments	Acidic substances enhance corrosion	Sherwood and Lipfert 1990

The cycling of calcium and other nutrient cations in forest ecosystems involves the inputs and losses of these materials (Figure 3.7). For most forest ecosystems the supply of calcium and other nutrient cations largely occurs by weathering (i.e., the breakdown of rocks and minerals in soil). Calcium and other nutrient cations may also enter forests by atmospheric deposition, although this pathway is generally much smaller than weathering. Losses largely occur by vegetation uptake and drainage waters. An important pool of ecosystem calcium and nutrient cations is the soil available pool or the soil cation exchange complex. Plants are generally able to utilize this source of nutrients. Forest ecosystems that are naturally sensitive to acidic deposition are generally characterized by low rates of weathering

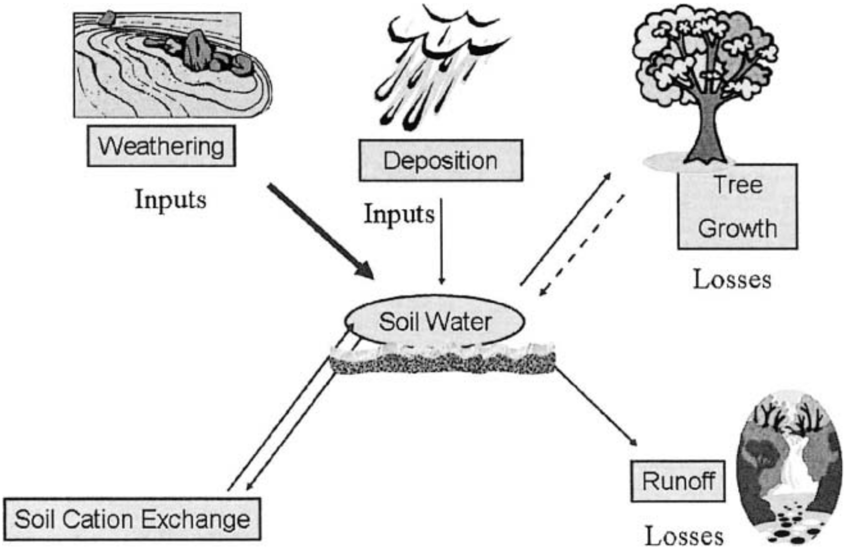


Figure 3.7. Conceptual diagram illustrating calcium cycle in forest watersheds. Inputs of calcium include weathering and atmospheric deposition; of these weathering is usually the greatest. Losses of calcium include tree accumulation and stream runoff. Under conditions of elevated acidic deposition stream losses increase, potentially depleting available calcium from the ecosystem, particularly from the soil exchange complex

and generally low quantities of available base cations (i.e., calcium, magnesium, sodium, potassium). Under conditions of elevated inputs of acidic deposition and subsequent transport of sulfate and nitrate in drainage waters, nutrient cations will be displaced from available pools and leached from soil (Ruess and Johnson 1986). This condition is not problematic for areas with high weathering rates and high pools of available nutrient cations. However, over the past century acidic deposition has accelerated the loss of large amounts of available calcium and magnesium from the soil in acid-sensitive areas (Likens et al. 1996; Kirchner and Lydersen 1995; Huntington et al. 2000). Depletion occurs when base cations are displaced from the soil by acidic deposition at a rate faster than they can be replenished by the slow breakdown of rocks or the deposition of base cations from the atmosphere. This depletion of base cations fundamentally alters soil processes, compromises the nutrition of some trees, and hinders the capacity for sensitive soils to recover from inputs of acidic deposition.

Dissolved inorganic aluminum is often released from soil to soil water, vegetation, lakes, and streams in forested regions with high acidic deposition, low stores of available calcium, high soil acidity and limited watershed retention of atmospheric inputs of sulfate and/or nitrate (Cronan and Schofield 1990). High concentrations of dissolved inorganic aluminum can be toxic to plants, fish, and other organisms. Concentrations of dissolved inorganic aluminum in streams and lakes in acid-sensitive regions receiving high inputs of acidic deposition are often above levels considered toxic to fish and much greater than concentrations observed in forested watersheds with low inputs of acidic deposition (Driscoll et al. 1988).

Acidic deposition results in the accumulation of sulfur and nitrogen in forest soils. As sulfate is released from the soil in response to decreases in emissions and atmospheric deposition of sulfur, it is transported to adjacent streams and lakes (Likens et al. 2000). The recovery of surface waters in response to emission controls has therefore been delayed and will not be complete until the sulfate left by a long legacy of acidic deposition is released from the soil.

Similarly, nitrogen has accumulated in soil beyond the amount needed by the forest and appears now to be leaching into surface waters in Europe and North America (Dise and Wright 1995; Aber et al. 2003; Nadelhoffer, this volume). Forests typically require more nitrogen for growth than is available in the soil. However, several recent studies suggest that in some areas, nitrogen levels are above what forests can use and retain. This condition is referred to as "nitrogen saturation" (Aber et al. 1989; 1998). Note that the levels at which atmospheric nitrogen deposition can result in elevated leaching losses of nitrate from forest watersheds appear to be higher in Europe (9–25 kg N/ha-yr) than eastern North America (7–8 kg N/ha-yr). The reason for this difference is not evident but may be due to a greater fraction of atmospheric nitrogen deposition occurring as ammonium in Europe; ammonium inputs are more readily retained in watersheds than nitrate. Alternatively, this difference may be due to highly managed forests in Europe and greater nitrogen retention compared to eastern North America.

2.2. Acidic Deposition Stress to Trees

Although it is difficult to separate the effects of air pollution from other stresses, recent research shows that acidic deposition appears to have contributed to the decline of red spruce trees throughout eastern North America and sugar maple trees in central and western Pennsylvania in the United States. Symptoms of tree decline include poor crown condition, reduced tree growth, and unusually high levels of tree mortality.

Red spruce and sugar maple are the tree species that have been most intensively researched, therefore they provide instructive case studies of the effects of acidic deposition on trees. Red spruce and sugar maple research has shown that acidic deposition has both direct and indirect effects on trees. In acid-impacted forests, acidic deposition harms trees directly by leaching calcium from the leaves and needles (i.e., foliage) of trees, rendering them more susceptible to winter injury. Acidic deposition can also affect trees indirectly by changing the underlying soil chemistry. In acid-sensitive soils, acidic deposition depletes available nutrient cations, such as calcium and magnesium, which are important plant nutrients that are necessary to maintain the health and vigor of trees. The depletion of nutrient cations also leads to soil acidification, which increases the availability of aluminum to the trees' roots thereby impairing the ability of trees to obtain necessary nutrients from the soil.

2.2.1. Red Spruce

Acidic deposition appears to be the major cause of red spruce decline at high elevations in North America. Since the 1960s, more than half of large canopy trees in the Adirondack Mountains of New York and the Green Mountains of Vermont and approximately one quarter of large canopy trees in the White Mountains of New Hampshire have died. Significant growth declines and winter injury to red spruce have been observed throughout its range, suggesting that damage from acidic deposition is likely widespread (DeHayes et al. 1999).

Recent research indicates that the decline of red spruce is linked to the leaching of calcium from cell membranes in spruce needles by acidic deposition (DeHayes et al. 1999). The loss of calcium renders the needles more susceptible to freezing damage, thereby reducing the tolerance of trees to low temperatures and increasing the occurrence of winter injury and subsequent tree damage or death. In addition, elevated aluminum concentrations in the soil, resulting from soil acidification, limits the ability of red spruce to take up water and nutrients through its roots. This limitation can lead to nutrient deficiencies that can lower a tree's tolerance to environmental stress and cause decline.

2.2.2. Sugar Maple

The decline of sugar maples has been studied in the eastern United States since the 1950s and there is growing evidence that sugar maple decline is linked to acidic deposition. Extensive mortality among sugar maples in Pennsylvania appears to result from deficiencies of base cations, coupled with other stresses such as insect defoliation or drought. Sugar maples are most prone to die on sites where base cation concentrations in soil or foliage are lowest (Horsley et al. 2000). Data from many acid-sensitive regions link the loss of soil calcium and magnesium with the leaching of these base cations by acidic deposition. Low levels of base cations can cause a nutrient imbalance and reduce a tree's stress tolerance. As such, acidic deposition is a predisposing factor in sugar maple decline. Under these conditions, the likelihood increases that stresses such as insect infestation and drought will cause dieback of a tree's crown or kill a tree.

Finally, there may be adverse effects on other tree species. For example, one might speculate that hardwood species such as white ash and basswood that prefer rich sites high in nutrient cations may experience problems in areas where nutrient cations have been depleted by acidic deposition. However, additional research is needed to assess more fully the response of these tree species to acidic deposition.

2.3. Effects of Acidic Deposition on Aquatic Ecosystems

Acidic deposition degrades water quality by lowering pH levels (i.e., increasing acidity); decreasing acid-neutralizing capacity (ANC; see Box 3.3); and increasing dissolved inorganic aluminum concentrations. While sulfate concentrations in lakes and streams have decreased over the last 20 years, they remain high compared to background conditions (< 20 μeq/L; Sullivan 1991).

An important characteristic influencing the acid-base status of surface waters is the supply of naturally occurring organic solutes. These materials include organic acids, which decrease the ANC of waters and mobilize aluminum from soil through complexation reactions. Watersheds with an abundance of wetlands typically have high concentrations of dissolved organic carbon and associated surface waters can be naturally acidic due to

Box 3.3. What is ANC?

Acid-neutralizing capacity, or ANC, is the ability of water from a lake or stream to neutralize strong acid (Stumm and Morgan 1996). ANC is an important measure of the impacts of acidic deposition as well as an indicator of chemical recovery from acidic deposition. Surface waters with ANC values below 0 μeq/L during base flow conditions are considered chronically acidic. Waters with ANC values ranging from 0–50 μeq/L are susceptible to episodic acidification. Waters with ANC values greater than 50 μeq/L are less sensitive to acidic deposition. The capacity of a watershed to prevent decreases in ANC and resist the effects of acidic deposition depends on many factors, including climate, soil conditions, surficial and bedrock geology, and land-use history.

organic acids. However in contrast to watersheds which are acidified by acidic deposition, waters that are acidic due to organic acids have high concentrations of aluminum which is largely complexed with organic solutes and therefore less toxic to aquatic biota (Driscoll et al. 1980; 1988).

Acidification of surface waters due to elevated inputs of acidic deposition have been reported in many acid-sensitive areas receiving elevated inputs of acidic deposition, including Great Britain, Nordic countries, Northern, Central and Eastern Europe (Evans et al. 2001), southwestern China (Seip et al. 1995), southeastern Canada (Jeffries 1991), the northeastern United States (Driscoll 1991) the Upper Midwest (Cook and Jager 1991) and the Appalachian mountain region of the United States (Elwood 1991). Large portions of the high elevation western United States are also potentially sensitive to acidic deposition (Fenn et al. 2003), however, atmospheric deposition to this region is relatively low. Concern over effects of acidic deposition in the mountain western United States may be overshadowed by potential effects of elevated nitrogen deposition, including eutrophication of naturally nitrogen-limited lakes.

One of the most highly impacted areas in North America is the Adirondack region of New York. A comprehensive survey of Adirondack lakes greater than 0.2 ha in surface area was conducted between 1984–87 to obtain detailed information on the acid-base status of waters in this region (Kretser et al. 1989). Of the 1469 lakes surveyed, 24% had summer pH values below 5.0. Also 27% of the lakes surveyed were chronically acidic (i.e., ANC < 0 μeq/L) and an additional 21% were susceptible to episodic acidification (i.e., ANC between 0 and 50 μeq/L; see Box 3.4). Note that 54% of these acid-sensitive lakes (733 lakes) are characterized by relatively low concentrations of dissolved organic carbon (i.e., < 6 mg C/L). The chemical composition of these lakes suggests that their acidity was largely derived from inputs of sulfate associated with acidic deposition (Driscoll et al. 2003). In contrast, 46% of the lakes are characterized by high concentrations of dissolved organic carbon (i.e., > 6 mg C/L) and naturally occurring

Box 3.4. Seasonal and episodic acidification

Seasonal acidification is the periodic increase in acidity and the corresponding decrease in pH and ANC in streams and lakes. Episodic acidification is caused by the sudden pulse of acids due to spring snowmelt and large rain events in the spring and fall. Increases in nitrate are important to the occurrence of acid episodes and tend to occur when trees are dormant and therefore using less nitrogen. Short-term increases in acid inputs to surface waters can reach levels that are lethal to fish and other aquatic organisms.

organic acids. These lakes are probably naturally acidic. While the contribution of naturally occurring acidity is greater in these lakes, sulfate was the dominant anion; the acidity of these lakes has been clearly enhanced by acidic deposition.

Decreases in pH and elevated concentrations of dissolved inorganic aluminum have reduced the species diversity and abundance of aquatic life in many streams and lakes in acid-sensitive areas (Table 3.2). Fish have received the most attention to date, but entire food webs are often adversely affected (Baker et al. 1990).

Decreases in pH and increases in dissolved inorganic aluminum concentrations have diminished the species diversity and abundance of plankton, invertebrates, and fish in acid-impacted surface waters. For example, in the Adirondacks a significant positive relationship exists between the pH and ANC levels in lakes and the number of fish species present in those lakes (see Figure 3.8). The Adirondack Lakes Survey showed that 24% of lakes (i.e., 346) in this region do not support fish. These lakes had consistently

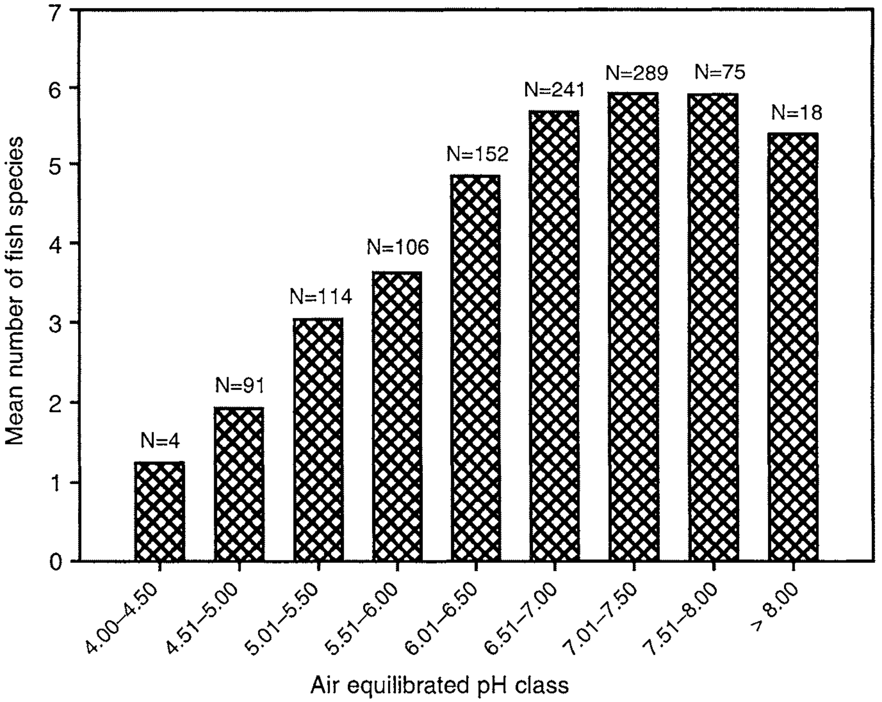


Figure 3.8. The mean number of fish species for pH classes from 4.0 to 8.0 in lakes in the Adirondack region of New York. N represents the number of lakes in each pH class (after Driscoll et al. 2001)

Table 3.2. Biological effects of surface water acidification (after Baker et al. 1990, an Adirondack lake survey)

pH Decrease	General Biological Effects
6.5 to 6.0	Small decrease in species richness of phytoplankton, zooplankton, and benthic invertebrate communities resulting from the loss of a few highly acid-sensitive species, but no measurable change in total community abundance or production Some adverse effects (decreased reproductive success) may occur for highly acid-sensitive species (e.g., fathead minnow, striped bass)
6.0 to 5.5	Loss of sensitive species of minnow and dace, such as blacknose dace and fathead minnow; in some waters decreased reproductive success of lake trout and walleye, which are important sport fish species in some areas Visual accumulations of filamentous green algae in the littoral zone of many lakes, in some streams Distinct decrease in the species richness and change in species composition of the phytoplankton, zooplankton, and benthic invertebrate communities, although little if any change in total community biomass or production
5.5 to 5.0	Loss of several important sport fish species, including lake trout, walleye, rainbow trout, and smallmouth bass; as well as additional non-game species such as creek chub Further increase in the extent and abundance of filamentous green algae in lake littoral areas and streams Continued shift in the species composition and decline in species richness of the phytoplankton, periphyton, zooplankton, and benthic invertebrate communities; decrease in the total abundance and biomass of benthic invertebrates and zooplankton may occur in some waters Loss of several additional invertebrate species common in oligotrophic waters, including *Daphnia galeata mendotae, Diaphanosoma leuchtenbergianum, Asplanchna priodonta;* all snails, most species of clams, and many species of mayflies, stoneflies, and other benthic invertebrates Inhibition of nitrification

table continues on next page

Table 3.2. (cont.)

5.0 to 4.5	Loss of most fish species, including most important sport fish species such as brook trout and Atlantic salmon; few fish species able to survive and reproduce below pH 4.5 (e.g., central mud minnow, yellow perch, and in some waters, largemouth bass) Measurable decline in the whole-system rates of decomposition of some forms of organic matter, potentially resulting in decreased rates of nutrient cycling Substantial decrease in the number of species of zooplankton and benthic invertebrates and further decline in the species richness of the phytoplankton and periphyton communities; measurable decrease in the total community biomass of zooplankton and benthic invertebrates in most waters Loss of zooplankton species such as *Tropocyclops prasinus mexicanus, Leptodora kindtii,* and *Conochilis unicornis;* and benthic invertebrate species, including all clams and many insects and crustaceans Reproductive failure of some acid-sensitive species of amphibians such as spotted salamanders, Jefferson salamanders, and the leopard frog

lower pH and ANC, and higher concentrations of dissolved inorganic aluminum than lakes that contained one or more species of fish. Experimental studies and field observations demonstrate that even acid-tolerant fish species such as brook trout have been eliminated from some waters.

Although chronically high acid levels stress aquatic life, acid episodes are particularly harmful because abrupt, large changes in water chemistry allow fish few areas of refuge (see Box 3.4). High concentrations of dissolved inorganic aluminum are directly toxic to fish and pulses of dissolved inorganic aluminum during acid episodes are a primary cause of fish mortality (Baker et al. 1996; van Sickle et al. 1996). High acidity and dissolved inorganic aluminum levels disrupt the salt and water balance in a fish's blood, causing red blood cells to rupture and blood viscosity to increase (MacAvoy and Bulger 1995). Studies show that the viscous blood strains the fish's heart, resulting in a lethal heart attack.

3. Ecosystem Recovery

Recovery from acidic deposition involves decreases in emissions resulting from regulatory controls, which in turn lead to reductions in acidic deposition and allow chemical recovery. The chemical recovery process is characterized by decreases in concentrations of sulfate, nitrate, and dissolved inorganic aluminum in soils and surface waters. If sufficient, these reductions will eventually lead to increased pH and ANC, as well as higher concentrations of base cations in water and on the soil exchange complex. As chemical conditions improve, the potential for the second phase of ecosystem recovery, biological recovery, is greatly enhanced.

An analysis of the scientific literature suggests that five thresholds can serve as indicators of chemical recovery (see Table 3.3). If chemical conditions in an ecosystem are above these thresholds, it is unlikely that the ecosystem has been substantially impaired by acidic deposition. Conversely, if chemical conditions are below these thresholds, there is a high likelihood that the ecosystem is vulnerable to acidic deposition.

The time required for chemical recovery varies widely among ecosystems, and is primarily a function of:

- the historic loading rate of sulfur and nitrogen oxides;
- the rate and magnitude of decreases in acidic deposition;
- the extent to which available base cations such as calcium have been depleted from soil;
- the extent to which sulfur and nitrogen have accumulated in the soil and the rate at which they are released as deposition declines;
- the weathering rate of the soil and underlying rock and the associated supply of base cations to the ecosystem; and
- the rate of atmospheric deposition of base cations.

As chemical conditions in soils and surface waters improve, *biological recovery* is enhanced. Biological recovery is likely to occur in stages, since not all organisms can recover at the same rate and may vary in their sensi-

Table 3.3. Indicators of chemical recovery from acidic deposition

Ecosystems	Indicators of recovery
Forest	Soil base saturation of 20% or higher (i.e., the percent of available cations in the soil that are bases) Calcium to aluminum molar ratio in the soil solution of 1.0 or greater
Aquatic	Stream and lake pH of 6.0 or higher (except where pH is lower under background conditions) Stream or lake ANC of 50 μeq/L or higher Stream or lake concentrations of dissolved inorganic aluminum less than 2 μmol/L

tivity to acidic deposition. The current understanding of the response of biological species to improvements in chemical conditions is incomplete, but research suggests that stream macro-invertebrates may recover relatively rapidly (i.e., within three years), while lake zooplankton may need a decade or more to fully re-establish. Fish populations in streams and lakes should recover in 5–10 years following the recovery of the macro-invertebrates and zooplankton, which serve as food sources (Gunn and Mills 1998). It is possible that, with improved chemical conditions and the return of other members of the aquatic food web, the stocking of streams and lakes could help to accelerate the recovery of fish.

Terrestrial recovery is even more difficult to project than aquatic recovery. Given the life span of trees and the delay in the response of soil to decreases in acidic deposition, it is reasonable to suggest that decades will be required for affected trees on sensitive sites to recover once chemical conditions in the soil are restored. Overall, the timing and extent of chemical and biological recovery depend on how soon and how significantly emissions that cause acidic deposition are reduced. Moreover, human influences (e.g., land disturbance, introduction of exotic or invasive species), in addition to acidic deposition, can delay biological recovery after chemical recovery has occurred.

Long-term stream data from Hubbard Brook reveal a number of long-term trends that are consistent with trends in lakes and streams across Europe and eastern North America (Stoddard et al. 1999; Evans et al. 2001; Stoddard et al. 2003; see Figure 3.9). Specifically, the concentration of sulfate in streams at Hubbard Brook declined 32% between 1963–2000. The pH of streams subsequently increased from 4.8 to 5.0. Although this represents an important improvement in water quality, streams at Hubbard Brook remain acidic compared to background conditions, when stream pH was estimated to be approximately 6.0. Moreover, the ANC at Hubbard Brook-a biologically important measure of a lake or stream's susceptibility to acid inputs-has remained acidic (ANC< 0 μeq/L).

Trends in surface water chemistry in Europe (Evans et al. 2001) and eastern North America (Stoddard et al. 1999) indicate that recovery of aquatic ecosystems impacted by acidic deposition has been occurring over a large geographic scale since the early 1980s. Some regions are showing rather marked recovery, while others exhibit low or non-existent increases in ANC. Based on long-term monitoring, virtually all surface waters impacted by acidic deposition in Europe and Eastern North America exhibit decreases in sulfate concentrations. This pattern is consistent with decreases in emissions of sulfur dioxide and atmospheric sulfate deposition. The exception to this

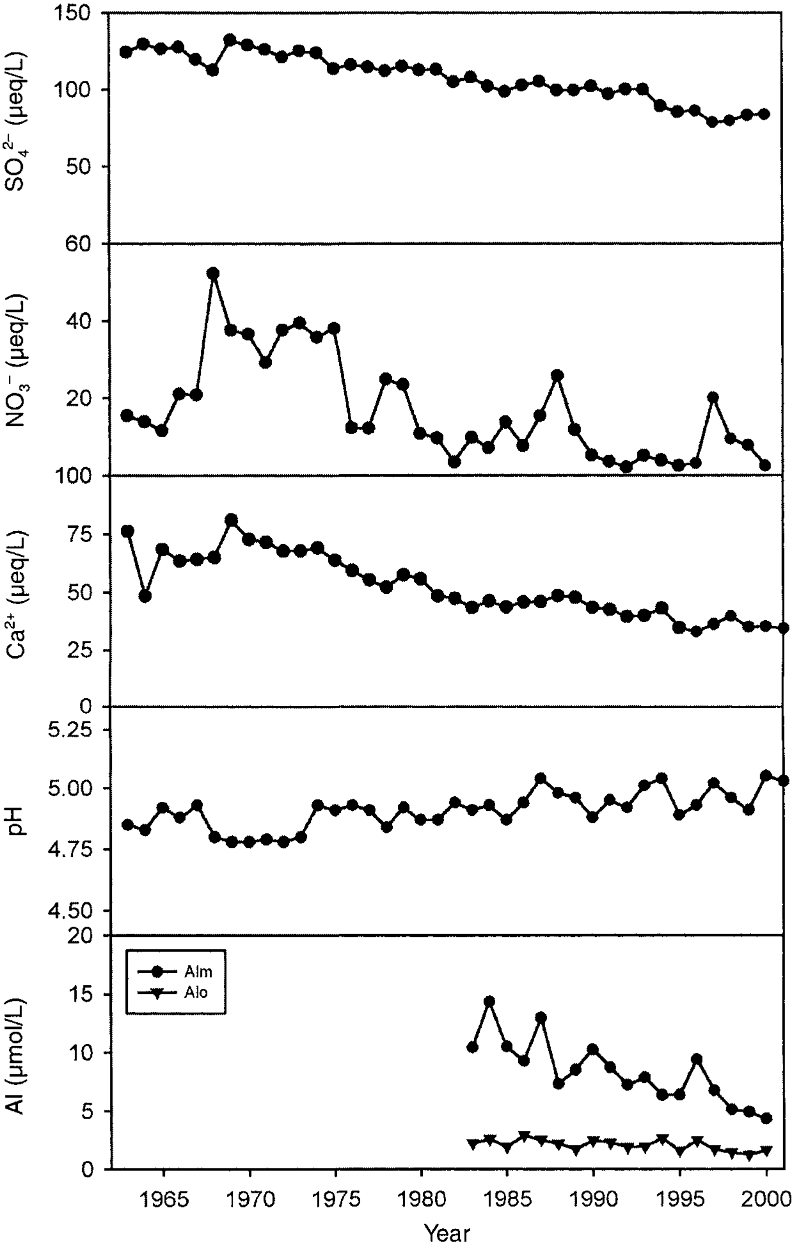


Figure 3.9. Annual volume-weighted stream water sulfate, nitrate, calcium concentrations, pH, and concentrations of total (Alm) and organic dissolved aluminum (Alo) at the reference watershed of the Hubbard Brook Experimental Forest from 1963-2000. Note that dissolved inorganic aluminum is the difference between total and organic dissolved aluminum

pattern is streams in un-glaciated Virginia. Watersheds in this region exhibit strong adsorption of atmospheric sulfate deposition by highly weathered soils. In Europe the most marked decreases in surface water sulfate have occurred in the Czech Republic and Slovakia, regions that experienced historically very high rates of atmospheric sulfate deposition. Somewhat more than half of the surface waters monitored in Europe are showing increases in ANC (Evans et al. 2001). The rate of ANC increase in Europe is relatively high. This pattern is due in part to the relatively high rates of sulfate decreases, but also the fact that decreases in base cations only account for about half of the decreases in sulfate plus nitrate, allowing for relatively large rates of ANC increases. In contrast in the United States only three regions are showing statistically significant increases in ANC; lakes in the Adirondacks and Upper Midwest and streams in Northern Appalachian Plateau (Stoddard et al. 2003). In the United States decreases in the sum of base cations closely correspond to decreases in sulfate plus nitrate, limiting rates of ANC increase.

Three factors have limited the recovery in chemical water quality at Hubbard Brook and other watersheds in acid-sensitive regions that have received elevated inputs of acidic deposition. First, levels of acid-neutralizing base cations in surface waters have decreased markedly due to the depletion of available base cations from the soil and, to a lesser extent, a reduction in atmospheric inputs of base cations. Second, as forest ecosystems mature, their requirement for nitrogen decreases (Aber et al. 1989; Aber et al. 1998). As a result, forested watersheds with limited disturbance that extracts nitrogen (e.g., tree harvesting, fire, agriculture) are expected to exhibit increasing losses of nitrate as forests develop. Finally, sulfur has accumulated in the soil under previous conditions of high atmospheric sulfur deposition and is now being released to surface water as sulfate, even though sulfate deposition has decreased.

While there is considerable information about the response of surface waters to decreases in acidic deposition under chronic conditions, much less is known about how episodic acidification responds to these changes. Laudon and Hemond (2002) reported decreases in episodic acidification following decreases in atmospheric sulfur deposition in northern Sweden. Unfortunately, comparable data sets have not been developed for other regions.

An alternative to recovery from controls on emissions of acidic or acid-forming substances is mitigation. Mitigation (base addition or liming) involves the application of basic materials directly to surface waters or watersheds to neutralize strong acid inputs (Olem et al. 1991). The most common material for mitigation is calcium carbonate (or limestone), although other materials have been effectively used. Mitigation has been practiced in Europe and North America to treat the effects of acidic deposition with some success. Direct application to lakes has been shown to neutralize acidity and allow for the survival of fish and other sensitive aquatic biota. This approach is less successful in lakes with short hydraulic residence times and in the recovery of a reproducing fish population (Driscoll et al. 1996). Alternatively, watershed treatment has been shown to be successful over the longer term by improving the base status of soil and allowing for a reproducing fish population. Note, mitigation is not an attractive alternative to source control. It might be implemented in areas that exhibit severe depletion of exchangeable nutrient cations and/or have biological species that are endangered due to acidic deposition. Acidic deposition impacts ecosystems in remote and wilderness areas that are difficult or inappropriate to treat by base addition. While no negative short-term effects of base treatment have been noted, long-term studies of the ecological response to base treatment have not been conducted.

4. Recovery of Acid-Sensitive Ecosystems with Future Decreases in Emissions

To date, emissions targets set in the United States and Europe have been met or exceeded. There are widespread decreases in surface water concentrations of sulfate and some waters are showing increases in ANC. Nevertheless, data suggest that these targets may not be sufficient to achieve the full recovery of sensitive ecosystems. In order to evaluate the extent to which historic and future emissions reductions will facilitate ecosystem recovery from acidic deposition, it is necessary to use acidification models to project the future relationship between emissions, deposition, and chemical recovery of acid-sensitive forest watersheds (see Box 3.5).

We used the model PnET-BGC (Gbondo-Tugbawa et al. 2001) to compare current emissions reductions required by the 1990 CAAA with an additional 55% and 75% cut in emissions of sulfur dioxide, and 20% and 30% decreases in nitrogen oxides by 2010. These scenarios are based on the electric utility emission reductions embodied in bills recently introduced to the U.S. Congress. PnET-BGC considered changes in sulfur dioxide and nitrogen oxide emissions. It was assumed that base cation and ammonium deposition and climate would remain unchanged.

According to the results of the computer model, the 1990 CAAA will have a positive effect on stream concentrations of sulfate at Hubbard Brook, but will not facilitate appreciable progress toward chemical recovery of key indicators of acidification stress, such as pH or ANC (see Figure 3.10). With moderate reductions in emissions (i.e., sulfur dioxide 55%, nitrogen oxides 20%) beyond the requirements of the 1990 CAAA, measurable chemical improvements occur. However, none of the five indicators reaches the threshold needed to support complete biological recovery at Hubbard Brook by 2050 (see Table 3.3). More aggressive reductions in emissions (i.e., sulfur dioxide 75%, nitrogen oxides 30%) beyond the 1990 CAAA hasten and promote more significant improvements in chemical conditions. For example, under this scenario streams in watersheds similar to Hubbard Brook would change from acidic to non-acidic in roughly 20–25 years. By 2050, the concentration of aluminum and the base cation content of the soil in these watersheds would begin to approach recovery thresholds or pre-industrial levels.

The model results suggest that full implementation of the 1990 CAAA will not bring about substantial improvements in chemical recovery at Hubbard Brook. The results further demonstrate that the process of recovery will be slow, particularly for sensitive systems such as Hubbard Brook. Similar analyses have been conducted at regional scales using PnET-BGC for the Adirondack region of New York (Chen and Driscoll 2005a) and northern New England (Chen and Driscoll 2005b), with similar results obtained. Other analyses have been conducted to evaluate the response of watersheds in Canada and Europe to future emission reductions (Wright 2003).

In sum, acidic deposition is a pervasive problem that has had a greater impact on soils, terrestrial vegetation, surface waters, and aquatic biota than previously projected. Although abatement strategies in Europe and North America have had positive effects, emissions remain high compared to

Box 3.5. Acidification models

Scientists have developed computer models that depict the physical, chemical and biological processes within forest watersheds. Watershed acidification models can be used as research and management tools to investigate factors responsible for the historical acidification of soil and water as well as the ecosystem response to anticipated future changes in acidic deposition. In order to effectively predict the pH, ANC and aluminum concentrations in streams, all major chemicals must be accurately simulated (e.g., sulfate, nitrate, calcium, magnesium). The acidification model PnET-BGC was used for this assessment because it has been rigorously tested at Hubbard Brook and other sites in the northeastern United States, and it allows the user of the model to consider the ecosystem response to multiple chemicals simultaneously. Other frequently used acidification models include MAGIC (Cosby et al. 2001), and NuCM (Lui et al. 1992).

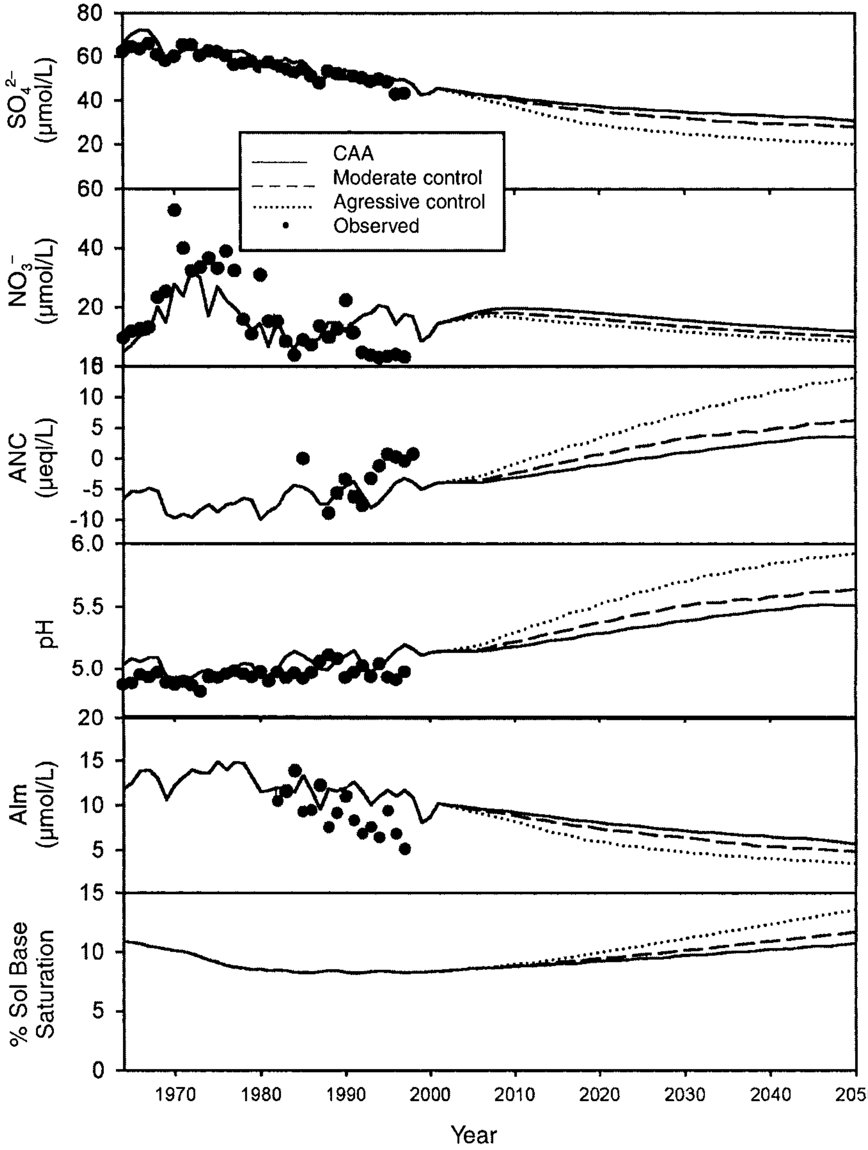


Figure 3.10. Time series of predictions with the acidification model PnET-BGC of changes in stream chemistry at Hubbard Brook to changes in past and potential future emissions of sulfur dioxide and nitrogen oxides, including the 1990 Amendments of the Clean Air Act and moderate and aggressive emission control scenarios. Shown are model-predicted stream concentrations of sulfate, nitrate, acid neutralizing capacity, pH and dissolved inorganic aluminum, and soil percent base saturation. Measured values are indicated for comparison

background conditions. Given the accumulation of acids and loss of buffering capacity in the soil, many areas are now more sensitive to acidic deposition and have developed an inertia that will delay recovery. Nevertheless, calculations from computer models show that deeper emissions cuts will lead to greater and faster recovery from acidic deposition.

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4. Long-Term Changes in Boreal Lake and Stream Chemistry: Recovery From Acid Deposition and the Role of Climate

Peter J. Dillon, Shaun A. Watmough, M. Catherine Eimers
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1. Introduction

In Canada, nearly 45% of the land area is considered sensitive to acid deposition. Lakes and watersheds located on the Canadian Shield are considered the most vulnerable, due to the low buffering capacity of the typically shallow soils that overlay the Shield bedrock. A large portion of eastern Canada, including much of Ontario and Quebec as well as parts of the Atlantic Provinces are underlain by silicate bedrock; these areas also receive the highest levels of acid deposition in the country.

The detrimental effects of acid deposition on both the aquatic and terrestrial environments in Europe and North America were documented extensively in the 1980s. As a result, sulfur (S) emission control programs were put into effect in both continents. In Canada, sulfur dioxide (SO_2) emissions declined from 3.81 million tons in 1980 to 2.52 million in 1990, 1.74 million in 1996 and 1.25 million in 2000, a drop of 68% over 20 years. Over the same time period, emissions of SO_2 in the United States dropped from 17.3 million tons in 1980, the baseline year for the Clean Air Act amendments of 1990, to 15.7 million in 1990, and 10.6 million in 2001, a total drop over 21 years of 39% (Stoddard et al. 2003). An additional drop of 3.3 million tons by 2010 will bring the total decrease to 10 million tons or almost 60%.

It is now widely accepted that S emission control programs have led to reductions in S deposition in most regions of Europe, the United States (Driscoll et al. 1995, Stoddard et al. 1999) and Canada (Bouchard 1997; Dillon et al. 2003a), and generally in lower concentrations of sulfate (SO_4^{2-}) in surface waters, the simplest and most direct measure of lake recovery, in both the United States (Driscoll and van Dreason 1993; Stoddard et al. 2003) and Canada (Dillon and LaZerte 1992; Jeffries et al. 2000; Dillon et al. 2003a). However, the declines in surface water SO_4^{2-} concentrations often have been much less than those anticipated based on the reduction in

S deposition (Driscoll et al. 1995; Dillon et al. 1997; Dillon and Evans 2001; Löfgren et al. 2001). Furthermore, other measures of chemical recovery, particularly increases in lake pH and acid neutralizing capacity (ANC), have provided conflicting evidence. In a few cases (Webster and Brezonik 1995; Stoddard et al. 1998), they have indicated limited evidence of chemical recovery, while in many others (Clair et al. 1995; Houle et al. 1996; Dillon et al. 1997), there has been little or no change in these parameters despite declining S deposition.

One factor that has been linked with long-term changes in the chemistry of surface waters and soils is the influence of variations in weather, particularly temperature and precipitation. The variability that fluctuations in weather induce in surface water chemistry (Hindar et al. 2003) may obscure the detection of trends resulting from reductions in SO_4^{2-} deposition. On the other hand, use of long-term data may obscure shorter trends occurring within the series, particularly if the trends are not uni-directional. In this case, the detection of trends may be dependent on the time window used in the analysis, and can affect both the detection of the presence and the direction of a trend (Clair et al. 2002). For example, Clair et al. (1992) found significant decreases in SO_4^{2-} concentrations in lakes in Nova Scotia, Canada, for the period 1983–1989, but with the addition of two more years of data, many of these trends were reversed (Clair et al. 1995).

The effects of intra- and inter-annual fluctuations in weather are further complicated by the effects of longer term climate and climate change. In North America, droughts in south and central Ontario (Dillon and LaZerte 1992; Dillon et al. 1997; Devito et al. 1999; Dillon and Evans 2001), in northwestern Ontario (Schindler et al. 1996) and in the upper midwest US (Webster and Brezonik 1995) were hypothesized to cause re-oxidation of reduced S stored in wetlands, which was followed by subsequent increases in SO_4^{2-} net export in wet periods to downstream lakes. It was also suggested that the frequency of these droughts was increasing (Schindler et al. 1996). Other reasons for the less-than-expected recovery have been proposed including: 1) desorption of previously adsorbed SO_4^{2-} in the soil (Driscoll et al. 1995; Löfgren et al. 2001; Alewell 2001), and/or 2) mineralization and oxidation of organically bound S (Löfgren et al. 2001; Mitchell et al. 2001) from the catchments.

Long-term climate patterns are also recognized as external factors that drive the behavior of lake and catchment biogeochemistry (Magnuson et al. 1990; Webster et al. 2000; Baron and Caine 2000). There are, however, very few studies that have addressed the impact of known long-term climatic patterns on surface waters. Anderson et al. (1996) related the dates of ice breakup in 20 Wisconsin lakes to the El Niño Southern Oscillation (ENSO)

Index (SOI) and George et al. (2000) found that winter temperatures in several Windermere catchment lakes were positively correlated to the North Atlantic Oscillation (NAO). These relationships reflect physical rather than chemical properties of the lakes. In North America, Dillon et al. (1997) and Dillon and Evans (2001) reported a connection between S redox reactions and ENSO episodes. They found that elevated stream SO_4^{2-} concentrations and export rates in south-central Ontario occurred in the autumn of years with prolonged severe drought, and that these droughts occurred in the years following strong El Niño events, i.e., when the SOI was strongly negative. A link between precipitation patterns in the North American Great Lakes basin and the two extreme phases of the SOI, i.e., El Niño and La Niña, was established by Shabbar et al. (1997) who reported a pattern of negative precipitation anomalies in the region during the first winter following the onset of El Niño events. Later, Dillon et al. (2003a, b) showed that climate indices including the SOI and the NAO were strongly correlated with SO_4^{2-} concentrations in lakes in Ontario and in Norway. These indices, in the case of Ontario, were also related to drought frequency (Dillon et al. 2003a; Eimers et al. 2004a), which was hypothesized to control SO_4^{2-} by altering redox reactions in the lakes' catchments. In addition, experimental manipulations at the ecosystem scale, e.g., the RAIN and CLIMEX projects (Wright and Jenkins 2001), have clearly shown that climate is a confounding factor in recovery from acidification.

In this chapter, we summarize the changes in SO_4^{2-}, alkalinity and pH in a set of eight lakes (nine basins in total) located in south-central Ontario that have experienced significant reductions in sulfate deposition over the past two decades. We chose lake SO_4^{2-} concentration as a simple measure of the lakes' recoveries following decreases in S deposition, and alkalinity and pH because they are parameters that are directly related to biological response. Because it has been noted that the change in SO_4^{2-} concentration in these lakes has been substantially less than anticipated (Dillon et al. 2003a), we have investigated the relationships between the observed patterns and external factors, particularly climate, which may have affected lake recovery.

We believe that the findings presented here are applicable to the majority of the hundreds of thousands of lakes located on the Precambrian Shield, i.e., softwater, under- to moderately- developed lakes that are remote from point source inputs of anthropogenic contaminants.

2. Study Area

The study lakes are the sites of a long-term study of the impacts of long range atmospheric transport, climate change and shoreline development on water quality in forested headwater catchments and lakes. The lakes are located in the District of Muskoka and the County of Haliburton in central Ontario on a southern extension of the Precambrian Shield, and are near the southern limit of the Boreal ecozone (Figure 4.1). Because the bedrock is composed of sparingly soluble silicate minerals, and because the overburden is very shallow (typically less than 1 m, often less than 0.3 m), this region is very sensitive to acid deposition. Furthermore, it has received levels of S deposition that are among the highest in eastern Canada, with deposition levels of between 80 and 100 meq m^{-2} yr^{-1} at the start of this study. Bedrock in the region is primarily granitized biotite and hornblende gneiss, with lesser areas of amphibolite and schist. The surface geology of the upland areas is dominated by basal tills and sand deposits; these are usually covered by peat in the lowland areas. Upland soils are primarily orthic humo-ferric and ferro-humic podzols and brunisols, while the lowland areas

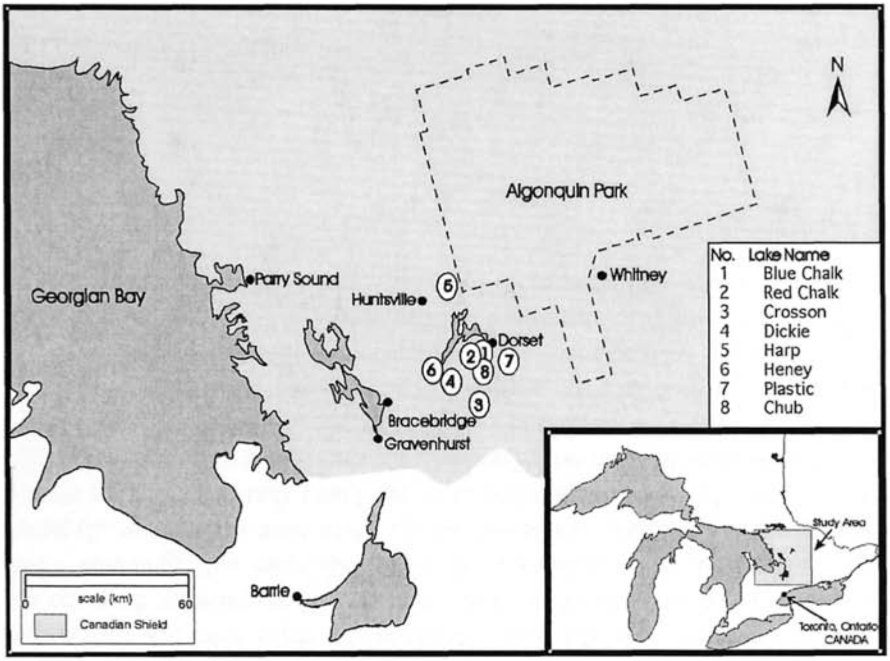

Figure 4.1. Study area showing lakes included in the study. Red Chalk Lake has two distinct basins, each with a separate hypolimnion; the two basins are treated individually here

have humisols and humic mesisols. Bedrock outcrops are found in most catchments, and small, shallow beaver ponds are found in half the catchments. Several of the catchments were partially cleared ca. 100–130 years ago, but the forests have regrown when farming was abandoned in the early 1900s. Descriptions of physiographic and meteorological features are given in Dillon et al. (1991).

The economy of the region is controlled largely by activities related to recreation and tourism, with shoreline development (i.e., the percentage of the shoreline occupied by cottages, houses, docks etc) extensive on two of the lakes (Harp, Dickie), light on three (Blue Chalk, Chub, Red Chalk) and absent on three (Heney, Plastic, Crosson). Selective forest harvesting has been practiced periodically at several of the sites, but there has been little in the past several decades. Upland portions of the lakes' catchments are dominated by sugar maple (*Acer saccharum* Marsh.) and red maple (*A. rubrum* L.) with some American beech (*Fagus grandifolia* Ehrh.), yellow birch (*Betula alleghaniensis* Britt.), red oak (*Quercus rubra* L.), eastern hemlock (*Tsuga canadensis* L.) and white pine (*Pinus strobus* L.). Nutrient cycling in the catchments of several of the lakes (Plastic, Harp, Chub, Red Chalk) has been studied extensively (Watmough and Dillon 2001, 2003a, b, 2004). Over the past 20 years the mean precipitation in the study area was 976 mm (30% as snow), with about 50–55% generating stream runoff, and the mean temperature was 5.3 degrees C.

Seven of the eight lakes are headwater lakes, while the eighth (Red Chalk) has one lake upstream; as a result catchment to lake area ratios are relatively low for this region, and water replenishment times are long, ranging from 1.6 to 5.7 years. The lakes are all dilute (conductivities of 22–35 μS cm^{-1}), with low levels of algal nutrients (total phosphorus of 4–14 μg/L), and with moderate concentrations of dissolved organic carbon (DOC 2–5 mg C/L) that depend on the relative importance of wetlands in each lake's catchment. Both the sensitivity of the lakes to acid deposition and the degree of acidification of the study lakes varies substantially, from lakes such as Plastic and Heney that have very low alkalinities and documented biological damage (Dillon et al. 1987), to others such as Harp and Blue Chalk that, because of their thicker surficial deposits, have higher alkalinities, base cations and pHs.

3. Methods

Atmospheric deposition was estimated as bulk deposition. Bulk precipitation chemistry and precipitation depth were monitored at a minimum of four stations in the study region. Methods are described in detail in Dillon et al. (1987).

Lake samples were collected as integrated, volume-weighted epilimnetic, metalimnetic and hypolimnetic samples, with the appropriate depths determined from temperature profiles taken during summer stratification, or in tube composites at all other times of the year. Whole-lake concentrations for any sampling day during stratification were calculated by mathematically weighting the concentrations in the epilimnion, metalimnion and hypolimnion according to the relative proportions of each lake's stratum. Sulfate was analyzed by ion chromatography and alkalinity by Gran titration.

Trends in measured parameters were compared using coherence analysis. Although Pearson product-moment correlations have been used to evaluate temporal coherence among lakes by comparing time series for all-possible pairs of lakes (Magnuson et al. 1990), we estimated temporal coherence between lakes using the intraclass correlation from a repeated-measures (or randomized block) analysis of variance (ANOVA, Somers et al. 1996). The ANOVA was a two-way model without replication, with year as one factor incorporating the repeated-measures nature of the data, and the pair of lakes as the second factor. The two-way interaction term in the ANOVA represents variation between lakes over years (i.e., interaction plus error given no replication). Here, lake is a blocking factor (i.e., this variation is factored out) such that differences in the overall means for each lake are partitioned from the year-to-year variation and ignored. Since this blocking does not control for differences in variances that are often correlated with the mean value, we standardized the variances for each time series to unity. Both the Pearson and intraclass correlations measure the parallel or synchronous nature of the two time series. However relative to the Pearson correlation, the intraclass correlation is a more-general type of correlation that is not restricted to pair-wise comparisons, is better suited when the variables are responses (i.e., measured with error), and it can accommodate missing data. More details are provided in Dillon et al. (2003a, b).

4. Results and Discussion

4.1. Deposition

Changes in SO_4^{2-}, nitrate (NO_3^-) and ammonium (NH_4^+) deposition in the study area for portions of the investigation period here have been discussed in Dillon et al. (1988, 1997), Dillon and Evans (2001), Eimers and Dillon (2002) and are only briefly summarized here. Deposition from 1976 to 2000 is shown in Figure 4.2. Sulfate deposition has declined by ca. 50% from that

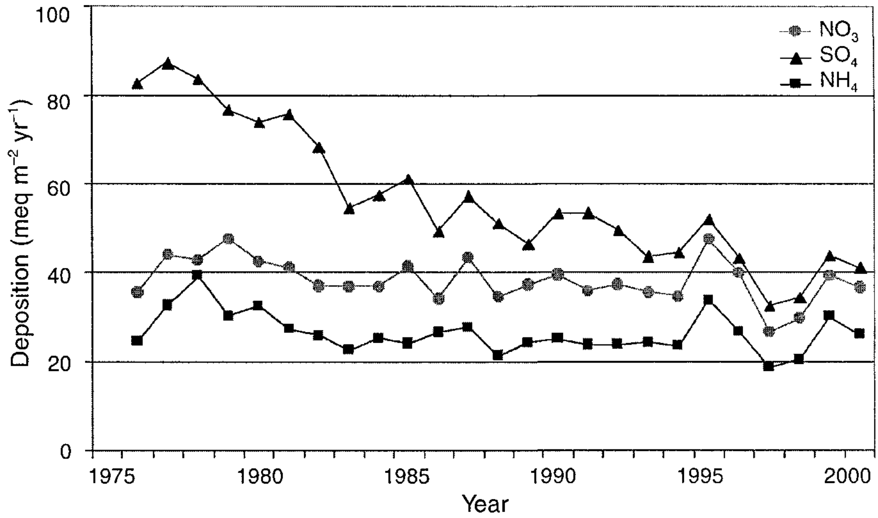

Figure 4.2. Sulfate, nitrate and ammonium deposition in the study area from 1976 to 2002. Deposition was measured in bulk samples and may underestimate the dry component

measured in 1976–78, and 45% from that of 1980, with most of the decrease occurring prior to 1986. Although there were several years in the 1990s with very low SO_4^{2-} deposition, these resulted from unusually low precipitation depths rather than reduced concentrations. Nitrogen deposition now exceeds S deposition in the study area, although NO_3^- and NH_4^+ deposition (and concentration) have changed little in 25 years, with no significant downward trends. At present, most of the inorganic nitrogen is retained in lakes (Kaste and Dillon 2003) and catchments (Watmough and Dillon 2003b, 2004), although in a few catchments NO_3^- leakage is relatively high (Schiff et al. 2002). High nitrogen leakage from catchments was linked with upland sites and summer droughts (Watmough et al. 2004). These decreases in SO_4^{2-} deposition are consistent qualitatively with the change in S emissions in eastern North America.

4.2. Lake Chemistry

Despite the substantial decline in S deposition over the past two decades, only about half of the lakes that have been surveyed in this region have shown decreasing SO_4^{2-} concentrations, while the remaining half have shown no positive response to changes in deposition (Jeffries et al. 2003).

When parameters more important to the biota of the lakes were considered (i.e., pH and alkalinity), recovery was evident in only a very small portion of the lakes. In addition, those that had shown some positive changes had improved considerably less than expected based on the change in deposition (Dillon and Evans 2001; Dillon et al. 2003a).

Although the eight study lakes receive equal SO_4^{2-} deposition, their SO_4^{2-} concentrations decreased over the 19-year duration of this study between 21 and 41%, averaging 27%, about half the decline in deposition over the same period. The biggest decline in SO_4^{2-} concentration (Heney) was similar in magnitude to the decrease in SO_4^{2-} deposition.

Changes in lake SO_4^{2-} concentrations that result from a change in deposition and subsequent input rates including terrestrial catchment sources will not be instantaneous, but rather will reflect the water replenishment rate (the hydrologic turnover time) of each lake. If it is assumed that SO_4^{2-} behavior in the lakes is conservative (the mean mass transfer coefficient or rate of removal from the water column by processes other than outflow for SO_4^{2-} in these lakes based on mass balance measurements over 18 years was not significantly different from 0.0 m yr^{-1}; Dillon, unpublished studies), then, based on a simple mass balance model, the half-life (years) for change in lake concentration is the natural log of 2 multiplied by the water replenishment time (i.e., $\ln 2 \times \tau$). Thus, in 3 half-lives, we expect the concentration to have changed ca. 90% of the way to the new steady-state level, i.e., in 3.3 to 12 years for these lakes. Thus, lakes with a short water replenishment time such as Heney should respond more quickly to decreasing inputs than those with long replenishment times, e.g., Blue Chalk. However, because almost all of the change in SO_4^{2-} deposition took place pre-1986, the current lake SO_4^{2-} concentrations should almost fully reflect the expected changes in all of the lakes if there are no lags in response.

The trends in SO_4^{2-} concentration in these lakes were not monotonic, and did not match the temporal pattern in changes in SO_4^{2-} deposition. During certain time periods such as 1983–84 and 1988–92, SO_4^{2-} in all of the lakes increased rather than declined. This is best seen when SO_4^{2-} concentrations are standardized (i.e., z-score transformed – Figure 4.3). Sulfate concentration declined and increased in all lakes at the same times over the 19 years, and the patterns were independent of the water replenishment time of the lakes. This synchrony in patterns was independent of the physical or chemical characteristics of the lakes or their catchments, although differences in these factors likely caused the differences in initial concentrations of SO_4^{2-} in these lakes.

Coherence analysis (Dillon et al. 2003a) demonstrated that, although there were two subsets of lakes (six lakes in one, four in the other) with

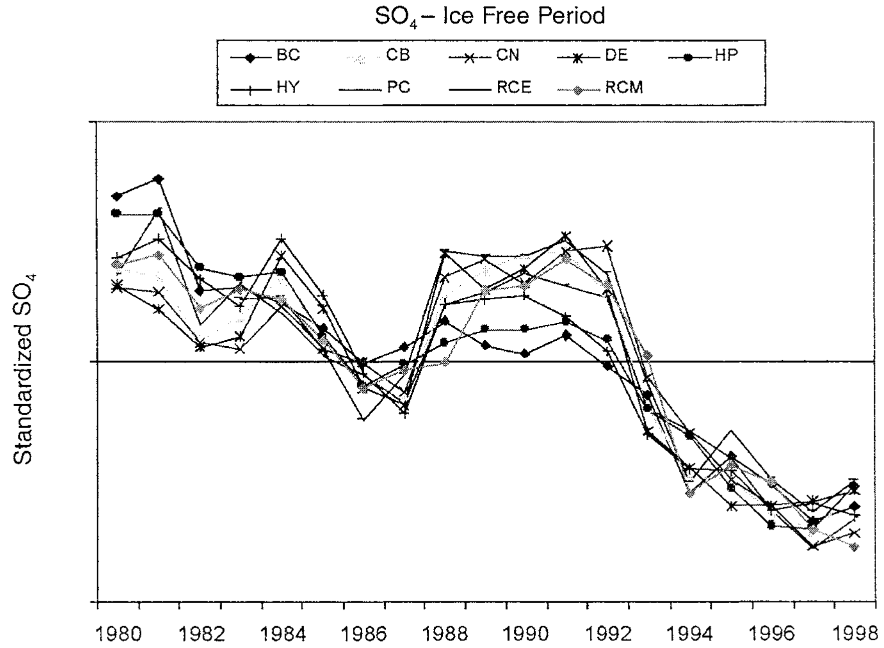

Figure 4.3. Sulfate concentration in the study lakes expressed as the z-scored value (difference from the mean divided by the standard deviation). All lakes have similar long-term patterns. Data are based on annual averages with n= 6–12 samples per lake per year

common temporal SO_4^{2-} patterns that were not significantly different, the average pattern of each of the two sets themselves were almost identical. The means of the z-scored concentrations of the lakes in each grouping were used in the subsequent development of predictive models. Although this analysis generated 2 models, they were almost identical, with the same independent parameters in each.

When a similar analysis was carried out with alkalinity and pH data (Figures 4.4 and 4.5 respectively), it was apparent that increases in each were proportionately much less than the observed decreases in SO_4^{2-}. There were small but non-significant increases in each over the study period. Coherence analysis suggested that the lakes did not group into a single or very few common patterns; this is a consequence of the fact that the changes in each parameter were small relative to the interannual variability within any lake.

The pattern in lake SO_4^{2-} concentration was compared to the long-term patterns in a suite of climate parameters and to the SO_4^{2-} deposition pattern.

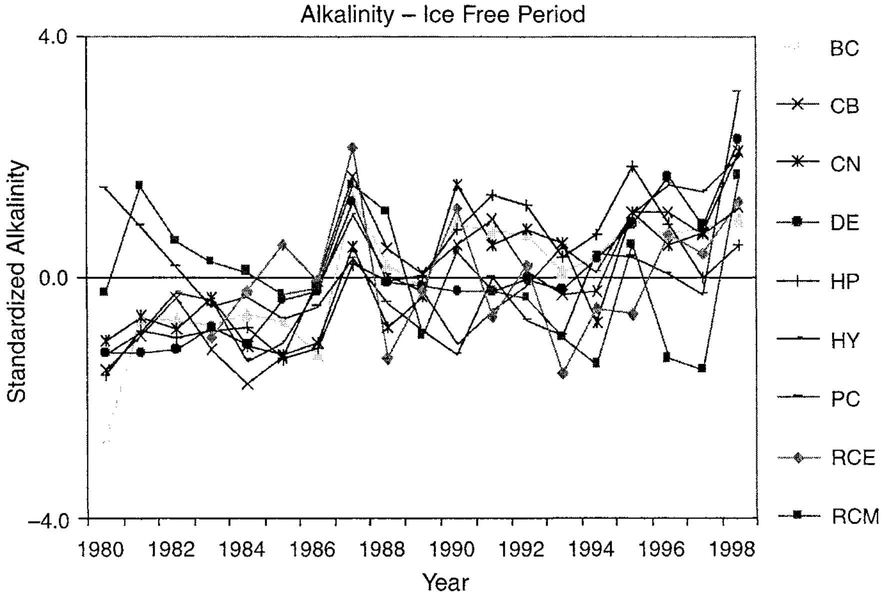

Figure 4.4. Alkalinity (by Gran titration) of the study lakes expressed as the z-scored value (difference from the mean divided by the standard deviation). Data are based on annual averages with n=6–12 samples per lake per year

Climate parameters included local and regional-scale data (precipitation, temperature, etc.) and large (global) scale indices including the SOI, the NAO and the Arctic Oscillation Index (AOI). Unlike the local/regional climate data, the SOI and the NAO were strongly correlated with the coherent SO_4^{2-} pattern in lakes (Table 4.1). A multiple regression model combining these two indices and measured SO_4^{2-} deposition as independent parameters gave the best model for SO_4^{2-} concentration in the study lakes.

Because of the lack of coherent and significant trends in alkalinity and pH, no models were constructed for these lake parameters.

4.3. Stream Chemistry

The fact that lake SO_4^{2-} concentrations have declined less than anticipated based on the decline in SO_4^{2-} deposition indicates that either processing of SO_4^{2-} within the lakes' catchments has changed or processing within the lake itself has changed or both. However, the latter can be ruled out because the mass transfer coefficient for SO_4^{2-} in the study lakes has remained at 0 m/yr (+/– 0.05), i.e., the loss (or production) of SO_4^{2-} within the lakes is insignificant relative to the flux of SO_4^{2-} through the lake (Dillon, unpub.

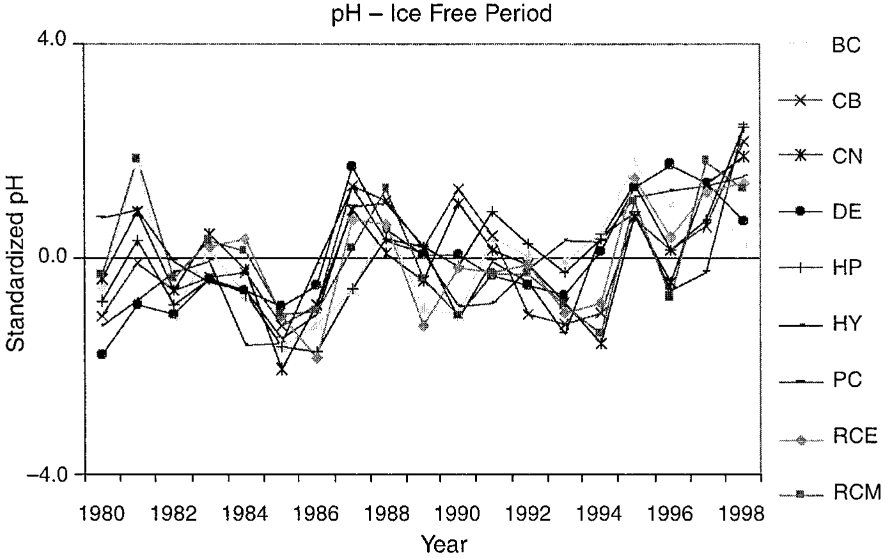

Figure 4.5. pH of the study lakes expressed as the z-scored value (difference from the mean divided by the standard deviation). Data are based on annual averages with n=6–12 samples per lake per year

studies). This is equivalent to stating that SO_4^{2-} is acting as a conservative parameter in the lakes.

The concentrations of SO_4^{2-} in streams in the study area provide strong evidence concerning the reasons for the less-than-anticipated changes in lake concentrations. In each stream studied, a consistent long-term pattern wherein concentration increased dramatically in specific years was observed; an example is shown in Figure 4.6. In some cases the increases reaching 10 to 20-fold. Coherence analysis demonstrated that all streams sampled in the study area had a consistent pattern (Figure 4.7); that is, the anomolous increases occurred simultaneously (Eimers and Dillon 2002). It was observed that the years with very high SO_4^{2-} concentrations in the streams were those where significant droughts were experienced (Dillon and LaZerte 1992; Dillon et al. 1997). When the length of the drought, expressed as days where flow dropped to 0, was compared to the net output of SO_4^{2-} from each catchment (Figure 4.8), it was apparent that the occurrence of drought was very strongly correlated with SO_4^{2-} flux from the catchments. It was also observed that the droughts followed the occurrence of El Niño events; each El Niño led to one or more years of drought, with low streamflow and increased SO_4^{2-} flux.

Table 4.1. SO_4^{2-} models (using backwards elimination stepwise multiple regression) for the study lakes based on suites of parameters. The global indices based on annual and seasonal data, with and without lags (1 and 2-year) and SO_4^{2-} deposition provided the best models. Local and regional-scale climate parameters (e.g., precipitation, temperature, etc.) did not provide significant models

Predictor Suite	r	F	p	Important Predictors
SOI indices	0.71	3.63	0.03	annual (lag1), annual (lag2), spring, summer (lag1)
NAO indices	0.57	3.92	0.04	annual (lag0), fall (lag1)
AO indices				none
SO_4^{2-} deposition	0.72	18.1	0.0005	fall (lag1)
combined	0.83	7.61	0.002	spring SOI, annual NAO (lag1), fall NAOI, fall SO_4 deposition

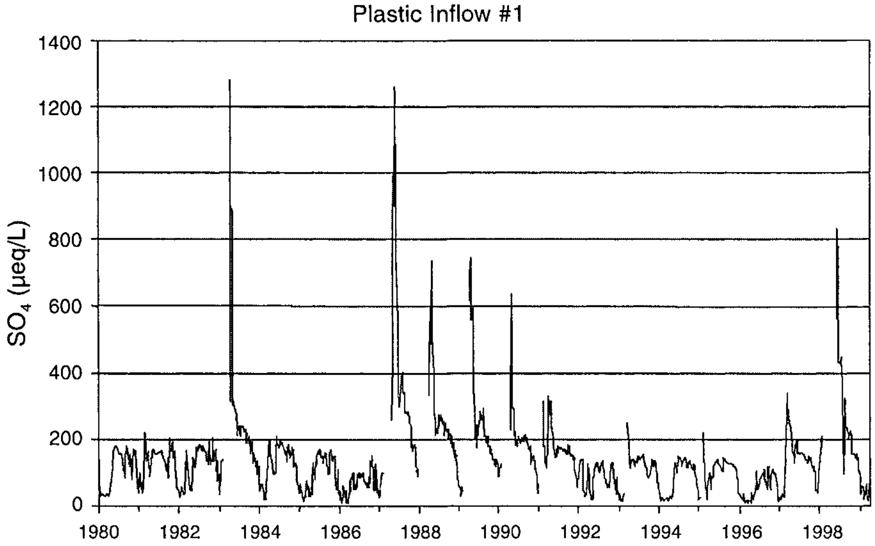

Figure 4.6. Sulfate concentration in the main inflowing stream to Plastic Lake (Plastic Inflow 1). Gaps in the data indicate periods when the stream was dry, i.e., flow was 0. Note extremely high concentrations following each drought

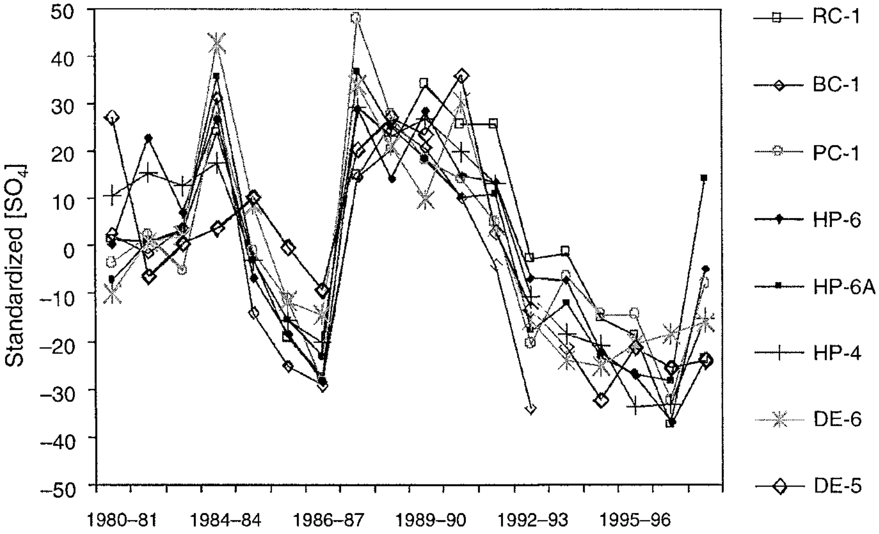

Figure 4.7. Sulfate concentration in eight streams, expressed as annual deviations (%) from the long-term mean

The link between drought and increased SO_4^{2-} flux is, at least in part, through the S redox cycle. Lower water tables in the wetlands and riparian areas, and even in sub-surface pools that result from bedrock topography, lead to re-oxidation of previously stored, reduced S. This has been demonstrated using stable S isotope methods (Eimers et al. 2004b); analysis of the SO_4^{2-} isotope signature in one of the study streams indicated that the SO_4^{2-} peaks observed following droughts had largely been previously reduced. Following the droughts, the subsequent re-wetting and resulting streamflow lead to export of re-oxidized SO_4^{2-} along with the acid produced during the oxidation reaction, or products of that acid's interaction with the soil, e.g., Ca. Despite this, base cations, particularly Ca, have declined in streams (Watmough et al. 2005) as the exchangeable bases in the terrestrial system continue to be depleted faster than they are renewed through primary weathering. It has also been shown that there is adequate reduced S in the catchments to support the net ouput of SO_4^{2-} for a long time, possibly centuries (Eimers et al. 2004c). If droughts increase in frequency or severity as anticipated because of climate change, the recovery of ecosystems from acid deposition will be further compromised.

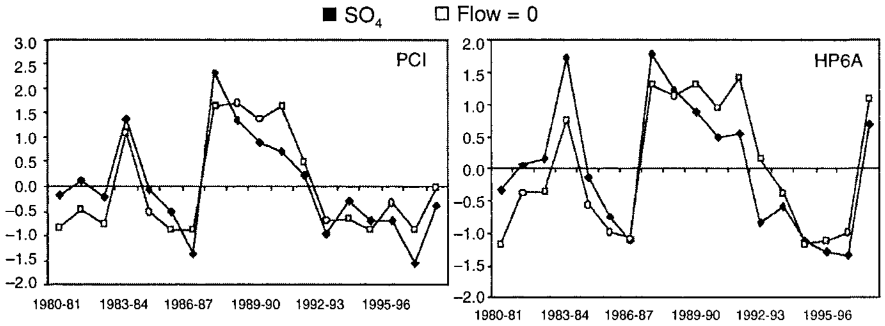

Figure 4.8. Sulfate retention and number of days where flow = 0 for the inflowing streams of two of the lakes, Plastic Inflow 1 and Harp Inflow 6A. Data are expressed as z-scored values

5. Summary

Over the duration of an almost three-decade study in south-central Ontario, Canada, sulfate deposition has decreased by ca. 40–45%. Nitrogen deposition has remained unchanged. and as a result total N deposition now significantly exceeds S deposition. Based on an evaluation of the response of lakes, streams and catchments assessed via analysis of trends in concentrations and in elemental budgets for 8 lakes and 20 sub-catchments, we have concluded that recovery has been compromised by climate. Lake recovery has been modest, as catchment-related processes have led to net output of sulfate from almost all of the catchments. Lake sulfate has decreased, but only about half as much as expected based on the deposition change, and lake alkalinity and pH have improved only marginally. Base cations have also declined in conjunction with the sulfate. Because acid deposition is still greater than the critical load at many of these sites, the possibility of further recovery without additional decreases in SO_4^{2-} deposition are limited. It is noteworthy, however, that the chemical changes are synchronous; that is the temporal pattern of change is identical in all lakes, as is the pattern in all streams. These patterns have a strong climate signal, notably a drought-related signal that is influenced strongly by the Southern Oscillation.

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5. Atmospheric Nitrogen Deposition: Implications for Terrestrial Ecosystem Structure and Functioning

Knute J. Nadelhoffer

1. Introduction

Acid rain research on terrestrial ecosystems has increasingly focused on the effects of inorganic nitrogen (N) deposition, both as nitric acid (HNO_3) and ammonium (NH_4^+, which can produce acidity in soils when oxidized). This is largely because acidification of sensitive catchments in the northeastern United States and elsewhere continued following the 1970 and 1990 amendments to the U.S. Clean Air Act (Likens and Lambert 1998). These amendments capped sulfur oxide (SO_x), but not N oxide (NO_x) emissions from electric utilities and industrial sources. Nevertheless, NO_x control programs, focused on utility and industrial sources as well as on vehicle emissions, have stabilized if not decreased NO_x emissions and resultant nitrate deposition in the eastern United States (See Driscoll et al., this volume). As a result, nitrate deposition has remained relatively stable in the northeastern United States and eastern Canada through the past decade, but has increased relative to sulfate deposition (Watmough et al. 2005). Ammonium deposition, due largely to ammonia (NH_3) emissions from fertilized agroecosystems and from concentrated animal feeding operations (CAFOs, *as in* Galloway et al. 2003), accounts for 30 to >50% of inorganic N deposition on the land surface in North American and other industrialized regions of the world (Galloway et al. 2004; Holland et al. 2005).

Because primary production, or plant growth, in ecosystems is often limited by the rate at which inorganic N forms become available for plant uptake, many temperate zone ecosystems are considered 'N limited' (Vitousek and Howarth 1991). Therefore, increased N deposition on forests and other temperate ecosystems could potentially increase primary production, thereby altering their structure (biomass amounts and species composition) and functioning (element cycling and retention). This contrasts with sulfur, which is almost always available in excess of biological demand. As a result, sulfur cycling and retention within terrestrial ecosystems is not strongly regulated by biological processes. This chapter will consider the effects of chronically elevated N deposition on forested ecosystems of the northeastern United States. Forest ecosystems dominate land cover in this region (Goodale et al. 2002) and atmosphere-plant-soil interactions in forests regulate drainage water chemistry and play an important role controlling N inputs to estuarine and coastal ecosystems (See Howarth, this volume).

2. Nitrogen Deposition and Ecosystem Acidification

Prior to the mid 20th century, most N inputs to temperate forests were small relative to plant N demands for growth (Galloway et al. 1982; Likens et al. 1987; Galloway 1998) and originated mostly as NO_x produced by lightning-driven N_2 oxidation and by biomass burning. Deposition of particulate organic N deposition (as dust) and low rates of N-fixation (reduction of atmospheric N_2 to amino-N by free-living microbes in soil or microbial symbionts within root nodules or legumes and a small number of other plant taxa) also contributed to N accumulation. However, rates of N-fixation in temperate forests are typically low and generally contribute little to plant N uptake (Zou et al. 1995).

Increasing emissions of NO_x and NH_3, accompanied by chemical transformations to NO_3^- (nitrate, the acidic anion of nitric acid, or HNO_3) and NH_4^+, and the transport of these ions from sources to forested and other receiving regions (Figure 5.1) has lead to increases in inorganic N deposition in excess of background levels. Across North America, total inorganic N inputs to forests now range from 4 to 30 kg N ha^{-1} yr^{-1} (Ollinger et al. 1993; Fenn et al. 1998), with inputs as high as 90 kg ha^{-1} yr^{-1} in "hotspots" downwind from CAFOs or heavily fertilized agricultural areas, near point sources of NO_x emission, or at high elevations prone to formation of acidified fog formation (Lovett and Kinsman 1990; Bytnerowicz and Fenn 1996; Fenn et al. 2003). Across the northeastern United States, N deposition is higher near emission sources to the south and west of the region and at high elevations (Figure 5.2).

The total range of N deposition rates of 4 to 30 kg ha^{-1} yr^{-1} (above) constitutes a large proportion of the estimated uptake of 40 to 130 kg N ha^{-1} yr^{-1} by forest vegetation (Reich et al. 1997). At the extremes, N deposition could conceivably provide between 3 to 75% of annual growth demands of forest vegetation. However, as discussed below, high rates of N input can alter N transformations and exchanges between soils and plants in forests. Chronically elevated N deposition can eventually lead to an "opening" of forest N cycling which can lead to losses of base cations from soils (e.g., Ca^{2+}, Mg^{2+}, K^+) and soil acidification, as well as exports of nitrate, base cations, and

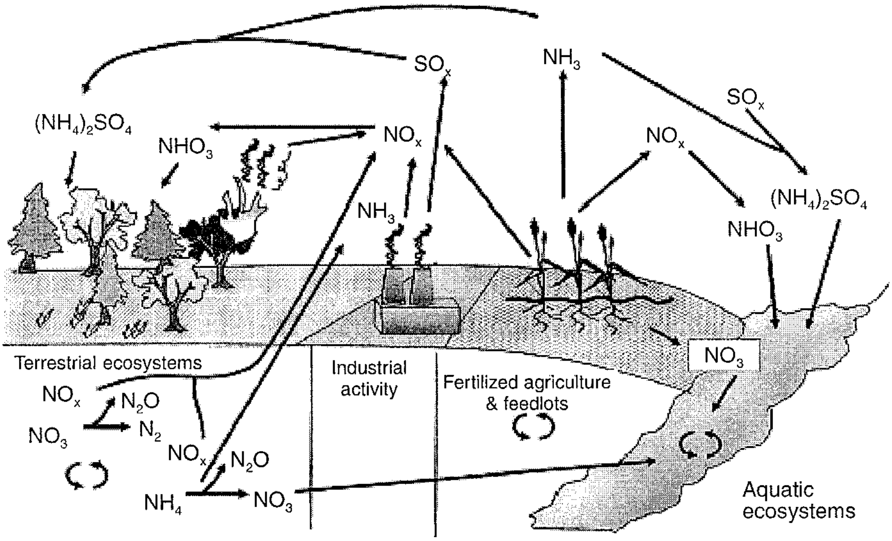

Figure 5.1. Sources, transformations, transport and deposition of inorganic nitrogen inputs to ecosystems. Fossil fuel combustion, biomass burning and microbial denitrification are the primary sources of NOx emissions, whereas fertilizer volatilization and livestock operations are the primary sources of NH_3 emissions. Closed circular arrows represent processes operating within ecosystems, including plant uptake, litterfall, mineralization, and nitrification. From Figure 4 in Matson et al. (2002), reprinted with permission from Royal Swedish Academy of Science

acidity (H^+ and Al^{3+}) to stream water. These changes in soil and drainage water chemistry can eventually result in changes in forest growth and plant composition.

2.1. Soil Processes

The dominant inorganic N ions entering ecosystems from the atmosphere, NH_4^+ and NO_3^-, are also generated naturally by microbial processes in soil via mineralization (the conversion of organic N to NH_4^+) and nitrification (the oxidation of NH_4^+ to NO_3^-) reactions (Figure 5.3). Mineralization of one unit of organically bound N in litter or humus to NH_4^+ consumes one unit of acidity (or H^+). If this NH_4^+ is subsequently taken up and assimilated into plant biomass, one H^+ unit is generated, yielding no net change in acidity. If, however, the mineralized NH_4^+ is nitrified, two H^+ units are produced, resulting in a net production of one H^+ for the combined mineralization and nitrification of one organic N unit. Importantly, if the NO_3^- produced by nitrification is taken up and assimilated into biomass, the H^+ produced from mineralization plus nitrification is consumed and the overall ecosystem acidity remains balanced. Likewise, if soil NO_3^- is denitrified, acidity is consumed. However, if NO_3^- is not assimilated into biomass or denitrified, the H+ generated during mineralization plus nitrification contributes to soil acidity. Atmospheric inputs of NH_4^+ and NO_3^- combine with soil N and H^+ dynamics as shown in Figure 5.3.

Nitrate ions, similar to the surface areas of clay-humus particles in temperate ecosystems, are negatively charged. As such, NO_3^- is readily leached with drainage water if it is not taken up by organisms or denitrified (Johnson and Lindberg 1992). The strong affinity of H^+ ions for negatively charged soil particles, can lead to the displacement of "base" cations such as Ca^{+2}, Mg^{+2}, and K^+ from particle exchange surfaces. These displaced bases are then leached together with unassimilated NO_3^- to maintain the charge balance of water draining soils. These exchange reactions can lead to partial depletion of base cations, followed by increasing exports of strongly acidic H^+ and Al^{+3} ions (van Breemen and van Dijk 1988; Reuss and Johnson 1989; Johnson et al. 1991).

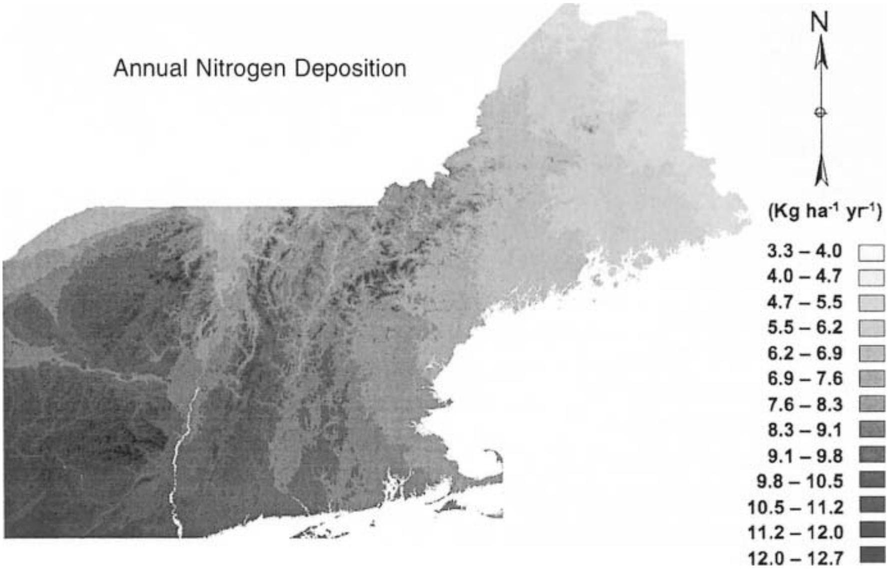

Figure 5.2. Total inorganic N (NH_4^+–N+NO_3^-–N) deposition in the northeastern United States. Patterns of N deposition were generated using a digital elevation map, precipitation data, and precipitation chemistry measurements. From Ollinger et al. (1993), reprinted with permission from Ecological Society of America

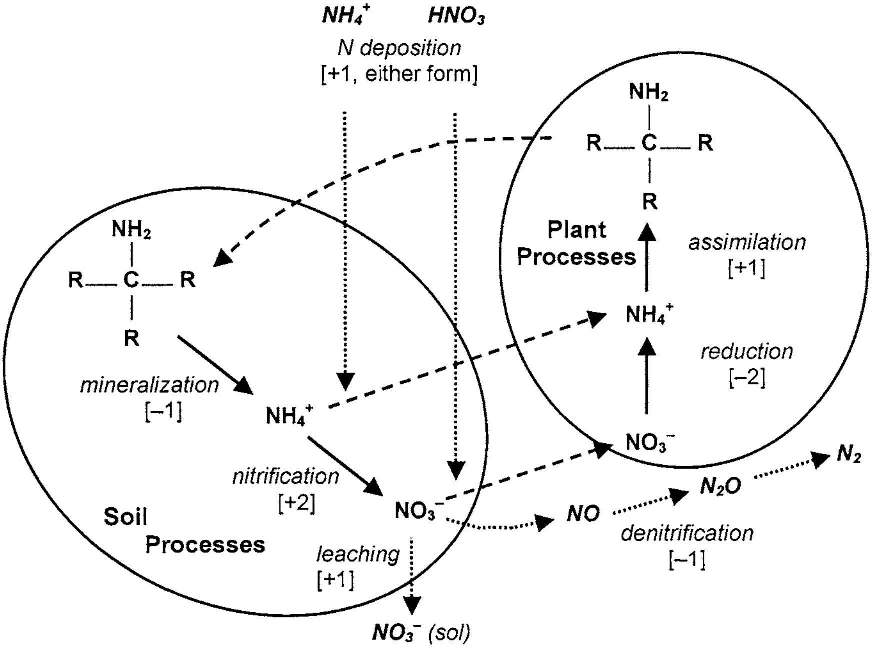

Figure 5.3. Forest N cycling and acidification. Atmospheric N inputs and N outputs to drainage waters are shown in *italics*. Soil processes (left) and plant processes (right) are clustered within ovals. Dashed lines indicate soil-plant exchanges (plant N uptake or organic N return to soil). Solid lines show processes within soils or plants. Dotted lines show fluxes into or out of forests. Values in brackets refer to net consumption [–] or production [+] of 1 mol H^+ associated with the transformation of 1 mol N. When forest N cycles are closed (small N inputs and outputs), the sum of H^+ consumed and produced by soil and plant processes is zero and no acidity is generated. When 1 mol of organic N is mineralized (1 mol H^+ consumed) and subsequently nitrified (2 mol H^+ produced), 1 mol H^+ remains to acidify soil or drainage water if nitrate is not removed from soil and converted to organic form by plants. Denitrification to any of three gaseous products consumes 1 mol H^+. Direct inputs of acidity can also result from ammonium and nitrate deposition. Reprinted from Nadelhoffer (2001) with permission from Elsevier

2.2. Plant-Soil Interactions

Elevated atmospheric N deposition can modify ecosystem N cycling by increasing ratios of N inputs to internal N cycling and by changing the amounts and forms of N made available to plants and microbes. In addition to these direct effects, N deposition can alter feedbacks between plants and soils. For example, increases in leaf or fine root N concentration occurring due to plant uptake of atmospheric N inputs could lead to higher N concentrations in plant litter, faster litter decomposition, and greater N mineralization (McClaugherty et al. 1985). Such changes in leaf and root litter chemistry and N mineralization could feedback to increase N availability to plants more than would be expected from elevated N inputs alone. Likewise, increased uptake of N inputs by decomposers (*microbial immobilization*) could increase microbial turnover, thereby increasing N mineralization and N cycling between plants and soils.

N deposition can stimulate N mineralization and nitrification, at least initially, in soils (McNulty and Aber 1993; McNulty et al. 1996b; Kjønaas et al. 1998). However, long-term studies suggest that N mineralization rates can eventually decline in response to N deposition, whereas nitrification rates can continue to increase or remain elevated above initial conditions (Magill et al. 2004). Increases in nitrification have important implications for soil acid-base relations, the availability of other nutrient ions to forest plants, and nitrogen outputs to the drainage water and the atmosphere (above).

Forest N cycling studies typically show foliar N concentrations increasing along atmospheric N deposition gradients and in response to experimental N additions (Burton et al. 1993; Magill et al. 2004; Magill et al. 1996; McNulty et al. 1991; McNulty et al. 1990). Although N concentrations often increase with N inputs, reports of decreased foliar Mg or Ca and increase foliar Al concentrations (Schulze 1989; Wilson and Skeffington 1994; Hutchinson et al. 1998; van der Eerden et al. 1998) suggest N deposition can lead to nutrient imbalances in trees. Such nutrient imbalances could explain why increases in primary production in forests subject to elevated N deposition are sometimes short-lived or not detectable. As with foliage, forest floor N concentrations can increase along gradients of N deposition or experimental N additions (Emmett et al. 1998). These increases in forest floor N concentration (often reported as decreases in forest floor C:N ratios) could result from higher N concentrations in litter and root inputs to soils.

3. Nitrogen Saturation Hypothesis

Responses of temperate forests in the northeastern United States and Europe to variations in Nitrogen deposition, both along gradients of atmospheric N deposition and in response to long term N addition experiments, have led to a conceptual model of ecosystem response to chronically elevated N deposition referred to as the *"Nitrogen Saturation Hypothesis"* (Aber et al. 1989; Aber et al. 1998; Aber et al. 2003). This model and various component mechanisms have been subjected to field tests during the past decade. In this model, ecosystem processes undergo transitions from relatively *closed N* cycles, with small inputs and outputs of N relative to internal cycling rates, to *open N* cycles with large exports of nitrate, cations, and acidity (Figure 5.4). The model predicts that initial ecosystem responses to elevated N deposition (*Stage 1*) are small. Any increases in primary production (or growth) due to greater N availability and plant uptake would likely be minor and beneath the detection limits of field measurements. The most likely "indicators" of response to N deposition would be increased N concentrations in leaves and fine roots. Higher N concentrations in these tissues would allow for greater resource uptake (carbon by leaves and nutrients by fine roots) and could possibly stimulate tree growth. *In Stage 1*, N mineralization in soils could increase due to greater N inputs from leaf litter and fine root turnover. Higher N mineralization rates, together with increased N deposition, could increase ammonium availability to nitrifiers and thereby increase nitrification (net NO_3^- production, above). Increased nitrification would (1) change the ratio of ammonium to nitrate available to tree roots; (2) stimulate denitrification (releasing and exporting N_2O, NO and N_2 gases), and (3) stimulate NO_3^- leaching losses, together with exports of base cations and acidic ions (H^+ and Al^{3+} ions). In summary, the possible initial responses to elevated N deposition include higher N availability, higher N concentrations in foliage and roots, greater production and turnover of foliar and root tissue, increased N cycling between vegetation and soils, increased nitrification and nitrate availability to roots, increased denitrification, and increased exports of nitrate, base cations and acidity to drainage waters.

The duration of *Stage 1* responses to elevated N deposition varies according to the intensity and duration of N deposition, land-use history and soil characteristics (Aber et al. 1995) and is not easily predicted. Forests subjected to chronically elevated N deposition, particularly those on infertile, base-poor soils (low exchangeable Ca^{2+} and Mg^{2+} levels) can progress to *Stage 2* (Figure 5.4). At this later stage, nutrient imbalances in plant tissues can lead to growth declines. Also, N availability in excess of plant demands can lead to even greater rates of nitrification, denitrification, nitrate leaching loss and soil acidification. *Stage 3* represents the end of this progression and is characterized by tree mortality and nitrate-N outputs to streams and groundwater eventually equaling or even exceeding N inputs. This stage is likely to be followed by major changes in plant species composition and a period of ecosystem re-equilibration to greater N cycling rates and increased soil acidity.

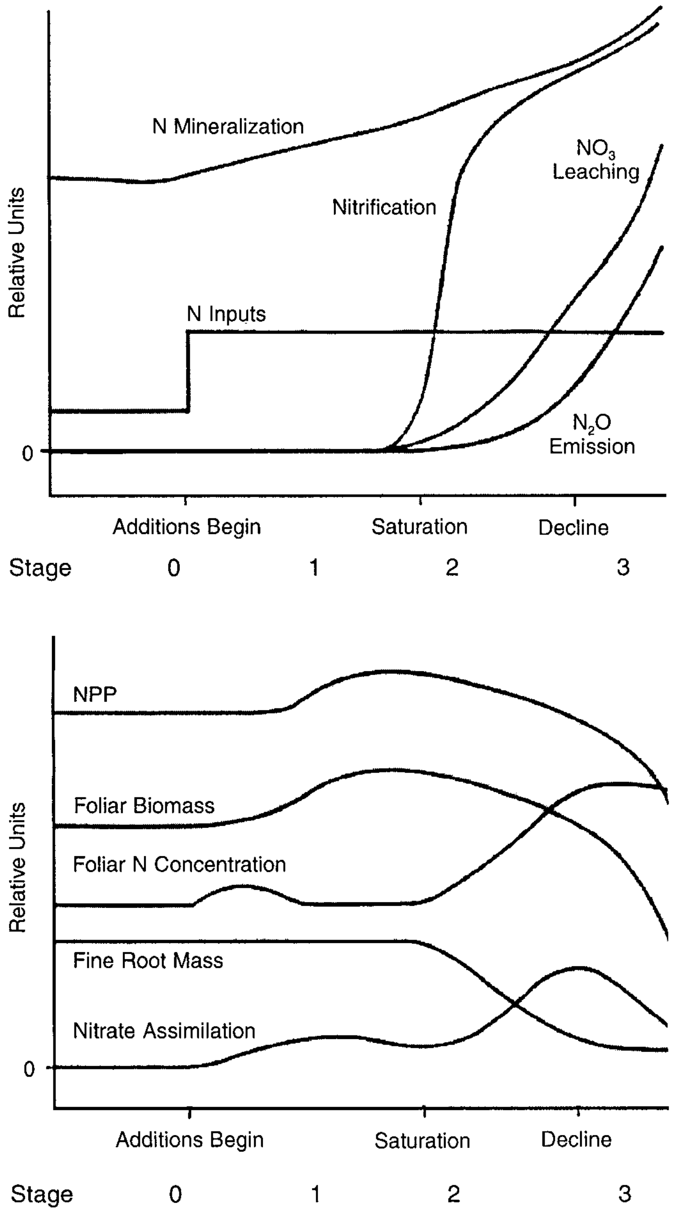

Figure 5.4. The *Nitrogen Saturation Hypothesis* (from Aber et al. 1989; 1998, reprinted with permission, copyright American Institute of Biological Sciences), a conceptual model of terrestrial ecosystem responses to chronically elevated nitrogen (N) deposition. This model identifies four stages of ecosystem response to N loading. Soil responses are shown in the upper panel, and plant responses in the lower panel. See text for discussion

4. Forest Responses to Nitrogen Deposition

Most temperate forests have not progressed to the advanced stages of N saturation. For example, forest biomass in eastern North America (Birdsey 1992; Brown et al. 1999) and Europe (Kauppi et al. 1992) has increased in recent decades despite elevated N deposition. In fact, increased forest growth accompanying increased N deposition has led to the hypothesis elevated atmospheric N inputs might be fertilizing forests. Some regional analyses report statistical relationships between N deposition and forest biomass accumulation (Nellemann and Thomsen 2001), but individual species growth can respond either positively or negatively to N deposition in mixed-species forests (Boggs et al. 2005) Some modeling studies have suggested that a fertilization effect of N deposition could contribute significantly to CO_2 uptake by forests in the North Temperate Zone. This requires, however, that most N deposition be taken up into plants and not into soil pools with long turnover times (Townsend et al. 1996; Holland et al. 1998). In contrast, field experiments simulating atmospheric N deposition by applying ammonium or nitrate fertilizers in small increments across annual cycles typically do not show increased tree growth when N additions are less than about 75 kg ha^{-1} yr^{-1}, the upper limit of N deposition reported in the temperate regions (Emmett et al. 1998; Schleppi et al. 1999; Magill et al. 2004). These studies, together with ^{15}N tracer experiments conducted in combination with low-level N additions (Nadelhoffer et al. 1995; Tietema et al. 1998; Nadelhoffer et al. 1999) suggest that most N deposition is immobilized in soils rather than taken up by trees. Therefore, field evidence suggests that increases in forest biomass and the consequent uptake of CO_2 is more strongly influenced by increases in forest area, improved forest management practices, and longer growing seasons associated with climate warming than by elevated N deposition.

A number of investigations have focused on potential plant and soil responses as indicators of N saturation stages. Analyses of results from multiple studies have shown that forest floor chemistry can vary with N inputs. For example, forest floor C:N ratios have been reported to decrease with N deposition in the northeastern United States (Aber et al. 2003) and Europe (Emmett et al. 1998; Gundersen et al. 1998). As discussed above, decreased forest floor C:N (or increased percent N) could result from increased decomposition of soil organic matter, higher concentrations of N in litter and root inputs to soils, greater incorporation of N inputs into microbial biomass, or any combination of these responses to N deposition. Forest floor C:N ratios typically vary with climate and tree species composition in the absence of elevated N deposition. For example, forest floors developing under conifer forests have higher ratios due to lower nutrient contents in litter and slowed litter decomposition as compared to deciduous litter. However, N deposition can increase N concentrations in litter and stimulate decomposition, leading to broad scale decreases in forest floor C:N ratios of both conifer and deciduous forests (Figure 5.5). This has led to the use of variations in C:N in foliage and forest floors as indicators of ecosystem responses to N deposition. Moreover, lower forest floor C:N ratios have been linked to increased nitrification in soils (Figure 5.6) and increased nitrate exports from forest soils to drainage water (Figure 5.7).

The process level studies discussed above are consistent with a number of other studies reporting increased nitrate concentrations in streams along N deposition gradients (Driscoll et al. 1987; Stoddard 1994; Nodvin et al. 1995; Fenn et al. 1998; Watmough et al. 2005) and in response to experi-

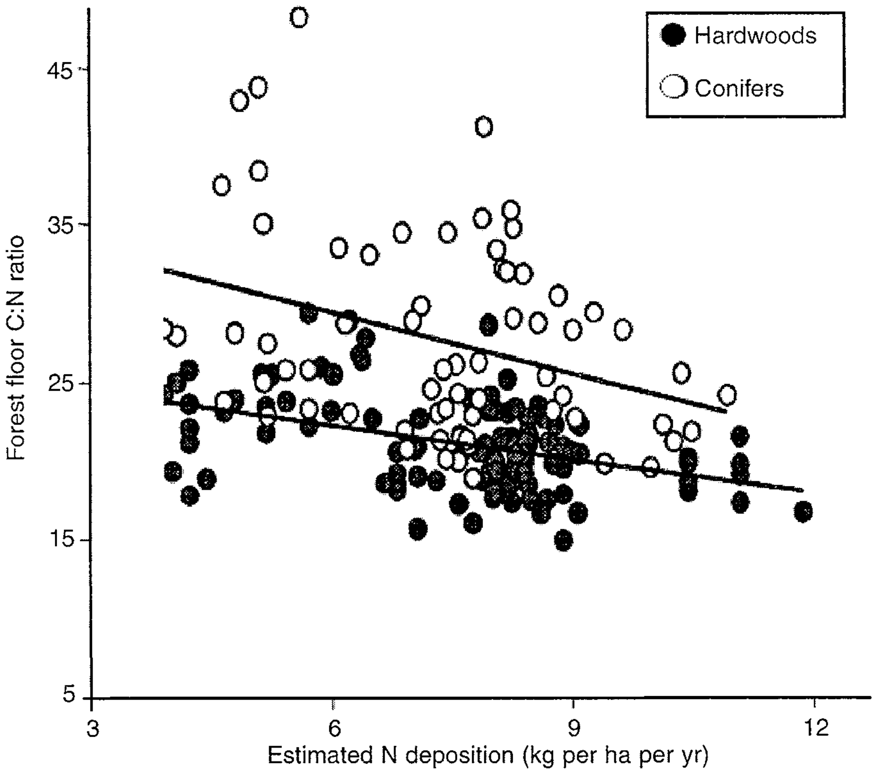

Figure 5.5. Carbon to nitrogen ratios (C:N) in the forest floor in relation to nitrogen deposition estimates in hardwood ($R^2=0.19$, $P<0.001$) and conifer ($R^2=0.27$, $P<0.001$) forests, showing different trends for hardwood and conifer stands (hardwood stands, $R^2=0.19$, $P<0.001$; conifer stands). From Aber et al. (2003), reprinted with permission, copyright American Institute of Biological Sciences

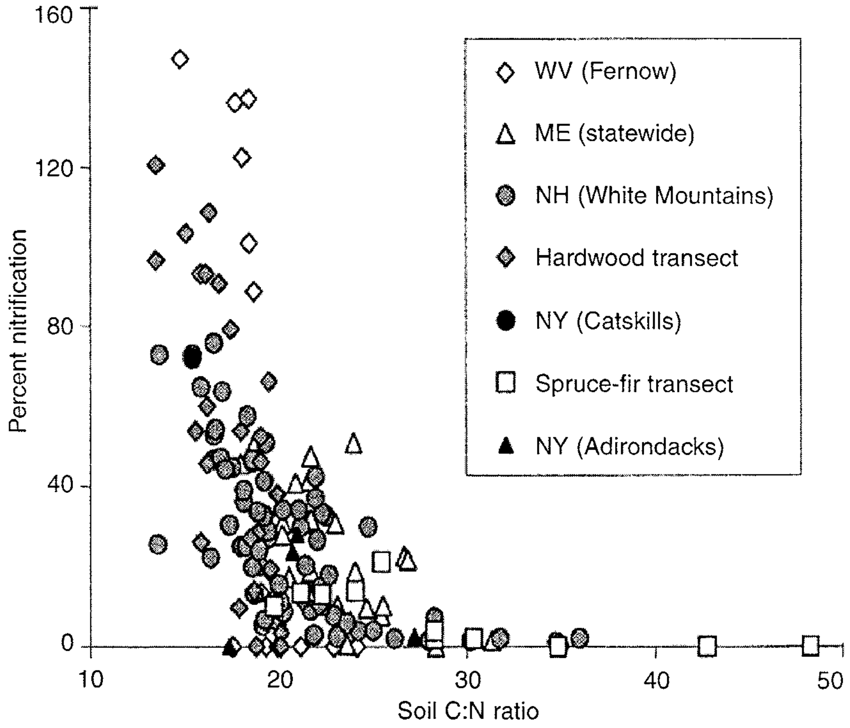

Figure 5.6. Nitrification as a percent of net N mineralization in combined organic and mineral soils in relation to soil C:N ratio. Trends were significant ($P<0.001$) in organic, mineral, and combined soil layers, but data for combined soils are shown here because several studies did not report nitrification rates for individual horizons. From Aber et al. (2003), reprinted with permission, copyright American Institute of Biological Sciences

mental N additions (Kahl et al. 1993). These studies, and others, have led to observations of elevated stream water nitrate to be considered an indicator of the onset of N saturation.

Exports of nitrate and acidity from N saturated forests can have toxic effects on freshwater organisms (Stoddard 1994; Baker et al. 1996) and can contribute to coastal eutrophication (Jaworski et al. 1997; Howarth, this volume). Stoddard (1994) linked nitrate concentrations in streams draining forested catchments to N saturation stages (as described in the previous section). Stoddard showed that nitrate concentrations in streams draining N-limited forests subject to low, background levels of N deposition were near zero throughout an annual cycle. Low nitrate concentrations under low levels of N deposition are due to nearly complete uptake of N deposition by vegetation and soil microbes during growing seasons and to minor exports

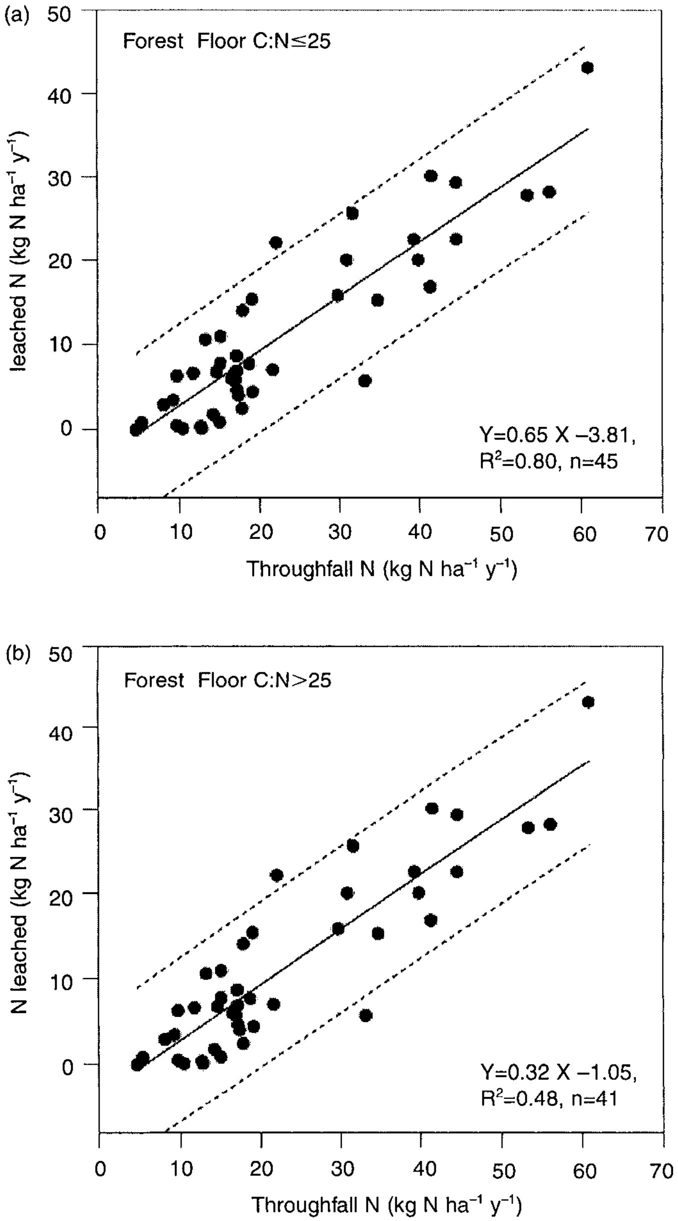

Figure 5.7. Nitrate N leaching losses from soils vs. throughfall N in 86 European forests. Up to 60% of N inputs to soils as throughfall were exported as nitrate leachate in field studies where forest floor C:N was less than or equal to 25. Nitrate losses were lower when forest floor C:N was greater than 25. From MacDonald et al. (2002), reprinted with permission from Blackwell

during winter thaw events and spring runoff. Following the onset of elevated N deposition (N Saturation *Stage 1*), the N cycle becomes more open and nitrate losses increase during spring. As the duration and magnitude of N inputs increase and forests progress to N Saturation *Stage 2*, smaller proportions of nitrate generated by nitrification and nitrate inputs (See Figure 5.3) are removed from soil solution by plants and microbes, and greater proportions are exported to drainage water. This results in increased nitrate losses to streams during storm events and baseflow. If catchment forests progress to *Stage 3*, N cycles become completely open as biological sinks for nitrate decrease and nitrate exports to streams increase.

In general, late stage N saturation with open N cycles is more likely when N deposition is high, forest stands are either old or late-successional, soils are poorly buffered (low levels of exchangeable Ca^{+2}, Mg^{+2}, K^{+}), and previous management practices have not removed large amounts of soil N from ecosystems. High elevation, steep topography, short growing seasons, and exposure to other stresses (e.g., elevated ozone) can also contribute to nitrate losses (Fenn et al. 1998). A major question regarding the long-term effects of elevated N deposition on forests is whether forests receiving low to intermediate levels of N deposition will eventually become N saturated with N cycles that undergo transition from closed to more open states such as has occurred in some forests subject to high levels of N inputs (~20 to >60 kg N ha^{-1} yr^{-1}). Answering this question will require continued long-term study of forest responses to N deposition and improvements in understanding of controls on N movements between plant, microbial and non-living components of forest ecosystems.

Atmospheric N deposition likely influences forest species assemblages as well as forest biogeochemistry. Changes in the availabilities of growth limiting resources such as inorganic N can ultimately lead to changes in plant community composition by altering competitive relations among species (Wilson and Tilman 1995). It is likely, therefore, that increased N cycling rates and nitrate availability resulting from elevated N deposition will lead to changes in forest species composition. This has been reported for ground flora and lichens in European forests (Buecking 1993; van der Eerden et al. 1998; van Dobben et al. 1999) and under experimental N additions in the United States (Rainey et al. 1999). Increased turnover of fine roots associated with increased N cycling and changes in soil chemistry and plant nutrient uptake associate with N deposition can also lead to declines in mycorrhizal fungi (Arnolds 1991; Ruehling and Tyler 1991; Wallenda and Kottke 1998). Changes in mycorrhizal species and abundances could have important implications for carbon storage and N retention in forest soils (Aber et al. 1998).

The progression of forests through various stages of N saturation likely leads to changes in forest tree species composition as well as in microbial and ground flora species. The most likely shifts are from conifer species with high N use efficiencies to deciduous tree species with lower N use efficiencies and higher N uptake requirements (McNulty et al. 1996). Results from long-term studies of forest responses to N deposition will be required to further identify how long-lived species such as dominate most forests will respond to N deposition.

5. Remaining Questions

The elevated levels of N deposition impacting forests and other terrestrial ecosystems began scarcely a half-century ago, and the long term impacts of this factor are not yet fully understood. A number of questions remain regarding the effects of N inputs on forest biogeochemical processes, species composition, and on interactions of forests with other terrestrial and aquatic ecosystems. Overall, regional analyses (See Howarth and Driscoll et al., this volume) indicate that North American temperate forest are retaining most of the N deposition they now receive, but can release significant amounts to downstream ecosystems if soils become acidified and N availability exceeds biotic demand. Will forests continue to retain and store most N deposition? Or, will N retention efficiencies decrease as deposition proceeds? What are the upper limits to N storage in forests? How do plant, microbial and physical processes interact to set limits of N storage in forests and other terrestrial ecosystems? How are plant communities and soil microbial communities changing with N deposition? How are changes in plant and microbial communities feeding back to alter N cycling and nitrate outputs? How will other changing climatic factors, such as increasing temperatures (winter and summer), longer growing seasons, moisture availability and pollutants (especially ozone) alter N cycling and retention in ecosystems. These, and related questions, will require our attention if we are to make more reliable predictions of ecosystem responses to N deposition and acidification.

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6. Atmospheric Deposition and Nitrogen Pollution in Coastal Marine Ecosystems

Robert W. Howarth

Since the first appreciation of the widespread occurrence of acid rain in North America (Likens and Bormann 1974), most public attention has focused on the acid component rather than effects from the associated elements in atmospheric deposition. The emphasis has been on freshwater ecosystems and forests in sensitive regions with relatively low buffering capacity. Effects of acid deposition on coastal marine ecosystems have usually not been considered, which makes sense in the context of acidity. Marine ecosystems are very well buffered, since they contain large amounts of dissolved carbonate and bicarbonate, and consequently are quite insensitive to acid inputs. Similarly, marine waters contain huge quantities of sulfate (~ 28 mM) and thus are not sensitive at all to inputs of sulfate associated with acid deposition. On the other hand, nitrogen (N) pollution can cause severe degradation in coastal marine ecosystems, and the role of atmospheric deposition as a contributor of nitrogen to coastal waters has received increasing scrutiny over the past 15 years since Fisher and Oppenheimer (1991) noted that the nitrate anion associated with nitric acid in acid rain may be a major source of nitrogen to Chesapeake Bay.

The effects of nitrogen in marine ecosystems are due to its fertilizing effect. In most coastal marine ecosystems, rates of primary production are limited by the supply of available forms of nitrogen (primarily nitrate and ammonium). As nitrogen availability increases, so do rates of primary production by phytoplankton (Nixon et al. 1996). In moderation, this can be viewed as beneficial, since it can also lead to greater rates of production higher up the trophic structure leading, for example, to greater fish production (Nixon 1988; Caddy 1993). However, excess inputs of nitrogen lead to eutrophication and associated deleterious ecological changes (Figure 6.1; Caddy 1993; Nixon 1995; National Research Council 2000; Rabalais 2002). These changes include hypoxia (low oxygen zones) and anoxia (zero oxygen zones), alterations in community structure, degradation of habitat quality, loss of biotic diversity and increased incidences and duration of harmful algal blooms (National Research Council 2000; Rabalais 2002). Two thirds of the coastal rivers and bays in the United States are moderately or severely degraded from nitrogen pollution (Bricker et al. 1999), and excess

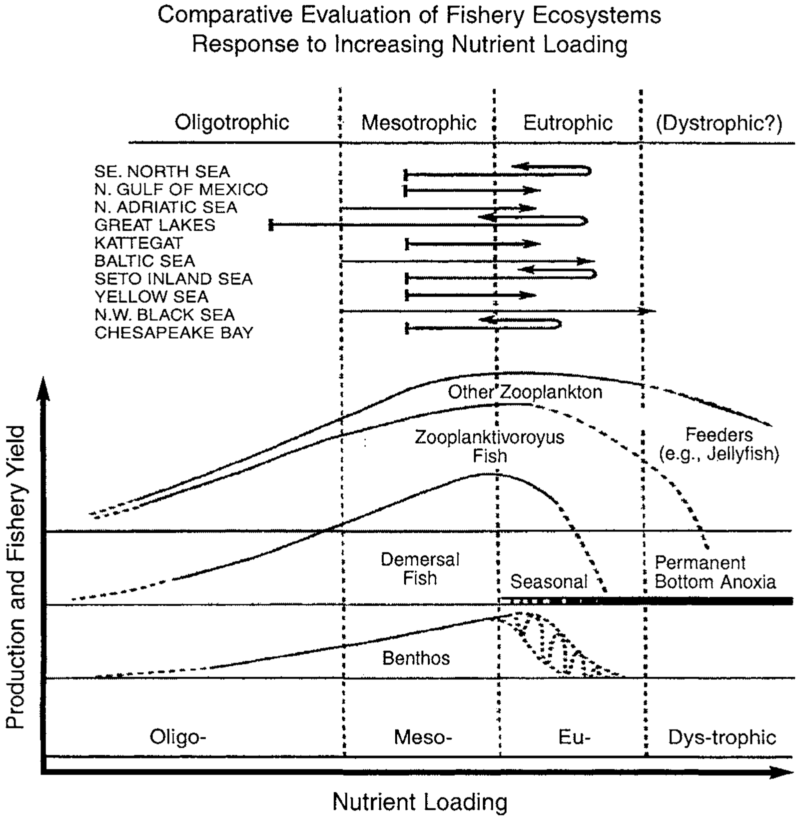

Figure 6.1. The response of secondary production and fishery yield in coastal marine ecosystems to increased nutrient loads (primarily nitrogen). Note the non-linear responses, with production at all trophic levels increasing as nutrient inputs increase at lower levels of loading, but declines in secondary production and fishery yields at higher nutrient loads. Demersal fisheries (those that are dependent upon the bottom waters and sediment) decline before fish dependent on zooplankton in the water column for their food. Modified from Caddy (1993)

Nitrogen inputs represent the largest pollution problem in the nation's coastal waters and one of the greatest threats to the ecological integrity of these ecosystems (National Research Council 1993, 2000; Carpenter et al. 1998; Howarth et al. 2000, 2002a, 2005). The inputs of nitrogen to the coastal waters of the United States are projected to continue to increase over future decades (Howarth et al. 2002a, 2005), in part due to rapid population growth in the coastal zone (Paerl 1997).

1. How Important is Atmospheric Deposition as a Source of Nitrogen Pollution to Coastal Waters?

For the United States as a whole, we have estimated that atmospheric deposition of nitrogen that originates from fossil-fuel combustion contributes on average 30% of the total nitrogen inputs to coastal marine ecosystems, while another 10% of these nitrogen inputs come from ammonia volatized into the atmosphere from agricultural sources (Howarth and Rielinger 2003). The rest of the nitrogen inputs to coastal waters come from runoff from agricultural sources and from municipal and industrial wastewater streams. The variation in both the fluxes of nitrogen to the coast and the source of the nitrogen is great across regions (see below), and in some regions and watersheds atmospheric deposition may be the single largest source of nitrogen.

Some of the nitrogen inputs from atmospheric deposition is nitrogen deposited directly onto the surface of coastal waters. This direct deposition to surface waters often contributes between 1% to 40% of the total nitrogen inputs to coastal ecosystems (Nixon et al. 1996; Paerl 1997; Howarth 1998; Paerl and Whitall 1999; Valigura et al. 2000). The direct deposition is most significant in very large systems, such as the Baltic Sea (Nixon et al. 1996) or in coastal systems which have relatively small watersheds in comparison to the area of the surface waters, such as Tampa Bay (Zarbock et al. 1996).

In most coastal marine ecosystems, the major route whereby atmospheric deposition contributes nitrogen is not the direct deposition onto surface waters but rather deposition onto the terrestrial landscape with subsequent downstream export in streams and rivers. As discussed below, these fluxes are difficult to measure, leaving significant uncertainty and debate about their magnitude. In the northeastern United States as a whole (Gulf of Maine through Chesapeake Bay), our studies have suggested that atmospheric deposition is the single largest source of nitrogen to coastal waters (Howarth et al. 1996; Jaworski et al. 1997; Boyer et al. 2002), while other studies have concluded atmospheric nitrogen deposition is the second largest source after wastewater discharges from sewage treatment plants (Driscoll et al. 2003). Our approach leads to the conclusion that atmospheric deposition of nitrogen onto the landscape—considering only the deposition of oxidized nitrogen compounds that originate from fossil fuel combustion (NOy)—contributes between 25% to 80% of the nitrogen flux in the different major rivers of New England (Figure 6.2; Boyer et al. 2002; Howarth and Rielinger 2003) and approximately 25% of the nitrogen flux in the Mississippi River (National Resource Council 2000; Howarth et al. 2002b). Using another approach—SPARROW, or Spatially Referenced Regressions on Watershed Attributes model—Alexander et al. (2000) concluded that atmospheric deposition onto the landscape contributed between 4% and 35% of the nitrogen flux in 40 major coastal watersheds across the United States, with the highest contribution in the northeastern and mid-Atlantic regions.

The uncertainty over the contribution of atmospheric deposition as a nitrogen source to coastal marine ecosystems stems from two issues: uncertainty over the magnitude of nitrogen deposition onto watersheds (particularly from "dry deposition"), and uncertainty over the amount of the deposited nitrogen that is subsequently exported downstream (National Resource Council 2000; Howarth et al. 2002b). Each of these is discussed in some detail in the following sections.

2. How Large is Dry Deposition?

The vast majority of measurements of nitrogen deposition in the United States—including those made by the National Atmospheric Deposition Program (NADP)—measure only "wet deposition" (i.e., N in rainfall and

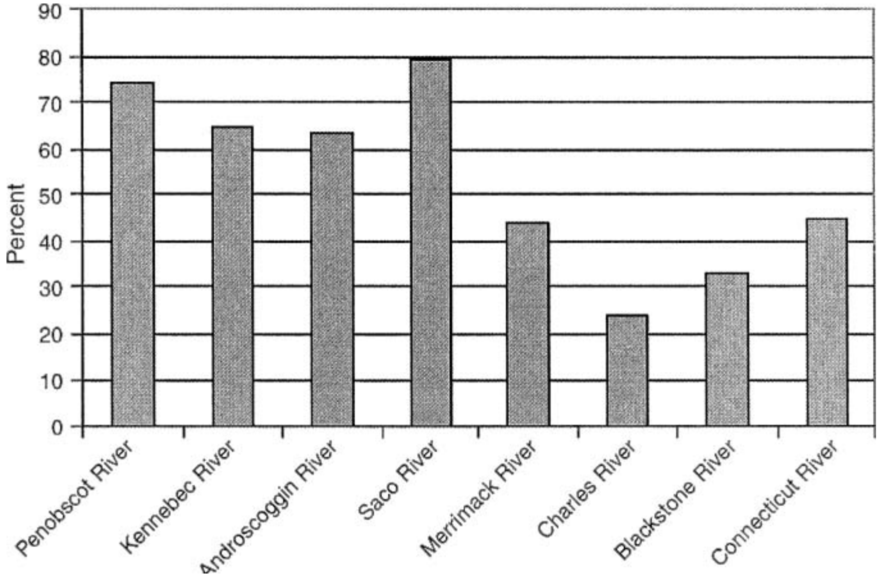

Figure 6.2. Percentage of nitrogen in major New England rivers that originates from fossil-fuel derived atmospheric deposition onto the landscape. Reprinted from Howarth and Rielinger (2003), based on data in Boyer et al. (2002)

snow). To estimate wet deposition onto an entire watershed, data at particular monitoring sites are extrapolated statistically considering factors such as local topography and precipitation (Ollinger et al. 1993; Grimm and Lynch 2005).

Substantial quantities of nitrogen can be deposited from the atmosphere as "dry deposition," which includes aerosols and other particles and uptake of gaseous forms of nitrogen by vegetation, soils, and surface waters. Both in the United States and Europe, the extremely sparse spatial coverage in networks for measuring dry deposition severely limits estimation of this process (Holland et al. 2005). In the United States, dry deposition is routinely measured only at sites that are part of the CASTNet and AIRMon-Dry programs. At the peak of these programs in the 1990s, these networks consisted of a total of 93 sites across the country, but the number is now substantially less (pers. communication, Richard Artz, Deputy Director of the NOAA Air Resources Lab). In the watersheds of Chesapeake Bay—an area of 165,000 km^2 that includes land in six states—there are only eight stations for monitoring dry deposition. In practice, given the sparse coverage of dry deposition monitoring, rates are often estimated as some percentage of wet deposition, based on comparisons of these two processes at a relatively few sites (Ollinger et al. 1993; Holland et al. 1999, 2005).

In addition to the limited spatial extent of the dry deposition monitoring networks, these networks do not measure all of the components that can be deposited. For example, particulate nitrate and particulate ammonium are routinely measured, as is nitric acid vapor. However, other gaseous nitrogen compounds that may play a significant role in deposition (i.e., nitric oxide, nitrogen dioxide, peroxy and alkyl based organics, and ammonia gas) are not measured. Nitric oxide and nitrogen dioxide are the major gases emitted from fossil fuel combustion, while ammonia is the major form of air pollution from agricultural sources. Ammonia is also released in vehicle exhaust, although at lesser amounts than for nitric oxide and nitrogen dioxide (Baum et al. 2001; Cape et al. 2004). To the extent these compounds are deposited, the dry depositional monitoring networks are underestimating total deposition. As currently measured, the dry deposition at the eight CASTNet sites in the Chesapeake Bay watershed ranges from 23% to 38% of total deposition, but the actual contribution when all forms of nitrogen gases are considered must certainly be higher.

There are other potential sources for the underestimation of nitrogen dry deposition. The manner in which dry deposition rates are calculated—multiplying concentration data obtained at the monitoring sites by "depositional velocities"—may also result in underestimation. For the AIRMon and CASTNet sites, these deposition velocities are estimated as a function of vegetation and meteorological conditions (Clarke et al. 1997). Our knowledge of depositional velocities is based on studies in flat, homogenous terrain; as noted by Bruce Hicks (former Director of the NOAA Air Resources Lab), when estimating dry deposition "we are simulating the world on the assumption that our understanding of [these] special cases applies everywhere ... We often display unwarranted confidence" in our estimates (Hicks presentation to the annual meeting of the American Society of Meteorology, October 2005). Complex terrain is likely to substantially increase depositional velocities. Vegetative cover is also important, and different models can vary in their estimates of spatial integrated dry deposition by more than five-fold depending upon different assumptions of the effect of vegetation (particularly coniferous forests) on depositional velocities (Weseley and Hicks 1999; Holland et al. 2005).

3. Two Approaches for Estimating Total Nitrogen Deposition in the Northeastern United States

Boyer et al. (2002) estimated the average deposition of oxidized nitrogen (NOy) onto the landscape of the major rivers of the northeastern United States, including both wet and dry deposition, following the approach of Ollinger et al. (1993) in using a statistical extrapolation of wet deposition monitoring data and relating dry deposition to wet deposition at particular sites. We found a range of values across these watersheds from ~ 360 kg N km^{-2} yr^{-1} in the Penobscot River basin in Maine to ~ 890 kg N km^{-2} yr^{-1} in the Schuylkill River basin in Pennsylvania (Boyer et al. 2002). The average value for this set of watersheds is ~ 680 kg N km^{-2} yr^{-1}.

Another approach for estimating nitrogen deposition onto the landscape can be obtained from models based on emissions to the atmosphere, with consideration of reaction and advection in the atmosphere, followed by deposition (Figure 6.3). We used one of these models (the GCTM model; Prospero et al. 1996) to estimate nitrogen deposition in all of the regions—including the northeastern United States—that surround the North Atlantic Ocean (Howarth et al. 1996). The GCTM model predicts depositional patterns globally at a relatively course spatial scale using emission sources as inputs and modeling atmospheric transformations and transport (Prospero et al. 1996). For the northeastern United States, the GCTM model yielded an estimated total nitrogen deposition (wet plus dry) of ~ 1,200 kg N km^{-2} yr^{-1}, a value 80% greater than that derived by Boyer et al. (2002) from extrapolation of wet-deposition monitoring data (Figure 6.4; Howarth et al. 2006). A similar, more recent emission-based model (TM3) developed by Frank

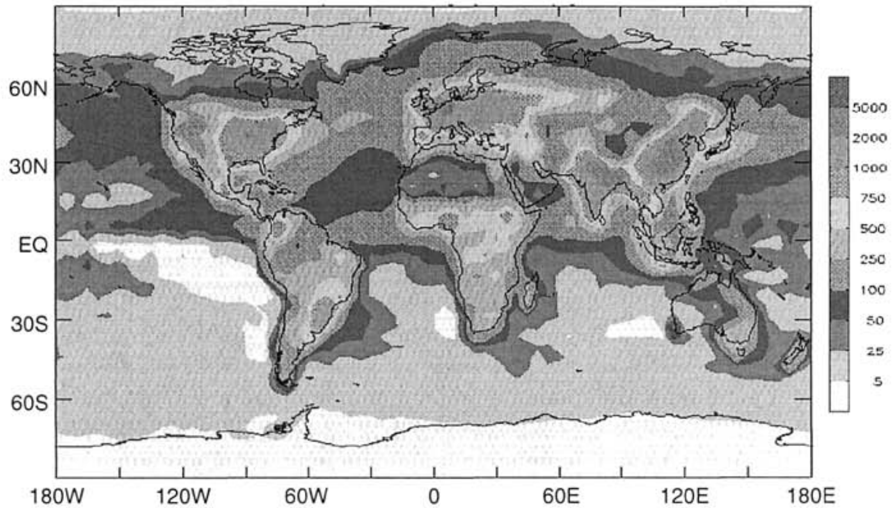

Figure 6.3. Pattern of average total deposition of inorganic nitrogen as of the early 1990s based on the TM3 model of Frank Dentener. Units are kg N km^{-2} yr^{-1}. Modified from a color graphic in Galloway et al. (2004)

Dentener and colleagues and used by Galloway et al. (2004) for their global and regional nitrogen budgets yields a comparable estimate for the northeastern United States as did the GCTM model (Howarth et al. 2006). These emission-based models are attractive, in that at least at very course spatial scales, they are as accurate as the emission data. However, they cannot easily be applied at a spatial scale fine enough to give estimates for the 16 individual northeastern watersheds.

Why is the estimate from the emission-based model (Howarth et al. 1996) so much greater than that from estimates based on extrapolation of the wet deposition monitoring data (Boyer et al. 2002)? There are three possible explanations, which are not mutually exclusive:

1. Deposition on the relatively urbanized coastal plain may be much greater than in the watersheds away from the coast. The watershed areas considered by Boyer et al. (2002) end upriver from the coast and tend to be more rural than is the coastal plain downstream (see Figure 6.4). Recent studies have found evidence that deposition near emission sources can be much greater than deposition away from emission sources; deposition within New York City was more than twice as high than in more rural areas to the North of the city (Lovett et al. 2000), and deposition in the immediate vicinity of

roads was much higher than a few hundred meters away (Cape et al. 2004; unpublished data of Howarth, Bettez, Marino, and Davidson);

2. The estimate based on wet deposition monitoring data (Boyer et al. 2002) may underestimate total deposition. This is of course likely, to the extent that dry deposition is underestimated. As noted above, not all of the important gases that may be deposited are routinely measured by the dry deposition monitoring networks, and depositional velocities may be underestimated in regions with major terrain features. Further, these networks are not designed to measure deposition in the immediate vicinity of emission sources (point #1, above). In fact, most of the NADP wet deposition monitoring sites are intentionally located far away from urban emission sources ; and
3. The estimate from emission-based modeling (Howarth et al. 1996) may overestimate total deposition. This could occur if emissions are overestimated, which may well be true for ammonia emissions but probably not for emissions of oxidized nitrogen to the atmosphere in

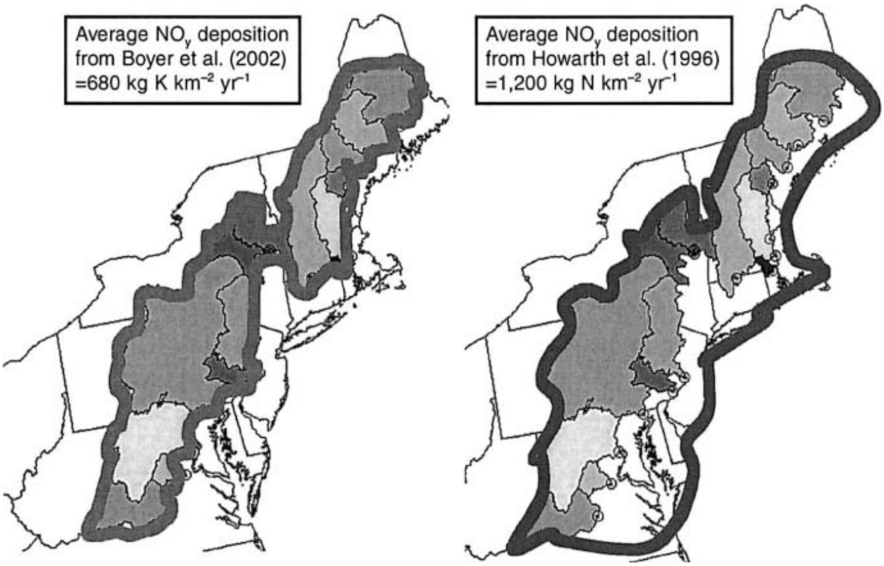

Figure 6.4. The geographic area considered by Boyer et al. (2002) was the area of 16 watersheds in the northeastern United States upriver from the lowest gaging station of the USGS (left). The area considered by Howarth et al. (1996) is somewhat larger, and includes the area on the coastal plain (right). Note that the average estimates for deposition of oxidized nitrogen pollution originating from fossil fuel combustion is ~ 80% greater in the Howarth et al. (1996) analysis, probably due to different approaches used for the estimation and/or the different area considered

> the United States (Holland et al. 1999). The difference between the Howarth et al. (1996) and Boyer et al. (2002) estimates highlighted in this chapter is for deposition of oxidized nitrogen. Alternatively, the emission-based modeling may not accurately capture the spatial pattern of the deposition. These models rely on a mass balance of nitrogen in the atmosphere, so global deposition estimates are as accurate as the emissions data that feed them. However, deposition may be underestimated in some regions and correspondingly overestimated elsewhere.

Obviously, significant uncertainty exists in the overall magnitude of total nitrogen deposition in an area such as the northeastern United States. When considering points #1 and #2 above, it is important to note that extrapolations based on wet-depositional monitoring (Ollinger et al. 1993; Grimm and Lynch 2005) do not appear to capture any evidence of higher deposition near urban centers and transportation corridors. For reasons discussed further below, I believe it likely that these traditional approaches that use NADP wet deposition monitoring data to estimate total nitrogen deposition result in substantial underestimates, especially for total nitrogen deposition in the urbanized portions of the northeastern United States.

4. Throughfall as an Approach for Estimating Total Nitrogen Deposition

The difficulty with measuring dry deposition of N (particularly of gaseous forms such as NO, NO_2, and NH_3) has led some investigators to use tree-canopy throughfall as a surrogate for total N deposition (Lajtha et al. 1995; Lovett et al. 2000; Weathers et al. 2005; Schmitt et al. 2005). Throughfall is the material that falls through the canopy of a forest, and so includes whatever is deposited on the canopy in both wet and dry deposition, plus the net exchange of material with the vegetation. Most studies have found that the assimilation of nitrogen from deposition into leaves of the canopy is generally as great as or greater than the leaching of nitrogen out of leaves (Lindberg et al. 1990; Johnson 1992; Lovett and Lindberg 1993; Dise and Wright 1995; Lajtha et al. 1995). Consequently, many experts on atmospheric deposition have argued that throughfall measurements provide a minimum estimate of total nitrogen deposition (Lindberg et al. 1990; Johnson 1992; Lovett and Lindberg 1993; Dise and Wright 1995; Lajtha et al. 1995; Lovett et al. 2000; Schmitt et al. 2005).

The estimation of total nitrogen deposition from throughfall measurements often yields much higher rates than those inferred from wet deposition data. For example, in a forest in Falmouth, MA, on Cape Cod, Lajtha et al. (1995) measured wet deposition of 420 kg N km^{-2} yr^{-1} and estimated a total deposition rate of 840 kg N km^{-2} yr^{-1} by assuming that dry deposition equaled wet deposition. This estimate is quite similar to the deposition predicted for that location by the spatial extrapolation of Ollinger et al. (1993). However, from their throughfall data, Lajtha et al. (1995) estimated that actual total nitrogen deposition at the site was 1,310 kg N km^{-2} yr^{-1}, or more than 50% greater. In a more recent study, Weathers et al. (2005) compared throughfall data with more traditional approaches for estimating nitrogen deposition in the Acadia National Park in Maine and in the Great Smoky Mountains National Park in North Carolina. In both locations, they found that total nitrogen deposition rates estimated from their throughfall data were 70% greater than those estimated from NADP and CASTNet wet and dry monitoring data. These throughfall estimates lend strength to the argument that the traditional approaches for estimating total deposition—such as we used in Boyer et al. (2002)—yield low values.

5. What is the Fate of Nitrogen Deposited onto the Landscape?

Forests are the dominant land cover in the northeastern United States (Boyer et al. 2002), and so much of the nitrogen deposition on the landscape falls on forests. Only a portion of the nitrogen is exported downstream, and much is retained in the forests or denitrified (converted to non-reactive, molecular N_2). Productivity of most forests in the United States is limited by the supply of nitrogen (Vitousek and Howarth 1991), so as forests receive more nitrogen from atmospheric deposition, production and storage of nitrogen in organic matter can be expected to increase. On average for the northeastern United States, approximately 60% to 65% of the nitrogen inputs to forests through natural nitrogen fixation as well as atmospheric deposition are retained in the forest (primarily accreted in woody biomass) or harvested from the forests in wood (Goodale et al. 2002; van Breemen et al. 2002). A little over 20% is exported from the forest in streams (primarily nitrate, but also dissolved organic nitrogen), with the rest denitrified (van Breemen et al. 2002). The ability of forests to store nitrogen, however, is limited, and forests can become nitrogen saturated when inputs exceed the needs of trees and the ability for soils to assimilate nitrogen (Aber et al. 1989; Gundersen and Bashkin 1994; Emmett et al. 1998). Nitrogen export downstream can then increase dramatically (Emmet et al. 1998; Howarth et al. 2002b; Aber et al. 2003).

A recent comparative study suggests that for the forests of northern New England, the nitrate concentrations in streams and small lakes just downstream increase dramatically as total nitrogen deposition increases above 600 to 800 kg N km^{-2} yr^{-1} (Figure 6.5; Aber et al. 2003), indicating a substantial increase in nitrogen export from the forests receiving the higher deposition. Figure 6.5 indicates the estimated average NOy deposition for the northeastern United States in the Boyer et al. (2002) and Howarth et al. (1996) studies. Note that total deposition, including ammonia, ammonium, and organic nitrogen, would be greater by 20 to 40% (Boyer et al. 2002; Howarth et al. 1996), but is also much more uncertain (Holland et al. 1999; Howarth et al. 2006), so I have chosen to illustrate just the NOy component. Note also that the deposition estimates used in the Aber et al. (2003) analysis may also be low, since these are based on extrapolation of wet-monitoring data. On the other hand, all of the data in the analysis of Aber et al. (2003) are from fairly rural sites, relatively far from emission sources;

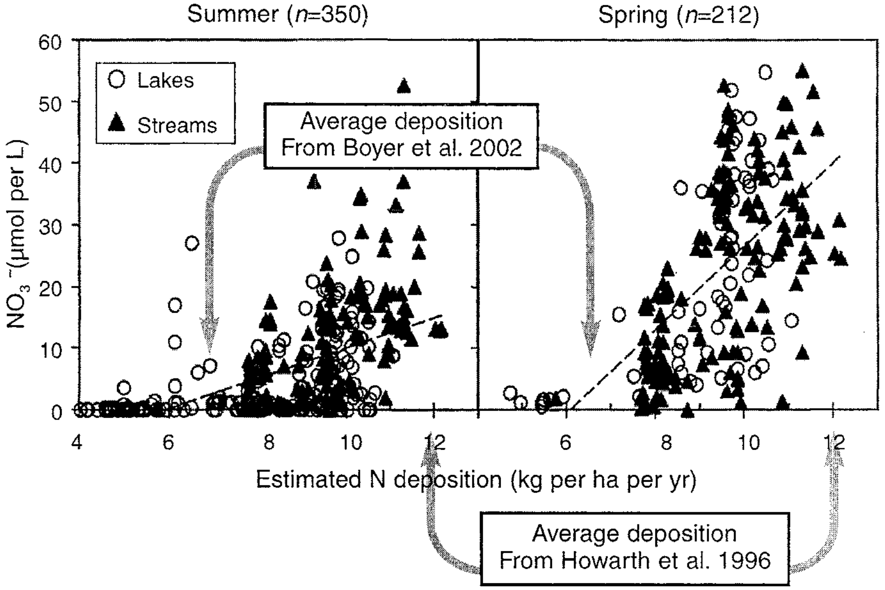

Figure 6.5. Concentrations of nitrate in small streams and lakes in forested catchments in northern New England in the spring (right) and summer (left) as a function of NOy deposition onto the landscape. Note the non-linear response, with nitrate concentrations tending to increase as deposition exceeds 6 to 8 kg N per hectare per year (600 to 800 kg N km^{-2} yr^{-1}). The arrows indicate the average deposition rates for oxidized nitrogen compounds (NOy) estimated for the northeastern United States in Boyer et al. (2002) and Howarth et al. (1996). Modified from Aber et al. (2003)

their deposition estimates may therefore be fairly reliable. In any case, Figure 6.5 suggests that nitrogen deposition onto the landscape on average in the northeastern United States is likely high enough to result in elevated losses of nitrogen from forests, particularly if the higher emission-based estimates used by Howarth et al. (1996) are valid.

While forests are often retentive of nitrogen, impermeable surfaces such as roads and parking lots are far less so. While not often studied, nitrogen runoff from these surfaces can be substantial. For example, runoff from highways near Providence, RI, is reported to be 1,700 kg N km^{-2} yr^{-1} (Nixon et al. 1995). Most if not all of this nitrogen likely originated from atmospheric deposition.

6. Case Study: How Important is Deposition to the Nitrogen Budget for Chesapeake Bay?

Chesapeake Bay is the largest estuary in the United States, and one of the most sensitive to nutrient inputs (Bricker et al. 1999; National Resource Council 2000). Nitrogen inputs to Chesapeake Bay have caused widespread loss of seagrasses (as they are shaded out by blooms of phytoplankton) and have greatly increased the volume of anoxic bottom waters. The role of atmospheric deposition as a source of nitrogen to the Bay apparently was not considered until Fisher and Oppenheimer (1991) suggested that it might contribute 40% of the total inputs. Their analysis was simple and preliminary, and was not believed by many scientists who worked on water quality in this estuary system. The most recent analyses by the Chesapeake Bay Program, while giving lower percentages, also suggest that deposition is important, contributing ~ 25% of the total nitrogen inputs to Chesapeake Bay (7% from direct deposition onto the surface waters of Chesapeake Bay, and 19% from deposition onto the landscape with subsequent export to Chesapeake Bay, using 2003 values; http://www.chesapeakebay.net/status.cfm?SID=126; see also http://www.chesapeakebay.net/nutr1.htm).

Two lines of evidence suggest that the Bay Program model may be underestimating the inputs of nitrogen from atmospheric deposition: 1) the model may be underestimating the magnitude of deposition onto the landscape; and 2) the model may be underestimating the percentage of deposition onto the landscape that is subsequently exported downstream. Each of these is discussed below.

The Chesapeake Bay Program model relies on an estimate of total nitrogen deposition of 1,210 kg N km^{-2} yr^{-1} (calculated from Figure A-4 Environmental Protection Agency 2003). The approach to derive this estimate is very similar to that used by Boyer et al. (2002): extrapolation from wet-deposition monitoring data for the 15 NADP sites in the watersheds of the Chesapeake (Lewis Linker, Bay Program modeling coordinator, PowerPoint presentation by conference call, January 9, 2006), although the Boyer et al. (2002) estimate is in fact somewhat lower (1,010 kg N km^{-2} yr^{-1} for the area-weighted mean for the watersheds of the Susquehanna, Potomac, Rappannnock, and James Rivers up river of the USGS gaging stations). If we assume that the Boyer et al. (2002) estimate underestimates by 80% (based on comparison with the global-scale emission-based model used by Howarth et al. 1996), then actual deposition on the Chesapeake watersheds may be as great as 1,550 kg N km^{-2} yr^{-1} (28% greater than assumed for the Chesapeake Bay Program model).

Perhaps of greater significance is the treatment of nitrogen retention in the landscape by the Bay model. The model treats retention as a function of land use, and on average assumes that 86% to 89% of total nitrogen deposition onto the landscape is retained, and only 11% to 14% is exported downstream to the Bay (calculated from Figure A-4, Environmental Protection Agency 2003). Most of this retention is assumed to occur in the 57% of the area of the watershed that is forested, with the export coming from deposition onto agricultural lands and urban and suburban areas with impermeable surfaces. The model assumes that most of the forests in the Chesapeake Bay basin are not nitrogen saturated, and therefore leak little if any nitrogen (Environmental Protection Agency 2003).

The average export of nitrogen deposition from all land uses (12%) seems low in comparison with the estimate that average forests in the northeastern United States export over 20% of nitrogen deposition (Goodale et al. 2002; van Breemen et al. 2002). If the deposition in the Chesapeake basin is evenly distributed over land uses, then 43% falls on other land uses where much higher rates of export would be expected. If much of the deposition from nitrogen pollution that originates from vehicles falls near these emission sources (either onto impermeable surfaces or onto vegetation where the rate of deposition would be very high), then very high rates of export might be expected. Obtaining better data on nitrogen retention in the landscape has been identified as a high national research need in a multi-agency federal planning document (Howarth et al. 2003). But given current knowledge, it is probably as reasonable to assume that the percent export from atmospheric deposition onto the landscape of the Chesapeake Bay basin—including all land uses—is 30% as to assume the 12% used by the Bay model. Ranges from 20% to 40% and even higher can be reasonably inferred from studies of large watersheds (National Research Council 2000; Howarth et al. 2002b, 2006; Boyer et al. 2002).

Table 6.1 illustrates the sensitivity of nitrogen loading to Chesapeake Bay to various assumptions on the rate of deposition and on nitrogen retention in the landscape. Within this range of reasonable assumptions, the total input of nitrogen to the Bay from atmospheric deposition (both directly onto the surface waters of the Bay and indirectly from deposition onto the landscape and subsequent export downstream) ranges from 34 to 92 thousand metric tons of nitrogen per year, and comprises from 25% to 50% of the total nitrogen load to Chesapeake Bay from all sources. Under the assumptions of greater deposition and lower retention in the landscape, the estimate for total nitrogen load to Chesapeake Bay increases substantially – from 130 to 188 thousand metric tons per year, or 45% greater total nitrogen load. Perhaps surprisingly, monitoring of the load of nitrogen to Chesapeake Bay is not adequate to constrain this total load estimate within this range of uncertainty. As with many other large coastal marine ecosystems, significant portions of the watersheds of Chesapeake Bay are not gaged because of the difficulty in gaging tidal streams and rivers (Valigura et al. 2000;

Table 6.1. Importance of atmospheric deposition as a source of nitrogen pollution to Chesapeake Bay under various assumptions. Fluxes are thousands of metric tons of nitrogen per year. Percentage values given in parentheses are percentages of total nitrogen load. The baseline run assumptions are from EPA (2003)

	Total Load to Bay	Input to Bay from Direct Deposition onto Bay Water Surface	Input to Bay from Deposition onto Watersheds	Total Input to Bay from Deposition
Chesapeake Bay model (2000 conditions)	130	9 (7%)	25 (19%)	34 (26%)
Deposition increased to 1,550 kg N km^{-2} yr^{-1}; no change in retention assumptions	140	12 (9%)	32 (23%)	44 (32%)
Chesapeake Bay model assumptions on deposition rate; assume 70% retention in landscape	168	9 (5%)	63 (38%)	72 (43%)
Deposition increased to 1,550 kg N km^{-2} yr^{-1}; assume 70% retention in landscape	188	12 (6%)	80 (43%)	92 (49%)

National Research Council 2000; Howarth et al. 2002b). These areas of the watershed are therefore not monitored for their nutrient inputs to the Bay. While the fluxes of nitrogen from the watersheds above gaging stations in the Chesapeake Basin are reasonably well known, the fluxes from the watershed in the more urbanized areas on the coastal plain near Chesapeake Bay—where nitrogen deposition may be much greater, and retention of nitrogen in the landscape much less—are estimated only from models and not from empirical monitoring data.

7. Steps Towards Effective Management

Despite the widespread damage to coastal waters from nitrogen pollution, for the most part governments have been slow to systematically apply effective policies for controlling this problem, in the United States or elsewhere (NRC 2000; Howarth et al. 2005). The reasons for this policy failure are many, but one major reason is that management of eutrophication or nutrient pollution generally has focused on phosphorus rather than nitrogen since the early 1970s (Howarth and Marino 2006; Howarth et al. 2005). While this is appropriate for freshwater lakes, nitrogen is the larger problem in most coastal marine ecosystems (National Research Council 2000; Howarth and Marino 2006). Although some local or regional agencies have addressed nitrogen pollution in coastal waters over the past two decades, even today no national standards for coastal nitrogen pollution exist (National Research Council 2000; Howarth et al. 2005). Scientific evidence for the necessity of phosphorus control on eutrophication in freshwater lakes and nitrogen control in coastal marine ecosystems has steadily accumulated for many decades, but only in the past 5–10 years has this evidence begun to be fully accepted by water quality managers. Even when managers have recognized that nitrogen is the prime cause of eutrophication in coastal rivers and bays, management practices for non-point sources of nitrogen often have remained focused on those proven effective for managing phosphorus pollution, with insufficient recognition that other practices may be needed for nitrogen because of its much greater mobility in groundwater and through the atmosphere (National Research Council 2000; Howarth et al. 2005; Howarth 2005).

Both fossil fuel combustion and agricultural practices contribute significantly to atmospheric fluxes of nitrogen but not phosphorus. The magnitude of the contribution of these atmospheric fluxes to coastal nutrient pollution remains uncertain, and understudied. Nonetheless, atmospheric deposition is clearly an important contributor to coastal nutrient pollution. This source demands more attention by water quality managers if the goal of reducing coastal nutrient pollution is to be met (National Research Council 2000).

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Part II

Acid Emissions Energy and Policy

7. The Politics of Acid Rain in Europe

Miranda A. Schreurs

1. Introduction

Acid rain has been a pivotal issue in the development of European environmental policies and programs. Appreciating the history of European responses to acid rain is also useful for understanding how the world has moved towards greater use of international environmental agreements to address transboundary pollution issues, such as stratospheric ozone depletion and global climate change. Indeed, the 1972 United Nations Conference on the Human Environment (UNCHE), the first truly global environmental meeting of the world's heads of state and government, was proposed in response to Sweden's concerns that acid rain originating in Great Britain and Germany (East and West) was responsible for the acidification and death of Scandinavian lakes. At the time of the UNCHE, there was still no scientific consensus or political acceptance of the idea that acid rain could fall as far as a thousand kilometers (600 miles) or more away from its pollution source. Nor was there much appreciation of the need for political action to address the transboundary and global nature of many pollution problems. Sweden used the UNCHE to bring international attention to the problem of transboundary acid rain and other increasingly pressing global environmental concerns.

It was also in response to acid rain problems in Europe that the world's first broadly regional international atmospheric pollution agreement—the Convention on Long Range Transboundary Air Pollution (CLRTAP)—was formulated. Acid rain became an issue taken up by the European Community (EC) in the late 1970s and 1980s, turning an issue that had once been defined as a problem for municipal governments to control into an issue of supranational importance.

Advances in scientific understanding of the causes and consequences of acid rain have transformed European approaches to environmental policy making from an early focus on reducing a single pollutant by a certain, generally politically determined amount (e.g., reducing sulfur oxides by 30% of current levels by a given year) to establishing legislation that targets a mix of pollutants and aims to reduce their levels to an amount that does not damage sensitive environments or human health. This is commonly known as a critical loads approach. As there are many sources of pollutants that contribute to acid rain and scientific understanding of the causes and consequences of acidic deposition has improved with time, there have been numerous changes to policy. Over time, Europe has learned that rather than addressing each atmospheric pollution issue separately, it makes sense to find more comprehensive approaches that address simultaneously multiple problems—such as ground level ozone that contributes to human health problems (especially respiratory problems) and premature death, global climate change, and acid rain.

European responses to acid rain are important as they have become a model for other regions suffering ecological damage from fossil fuel emissions, including East and Southeast Asia and Antarctica. Europe was in many ways the international leader in responding to regional acid rain problems. This does not mean, however, that Europe has been united in its acceptance of acid deposition science and findings of damage or in how best to respond politically to scientific warnings. Nor does it mean that European states have been equally aggressive in their responses or effective in their implementation of policy. Still, there are many lessons that European experiences provide on how states can cooperate to address pollution more effectively.

2. Acid Rain

Acid rain is the common term for a complex chemical and atmospheric process related to the deposition of acidic compounds released from the burning of fossil fuels. Acid rain is but one way in which acid deposition can happen. Acid deposition can occur with precipitation in the form of rain, snow, sleet, or fog (and hence the terms, acid rain, acid snow, and acid fog), but also as dry deposition. Acidic compounds, in other words, can be carried in the atmosphere for long distances and then fall to the earth even without precipitation. Although the term acid rain is used throughout this chapter, it should be understood to refer to the more general phenomena of the wet and dry deposition of air pollutants, including sulfur dioxide (SO_2), nitrogen oxides (NO_x), and volatile organic compounds (VOCs) that can cause acidification or oxidation.

3. Brief Historical Background of Responses to Air Pollution and Acid Rain

The scientific community has long known that SO_2, emitted when burning coal and other fossil fuels, could produce sulfuric acid that could fall back to earth when it rained. In fact, Britain's first Alkali Inspector, Robert Angus Smith, coined the term "acid rain" in his 1872 book, *Air and Rain: The Beginnings of Chemical Climatology.*[1] For the best part of the next century, however, to the extent that acid rain received any attention, it was viewed as a local pollution problem. Air pollution more generally was the main problem in people's minds and even that received but scant attention from national authorities. At times, communities that suffered from factory emissions pressured polluting firms to do something to control their smoke and soot. The response by factories was often simply to build taller smoke stacks, the idea being that this would help to disperse their emissions over wider areas and reduce their concentration at the source. While these measures were successful at reducing somewhat pollution close to factories, they contributed to the regional acid rain problems of Europe.

Before discussing how Europe reacted to its growing acid rain problem, it is useful to step back and consider how Europe responded to air pollution problems more generally. Air pollution was particularly bad in nineteenth century Great Britain, where the Industrial Revolution had been embraced most enthusiastically and where coal became the main fuel driving the engines of modernization. Charles Dickens' stories vividly portray the terrible pollution that shrouded England's cities. He opens his novel, *Bleak House,* with a poignant description of London's pollution: "Smoke lowering down from chimney-pots, making a soft black drizzle, with flakes of soot in it as big as full-grown snowflakes-gone into mourning, one might imagine, for the death of the sun."[2] His book, *Hard Times,* opens with a description of Coketown, a mill town choking in its own pollution: "A SUNNY midsummer day. There was such a thing sometimes, even in Coketown. Seen from a distance in such weather, Coketown lay shrouded in a haze of its own, which appeared impervious to the sun's rays. You only knew the town was there, because you knew there could have been no such sulky blotch upon the prospect without a town. A blur of soot and smoke, now confusedly tending this way, now that way, now aspiring to the vault of Heaven, now murkily creeping along the earth, as the wind rose and fell, or changed its quarter: a dense formless jumble, with sheets of cross light in it, that showed nothing but masses of darkness: – Coketown in the distance was suggestive of itself, though not a brick of it could be seen."[3]

The seriousness of the smoke problems in England (as well as on the continent) led to numerous efforts by local governments to control smoke, such as the 1853 Smoke Nuisance Abatement Act in London and the 1857 Smoke Abatement Act in Scotland, which both defined smoke from furnaces as a nuisance. In 1863, the British Parliament passed the Alkali Act, which set up an Inspectorate to control industrial emissions of hydrogen chloride. Its mandate was expanded over time to deal with other industrial pollutants. In 1887, the Select Committee on Smoke Nuisances determined that fog in London had increased significantly and was as dangerous as any epidemic. In 1926 Parliament passed a Smoke Abatement Act, enabling local governments to pass by-laws to control smoke from industrial sources.[4]

While efficiency improvements, structural shifts to the economy, greater reliance on natural gas and oil (although coal use remained abundant), and local pollution ordinances relieved air pollution levels somewhat, air pollution remained a serious problem and one that took on new dimensions with the advent of the automobile society. The destruction of World War II brought many of continental Europe's economies back to pre-war levels. In the next decades, rapid industrialization based on coal as a primary fuel in both East and West Germany and in neighboring Poland, Czechoslovakia, and Hungary worsened air pollution problems on the continent as well. Coal was relatively abundant and cheap and was a primary source of fuel throughout northern continental Europe and was still widely used in Great Britain. The Black Triangle, an area of concentrated lignite-based power generation in Poland, East Germany, and what is now the Czech Republic, became one of the most densely industrialized, and heavily polluted places on the European continent.

While some measures were taken to control the worst of the smoke dust, soot, and particulate matter, air pollution was considered a matter for local governments to control. It was not until well after World War II, that air pollution control became an accepted area of national governmental jurisdiction in Europe. Priorities were with economic development and air pollution was considered something of an unavoidable by-product of modernization.

Several particularly deadly air pollution incidents in Europe and North America were to help change attitudes about air pollution and to motivate national governments to introduce framework air quality laws. The first incident to gain wide-spread media coverage occurred in December 1930 when over 60 people died from pollution-related respiratory problems in the Meuse Valley in Belgium, an area of steel mills, power plants, glassworks, zinc refining plants, a coking plant, a sulfuric acid plant, and a fertilizer plant. Sulfur compounds were considered the primary culprit. Years later, in 1948, Donora, Pennsylvania was hit with an air inversion, which trapped the region's industrial pollutants. Twenty died and over 6000 became ill. Two years later, in 1950 in Poza Rica, Mexico a low altitude air inversion trapped pollutants, leading to the deaths of 22 individuals and hospitalization of over 300. Recognition grew that air pollution was more than simply a "nuisance."

It was the "killer fogs" of London that attracted the most attention worldwide. London is known and even loved for its fogs. The fogs can become deadly, however, when mixed with pollutants, and especially compounds of sulfur, producing smoke-filled and acidic fogs. The word "smog" was derived from the terms smoke and fog. London experienced severe smogs with high mortality rates in the winters of 1873, 1880, 1882, and 1892, but then there was a gap of over forty-years in occurrences of such deadly fogs, probably the result of a reduction in domestic coal consumption after the introduction of gas heating and lighting and later electricity. The deadly fogs reoccurred, however, in the late 1940s and 1950s, most likely a combination of the pollution from coal burning for heating, which was common in the years after the war, and automobile emissions. There were three particularly serious smog incidents in 1948, 1952, and 1956. The fog of 1948 was linked to over 300 deaths, that in 1956 to over 1000. The December 1952 "Great Smog of London" was by far the worst. Sulfur dioxide and particulate levels were 10–15 times the level they had been in the winter of 1951, which had been fog-free. Sulfur dioxide, coming into contact with the water droplets in the fog, produced sulfurous acid, which is highly irritating to the respiratory and gastrointestinal tracts. Approximately 4,000 people died of respiratory ailments during or just after the smog event. Another 8,000 deaths occurred in the next ten weeks.[5] During the four-days of December 5–9 the city was covered with such a dense layer of smog that all transportation was stopped except the subways.

The London smog incidents were the crises that it took to force the British Parliament into action. In 1956, the Clean Air Act, the world's first national air quality legislation was formulated. It was revised in 1968, introducing smokeless zones in urban areas and promoting a tall chimney policy. The British example was followed by other states, where concern with pollution was also on the rise. The United States (U.S.) Congress introduced the Clean Air Act in 1963, strengthening this legislation with amendments in 1965, 1967, 1970, 1977, and 1990. The German Diet introduced a comprehensive Environment Program that addressed air, water, soil, and noise pollution in 1971.

4. Actors and Arenas in European Acid Rain Politics

The politics of acid rain in Europe are complicated by the large number of countries that are involved. At one level, acid rain politics in Europe have been a matter of inter-state negotiation among a large number of European states, Canada, the United States, and the Russian Federation. Much of the legislation governing acid rain in Europe has been formulated through the United Nations Economic Commission for Europe. This means that at times more than 40 countries have been involved in negotiations, even though the states of Europe have been the primary drivers of the negotiations. Each state has come into these negotiations with its own domestic political, economic, and environmental realities. Some states entered the negotiations knowing that, as large producers of emissions from fossil fuels, they were viewed the primary culprits. Others have come into the negotiations viewing themselves primarily victims because prevailing wind patterns have meant that they have received the pollution caused by others. Yet others realized that they were both emitters and victims.

It should be remembered too that there are great differences in the economic characteristics of the states of Europe with some being very wealthy but without much heavy industry or only moderate reliance on heavily polluting fossil fuels (e.g., Scandinavia, Switzerland, Austria, Luxembourg, France), others being wealthy but with large populations and relying heavily on fossil fuels (Great Britain, Germany), others still in developing stages and dependent largely on coal for their industry (many of the Central and Eastern European states, such as Poland, Hungary and the Czech Republic), and yet others that are developing economies but without as much large industry (much of southern Europe). There also are major differences among the states of Europe in terms of their ecological vulnerability to acid rain.

Acid rain as an issue has also been taken up at different political levels. As noted above, the United Nations Economic Commission for Europe has been a central arena for inter-state negotiations. Thus, within this arena, the states involved have included not only Western European states, but the entire European continent (including Russia), Canada and the United States. The acid rain problem has also been taken up by the European Union (EU) as a transboundary matter affecting its member states, and thus, requiring the involvement of the supranational institutions of the EU.[6] Over the course of the thirty plus years that acid rain has been recognized as a political matter within Europe, the size of the EU has grown substantially. Thus, in 1972 at the time of the UNCHE, the European Economic Community, the predecessor to the European Union, included only 6 states. The United Kingdom (UK), Ireland, and Denmark were granted membership the following year. Greece joined the Community in 1981; Spain and Portugal in 1986; Austria, Finland and Sweden in 1995, and; the Czech Republic, Poland, Hungary, Slovakia, Latvia, Lithuania, Estonia, Malta, Cyprus, and Slovenia in 2004. With each wave of accession, the politics of the EU has changed significantly. Thus, for example, states such as Greece, Portugal and Spain, which were less developed than their northern neighbors, were reluctant to mandate emissions reductions, which they felt would hamper their economic development. In contrast, when Sweden joined in 1995, it used its new membership status to push for more aggressive and comprehensive policies for acid rain prevention.

Finally, over the thirty years of negotiations on acid rain, the positions of states have shifted as new scientific evidence has shown that some states that did not consider themselves to be victims were suffering from considerable acid rain damage domestically. The UK, for example, did not initially consider itself as a victim until it was discovered that Scotland and Wales were suffering from acid rain. Changes in energy policies and economic structures have also changed the calculations that states have used when establishing their positions in the negotiations.[7] Thus, as the UK shifted its energy dependence from coal, increasingly to oil and natural gas, its cost-benefit calculations on action changed.

4.1. Laggard and Lead States

For years, Scandinavia, and especially Sweden and Norway, as downwind victims of continental pollution, were the lone European voices calling for political measures to respond to acid rain. Their demands for remedial action were met with the resistance of the more industrialized and polluting states of Europe—the UK, West Germany, East Germany, Poland, the former Czechoslovakia, and Hungary—that were the primary sources of emissions. It was only when discoveries were made that acid rain was more wide-spread and damaging than originally thought and that upwind states were also suffering from acid rain that attitudes towards environmental cooperation on transboundary acid rain changed in a more positive direction on the continent. The UK was slower than Germany to convert on the issue and this affected the nature of early European responses to acid rain, the effectiveness of those responses, and the speed with which international agreements entered into force.

As will be discussed in more detail below, changes in scientific understanding of the causes and consequences of acid rain, the extent to which various areas were suffering from acidic deposition, and what the best policy measures would be, greatly influenced political debate. Scientific findings alone, however, cannot explain the position of European states on acid rain mitigation. Also important were domestic political factors, such as the emergence of Germany's Green Party, Margaret Thatcher's concerted effort to weaken the power of the coal lobby in Great Britain, and Leonid Brezhnev's interest in East-West détente. The coalitions that formed for and against European-wide strategies for acid rain abatement shifted with time, influencing the shape and timing of policy responses.

Table 7.1 shows that there has been considerable variation among European states in the extent to which they have embraced cooperative approaches to acid rain abatement. While a large number of states have signed the CLRTAP, many have failed to sign or ratify its many protocols establishing specific emission reduction goals. It should be remembered that while signing an international agreement is a signal of a state's intent to comply with its contents, it only becomes law within a state once that state's parliament has ratified the agreement. Thus, states that have signed but not ratified an agreement are not bound by its obligations. It should also be remembered that a state can become a party to an agreement through parliamentary ratification without first signing an agreement. Thus, for example, the new members of the European Union were required to transform into national law all existing European Union laws and international agreements. This has meant that many of the new member states have joined international agreements negotiated by other member states as a result of their desire to join the EU.

4.2. Industry

Generally, European industry has been opposed to stringent measures to reduce emissions that contribute to acid rain arguing that it hurts their competitiveness. This has been particularly true of the fossil fuel industry (and especially, coal) and the automobile industry. These industries have lobbied decision-makers and succeeded in weakening the stringency of the regulations adopted at the international level despite the best efforts of environmental advocates and pollution victims. Especially vocal opponents of regulations were the Central Electricity Board in the UK into the late 1980s and the German coal industry into the early to mid-1980s.

Over time, however, the power of the coal industry in Europe has waned, reducing somewhat the opposition of the fossil fuel industry to acid rain abatement measures. This has occurred both because of changes in European energy policies and programs as European countries have diversified their energy sources and because the relatively high prices of energy in Europe have pushed industries to improve their energy efficiency. Together, moreover, these developments have helped to make Europe a leader in technological innovation related to energy efficiency improvements, emissions reduction, and emissions capture. This means that many European industries now see being on the technological cutting edge a benefit as they can lead world markets in energy efficient and environmentally-friendly technologies. Thus, while they often still lobby against stringent emission reductions, there are also numerous industries that are more supportive of

Table 7.1. Participation in the CLRTAP and its protocols (S = signed; R = ratified; * = member of the European Union as of 2005; **scheduled to join EU in 2007) (Data as of September 2005)[8]

	1979 CLRTAP	1984 EMEP	1985 Helsinki (SO_x)	1988 Sofia (NO_x)	1991 Geneva (VOCs)	1994 Oslo (SO_x)	1998 Aarhus (Heavy Metals)	1998 Aarhus (POPs)	1999 Gothenburg (SO_x, NO_x NH_3, VOCs)
Date of Entry Into Force	1983	1988	1987	1991	1997	1998	2003	2003	2005
Armenia	R						S	S	S
Austria*	SR	R	SR	SR	SR	SR	SR	SR	S
Azerbaijan	R								
Belarus	SR	SR	SR	SR					
Belgium*	SR	SR	SR	SR	SR	SR	SR	S	S
Bosnia and Herzegovina	R	R							
Bulgaria**	SR	SR	SR	SR	SR	SR	SR	SR	SR
Canada	SR	SR	SR	SR	S	SR	SR	SR	S
Croatia	R	R				SR	S	S	S
Cyprus*	R	R		R			SR	SR	
Czech Republic*	R	R	R	R	R	SR	SR	SR	SR
Denmark*	SR	SR	SR	SR	SR	SR	SR	SR	SR
Estonia*	R	R	R	R	R			R	
Finland*	SR	SR	SR	SR	SR	SR	SR	SR	SR
France*	SR	SR	SR	SR	SR	SR	SR	SR	S
Georgia	R								
Germany*	SR	SR	SR	SR	SR	SR	SR	SR	SR

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Table 7.1. (cont.)

	1979 CLRTAP	1984 EMEP	1985 Helsinki (SO_x)	1988 Sofia (NO_x)	1991 Geneva (VOCs)	1994 Oslo (SO_x)	1998 Aarhus (Heavy Metals)	1998 Aarhus (POPs)	1999 Gothenburg (SO_x, NO_x NH_3, VOCs)
Greece*	SR	R		SR	S	SR	S	S	S
Holy See	S								
Hungary*	SR	SR	SR	SR	SR	SR	SR	SR	S
Iceland	SR						S	SR	
Ireland*	SR	SR		SR		SR	S	S	S
Italy*	SR	SR	SR	SR	SR	SR	S	S	S
Kazakhstan	R								
Kyrgyzstan	R								
Latvia*	R	R					SR	SR	SR
Liechtenstein	SR	R	SR	SR	SR	SR	SR	SR	S
Lithuania*	R	R					SR	S	R
Luxembourg*	SR	SR	SR	SR	SR	SR	SR	SR	SR
Malta*	R	R							
Monaco	R	R			R	R	R		
Netherlands*	SR	SR	SR	SR	SR	SR	SR	SR	SR
Norway	SR	SR	SR	SR	SR	SR	SR	SR	SR
Poland*	SR	R		S		S	S	S	S
San Marino	S								
Serbia and Montenegro	R	R							

table continues on next page

Table 7.1. (cont.)

	1979 CLRTAP	1984 EMEP	1985 Helsinki (SO_x)	1988 Sofia (NO_x)	1991 Geneva (VOCs)	1994 Oslo (SO_x)	1998 Aarhus (Heavy Metals)	1998 Aarhus (POPs)	1999 Gothenburg (SO_x, NO_x NH_3, VOCs)
Portugal*	SR	R			S		S	S	SR
Rep. of Moldova	R						SR	SR	S
Romania**	SR	R					SR	SR	SR
Russian Federation	SR	SR	SR	SR		S			
Slovakia*	R	R	R	R	R	SR	SR	SR	SR
Slovenia*	R	R				SR	SR	S	SR
Spain*	SR	R		SR	SR	SR	S	S	SR
Sweden*	SR	SR	SR	SR	SR	SR	SR	SR	SR
Switzerland	SR	SR	SR	SR	SR	SR	SR	SR	S
The Former Yugoslav Rep. of Macedonia	R								
Turkey	SR	SR							
Ukraine	SR	SR	SR		S	S	S	S	
United Kingdom*	SR	SR		SR	SR	SR	SR	SR	S
United States	SR	SR		SR	S		SR	S	SR
European Community	SR	SR		R	S	SR	SR	SR	R
TOTAL	S=32 R=49	S=22 R=41	S=19 R=22	S=25 R29	S=23 R+21	S=28 R=26	S=36 R=27	S=36 R=24	S=31 R=18

regulatory initiatives. The fast growing renewable energy industry in Europe is also a supporter of environmental regulations.

One of the biggest problem industries today remains the automobile industry. While the industry has made strides in improving energy efficiency and reducing harmful emissions, there is much more that could still be done to reduce harmful emissions. Moreover, the continuously growing number of cars on the road has cancelled out many of the positive benefits that have been achieved to date. Another major problem industry that has until recently gone largely unnoticed is the marine shipping industry. With Europe heavily reliant on marine transport of goods, the heretofore largely unregulated emissions of marine ships are becoming an increasingly important area of concern.

Industries facing domestic regulations have tended to favor regulatory change at the European level as well. Thus, German industry faced by the prospects of strict domestic regulations on large emitters of SO_2 pushed for the introduction of similar regulations at the European level. Similarly, German automobile manufacturers favored the introduction of catalytic converters to control NO_x emissions and pushed for a similar approach within the EC.

4.3. Expert Communities

The scientific community was pivotal in bringing international attention to transboundary acid rain and in deepening understanding of the ecological impacts of acid deposition. It was the research of individual and teams of scientists that sounded the warning call that emissions from fossil fuel burning in Europe were causing damage to Scandinavia's sensitive water systems, Germany's forests, and Greece's archaeological wonders.

Expert communities of scientists, economists, and ecologists also helped to transform acid rain policy making in Europe. With the development of new theories about environmental protection (such as the concept of critical loads discussed more fully below), they succeeded in convincing policymakers in national capitols and in Brussels to shift from the traditional use of flat emission reduction targets, to developing differentiated national policy goals focused on reducing harm to sensitive ecological systems. They also developed elaborate systems to monitor emissions and ecological impacts and to track the long-distance movement of emissions.

4.4. Non-Governmental Organizations

Non-governmental organizations (NGOs) played an important role in raising public awareness of acid rain in Europe and in lobbying governments to take policy action, especially in the early and mid-1980s. Some of this was done through attention grabbing protest activities. In 1984, for example, Greenpeace launched coordinated acid rain campaigns across Europe. On April 2 Greenpeace teams simultaneously unfurled banners from the top of smokestacks, each with a single letter on them. Photographs of all the climbers with their banners hanging from the top of factory smokestacks in Belgium, West Germany, Austria, the UK, Denmark, the Netherlands, France and Czechoslovakia showed that together they had written the word, "STOP" twice over.[9]

Other NGOs focused attention on awareness raising through information dissemination. The Swedish NGO Secretariat on Acid Rain, a coalition of five Swedish NGOs, for instance, began publishing a regular newsletter, entitled *Acid News,* and developing a web page that provides information on policy initiatives, events, and terminology.[10]

NGOs have worked to lobby for legislative change as well. The Swedish NGO Secretariat on Acid Rain, the European Environment Bureau, the European Federation for Transport and Environment, and Seas at Risk jointly produced a recommendation that was presented to the European Commission regarding the Commission's plans to introduce measures to reduce the sulfur content of marine fuels, noting that sulfur dioxide emissions from these fuels by 2010 were likely to reach levels equivalent to three-quarters of all EU land based sources. The group of NGOs in their recommendation urged more stringent measures than those that were being floated by the Commission.[11] On the whole, NGO attention to acid rain, however, subsided after the formation of the CLRTAP and its first few protocols (Helsinki and Sofia). By the late 1980s, their attention had shifted in great part to other atmospheric pollution problems, including stratospheric ozone depletion and global climate change. As a result, policy making on acid rain has been increasingly left to specialized expert communities.

4.5. The European Commission, the European Council of Environmental Ministers, the European Parliament and the European Environment Agency

Since the late 1980s, acid rain politics has been a matter taken up increasingly aggressively by the European Communities, the pillar of the European Union that addresses economic and environmental cooperation. The European Council of Environmental Ministers and the European Commission have come to play increasingly important roles in shaping acid rain policy measures in Europe. The European Council of Environmental Ministers brings together the environmental ministers of the member states to set the policy agenda on environmental matters for the European Union. The European Commission, which is located in Brussels, is tasked with drafting legislation based on Council recommendations and implementating those regulations. Within the European Commission, the Directorate-General Environment, somewhat akin to an environmental ministry, has the primary role of drafting and enforcing environmental legislation. In recent years, the European Commission has sought to alter how acid rain is treated as a problem within Europe by focusing less on acid rain as a separate problem and more on acid rain as part of the larger air quality issues affecting the Community.

While the European Commission is the most important institution pushing forward changes to existing European laws addressing emissions that contribute to acid rain, its actions are watched over closely by the European Parliament. The European Parliament, the only elected body of the European Union, has a large percentage of pro-environment parliamentarians. The Parliament is the greenest of all EU institutions but has traditionally also been the weakest among them. With new powers they have achieved as a result of changes to the treaties that govern the workings of European institutions, however, they have started to push the European Commission to strengthen various draft environmental regulations. For example, in 1999 the European Parliament voted to strengthen two draft laws dealing with air pollution from large power plants and waste incineration facilities that the Commission presented to them. Due to domestic opposition within Germany and Great Britain to regulation that could economically harm coal-fired power plants, the Commission's original proposal only addressed plants licensed after 2000. The Parliament demanded that the regulations cover all existing plants, new and old.[12]

In addition, a separate European Environment Agency (EEA), located in Copenhagen, that includes all of the members of the EU plus several other countries that cooperate closely with the EU economically, such as Norway and Switzerland, has the role of information gathering and dissemination related to acid rain and other European environmental problems. Many of the reports assessing the progress of Europe in meeting its acid rain goals have been conducted by the EEA.

Over time, acid rain policy making has gone from being highly politicized and contentious, to being more widely accepted as a problem affecting wide swaths of Europe. Efforts to address the sources of acid rain have at the same time become increasingly specialized, focusing on, for example, emissions from marine ships and levels of sulfur in fuels, and more encompassing, linking acid rain policy to policies to address related air pollution issues.

5. Scandinavian Sounds the Warning Bells on Acid Rain

By the end of the 1960s, awareness was growing throughout Europe of the health and environmental problems that could be associated with not only smoke, dust, and soot, but other air pollutants, including sulfur dioxide, nitrogen oxides, carbon monoxide, and lead. The idea that air pollution was a matter that required national government intervention was slowly gaining acceptance across Western Europe.

There was also growing appreciation that pollution could be a matter of international scale and scope although acid rain was not yet a widely accepted idea and scientific understanding of transboundary pollution issues was still in its infancy. As an indication of the level of awareness of this environmental matter, there was no entry for acid rain in the 1968 Encyclopedia Britannica. The main entry dealing with pollution, "Pollution, Environmental", does mention that "polluted air or water does not respect city, state or provincial, national, or even continental boundaries" but only makes reference to the Ruhr River regional pollution abatement plan, as a specific example of a transboundary pollution issue.[13] There is no mention of acid rain in this entry either.

Svante Odén, a Swedish scientist, was to help bring international attention to the phenomenon of transboundary acid deposition in the late 1960s with his research that linked the acidification of Swedish lakes to pollution emitted abroad. As a relatively sparsely populated region with low levels of industrialization, domestic emissions could not explain the damage that Odén found occurring in southern Sweden's water bodies. Thus, he hypothesized that the source of the pollution was upwind – that is from Great Britain and the European continent.[14] His theory gave birth to the modern acid rain movement.

Growing national concern that acid rain was killing off plant and aquatic life in Scandinavia's lakes and streams was a critical reason why Sweden initiated efforts to organize the first UNCHE and why Sweden became a pioneer in environmental reforms, opening the world's first Environment Agency in 1967. At the UNCHE, Swedish scientist Bert Bolin presented a case study of the effects of acid rain on Sweden.

Although concern with air pollution domestically had led to legislative changes in both the UK (1956 and 1968) and Germany (1971), neither country showed much concern with Scandinavia's plight. Both countries resisted Swedish and Norwegian calls that they restrict their harmful emissions, calling for further research into the matter. Neither Germany's coal and auto industries nor the UK's Central Electricity Board and British Coal had much incentive to restrict their sulfur dioxide emissions as there was little national concern with acid rain. When Sweden first raised its concerns about acid rain at the Organization for Economic Cooperation and Development (OECD) Air Committee meeting held in Paris in 1969, it met with no response although three years later the OECD did establish a Cooperative Technical Program to Measure Long-Range Transport of Air Pollutants. Scientific research conducted in this and other programs over the next decade helped to confirm Odén's hypothesis.[15]

6. An Odd Couple: The Soviet Union Joins Scandinavia in Calling for International Action on Acid Rain

In the mid-1970s the Soviet Union joined Sweden and Norway in calling for action on transboundary air pollution. This unusual coalition brought together downwind victims of air pollution. The Soviet Union was a net recipient of air pollutants emanating from its neighbors to the west, and especially Poland, Czechoslovakia, Germany, Hungary and Finland.[16] Although the Soviet leadership was not very concerned with acid rain as an environmental matter, they saw it as a useful issue upon which to promote their East-West détente initiative. Thus, at the initiatives of Scandinavia and the Soviet Union, efforts began under the auspices of the United Nations Economic Commission for Europe to establish a Convention on Long-Range Transboundary Air Pollution. Although Scandinavia was pushing for an agreement that would have required sulfur dioxide emissions reductions of at least 30%, the UK and Germany remained opposed to this idea.[17] By this time, however, scientific evidence was accumulating that air pollutants could indeed travel thousands of kilometers before deposition occurred.

After long negotiations, a least common denominator compromise was reached. The CLRTAP called upon states to reduce their transboundary air pollution as much as was economically feasible, report on their efforts to control emissions, and cooperate on research and monitoring of acid rain. All major western European states, the former Soviet Union and many of its satellite states, the EC, Canada, and the United States signed the convention in 1979. The CLRTAP came into force in 1983 and has been extended by eight specific protocols. As of September 2005, it has 49 parties. While little more than a least common denominator agreement and a disappointment to Scandinavia, the CLRTAP was significant as the first internationally legally binding agreement to deal with air pollution at a broad, regional level.[18]

7. The Greening of Germany: A Turning Point for European Cooperation on Acid Rain Abatement

In 1982, eager to convince its neighbors of the need to take more significant steps to curtail emissions, Sweden hosted the Stockholm Conference on Acidification for the Environment. International experts from around the world convened in Stockholm to debate the impacts of acid rain. While it was generally recognized that acid rain could damage aquatic systems, its impacts on other ecological systems were not well understood. At the conference, German scientist, Bernhard Ulrich presented his research that showed that forests across Europe were suffering as a result of acid rain.[19] His research played a central role in widening the debate within Europe on the consequences of acid rain.

Ulrich's research on *Waldsterben* (forest death) had already been picked up a year earlier (in November 1981) by the widely-read *Der Spiegel,* and it immediately struck a cord of concern with the German public.[20] It is often said that Germans have a special love for forests.[21] That Germany's forests, the backdrop for many of the fairy tales collected and published by the Brothers Grimm, were being threatened by acid rain helped to build public support for emissions reductions. *Waldsterben* resonated with a public that was becoming increasingly environmentally-sensitized and concerned with what modernization and industrialization were doing to their living environment.

Ulrich's research on *Waldsterben* coincided with growing support for the young German Green Party. The Green Party gained sufficient votes in the 1982 federal elections to overcome the 5% hurdle that had been established precisely to discourage the formation of small, single issue parties. The other major parties were sufficiently worried by the impact that the Greens had on their own share of the vote that they began to green their party platforms in the hopes of recapturing voters that had shifted their allegiance. The greening of the political landscape led to major changes in German environmental laws and among the first issues to be tackled were air pollution and acid rain. In 1983, the German Diet passed the Large Combustion Plant Ordinance (*Grossfeuerungsanlagen-Verordnung* (GFAVo)), which established stringent sulfur dioxide emission controls on large polluters.

This shift in German domestic politics was important at the European level as well since Germany, the largest economy in Europe, came to share Scandinavia's concern with acid rain. This enlarged coalition made possible the establishment of several important protocols to the CLRTAP as is discussed below. It is also noteworthy that German industry was eager to make sure that its competitors in Europe would face the same environmental regulations it was confronting so that they would be working on a level playing field. German political leaders were persuaded to push within the EU for the establishment of a Large Combustion Plant Directive, modeled on the German GFAVo. The Directive was adopted in 1988 (and replaced by a revised, more stringent Directive in 2001).

Germany's change of heart on acid rain meant that it was Europe that took the lead internationally in responding to regional acid rain problems. At the same time that Europe was formulating Protocols to the CLRTAP aimed at cutting sulfur dioxide emissions, the United States was resisting Canadian calls to address transboundary acid rain in North America. It was not until the passage of the 1990 amendments to the Clean Air Act, which introduced an emissions trading system aimed at reducing sulfur dioxide emissions that the United States responded to Canadian pleas for action.

Also significant was the trans-Atlantic coalition that began to form between Canada and the Scandinavian countries. Canada was a victim of acid rain, some of it of its own making, but also considerable portions of which came from the United States. Much like Scandinavia was having troubles convincing the UK and the European continent to take decisive measures to reduce the pollutants causing acid rain, Canada could do little more than convince the United States to agree to conduct research on the causes and consequences of acid rain.

8. The Greening of the United Kingdom

Much like the rise of the Green Party in Germany led to major changes in Germany's domestic political situation and resulted in Germany becoming a more proactive player in the international acid rain negotiations, the UK became a less obstructive and more cooperative player in the international negotiations in the late 1980s. The year 1988 was something of a turning point in the UK's position on international atmospheric pollution matters, ranging from acid rain to global climate change. Although the UK had been a pioneer in early domestic air pollution legislation, it was a foot dragger in the international negotiations to address acid rain. The UK's opposition to meaningful emissions reductions, and its failure to agree to the Helsinki Protocol discussed below, led to strong criticism from Scandinavia.

In 1988, however, Thatcher began to be more amenable to cooperating with Europe on acid rain abatement, transboundary air pollution control, and climate change mitigation. This may have had to do in part with the success of her efforts to weaken the coal industry through the promotion of the use of natural gas as well as the privatization of the Central Electricity

Board. It may also have had something to do with changing environmental ideas in the UK after the discovery of the ozone hole by two British scientists in 1985, the Chernobyl nuclear accident in 1986, and the sense of an impending climate crisis after record hot summers in Europe in the late 1980s. After this point, the UK began to exhibit a more cooperative stance within the CLRTAP process. This was critical as now the two largest and most polluting economies in Western Europe—the UK and Germany—were willing to work with Scandinavia on addressing Europe's acid rain problems.

9. Negotiating the First Generation of Protocols to the Convention on Long-Range Transboundary Air Pollution

In 1984, the first protocol to the CLRTAP was reached.[22] It was a relatively non-controversial agreement as it simply dealt with financing of research, monitoring and evaluation. Technically called the Protocol on Long-term Financing of the Co-operative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe, or more simply, EMEP, it has 41 parties to it, including all major European states, the EC, the United States and Canada. It finally received a sufficient number of ratifications to enter into force in 1988. EMEP has provided critical financial resources for European-wide monitoring of acid rain. It has also been central to the development of a general scientific understanding of the nature and extent of acid rain in Europe and mapping of pollution flows.

Also in 1984, an informal 30% club was established by 10 countries that agreed to reduce their sulfur dioxide emissions by that amount as Sweden had hoped to achieve already in the initial CLRTAP. The 30% Club was initiated by the Scandinavian countries. Germany was persuaded to join after the 1982 federal elections. In Ottawa, Canada, Switzerland, Austria, the Netherlands and France agreed to the idea as well. The Soviet Union also showed support for the initiative but was not present at the meeting. Noticeably absent from this agreement were the UK and the United States. The 30% Club became the basis for the second protocol to the CLRTAP, negotiated in Helsinki in 1985. The Protocol on Reduction of Sulfur Emissions or their Transboundary Fluxes by at least 30%, required states to reduce their SO_x emissions or transboundary fluxes by 30% of 1980 levels by 1993.[23] At the negotiations, Germany had also argued for the extension of the protocol to include NO_x emissions, but this was opposed by the UK. In the end, 22 countries (but not the United States, the UK, or Poland) ratified the protocol and it entered into force in 1987. Marc Levy argues that while some of these 22 signatories agreed to the protocol based on their concerns that their own environments were being adversely affected by transboundary pollution, many also signed because they calculated that the reduction requirements would require little or no domestic policy action as changes in domestic energy policies throughout Europe were already expected to lead to a reduction in sulfur oxide emissions. Furthermore, he suggests that many of the east bloc states that signed the agreement did so to one-up the United States and the UK; there is, moreover, little indication that they ever seriously intended to abide by the protocol's reduction requirements.[24] Nevertheless, the Helsinki Protocol, as it is commonly referred to, was a critical step forward in Europe's efforts to address a common pool resource problem. The protocol set European states in motion towards a codification of emission reduction goals that were subsequently to be extended to other pollutants through additional protocols.

Soon after the Helsinki Protocol was completed, negotiations began on controlling other acid rain precursors. The 1988 Sofia Protocol extended controls to nitrogen oxides and targeted nitrogen compounds, including ammonia and volatile organic compounds.[25] Due to political resistance, this protocol simply required a freeze in emission levels of nitrogen oxides at 1987 levels by 1994. It did, however, also call upon states to apply national emissions standards to major new stationary and mobile sources and to commence negotiations on further measures that could be taken to reduce their emissions of nitrogen oxides based on a multi-pollutant, multi-effect critical loads approach. It is significant that the Sofia Protocol did not simply stipulate a common reduction goal for NO_x emissions as had been done in the Helsinki Protocol, but also introduced the critical loads concept, shifting policy making towards measures more suitable to reducing the ecological impacts of emissions. The critical loads concept essentially refers to the highest load of a pollutant or mix of pollutants that sensitive ecological systems can absorb before there are chemical changes that result in long-term damage or harm to the system.

Thus, the protocol includes both a fixed policy goal of a freeze in emissions, but also called for the development of national policies based on the critical load concept. In contrast with the 1985 Helsinki Protocol, the UK and the United States both signed the Sofia Protocol. As with the case of the Helsinki Protocol, the Sofia Protocol was something of a least common denominator response. Scandinavia's push for a NO_x reduction target failed. Instead, as with the case of SO_x before it, a group of 12 countries (Austria, Belgium, Denmark, Finland, France Italy, Liechtenstein, the Netherlands, Norway Sweden, Switzerland, and West Germany) agreed to reduce their

NO_x emissions by 1998 by 30% relative to a base year between 1980 and 1986; this became known as the Sofia Declaration.[26]

An assessment conducted by the United Nations Economic Commission for Europe found that by 1993, all 21 parties to the Helsinki Protocol had met its sulfur reduction targets, reducing their 1980 sulfur emissions by 48%. As of 1993, 18 of 25 parties to the Sofia Protocol had fulfilled its emission target, three had failed to submit data, and four had emissions that were 4–41% above their 1987 NO_x limits, largely because of increased urban traffic.[27]

Slowly, over the course of the next decade, European policy making was to move towards more equitable and effective approaches to dealing with Europe's acid rain problems, allowing for differentiated emission reduction targets that took into account different state's ecological vulnerability to acid deposition, contributions to emissions, their respective costs for taking control measures, and their expected benefits to be derived from various pollution control measures. Increasingly sophisticated critical loads maps of Europe were developed and the RAINS Model, a computerized assessment tool developed by the International Institute for Applied Systems Analysis (IIASA) was used to help negotiators determine optimum policy responses. The model integrated critical loads information, data about emissions flows, and cost data for emissions reduction measures.

10. Negotiating the Second Generation of Protocols to CLRTAP

In the 1990s, with the critical loads concept as a driving framework and evidence that acid rain problems in Europe were more serious than previously realized, several additional protocols were negotiated extending the CLRTAP to a wider range of pollutants, including volatile organic compounds (VOCs), heavy metals and persistent organic pollutants. Here two of these protocols will be briefly discussed: the 1994 Oslo Protocol requiring further SOx reductions based on national critical load assessments and the 1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone.[28] The other protocols are described in Appendix 7.1.

Despite the measures that were taken to control SO_x and NO_x emissions in the first generation of protocols, scientific evidence showed that large parts of Europe were still under threat of acid rain damage and that there were far more ecologically sensitive regions than had been first understood. Further measures would have to be taken to reduce harmful emissions. This led to the negotiations for the 1994 Oslo Protocol. The Oslo Protocol essentially extended the critical loads approach to sulfur oxides. By the end of the negotiations, Europe as a whole had agreed to reduce their combined SO_x emissions by 35% of 1985 levels by 2000 but with differentiated targets based on states' contributions to pollution and costs of abatement.[29] The Oslo Protocol proved easier to negotiate as the UK was no longer adamantly opposed to regional cooperation on acid rain as it had been throughout much of the previous decade. While there was still considerable acrimony in the negotiations regarding what measures states should be required to take, the protocol was a significant step forward in establishing national goals for reducing harmful emissions.[30]

The Gothenburg Protocol reflected the transition in Europe towards dealing not with individual pollutants but with groups of pollutants recognized as having similar ecological impacts. It set emission ceilings for four pollutants: sulfur, NO_x, VOCs, and ammonia for 2010 based on scientific assessments of pollution impacts and abatement options and costs. The Gothenburg Protocol is expected to reduce Europe's sulfur emissions by at least 63%, its NO_x emissions by 41%, its VOC emissions by 40% and ammonia emissions by 17% relative to 1990 levels. Commitments vary considerably among countries. Thus, for example, on the high end, Germany and Hungary both committed to 90% reductions in their sulfur emissions relative to 1990 levels by 2010; the Czech Republic, Slovakia, Slovenia, and the UK by figures in the 80% range; the EC as a whole by 75%; Belgium, Denmark, Ireland, Italy, Luxembourg, and the Netherlands by figures in the 70–79% range; and Austria, Bulgaria, Croatia, Finland, France, Norway, Poland, Portugal, and Spain in the 50–69% range. On the other hand, Greece only committed to holding increases in its emissions to 7% in the same time frame and Armenia to holding its emissions constant. No country committed to equally strong reductions in NO_x emissions with the highest reduction commitments coming from the Czech Republic (–61%); Germany (–60 %), and Denmark, Luxembourg, the Netherlands, Sweden, and the UK agreeing to reductions in the 50–59% range. Greece committed to stabilize its emissions. Denmark and the Netherlands, which have very intensive dairy industries, both committed to reduce their ammonia emissions by 43%. Other countries' commitments were considerably smaller and the best Spain and Portugal could commit to was a small increase in their emissions, 1% and 10%, respectively. The EC as a whole, committed to a 15% reduction. In terms of volatile organic chemicals, reduction commitments ranged from maximum pledges of 72% for Ireland, 69% for Germany and 68% for Portugal to only a stabilization commitment from Armenia and a small 4% reduction from Poland.[31]

The 1999 Gothenburg Protocol led to the establishment of new European Union legislation, Directive 2001/81/EC on National Emissions Ceilings for Certain Pollutants. The Directive set upper limits for each member state for the total emissions of SO_2, NO_x, VOCs, and ammonia, four pollutants responsible for acidification, eutrophication, and ground-level ozone pollution. Member states are allowed to determine which measures they will take to comply with the directive, but are required to report annually on their national emission inventories and projects for 2010.[32]

10.1. Beyond CLRTAP

The European Union has introduced a number of measures complementing and going beyond the CLRTAP. As a whole the measures aim to enhance overall air quality and reduce harmful deposition. In addition to the Directive for National Emission Ceilings described above, there have been several other noteworthy policy developments. Numerous new directives have been issued in an effort to reduce emissions of harmful air pollutants that contribute to acid rain. For example, there have been various directives aimed at reducing the emissions of VOCs from storage of gasoline and during its use at service stations and in solvents; reducing the sulfur content of gasoline and in certain liquefied fuels; and limiting sulfur emissions from ships.[33] In 2001, a Revised Large Combustion Plants Directive (LCPD, 200180/EC) was instituted aiming to reduce emissions of SO_2, NO_x, and dust from large combustion plants with a thermal output of over 50MW. It introduced emission limit values for these pollutants that were twice as strict as existing ones and also established new requirements for older plants licensed before 1987 that were exempted from regulation in the previous directive. Interestingly, the new directive also encourages combined generation of heat and power and promotes the use of biomass as fuel.[34] In September 2005 the European Commission proposed a clean air strategy, which calls for further measures to clean the air to reduce premature deaths from air pollution and reduce the area of forests and other ecosystems experiencing harm. The proposal focuses on fine particulates and ground-level ozone and calls for the streamlining of air quality legislation by merging existing legal instruments into a single Ambient Air Quality Directive.[35]

10.2. Assessing the Impacts of the CLRTAP and its Protocols and European Clean Air Policies

The United Nations Environment Program's Global Environment Outlook (2002) found that air quality in Europe was on the mend. Emissions of air pollutants have declined on the whole in Europe since the CLRTAP was introduced. Reductions in SO_2 emissions have led to substantial recovery of a natural pH level in water and soils in Europe although emissions are still too high for the most sensitive ecosystems. Improvements have not been universal throughout Europe, however. Thus, while there were major reductions in SO_2 levels in Germany, SO_2 emissions rose in Greece and Portugal between 1990 and 1998. Moreover, except for Germany, the UK, and many of the Central and Eastern European countries, NO_x levels have not declined. Ammonia (NH_3) levels remain high throughout Europe as well. The report concludes: "It is clear that the reductions in emissions are at least partly due to national and local measures that have been taken to achieve targets set by CLRTAP and its Protocols, and to EU Directives linked to air emissions such as the Limitation of Emissions of Certain Pollutants into the Air from Large Combustion Plants Directive (1988) and various directives on vehicle emissions, the change to unleaded petrol and higher quality diesel fuels and improved engine design. Despite this clear progress, many air pollution reduction targets have still not been met. In Western Europe, only the EU and CLRTAP targets for SO_2 were met well before the target date (the end of 2000) with less progress on NO_x, NH_3 and VOCs."[36]

An assessment conducted by the European Environment Agency (EEA) of air pollution levels between 1990 and 2000 found that NOx emission levels had declined by 27% across the 31 member countries of the EEA, SOx emission levels by 60%, and non-methane VOCs by 29%. Acidifying deposition, however, remains above critical loads throughout much of central and Northwestern Europe, eutrophying deposition remains widespread, and reductions in nitrogen deposition have been limited. The EEA assessment concluded: "The improved energy intensity and fuel intensity of the European economies, and the shifts towards lighter fuels, both in the EU-15 and to a lesser extent in the accession countries, cannot explain the observed downward trends in emissions. It must be concluded that a substantial contribution to decreased emissions in Europe is due to the successful abatement of emissions from various sources, in particular large combustion plants and passenger cars. This contribution is probably mainly an effect of both EU legislation as well as UNECE CLRTAP agreements."[37]

11. Conclusion

Europe has done much to reduce its emissions of pollutants that contribute to acid rain and European approaches to acid rain abatement and air quality control are being studied by other regions of the world where air pollution and acid rain have become major problems. Thus, for example, a RAINS-

Asia modeling system has been developed by IIASA to assess acid rain problems and mitigation options in East Asia. East Asian states have also developed an East Asian Acid Rain Monitoring Network based on early European examples. The European Union is also working to export its lessons to the region, both in an effort to help Asia reduce its contributions to global atmospheric pollution and in the interest of exporting European technologies and know-how to the region.

Enlargement of the European Union to the states of Central and Eastern Europe provides new challenges as well as opportunities for improving air quality in Europe and reducing acid rain damage. Already, there has been a substantial improvement in air quality in the new member states as a result of the collapse of their Soviet-styled economies and because of the adoption of the stringent environmental laws of the European Union. While many problems remain in Europe, including the continued heavy reliance on automobiles, the continued operation of some old and inefficient power plants, and pollution from ships and airplanes, progress in improving air quality has been substantial. Europe has many lessons it can provide to other regions on how to cooperate to improve environmental conditions.

Acronyms

CLRTAP Convention on Long-range Transboundary Air Pollution
EC European Community
EEA European Environment Agency
EMEP Co-operative Programme for Monitoring and Evaluation of the Long-range Transmissions of Air Pollutants in Europe
EU European Union
GFAVo Grossfeuerungsanlagen-Verordnung
IIASA Institute for Applied Systems Analysis
NGO Non-governmental Organization
NH_3 Ammonia
NO_x Nitrogen Oxides
OECD Organization for Economic Cooperation and Development
POPs Persistent Organic Pollutants
R Ratified
S Signed
SO_x Sulfur Oxides
SO_2 Sulfur Dioxide
UK United Kingdom
UNCHE United Nations Conference on the Human Environment
UNECE United Nations Economic Commission for Europe
U. S. United States
VOCs Volatile Organic Chemicals

Appendix 7.1. Protocols to the Convention on Long Range Transboundary Air Pollution (information as of April 2006)[38]

1984 Protocol on Long-term Financing of the Cooperative Program for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP) (41 parties; entered into force in 1988)

- Agreement on financing of monitoring programs for acid rain
- Important as a basis for developing general scientific understanding and at times consensus within Europe on the nature and extent of acid rain problem
- Supported by EC and United States. Entered into force in 1988. 41 parties

1985 Helsinki Protocol on Reduction of Sulfur Emissions or their Transboundary Fluxes by at least 30% (22 have ratified; entered into force 1987).

- Stipulated a reduction of emissions of SO_x by 30% of 1980 levels by 1993
- Opposed by UK, Poland, United States
- Entered into force in 1987

1988 Sofia Protocol concerning the Control of Nitrogen Oxides or their Transboundary Fluxes (29 have ratified; entered into force February 1991)

- Calls for freeze of nitrogen oxide emissions or their transboundary fluxes relative to 1987 (or 1978 in the U.S. case)
- Makes use of an effects-based approach for the second step of the protocol, which aims at further reductions of nitrogen compounds, including ammonia and volatile organic compounds
- Germany pushed to include NO_x emissions controls already in Helsinki Protocol but failed due to UK opposition
- Countries did agree to freeze NO_x emissions at 1987 levels by 1994
- United States, UK signed (already had national programs and technologies that would make this possible)

1991 Geneva Protocol concerning the Control of Emissions of Volatile Organic Compounds or their Transboundary Fluxes (21 have ratified; entered into force September 1997)[39]

- Specifies three options parties can take to reduce their emissions
- The dominant option being followed is a 30% reduction in emissions of volatile organic compounds (VOCs) by 1990 using a base year between 1984 and 1990 (the option chosen by Austria, Belgium, Estonia, Finland, France, Germany, the Netherlands, Portugal, Spain, Sweden and the UK with 1988 as a base year; Denmark with 1985; Liechtenstein, Switzerland, and the United States with 1984; and by the Czech Republic, Italy, Luxembourg, Monaco and Slovakia with 1990 as a base year)

1994 Oslo Protocol on Further Reduction of Sulfur Emissions (25 have ratified; entered into force August 1998)

- Based on concept of critical loads: "a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specific sensitive elements of the environment do not occur according to present knowledge"
- Led to formation of national policies based on critical load base lines
- Led to development of EU Directive establishing binding national emissions ceilings for SO_x, NO_x, and particulates to be achieved by 2010

1998 Aarhus Protocol on Heavy Metals (24 countries have ratified; entered into force December 2003)[40]

- Targets three chemicals: cadmium, lead and mercury.
- Parties must reduce their emissions for these metals below 1990 (or an alternate year between 1985 and 1990) levels
- Establishes stringent limit values for emissions from stationary sources and suggests best available techniques for these sources, including filters and scrubbers for combustion sources and mercury-free processes
- Targets other products, such as batteries, electrical components, measuring devices and fluorescent lamps

1998 Aarhus Protocol on Persistent Organic Pollutants (POPs) (22 have ratified; entered into force October 2003)[41]

- Focuses on a list of 16 substances singled out with goal of eliminating discharges, emissions and losses of these POPs
- Bans the production and use of aldrin, chlordane, chlordecone, dieldrin, endrin, hexabromobiphyenyl, mirex, and toxaphene
- Schedules the phase out of DDT, heptachlor, hexaclorobenzene and PCBs
- Severely restricts the use of DDT, HCH (including Lindane) and PCBs
- Obliges parties to reduce emissions of dioxins, furans, PAHs and HCB below their 1990 levels (or an alternative year between 1985 and 1995)

1999 Gothenburg Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (16 have ratified; entered into force May 17, 2005)

- Sets emission ceilings for 2010 for sulfur, NO_x, VOCs and ammonia.
- Ceilings were negotiated based on scientific assessments of pollution effects and abatement options
- Parties whose emissions have greater environmental or health impacts and whose emissions can most cheaply be reduced are required to make the biggest cuts
- Expectations are that this will cut Europe's sulfur emissions by 63%, NO_x emissions by 41%, VOC emissions by 40%, and ammonia emissions by 17% compared to 1990
- Sets tight limit values for specific emission sources and requires use of best available techniques

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8. Acid Rain in a Wider Europe: The Post-Communist Transition and the Future European Acid Rain Policies

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1. Introduction

Air pollution was among the most salient environmental problems in Central and Eastern Europe (CEE) in the aftermath of communism.[1] Photographs of damaged monuments, forests defoliated from acid rain, and degraded landscapes became emblematic of the environmental burden associated with centralized planning. CEE countries contributed significant amounts of transboundary acidification in Europe due to prevailing atmospheric and geographic patterns. The area bordering the Czech Republic, East Germany, and Poland—a region of high concentration of industrial enterprises and acidification—became know as the "Black Triangle" and was just one example of a regional pollution "hot spot." The period of democratic transition brought promise as well as international pressure to tackle domestic and transboundary air pollution in transition countries (Jancar-Webster 1993; Regional Environmental Center 1998; World Bank 1994).

This chapter examines how ten CEE states dealt with the problem of acidification in the 1990s, during a period of transition and political integration with the European Union (EU). All of these countries applied for EU membership in the early to mid 1990s. As a consequence, they faced strong international pressure to reduce their transboundary emissions, and to comply with EU regulations and the protocols of the Convention on the Long Range Transboundary Air Pollution (LRTAP). At the same time, these societies underwent reforms of democratization and market restructuring. In the course of the transition, concerns about unemployment and economic growth took precedence and environmental protection fell low on the political agenda. This sample of countries, thus offers a unique opportunity and almost an experimental setting, to examine how countries facing difficult

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economic circumstances have succeeded in adopting and applying international air pollution standards.

Acid rain policies in post-communist Europe present a story of qualified success that can be understood only if we examine the interplay between international and domestic politics. While significant reductions in acidifying emissions resulted from economic downturn in the late 1980s and early 1990s, these environmental gains would have been difficult to sustain later in the decade without concerted policy action. Under pressure from international institutions, and the EU in particular, most CEE countries reformed their air pollution regulations considerably. The implementation of the new regulations, in turn, depended to a large degree on domestic concern and capacity.

This chapter develops the argument as follows. The next section examines the participation of CEE states in the European acid rain regime and the role of European institutions in promoting the air pollution reforms. The chapter then looks at how domestic politics have shaped the responses to international regulations. A comparative analysis of the acid rain policies of Bulgaria, the Czech Republic, and Poland reveals that countries such as the Czech Republic that relied on domestic support and capacity to implement acid rain policies have achieved most dramatic improvements in air quality and reduction of acidification. At the same time, other countries, such as Bulgaria, have relied primarily on the emission reduction as a by-product of economic decline and restructuring and are facing a more difficult task of implementing tougher air pollution standards in the future. On the basis of this analysis, the conclusion discusses the likely influence of CEE states, most of which are now members of the EU, on the future cooperation to reduce acidification in Europe.

2. CEE Countries in the European Acid Rain Regime

Concern about acid rain has been to a large degree "imported" to CEE countries from the rest of Europe. During the 1970s, when Norway and Sweden pushed for an international agreement to control transboundary air pollution, there was relatively little public information on acidification in communist Europe and the Soviet Union. Most CEE countries were net exporters of acid pollution and to the extent that communist governments were interested in environmental protection, they sought to control mostly local problems. In 1979, Soviet leader Leonid Brezhnev supported the conclusion of the LRTAP Convention primarily as a convenient platform to advance the objective of Cold War détente between East and West. The East

European satellites followed the Soviet leadership and joined the treaty (Darst 2001; Levy 1993). All communist countries except Poland ratified the First Sulfur Protocol (1985), which required 30% reduction of 1980 levels of SO_x emissions, but did little in terms of implementation during the 1980s beyond measures that addressed local air pollution. In 1988, CEE countries became parties to the Protocol on Nitrogen Oxides (NO_x), which required stabilization of NO_x at 1987 levels by 1994.

The collapse of the communist regimes after 1989 brought hopes for more active policies and the tackling of transboundary flows of pollutants from CEE. The sharp economic downturn of the early transition period contributed to a decline in acidifying emissions. Environmental advocates and policy analysts feared, however, that the decline in acidifying emissions could be reversed as post-communist economies regain growth (McCormick 1998).

Indeed, during the negotiations of the Second Sulfur Protocol (1994), the transition countries were pressed to accept commitments for reductions of their emissions ranging from 33% to 50% of 1980 levels by 2000, and yet further tightening of these ceilings by 2005 and 2010 (United Nations Economic Commission for Europe 1994). The differentiated country ceilings under the Protocol were negotiated on the principles of minimizing ecological damages to the most-sensitive ecosystems and cost-effectiveness. Relatively high reduction targets were assigned to CEE countries partly because these countries sent acidifying emissions northward affecting sensitive ecosystems. It was also estimated that the relative cost of sulfur emission reductions would be lower in CEE, where few controls had been implemented during the 1980s (Levy 1995). The Protocol also included emission standards for large utilities based on the application of Best Available Technology.

The standards of the Second Sulfur Protocol thus sought to assure that the environmental gains from the early transition period would not be lost once growth resumed. The transition countries bargained hard for an agreement on financial transfers to help with the investment costs associated with requirements of the Second Sulfur Protocol, but got little, if anything, in terms of targeted assistance (Levy 1995). In 1999, all European countries except Russia agreed to a new Multi-Pollutant, Multi-Effect Protocol under the LRTAP Convention. The Protocol tackles simultaneously acidification, eutrofication, and ground level ozone, recognizing the complexity and linkages between precursor pollutants and environmental problems. It tightened further the 2010 emission ceilings for CEE countries (United Nations Economic Commission for Europe 1999, see also Box 8.1 and Table 8.1). The high emissions reduction targets combined with limited domestic capacity and little financial assistance raised concerns about compliance in CEE and potential unraveling of the effectiveness of LRTAP protocols (McCormick 1998).

By the mid 1990s, however, the EU was beginning to exert an increasingly strong influence over environmental reforms in CEE (Andonova 2004). Lead by their ultimate foreign policy objective of unification with Western Europe, CEE states declared early in the 1990s their desire to join the EU. The 1993 Copenhagen Council of EU Ministers set three general membership criteria: democratic institutions, functioning market economy, and the adoption of the EU body of legislation. Thus, as a condition for membership, CEE countries had to adopt all environmental regulations of the EU including the regulations pertaining to acid rain.

The EU acid rain standards had developed in parallel to the LRTAP regime. The 1988 Large Combustion Plant Directive of the EU set standards for large utilities very similar to those later adopted by the 1994 Second Sulfur Protocol. In 1997, the European Commission, with support of key EU member states, adopted a Community Strategy to Combat Acidification. The Strategy called for national emission ceilings for EU member states; an amendment of the Large Combustion Plant Directive to tighten emission standards for large utilities; and limits of the sulfur content of liquid fuels. A key component of the EU Acidification Strategy was also the ratification of and compliance with the Second Sulfur Protocol by CEE accession countries as a precondition for achieving the EU acidification reduction targets (Commission of the European Communities 1997).

Box 8.1. LRTAP protocols on acid rain emissions

- The 1999 Protocol to Abate Acidification, Eutrophication and Ground-level Ozone (Multi-Pollutant Protocol)
- The 1994 Protocol on Further Reduction of Sulphur Emissions (Second Sulfur Protocol)
- The 1988 Protocol concerning the Control of Nitrogen Oxides or their Transboundary Fluxes
- The 1985 Protocol on the Reduction of Sulphur Emissions or their Transboundary Fluxes by at least 30 per cent
- The 1984 Protocol on Long-term Financing of the Cooperative Programme for Monitoring and Evaluation of the Long-range Transmission of Air Pollutants in Europe (EMEP)

Source: Convention on Long-Range Transboundary Air Pollution, accessed via http://www.unece.org/env/lrtap/status/lrtap_s.htm; April 2006.

Table 8.1. LRTAP commitments and national emissions (Gg/year) of CEE countries

Party	SO_2 Ceilings 2000	SO_2 Ceilings 2010	SO_2 Emissions 1990	SO_2 Emissions 1999	SO_2 Emissions 2002	NO_x Ceilings 2010	NO_x Emissions 1990	NO_x Emissions 1999	NO_x Emissions 2002
Bulgaria	1375	856	2008	943	940	266	361	202	197
Czech R	1128	283	1876	268	237	286	742	313	318
Hungary	898	550	1010	590	359	198	238	201	180
Latvia	NA	107	119	29	12	84	93	40	37
Lithuania	NA	145	222	70	43	110	158	54	51
Poland	2583	1397	3210	1719	1564	879	1280	951	796
Romania	NA	918	1311	912	912	437	546	319	349
Slovakia	337	110	543	171	102	130	225	118	105
Slovenia	130	27	194	104	104	45	62	58	60

Source: Second Sulfur Protocol for SO_2 2000 ceilings, Gothenberg Multi-Pollutant Protocol for SO_2 and NO_x 2010 ceilings; Convention on Long-Range Transboundary Air Pollution 2005 for emissions data (accessed via http://webdab.emep.int/). Estonia has no commitments under the Second Sulfur Protocol or the Gothenberg Multi-Pollutant Protocol and is not included in the table.

The EU Acidification Strategy started a period of closer coordination between LRTAP and EU acid rain policies. In 1999, shortly after the adoption of the LRTAP Multi-Pollutant Protocol, the EU adopted a similar multi-pollutant directive, setting national ceilings for the then 15 EU member states for sulfur dioxide, nitrogen oxides, particulate matter, and lead. In 2001, the EU amended the Large Combustion Plant Directive raising air emission standards for large utilities. Most CEE countries requested transition periods for compliance with the 2001 Amendment of the EU Large Combustion Plant Directive, arguing that due to its acceptance shortly before their accession, they needed time to prepare for its implementation. Boxes 8.1 and 8.2 summarize in chronological order the major LRTAP and EU regulations on acidifying emissions.

EU and LRTAP institutions have been instrumental in keeping acid rain on the policy agenda of CEE states in the period of difficult economic transition. This international influence materialized through a number of mechanisms. Most importantly, in both EU and LRTAP institutions, the linkages between acid rain cooperation and broader foreign policy objectives for East-West cooperation have played an important role. The EU conditionality and pressure was particularly important in pushing acid rain policies forward in post-communist Europe after political attention shifted to more urgent economic problems.

A second mechanism of international influence embedded in these regimes was growing East-West scientific and technical collaboration, which in the 1980s and early 1990s armed environmental advocates and dissident groups with information on the problem and the extent of health and

Box 8.2. Selected EU directives on acidifying emissions

- Directive 2001/80/EC of the European Parliament and of the Council on the limitation of emissions of certain pollutants into the air from large combustion plants
- Council Directive 1999/30/EC relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air
- Council Directive 93/12/EEC of 22 March 1993 relating to the sulphur content of gas oil
- Council Directive 88/609/EEC on the limitation of emissions of certain pollutants into the air from large combustion plants

Source: European Commission accessed via http://europa.eu.int/scadplus/leg/en/s15004.htm; April 2006.

ecological damage. In some part thanks to the evolution and the consensual nature of acid rain science, the issue of air pollution and acid rain emerged at the top of the environmental policy agenda of transition countries in the early 1990s (Regional Environmental Center (REC) 1998). EU technical assistance during the late 1990s, in turn, increased the capacity of domestic institutions and the ministries of the environment to monitor the implementation of acid rain regulations. The exchange of information among transnational scientific and bureaucratic networks as part of these regimes, has also facilitated learning of the policy options to combat acidification ranging from fuel substitutions, to economic instruments, to strict technology standards of the type required by the Large Combustion Plant Directive and the LRTAP protocols (Andonova 2006).

Finally, international assistance has played some role, albeit not a central role, in promoting compliance with the international regimes. Significant international assistance was allocated to the Black Triangle Region, where acidifying pollution was most visible and concentrated. Development banks, including the European Bank for Reconstruction and Development and the World Bank, were also eager to offer technical assistance and loans for infrastructure investments associated with air pollution abatement. However, the larger share of the financing for acidification abatement was generated through domestic sources such as the special environmental funds that were capitalized through environmental fees, taxes and in some cases grants, or through innovative policies that minimize the cost of abatement (Andonova 2004; Klarer and Moldan 1997; Regional Environmental Center 2001a).

While the mechanisms of international influence on the acid rain policies of CEE countries were broadly similar, these states responded with different compliance strategies depending on a range of domestic political and economic factors. In order to understand the sources of success and failure of acid rain policies in the region, we need to look also at the interplay between international and domestic policy incentives.

3. Compliance with International Commitments

Since the beginning of the transition period in 1989, aggregated acidifying emissions in CEE declined steadily (Figure 8.1). By 2000, most CEE countries *over-complied* with their emission ceilings set by the 1994 Sulfur protocol, which they argued back in 1994 would be costly and close to impossible to achieve (Table 8.1). Data as of 2002 also indicates that most CEE countries are either already compliant, or on a steady path toward compliance with the 2010 ceilings set by the Gothenburg Protocol for acidifying emission (Table 8.1).

What explains the "star" performance of the relatively poor CEE countries in reducing acidifying emissions and complying with LRTAP protocols? The most dramatic decline occurred in the period 1989–1993 and can be attributed to the economic downturn, reflected in Figure 8.1 as a reduction of about 35% in the total GDP of the ten countries in the region. More interestingly from a policy perspective is, therefore, the period after 1993, when CEE economies resumed growth, while SO_2 emissions in most countries continued to decline, while NO_x emissions stabilized and then declined albeit at a much slower rate than sulfur. The period after 1993 saw the decoupling of economic growth from acidifying emissions in the region.

Economic factors continued to play a role in curbing acidifying emissions even after economic growth resumed in CEE. Throughout the 1990s these countries continued to restructure and liberalize their economies. These reforms promoted efficiency gains, which helped keep emissions low, while the economies were growing. In addition, the restructuring process

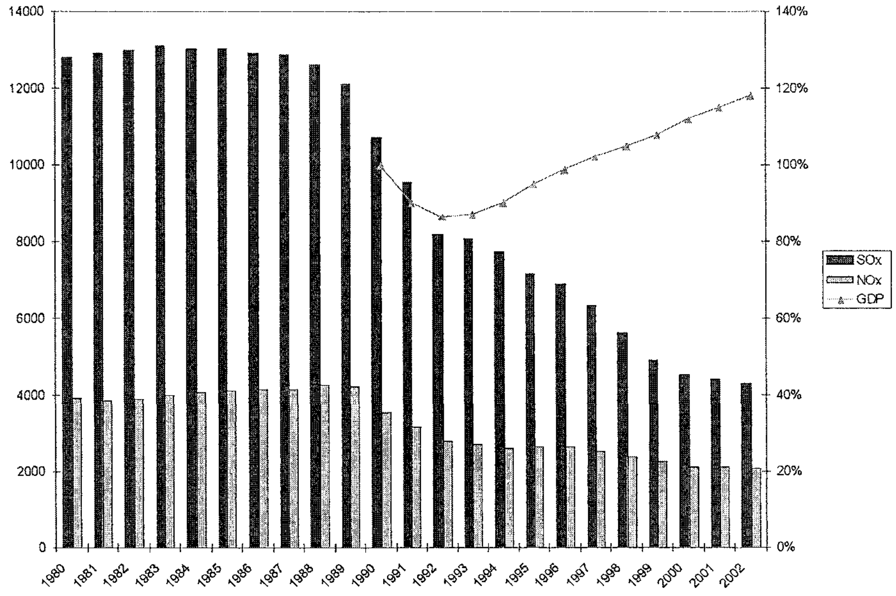

Figure 8.1. Total acidifying emissions (Gg/year) and GDP (percent of 1990 level) in Central and Eastern Europe, 1980–2002

Source: Convention on Long-Range Transboundary Air Pollution 2005 for SO_x and NO_x data (accessed via http://webdab.emep.int/); World Bank 2005 for GDP data in constant 2000 US$.

resulted in reducing the relative share of industry (particularly heavy industry) in the economy, while expanding the share of light industry and the service sector, which contribute less acidifying emissions. In 1990, for example, the share of industry in CEE economies was on average 47% and the share of services 37%. By 2002, the share of industry dropped to 31% on average, while the share of services increased to 62% on average in the region (World Bank 2005).

In addition to economic restructuring, air pollution policies adopted in the early transition period contributed to a steeper decline in emissions by the end of the 1990s than was anticipated by CEE governments as economically feasible. This allowed transition countries to agree under the 1999 Multi-Pollutant Protocol to high emission reductions compared to 1990 levels, which however did not require huge reductions of the then current 1999 levels, or policies significantly different to those already adopted or slated for implementation (Table 8.1).

The application of progressively stricter LRTAP and EU regulations, which target primarily large utilities, have contributed to a path of steady decline in acidifying emissions in Central and Eastern Europe. If we compare the paths of SO_x and NO_x emissions in the region (Figure 8.1), the steeper SO_x decline suggests that more consistent policies have been implemented to control sulfur. In the case of NO_x emissions, the growth of the vehicle fleet, coupled with relatively weak limits on emissions from mobile sources goes a long way in explaining the stagnating levels of NOx emissions and the continued problems with air quality in major metropolitan centers in the region.

If we disaggregate the data further, we would see that even though CEE countries faced similar international pressures and commitments, they differed considerably in their pollution abatement strategies and responses to international regulation. If we take three of the biggest CEE contributors to the transboundary flow of acidifying emissions—the Czech Republic, Poland and Bulgaria—we see very different patterns of emission reductions (Figures 8.2–8.4). The Czech Republic reduced sulfur and nitrogen emissions quickly and dramatically. Poland took a much more gradual approach of emission reductions. Emissions in Bulgaria followed closely the decline and growth of the economy before a steadier pattern of stabilization and reduction was established. The variety of national policy responses suggests the need to examine more carefully the role of domestic political and economic factors that contributed to the different paths of compliance with the LRTAP protocols.

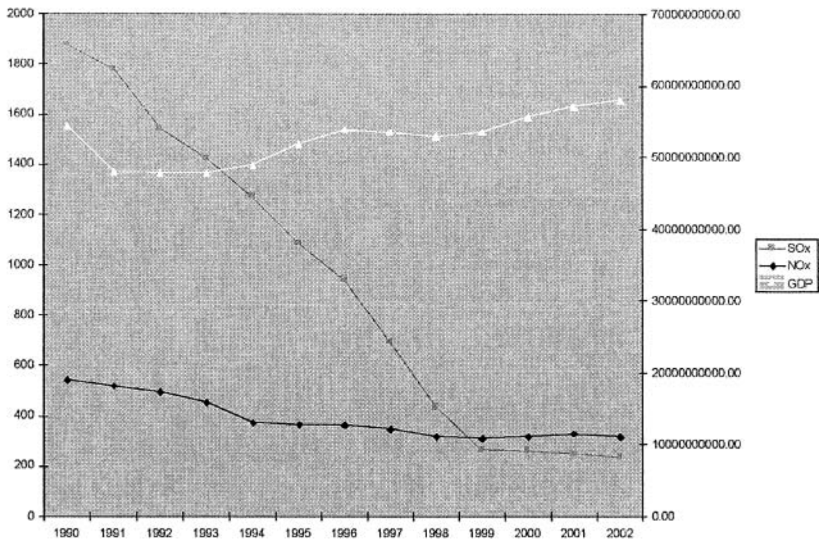

Figure 8.2. Acidifying emissions (Gg/year) and GDP (constant 2000 US$) in the Czech Republic, 1990–2002

Source: Convention on Long-Range Transboundary Air Pollution 2005 for SO_x and NO_x data (accessed via http://webdab.emep.int/); World Bank 2005 for GDP data.

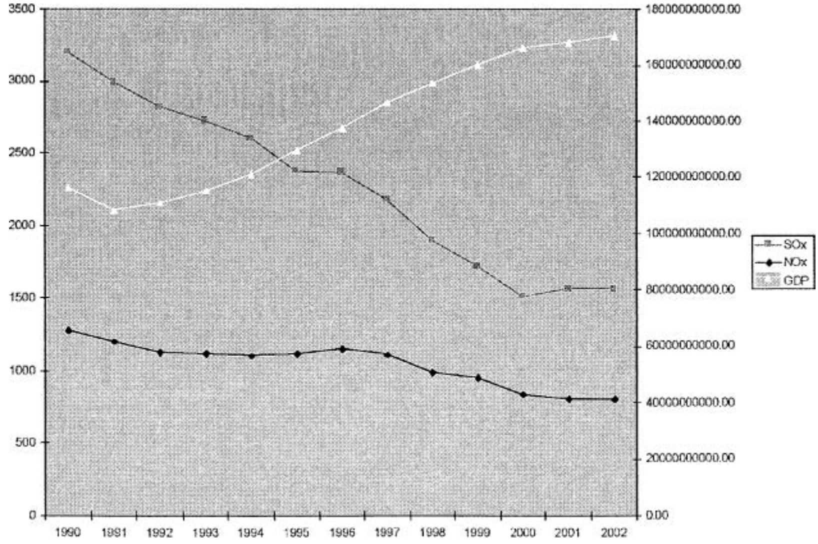

Figure 8.3. Acidifying emissions (Gg/year) and GDP (constant 2000 US$) in Poland, 1990–2002

Source: Convention on Long-Range Transboundary Air Pollution 2005 for SO_x and NO_x data (accessed via http://webdab.emep.int/); World Bank 2005 for GDP data.

4. Domestic Responses to International Norms: Bulgaria, the Czech Republic, and Poland

In the 1980s and early 1990s, Bulgaria, the Czech Republic, and Poland were among the largest emitters of acidifying substances in CEE. Their national air pollution strategies during the transition period, however, exemplify three very different responses to LRTAP and EU standards. The Czech approach can be characterized as *anticipatory over-compliance* with LRTAP and EU regulations; Poland embarked on a path of *gradual implementation* of LRTAP and EU standards; while Bulgaria adopted a *wait-and-see* approach to compliance with the same standards. Understanding these responses, which are rooted in the domestic political process, can tell us much about the conditions under which countries facing similar international circumstances are likely to implement international standards.

The Czech Republic's early over-compliance with international acidification norms was rooted in strong domestic demand for cleaner air and a proactive environmental administration in the early 1990s (Andonova 2004; Andrews 1993; Klarer and Moldan 1997). Immediately after the 1989 "velvet revolution" that ended communism in Czechoslovakia[2], the first

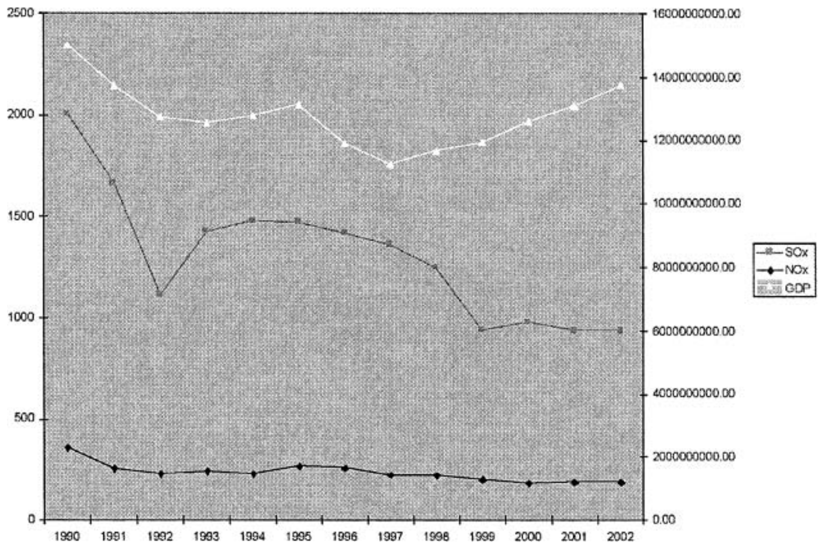

Figure 8.4. Acidifying emissions (Gg/year) and GDP (constant 2000 US$) in Bulgaria, 1990–2002

Source: Convention on Long-Range Transboundary Air Pollution 2005 for SO_x and NO_x data (accessed via http://webdab.emep.int/); World Bank 2005 for GDP data.

post-communist government of the country declared environmental protection as one of its priorities along with economic restructuring and integration in European institutions. The leadership of the Ministry of the Environment, then headed by former dissident and environmental advocate, Josef Vavrousek, pressed for timely environmental reforms to capitalize on the pro-environmental sentiment of the population. Air pollution, both local and transboundary, was identified as one of the most urgent and damaging problems.

As a result of domestic activism, Czechoslovakia adopted a Clean Air Act in 1991, which required considerable emission reductions and technical standards of desulfurization from all power plant facilities to be implemented by 1998. By the time of the negotiations of the Second Sulfur Protocol in 1994, the implementation of the 1991 Clean Air Act was already under way in the successor states of Czechoslovakia—the Czech Republic and Slovakia—contributing to significant declines in both sulfur and nitrogen emissions (Figure 8.2). It is thus no surprise that the Czech Republic as well as Slovakia committed to high levels of emission reductions under the Second Sulfur Protocol, as their domestic legislation demanded similar if not stricter standards.

What is surprising, however, is that the Czechoslovak and later the Czech authorities succeeded in forcing tough environmental standards on their power sector in a period of economic and political transition. The strong public demand for cleaner air in the regions where these industries are located clearly played a role in supporting the new legislation, but cannot fully explain the high level of industrial compliance. The government also needed to gain the cooperation of its electricity generation sector. One of the main mechanisms, which the Czechoslovak environmental administration used to promote strict air pollution standards early on, was to tie its hands behind the norms adopted by West European states and link air pollution reform to the broader objective of European integration. Minister Vavrousek argued very forcefully in the executive branch, in parliament, and in negotiations with the industry, that for the Czech Republic to achieve its ultimate objective of reintegration in Europe, it had to clean up its image, reduce its transboundary air pollution, and achieve environmental standards compatible with those of the West (Andonova 2004; Federal Assembly of the Czech and Slovak Federal Republic 1991).

The technology-based model of direct regulation, promoted by Germany in the 1980s and adopted by European Community in the 1988 Large Combustion Plant Directive, was also deemed suitable for Czechoslovakia. The Czechoslovak Clean Air Act of 1991 adopted air emission regulations for large combustion installations almost identical to those of the 1988 Large

Combustion Plant Directive and German air pollution standards. The Czech ministry of the environment argued that this approach was suitable for the country as it strived for technological excellence similar to that of Germany, and sought a rapid and effective solution to the air pollution problem (Andonova 2004).

With respect to the cost of regulation, the ministry of the environment argued that the country would have to adopt European regulations sooner or later, as it prepares for European integration, and the costs would be less if it prepares early on. In effect, it forced the then state-owned power sector to adopt a long-term view of air pollution regulations as inevitable in the medium- and long-run. With that long-term perspective, industry focused its political strategy on lobbying for policy concessions and indirect subsidies that would allow it to offset the cost of direct air pollution regulations and pass some of that cost on to consumers.

The Czech Electricity Company (CEZ) used the argument of excessive environmental compliance cost to keep its monopoly position on the electricity markets through much of the transition period. The government also supported the CEZ project for a second nuclear power plant, which facilitated the retirement of the oldest and most polluting coal fired plants. The Czech government helped the company secure international investment in desulfurization equipment. It also supported investments in air pollution abatement through the system of environmental funds, capitalized through the collection of environmental taxes, fees, and fines. As a result of both government and industry policies, CEZ cut drastically its air emissions, achieving close to full compliance with the Czech Clean Air Act standards by 1999 (Andonova 2004; Czech Electricity Company 2002). Similar to LRTAP and EU regulation, the Czech air pollution legislation targeted mostly stationary sources, and as a result, the gains in reducing NO_x emissions have been less dramatic than those in sulfur reduction. Since the late 1990s, NO_x emissions largely stagnated in the Czech Republic (Figure 8.2) due to increased traffic and limited possibilities for further reductions from stationary sources already compliant with the Clean Air Act.

In sum, the zealous over-compliance with international acidification norms by the Czech Republic was driven by strong domestic concern and policies that mandated strict emission standards and neutralized the opposition of industry through policy concessions and subsidies. International norms provided a commitment mechanism for domestic reformers and a model for regulations. Beyond these facilitating factors, the close correspondence between domestic preferences and international commitments was the single most important factor that explains the high compliance with European acid rain standards in the Czech Republic. The significant acidifying emission reductions achieved in Slovakia, similarly suggests that the reforms and the 1991 clean air legislation of Czechoslovakia had a lasting impact on the acidification policies of its successor states.

In the case of Poland, there was a more significant gap between the requirements of international regulations and the priorities of national air pollution reforms. Reduction of air pollution, particularly in hot spot areas, was a priority of the 1990 National Environmental Plan of Poland. Poland, however, chose to focus its regulatory efforts on achieving ambient air quality standards, rather than on specific technology-based standards for large utilities embedded in EU and LRTAP regulations. It also argued that that objective of reducing transboundary air pollution should be achieved with foreign assistance (Andersson 1999; Andonova 2004; Karaczun 1996; Ministry of Environment, Natural Resources and Forestry of the Republic of Poland 1991).

Poland relied strongly and to a greater extent than any other CEE countries on its domestic coal sector for electricity production. Nuclear energy was not publicly accepted as a viable energy source. The construction of the only nuclear power plant was halted shortly after the Chernobyl disaster, under the pressure of the Polish environmental and peace movements (Hicks 1996). Poland was not interested to increase substantially the share of natural gas, as it wanted to avoid at any cost energy dependence on Russia. Thus the domestic environmental policy of Poland in the early 1990s, while strongly focused on air pollution reduction was also very concerned of the cost of regulation that might hurt its coal and electricity sectors (Andersson 1999; Andonova 2004; Financial Times 1998).

The post-communist environmental administration of Poland was keen to experiment with flexible, cost-minimizing policy instruments based on economic incentives, rather than commit to command-and-control regulations embedded in the LRTAP protocols and EU directives. The Polish Ministry of the Environment was one of the first in the region to receive substantial technical support from the World Bank and US research organizations. It was well staffed with environmental economists, and promoted the use of a variety of economic instruments ranging from price liberalization of coal, experimentation with tradable permits, introduction of high pollution fees and penalties for air pollutants, and the implementation of the strongest system of environmental funds in the region (Andersson 1999; Andonova 2006; Bluffstone and Larson 1997; Regional Environmental Center 2001a).

At the end of the 1990s, the sulfur emission charge in Poland was one of the highest in Europe at EUR 80 per ton, the non-compliance penalty was EUR 800 per ton, and the collection rate was close to 99% (Regional Environmental Center 2001b). The Polish National Environmental Fund as well as the regional environmental funds mastered considerable resources from the enforcement of pollution fees and fines and in turn invested a large share of these resources in the subsidization of pollution abatement (Andonova 2004, Regional Environmental Center 2001a).

As a result of these policies, Poland's acidifying emissions steadily declined during the 1990s despite the growth of its economy for the most part of the decade (Figure 8.3). Poland, however, did not ratify the Second Sulfur Protocol. It argued that strict compliance with technology-based standards for individual utilities would impose unnecessary economic burden on its industry. The pull of EU membership conditionality along with the realization that pollution abatement costs were less than anticipated thanks to restructuring and early investment in desulfurization, eventually led Poland to adopt air emission standards compatible with the 1988 Large Combustion Plant Directive of the EU. It bargained for and received a transition period for compliance with the much stricter version of the directive adopted in 2001 (Andonova 2004). As Figure 8.3 indicates Poland's sulfur emissions stabilized and even increased slightly after 2000, a trend that most likely reflects the delayed implementation of some of the most costly emission standards for older large utilities.

Poland exemplifies a case of some correspondence between the objectives of improving local air quality and reducing transboundary acidifying emissions, but also a dissonance between the policy approaches deemed suitable for the domestic context and those required by LRTAP and EU regulations. This resulted in a much more gradual and less willing adjustment to international norms. Poland reduced emissions and complied with the spirit of the Second Sulfur Protocol, but paradoxically did not ratify the actual document. It adopted EU technology-based regulations unwillingly, only because of powerful political influence of the EU on the domestic policy agenda, and delayed their full implementation as much as was politically feasible.

In Bulgaria, the third country examined here, acid rain policies of the 1990s represent the all-too-familiar case in international cooperation of a dissonance between the goals of international regimes and the immediate domestic priorities and capacity. This resulted in slow air pollution reforms and eventually the formal adoption of international rules, with delayed implementation. I call this a *wait-and-see approach* to international regulations, as the country postpones compliance waiting for circumstances that could reduce the cost of international regulations to make them more palatable to domestic constituencies.

As in the other post-communist countries, there was visible public concern about air pollution in Bulgaria at the beginning of the transition period. One of the main opposition groups to the communist government, Ekoglastnost, had formed in the late 1980s to protest the devastating air pollution conditions in the northern town of Ruse. Ruse's environment and the health of its citizens had been repeatedly assaulted by uncontrolled pollution from a Romanian industrial complex across the Danube, while the communist government took little action to remedy the situation. There was also awareness of local air pollution problems, particularly around industrial installations, where ambient standards were exceeded regularly.

In Bulgaria, however, the public concern about air pollution did not trickle up to the top levels of politics in the early 1990s as it did in the Czech Republic and Poland. The first post-communist elections brought the reformed former communist party to power. The government was forced to resign only 11 months after it assumed power, and its resignation was followed by a period of frequent changes of governments and policies. In the context of political instability, continued economic decline, and double digit unemployment, environmental policies never rose to the top of the policy agenda. The power sector, which was the largest source of air pollutants, remained unreformed and politically influential, and argued that any imposition of tough air pollution standards would further undermine the competitiveness of the economy.

In this context of low political prioritization for the environment, international conventions played in some sense a critical role. In 1996, when Bulgaria hosted the second pan-European conference of European Environmental Ministers, the environmental ministry and other advocates of air pollution reforms argued that in order to save face in front of the international community, the Parliament should at least ratify the 1994 Second Sulfur Protocol and adopt a Clean Air Act in the spirit of the Protocol. This was indeed achieved, thanks in part to the fact that the 1996 Clean Air Act did not set specific emission requirements for pollution sources, which were to be specified later on in executive orders. The 1996 legislation also stipulated that utilities that had difficulty complying with the provisions of the Clean Air Act because of economic hardship could apply for "temporary," higher emission standards (National Assembly of the Republic of Bulgaria 1995). Thus, Bulgaria moved toward formal compliance with the LRTAP convention, but with weak provisions for implementation. As a result of the weak air pollution policies of the early 1990s, acidifying emissions started to increase after 1992 as the economy regained growth (Figure 8.4).

At the end of the 1990s, the foreign policy objective of advancing EU accession negotiations motivated the right-of-center government of the

Union of Democratic Forces to finally adopt almost wholesale the implementing regulations of the 1996 Clean Air Act. These implementing regulations followed closely the requirements of the EU Large Combustion Plant Directive (1988), as well as other LRTAP and EU standards. The Clean Air Act was also amended to correspond more closely with EU law, but the government nevertheless kept the provision of temporary exemptions for utilities from the strictest emission requirements. As a consequence, during much of the 1990s, there was very little investment in the type of desulfurization equipment implemented in the Czech Republic and Poland. Bulgaria's system of environmental taxes and funds was considerably weaker, and as a consequence, the government had little capacity to support environmental infrastructure investment similar to that in the economically and institutionally more advanced Poland and the Czech Republic.

The stabilization and gradual decline of sulfur emissions in Bulgaria during the second half of the 1990s was achieved chiefly through switching to fuels with lower content of sulfur in district heating, which was a policy measure of great significance, both for local air quality and acidification (Andonova 2004). As the government initiated the liberalization and privatization of electricity utilities after 2001, one of the requirements for future investors was compliance with compatible EU standards (Andonova 2002). Thus, by adopting a Clean Air Act and subsequently its implementing regulations, Bulgaria achieved formal compliance with European norms, but postponed compliance with the most costly, technology-based EU and LRTAP standards. It achieved reduction of acidifying emission compatible with the requirements of the Second Sulfur Protocol and the Gothenburg Protocol thanks, on one hand, to economic restructuring and, on the other hand, to relatively low-cost policy measures such as fuel switching that benefited local air quality. Compliance with costly emission standards for power utilities was postponed as successive governments waited for a political opportunity to shift the cost of compliance to future owners in the process of electricity sector privatization, and indirectly to consumers through the liberalization of electricity prices. It remains to be seen to what extent this *wait-and-see approach* would bear fruit. Enterprises continue to voice their concern about the high costs of EU environmental standards, as the EU tightens its monitoring prior to Bulgaria's 2007 accession.

This brief history of the acid rain policies of three transition countries demonstrates that the rules set by international regimes tend to be reshaped by different domestic circumstances. When there is a high correspondence between domestic priorities and international norms as in the Czech Republic, there is high level of implementation and even anticipatory over-compliance with international standards. The interplay between domestic and international politics is far more complex when there is a dissonance between domestic and international regulatory approaches as in the cases of Poland and Bulgaria. In the case of Poland, the strong bargaining leverage of the EU moved Polish air pollution standards closer to the standards of the international regime, even though it is not clear that these international instruments are any more effective in terms of environmental impact than Poland's preferred approach of flexible regulation. Finally, Bulgaria exemplified the case of a weak political concern and capacity to deal with acidifying emissions, which resulted in delayed and to some extent coincidental compliance with LRTAP emission ceilings, and a larger risk of non-compliance with the tougher EU and LRTAP technology-based emissions standards for utilities.

Although each country faced different circumstances and propensity to comply with international standards, international institutions played a role in stimulating the domestic air pollution and acid agenda in each of the three cases. In the case of the Czech Republic, international commitments strengthened the bargaining leverage of domestic advocates of policy change and forced industry to look for more creative solutions to reduce acidifying emissions. In Poland, international pressure and leverage influenced not so much the pace of reforms but ultimately the mixture of policy instruments used. In Bulgaria, international commitments were the single most important factor keeping the acidification policy agenda alive in the midst of the turbulent politics of the early and mid 1990s.

These three cases also illuminate some of the conditions that facilitate policy reform and compliance with international norms. The cases of the Czech Republic and Poland demonstrate that the ability of the government to work out an acceptable bargain with industrial actors bearing the cost of regulations is critical for implementing reforms. Policy compromises that could further multi-sector agreements and at the same time the public agenda seem critical for successful environmental reforms. The capacity of environmental administrations to identify and support such political bargains with industry seems crucial for tackling air pollution in emerging markets that seek rapid growth. Pro-active, highly qualified environmental ministries, supported by international funding and know-how, were critical players in the early environmental reforms of the Czech Republic and Poland. As the Bulgarian Ministry of the Environment gained strength in the late 1990s with the support of EU twinning programs and other technical assistance, it also gained capacity to move the air pollution agenda forward and lock-in environmental requirements in privatization and restructuring policies. Finally, the capacity of government institutions in transition and developing countries, not only to legislate but to support the initial investment in air pollution abatement through innovative financial mechanisms such as environmental funds, soft loans, or international financing also seems critical for the success of the air pollution reforms as evidenced by the cases of the Czech Republic and Poland. While international institutions such as the International Monetary Fund and the World Bank have often argued that government spending is not necessary and could be counterproductive for the environment, it is necessary to recognize that governments could play a critical role in stimulating environmental investment by providing targeted incentives and correcting for weakly developed financial institutions in emerging and developing markets.

5. Conclusion: Acidification Policy in a Wider Europe

What is in stock for European acid rain policies as the EU widens to include the CEE states? Skeptics might fear the weakening of the EU environmental regime and acidification standards in particular, as a result of the inclusion of poorer, yet relatively highly industrialized member states. In fact, this was one motivation for greener members of the EU to push for the adoption of the 2001 amendment of the Large Combustion Plant Directive prior to the first wave of Eastern enlargement. Polish negotiators have also made it clear that once the country is a member of the EU, they would bargain to make sure that regulations do not impose an undue burden on the Polish industries and that the cost-effectiveness principle of regulation is applied (Financial Times 1998). During the period of accession negotiations, CEE industries had limited leverage over EU acid rain policies as they were strictly on the receiving end of regulation. Now, these industries are voicing their concern about the cost of air pollution standards, and are likely to have more influence over EU policies. It is also not clear, as indicated by the Polish and the Bulgarian cases, that the full implementation of the most costly, source specific EU emissions standards would be readily achieved in some of the new member states. As a result, the European activism (both within the EU and LRTAP) in the area of acid rain policies between 1997 and 2001 might be followed by a relative lull in policy activity on the acidification front. The European Commission is likely to focus on measures of implementation, as problems with imperfect compliance with EU regulations persist, as well as tackling other urgent issues such vehicle emissions and ground level ozone.

At the same time, there are also reasons for optimism about the future of acid rain abatement in Europe. Very importantly, the EU and LRTAP policies of 1999–2001, have committed European countries to policy changes that would result in important reductions in acidification during the first decade of the new millennium. Estimates of the European Commission suggest that the new CEE member states are likely to continue to comply or over-comply with the emission ceilings set by the Gothenburg Protocol as a result of adopted policy measures and continued economic restructuring and modernization (Amann et al. 2005).

Moreover, EU leadership in climate change policies is likely to result in some important co-benefits in terms of reducing acidifying emissions (Amann et al. 2005), much like economic restructuring in transition countries assured greater reduction in acidifying emissions than anyone expected. The EU Greenhouse Gas Emission Trading Scheme regulates CO2 emissions from large utilities and enterprises, many of which are responsible for the bulk of acidifying emissions from stationary sources. Thus, increased efficiency and a shift to cleaner fuels, which are likely to be spurred by the Emission Trading Scheme, would support the objective of further reduction of acidifying compounds particularly in CEE countries where the levels of desulfurization are not as high as in Western Europe. Since the CEE states are eager to attract Joint Implementation projects as part of their climate change policies under the Kyoto Protocol, such projects could help maintain the path of declining emissions through technology innovation and efficiency gains.

Another important co-benefit likely to come from climate change policies is related to emission reductions from the transportation sector. So far, the focus of both acidification policies and climate change policies has been on large stationary sources. As options for emission reduction in those sectors become more expensive, and the size of the transportation sector grows in CEE, policy attention increasingly shifts to mobile sources of emissions contributing to acidification, as well as climate change and urban air pollution.

Uncontrolled urban air pollution is arguably one of the most health threatening and visible environmental problems in CEE. The slow decline of NO_x emission in CEE since 1992 (Figure 8.1) reveals the difficulty of dealing with nitrous oxides without meaningful policies on automobile emissions. Domestic acid rain and air pollution policies in CEE, as the three country case studies revealed, focused on large stationary sources covered by LRTAP protocols and EU regulations. If anything, CEE governments should have incentives to support European initiatives that promote cleaner vehicles, fuels, and sustainable transportation policies as this is likely to be in tune with domestic environmental and health priorities.

Finally, another possible co-benefit of the Eastern enlargement and the EU CO2 trading scheme, might be an increased willingness of EU institutions to consider a larger menu of more flexible regulatory instruments for acid rain. Regulations that stimulate further reduction of acidification, through emissions trading within the EU, would be beneficial for both East and West. Such a scheme would reduce the overall cost of compliance with the next generation of acid rain policies for West European states and industries, while providing the necessary capital for their CEE counterparts to explore further cost-effective and efficiency enhancing options for emission reductions.

In sum, the widening of the EU brings cautious optimism about the future effectiveness of pan-European efforts to reduce acidification. Cautious, because the new member states still face considerable economic problems and may be willing to use their political voice in the EU to block new regulations. The optimism comes in part from the history of the overall successful reduction of acidifying emissions since the 1990s in both parts of the continent, and the lock-in of standards that commit Europe to a similar course in the near future. More importantly, however, the hope is that expanded membership would enhance the opportunities for creative policy making and synergies among a variety of policy objectives. To sustain strong acid rain policies, European countries would have to increase their attention to cleaning up the vehicle sector, to innovation and flexibility in policy instruments, and to exploiting the synergies between acidification and climate change policies. CEE members are likely to be a positive force in this new agenda.

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Notes

1. The term Central and East European (CEE) states refers here to the ten countries from Central and Eastern Europe that applied for membership in the European Union (EU) during the 1990s: Bulgaria, the Czech Republic, Estonia, Hungary, Latvia, Lithuania, Poland, Romania, Slovakia, and Slovenia.
2. In 1992, former Czechoslovakia split into two countries: the Czech Republic and Slovakia. I refer to Czechoslovakia when discussing policy developments that took place up to 1992.

9. Acid Rain Politics in North America: Conflict to Cooperation to Collusion

Don Munton[1]

1. Introduction

Acid rain emerged in the late 1970s both as a domestic issue within the United States and Canada and as a contentious problem between the two neighbors. A decade of debate over the 1980s led eventually to controls on the emissions that cause acid rain, a step that required a new federal-provincial agreement in Canada and significant amendment of the U.S. Clean Air Act, and led also to an international agreement—the bilateral Air Quality Agreement of 1991. In the decade that followed, these efforts were praised as highly successful, especially in Ottawa and Washington. While other transboundary issues have achieved more prominence in recent years, particularly ground-level ozone, acid rain remains on the bilateral agenda.

This chapter represents an overview of the politics of transboundary acid rain in North America. It will examine the conflict that emerged in the 1980s between the United States and Canada after scientists went public about the environmental problems caused by acidification. The chapter then looks at a somewhat briefer period in the first half of the 1990s when, after a meeting of political minds, bilateral cooperation became the watchword. Finally it examines the record of implementation of the Air Quality Agreement in recent years and comes to the unsettling conclusion that cooperation has become collusion. Both governments have failed to meet many of the commitments they made in that Agreement; both seem determined at present to pretend, and allow the other to pretend, that all is well.

2. Nature of Acid Rain

What became popularly known as "acid rain" is a complex set of physical and chemical phenomena by which gases—especially sulfur dioxide (SO_2) and nitrogen oxides (NO_x)—are created and emitted, mostly by industrial processes, transported through the atmosphere, transformed into acidic compounds, and deposited onto land and water surfaces, often with serious negative effects for aquatic and terrestrial ecosystems. Acidification of lakes and streams kills fish and other aquatic species. The impact is particularly strong during the spring run-off period when melting acidic winter snows produce an "acid shock." Acid deposition can cause long-term damage to some soils and has deleterious effects on various crops and on the growth of certain types of trees, particularly at high altitudes.

The major sources of acid rain in North America are SO_2 emissions from coal-fired electrical generation and metal smelting. Motor vehicles and various industries produce NO_x. There are, however, dramatic differences between the United States and Canada in terms of sources. In the United States during the 1970s and 1980s the bulk of the SO_2 came from hundreds of thermal power plants particularly concentrated in the American Midwest. These plants tended to be old, large, and burning medium or high-sulfur coal. For Canada, the major SO_2 sources were a small number of non-ferrous smelters, particularly one located outside Sudbury, Ontario, owned by mining giant International Nickel (Inco). There were (and are) only a handful of coal-fired power plants in Canada.

A Swedish scientist, Svante Odén (1968), is widely recognized as the person who synthesized the phenomenon of acid rain in the late 1960s. The scientific pioneers, before Oden, were a Scottish chemist, Robert Angus Smith, who first identified acidic rainfall in Manchester, England, 150 years ago, and a Canadian ecologist, Eville Gorham, who observed acid rain in eastern Canada in the early 1950s (Smith 1852, 1872; Gorham 1955, 1957). The impact acid rain was having on the North American environment (and that of Europe) was documented in the 1970s by a loosely linked group of dedicated researchers.[2] It was these scientists, and not environmentalists, who brought the problem to the attention of politicians.

Canadian concern with acid rain was aroused by two sets of scientific studies. On the one hand, Ontario government scientists showed in the mid-1970s that lakes were being acidified in cottage country just north of Toronto, hundreds of miles away from the well-known Inco smelter (Dillon et al. 1978a; Dillon et al. 1978b).[3] On the other hand, meteorological research being conducted by Environment Canada scientists indicated that the acidity in precipitation falling in eastern Canada was not solely due to such domestic sources as the Sudbury smelter. The long-range transport of pollutants from the United States was a significant part of the acidification problem in Canada (Summers and Whelpdale 1976; Galloway and Whelpdale 1980). The combined message from these studies was clear: Canada had a serious and widespread environmental problem, one that was international as well as domestic.

3. Acid Rain and the Bilateral Agenda

The Canadian government's view of the problem was first set out in 1977 by the federal environment minister, Romeo LeBlanc, who termed acid rain an "environmental time bomb." Noting that transboundary acid rain originated in both the United States and Canada, LeBlanc called for immediate negotiations with the United States to "draw up new rules which could allow one nation to tell the other to turn off the pollution at the source" (Howard 1977; Malarek 1977). Shortly afterward, a 1978 "sense of the Senate" resolution formally called upon the Carter administration to negotiate a bilateral "cooperative agreement" on air pollution with Canada. This move, however, was less concerned with long-range transport of pollutants or with acid rain than with potential impact of two Canadian coal-fired power plants planned for locations near the U.S. border (United States Senate 1978). The resolution was quickly forgotten when the former issues were raised.

The two governments could only agree to form a scientific fact-finding committee. The body, ungainly dubbed the Bilateral Research Consultation Group on the Long-Range Transport of Air Pollutants (BRCG), produced an interim report in 1979 and, after some disagreements, a final report in October 1980 (Canada-United States Research Consultation Group 1979, 1980). The scientists warned that acidification was widespread and doing "irreversible" damage to lakes, rivers and fish. They also pointed to possible damage to soils, high elevation forests and man-made structures. Their report also confirmed Canadian suspicions about the major sources of the problem. Given the larger population and industrial base in the United States, American emissions of sulfur dioxide were five times greater than the Canadian totals and the United States produced overall about 70–80% of the pollutants that moved across the boundary. Canada's contribution to acidic deposition in the United States was minor in comparison.

In 1979, the United States and Canada also joined a large group of European countries in signing the Convention on Long-Range Transboundary Air Pollution (LRTAP), negotiated under the auspices of the United Nations Economic Commission for Europe (Convention 1979). This Convention acknowledged the problems of acid rain and acidification but stopped short of requiring firm targets for reducing long-range transport let alone setting out timetables for emission reductions. In August 1980, Canada and the United States signed a bilateral "Memorandum of Intent" (MOI), in many respects a reflection of the LRTAP Convention. The MOI committed both countries to negotiating a transboundary air quality agreement (but set no deadline) and to pursuing vigorous enforcement actions under existing statutes.[4] At this point, however, neither country had control programs or legislation dealing explicitly with acid rain.

4. Bilateral Conflict in the 1980s

Working groups under the MOI picked up where the BRCG had left off, and eventually produced reports (United States-Canada 1983). The MOI, however, ultimately proved not to be the kick-start to the negotiation process that Ottawa, at least, had intended it to be. In early 1982 Canada proposed formally that SO_2 emissions be reduced by 50% in both countries. This target was designed to achieve a level of deposition of no more than 20 kg of wet sulphate per hectare per year in areas sensitive to acidification. The United States rejected the proposal, firmly. The Reagan administration made clear its conviction that action was premature and, for the rest of the decade, consistently took the position that more research had to be done on the problem of acid rain before control actions could be considered.

The Canadians were frustrated but did not go away quietly. Following an inconclusive negotiation session in June 1982, Canada's environment minister, John Roberts, accused the United States of adopting a "stuck in the mud stance" and of "stalling." Shortly afterward, Canada announced it would withdraw from the formal negotiation process. A senior Canadian official subsequently charged that political appointees in the Reagan administration were interfering with the MOI scientific work groups (Robinson 1982; BNA 1982). Given the realities of North American transboundary pollution, however, Canada had no alternative but to take part in informal talks off and on over the coming years and to keep pressing Washington.

Canada publicized the acid rain problem in the United States by somewhat unorthodox, non-diplomatic means.[5] Ministerial and official speeches to U.S. audiences were a favorite route. The phrase "environmental aggression" found its way into at least one of these (Roberts 1980).[6] The Canadian embassy in Washington cooperated closely with such groups as the National Wildlife Federation on its public information campaign, and promoted a Canadian film on acid rain that the administration labelled "foreign propaganda." The Canadian province of Ontario also joined with some American state governments in legal action against the EPA's lack of enforcement of American air pollution laws, much to the annoyance of the head of the Agency (Garland 1988). And Canada doggedly pursued its bilateral acid rain agenda through multilateral channels, especially the 1979 LRTAP Convention, although Ottawa knew the United States was reluctant to go along.[7] The impact of the LRTAP Convention and subsequent activities, which always had a European focus, was however never particularly significant in North America.[8] The bilateral rhetoric softened somewhat during the latter 1980s but was still not always diplomatic. In 1987, environment minister Tom Macmillan, branded an official Reagan administration report as "voodoo science," because it downplayed the seriousness of acid rain, and referred to American politicians who opposed SO_2 controls as "Neanderthals."

The acid rain wars of the 1980s were rooted in conflicting interests and domestic politics, and were thus a departure from what some observers see as a typical "problem-solving" approach in Canada-U.S. relations (Holsti 1971). The two countries' goals differed markedly. Canada had been committed since 1977 to tackling the problem of acid rain which Canadians and Canadian governments perceived to be both a domestic and transboundary problem. Just as Canada is dependent economically on the United States, so it is in what I have termed a position of *environmental dependence* with respect to acid rain.[9] Most, but not all, of the emissions that lead to most of the acid rain in much of Canada originate in the United States. And much of eastern Canada is particularly sensitive to acidic deposition. The United States, on the other hand, is not much affected by acid rain from Canada. And, during the Reagan years, America had other priorities (Munton 1983).

5. Cooperating through Domestic Acid Rain Programs

The acid rain log jam began to clear in Washington only toward the end of the 1980s. A significant domestic political realignment, and a change of leadership, brought about the legislation America needed to reduce long-range transport, particularly of SO_2. That legislation in turn enabled the United States to resolve the longstanding conflict with Canada. In this shift, Canada's influence was decidedly secondary to American domestic politics.

The weight of scientific evidence in favour of action on acid rain had mounted during the decade. Following on from the bilateral reports out of the MOI process came additional investigations by the National Academy of Sciences, by congress and by the president's own Science Advisor, as well as a steady stream of evidence from LRTAP-related studies in Europe.[10] When Margaret Thatcher's Britain, long the holdout in Europe, announced an acid rain control program, America stood alone amongst industrial countries. U.S. environmental groups became more united in pressing for a stronger Clean Air Act. Signs of movement appeared in Washington, at least outside the Reagan White House. Debate in the U.S.

Congress had shifted in focus from whether or not acid rain was a problem to how to control acid rain and how to apportion the costs of doing so. And, sensitive to the direction of that debate, the once highly effective opposition alliance of electric utilities, coal companies, and coal miners began to splinter.[11]

The *coup de gràce* came during the 1988 presidential primaries when candidate George H. Bush promised to overhaul the Clean Air Act, and adopt new SO_2 emission controls.[12] Following the election, his administration set to work with congress on what became the Clean Air Act Amendments (CAAA) of 1990, which aimed to reduce SO_2 emissions in the United States by about 50%—the target Canada proposed in 1982 and adopted domestically. Meanwhile, Canada's own acid rain control program had been in place for awhile. When bilateral negotiations broke down in the early 1980s Canadian officials had reconciled themselves to the fact that the international war on acid rain required more of a domestic front. Pushed by Charles Caccia, a crusading federal environment minister, and solid public support, the seven eastern provinces agreed in 1985 on province-by-province allocations of reductions and on cost sharing arrangements.[13]

6. Cooperating through the Air Quality Agreement

Passage of the 1990 Clean Air Act Amendments provided the United States with both the legislation to deal with its own acid rain problem but also the needed basis for the international agreement Canada had long sought. Talks between Washington and Ottawa had actually begun in the summer months of 1990, before the Congress took its final votes on the Clean Air Act Amendments. The informal talks produced an agreed list of the elements of the prospective accord, a list that looked rather like the one embodied in the 1980 MOI. Later formal negotiations did not take long but did feature initially differing views over what was being negotiated. A compromise was eventually worked out,[14] and the Canada-United States Air Quality Agreement was signed in March 1991. A decade of bilateral conflict over acid rain was over.

In addition to enshrining commitments both to reduce acid rain and to cooperate on scientific and other matters (specifics of which will be discussed below), the Agreement had a broader goal. It was the vision of at least some if not most of those involved in the negotiations that the AQA would become a central tool or key "regime" in the management of air quality in North America. The Agreement was not to be merely an acid rain accord but rather was intended to establish an institutional framework within which the two countries could and would manage an ever wider range of ongoing air quality issues, an institutional framework that would be broad, strong and enduring. Thus, the AQA created a permanent bilateral coordinating body, the Canada-United States Air Quality Committee (AQC) – a mechanism notably lacking in the 1970s and 1980s. It was to meet regularly and produce biennial reports. The AQA also gave the International Joint Commission (IJC) a very modest responsibility as a conduit for public input.[15]

7. Cooperating through Emission Reductions

The SO_2 reductions mandated by the federal-provincial programs in Canada and the revised U.S. Clean Air Act, as well as promised in the AQA, proceeded with relative speed.[16] The Clean Air Act amendments re-wrote the rules for industrial SO_2 emissions in the United States, particularly from coal-fired power plants. For the first time it set emission standards for many large existing plants, most of them dirty and almost all of them east of the Mississippi River. Not incidentally, these were the sources of most relevance to deposition in Canada.[17] Unlike its 1977 predecessor, the 1990 Act adopted a market-oriented approach. It neither required the retrofitting of scrubbers nor generally specified how the emission limits were to be achieved. It allowed the utility companies to choose to meet these new standards using whatever viable means they chose.[18] It also established an allowances market for SO_2 emissions from utilities. An allowance entitled the holder to emit one ton of SO_2 per year. The number of allowances granted to each utility was calculated on the basis of cuts from historical emission levels. The affected power plants were required either to cut emissions down to the level covered by the allocated allowances or to purchase sufficient extra SO_2 allowances to cover their excess emissions, or adopt some combination thereof. The tradable allowance system was in principle designed to reduce the costs of the SO_2 program by encouraging emission reductions where control costs were lowest. Since such a system can only work successfully with an overall ceiling on emission, the CAAA set a nationwide cap on utility SO_2 emissions of 8.9 million tons.[19]

Acid rain controls in the United States proved to be not only a success but a bargain, replete with surprises. All the electrical power plants targeted by the 1990 CAAA met the Phase I deadlines for SO_2 and NO_x reductions in 1995. Indeed, there was initially substantial over-compliance. Affected emission sources also met the tighter Phase II standards in 2000. The utility industry accomplished its reductions not by widespread adoption of expensive scrubber technology (as expected) but largely by switching to low-sulfur coal (Munton 1998). The costs involved have been a fraction of most estimates. Low-sulfur coal was readily available, and became more so. It was cheap, and became significantly cheaper. Consequently, few plants installed scrubbers to meet the new standards, and only a few bought allowances to cover emissions.[20] The extent of actual allowance trading was therefore low, at least initially, and it played a very minor part in the reductions. While the allowance trading system was often portrayed as the "centerpiece" of acid rain controls, it was in fact more the icing on the cake for most of the 1990s.

Canada also reduced its emissions of SO_2 on schedule and at lower than expected costs, albeit without an equivalent to the much-heralded U.S. allowance system. As of 2001, SO_2 emissions in eastern Canada were down more than 50% from 1980 levels and were almost 30% below the cap set for the region.[21] Most of the SO_2 reductions resulted from extensive modernization measures at the Inco smelter near Sudbury, Ontario. Further reductions were achieved at power plants in Ontario and elsewhere.

The impact of the emission cuts is beginning to be noticed, although more with respect to deposition than ecosystem recovery. The U.S. Environmental Protection Agency soon reported declines in wet sulphate deposition in the order of 10 to 25% across the eastern United States.[22] There are also possible improvements with respect to dry deposition of sulphates, fine particulate matter concentrations, and visibility. Aquatic and terrestrial ecosystems respond slowly, and it is not surprising that evidence of ecosystem improvements would trail behind trends in reduced atmospheric transport and deposition. The EPA, while modest in its assessment of the environmental benefits to be accrued from reduced acidification, has been rather bullish about the investment in human health benefits represented by "acid rain" controls. It expects SO_2 emission cuts to reduce American health costs by $10 billion *annually,* through reduced morbidity and mortality, and these savings are projected to rise to $40 billion annually by 2010 (EPA 1995).

Both Ottawa and Washington have acknowledged that further emission reductions will be necessary in the future to solve the problem of acidification. A scientific report released by Canada's Atmospheric Environment Service in September 1997 suggested acidic deposition remained a problem (Environment Canada 1997). A second report from a multistakeholder group argued for more extensive cuts in SO_2 emissions (Environment Canada 1997; Acidifying Emissions Task Group 1997). Similarly, more recent assessments have reinforced the Environmental Protection Agency's 1995 report to Congress that further reductions would be required beyond those of the 1990 CAA to allow sensitive Adirondack Lakes to recover (EPA 1995). As a joint bilateral review said: "both countries recognize that control of transboundary pollution still has not occurred to the extent necessary to fully protect the environment, particularly in highly sensitive areas" (AQA Review 2002, p. 6). Thus, there is a clear consensus that the acidification problem will only be solved by further emission reductions to the relatively modest cuts made so far.

8. Collusion in Implementing the Agreement

The bilateral cooperativeness engendered by the Clean Air Act and the Air Quality Agreement has been reflected in official statements since the early 1990s. As required by the AQA, Canada and the United States have provided biennial progress reports from 1992 through 2004, and have conducted mandated five-year reviews of the implementation of the Agreement, in 1996 and 2001–2.[23] The biennial reports and the two five-year reviews have consistently found the governments' performances to be quite satisfactory. The 2002 review, for example, concludes that "Canada and the United States continued to successfully fulfill the obligations set forth in the Air Quality Agreement."[24] The overall impression, continually created, is that all is well. These reviews and progress reports have been largely ignored by the media and by most observers. Good news, it seems, is not news. If these official documents are considered carefully, however, they do not support the official claims. In fact, the biennial reports provide a basis for arguing that the record is not all good news, that all obligations have not been fulfilled, and that cooperation has taken on a new character – collusion.

The fact that the regular reports under the AQA have been overwhelmingly rosy is perhaps not surprising. Both the progress reports and reviews have been done by the very same officials responsible for the implementation of the Agreement – individuals who are thus in the enviable position of reviewing their own work.[25] Suffice it to say, as international accords go, this is not an agreement with a great deal of transparency.

The key commitments of the Agreement are contained in Annex 1 ("Specific Objectives Concerning Sulphur Dioxide and Nitrogen Oxides") and Annex 2 ("Scientific and Technical Activities and Economic Research"). (The relevant provisions of Annex 1 and all of Annex 2 are reproduced here in Appendix 1.) A close examination of these provisions shows clearly that Canada and the United States are not meeting all their self-assumed obligations. Space constraints necessitate but a brief summary of two main arguments here: (i) both parties have failed to meet key commitments with respect to improving air quality, and (ii) both parties have failed to meet most of the requirements of Annex 2 with respect to scientific and technical cooperation.

The governments' progress reports have consistently expressed particular satisfaction over the record of SO_2 emission reductions on both sides of the border.[26] While it is abundantly clear both countries have reduced emissions, it is also less obviously true that the American government missed a key AQA deadline in 2000. Annex 1 of the Agreement stipulates that the United States would cut SO_2 by "approximately" 10 million tons by 2000. Total SO_2 emissions from electric power generation in the United States in 2000 however were 11.2 million tons. Thus, the United States did not meet the target of a 10 million ton reduction from 1980 levels of about 20 million tons.[27] Indeed, according to recent figures, that target remained unmet in the subsequent three years. Emissions in 2003 actually increased over 2002 levels.[28] (Emissions had also increased over the three-year period following the initial 1995 Phase I deadline.) The "how" behind this recent increase (as in, how could this occur?) matters here more than the "why" (as in, why did emissions increase?)[29] The higher emission levels were legally possible under the CAA because utility companies held unused, or "banked," allowances which they chose to use in 2003–04 in order to render legal otherwise illegal plant emissions.

What is legal under American law, however, is not necessarily consistent with the AQA. The American emissions reduction commitment in the Agreement was qualified, but very precisely: the emissions target there exempts sources adopting "clean coal technology" and sources receiving "bonus allowances." These two exceptions were based on special concessions, or rewards, provided by Congress, and written into the CAA, and are in addition to the normal allowance allocations and the trading system. The Agreement, however, does not qualify the achievement of the 10 million ton target for the year 2000 by providing for the use of banked allowances, an option inherent to that trading system.[30] That being the case, the United States violated the terms of the AQA with respect to the year 2000 SO_2 emissions target and, by implication, in the years following.[31] Neither the United States nor Canada has explicitly noted this violation in bilateral reports or in official statements.

Canada for its part met its SO_2 emissions target.[32] It has, however, failed to comply with another provision of Annex 1, namely its obligation to "develop and implement" programs to prevent "significant air quality deterioration" in pristine areas.[33] The American side has, from time to time, pointed to this Canadian failure. With respect to Annex 2, Canada and the

United States have, similarly, and continually, attempted to create the impression they are meeting their commitments. In 1996 their officials concluded that "the level of scientific and technical activities and economic research has been adequate to meet the reporting needs of the Agreement." The Parties also found that the Air Quality Committee was fulfilling the objectives of the Agreement because it was meeting annually and publishing biennial reports. That is one measure, to be sure, but at best a rather minimalist one. In fact, once again, the reality is otherwise. The failures to implement Annex 2 have been substantive and numerous rather than technical and specific. Stated briefly, the parties committed themselves in Annex 2 to coordinate emission and ecosystem monitoring, and to *cooperate* on research programs with respect to atmospheric modeling, environmental effects (aquatic and forest ecosystems, visibility, human health, and others), control technologies, economic measures, and other aspects of transboundary air pollution problems (see Appendix 2).

The regular biennial reports of the Canada-United States Air Quality Committee (AQC) provide a basis for assessing the record of the governments' implementation of Annex 2, as they did for Annex 1. The reports employ the activity categories listed in the Agreement itself, namely: emissions inventories; atmospheric modeling; deposition monitoring; aquatic ecosystem, forestry, agricultural, materials and health effects; control technologies; and mitigative measures. I will summarize the findings, in this order. (Appendix 2 to this paper provides the details.)

The AQC reports indicate the governments have engaged in adequate if not significant coordination of SO_2 and NO_x emissions data as well as of atmospheric model development, or at least they did up to 1995. There has, however, been little cooperation and little or no coordination on monitoring acidic deposition. (More will be said below on this gap, as it is a particularly serious one.) The governments have exchanged information but done little or no cooperative research on the effects of acidic deposition on aquatic ecosystems, forest ecosystems, visibility or materials. Canadian and American scientists have done some coordinated and cooperative work on human health effects; the AQC recognized this work but provided no funding or support. The governments have not engaged in significant cooperation on control technologies or market-based mechanisms, and report no consultation on mitigative measures.

To summarize the record quantitatively, there are 19 distinct commitments in Annex 2. Canada and the United States have fully met four of these 19 commitments. They have implemented eight only partially and have failed to implement the remaining seven to any significant degree. The few fully implemented commitments tend to be scientific activities that were already well coordinated at the time of the negotiation of the Air Quality Agreement (for example, atmospheric modelling). The most significant commitments the governments have not implemented to any significant degree relate to deposition monitoring and ecosystem effects research – activities critical to assessing the extent to which the reductions of SO_2 and NO_x emissions have accomplished the aims of the 1991 Agreement. This failure has persisted despite the fact that the governments claim "tracking progress made by emission reductions ... is important" (2002 Review, p. 1). While recent American studies have begun to provide a picture of declining acidic deposition in that country,[34] the Canadian performance on monitoring deposition and effects has been particularly lacking.

The absence of cooperative deposition and effects monitoring is not merely a violation of the AQA. The lack of such research almost certainly means there is no scientific basis for long-term assessment of the success or failure of current air pollution control programs. The extent of monitoring efforts is certainly inadequate to the task of fully assessing the environmental response in Canada to emission reductions in the 1980s and 1990s. In other words, the parties and particularly Canada have not ensured an adequate scientific basis on which to judge conclusively if the acid deposition goals of the Agreement are being achieved.[35] The governments' own progress reports thus reveal at best a very mixed record on Annex 2 commitments. Generally, while they have engaged in a reasonable degree of information sharing, the governments have not cooperated or coordinated on a wide range of scientific and technical matters to the extent mandated by the AQA. The level of Annex 2 activity reported since 1992 has not reflected the original intent and desire of the Parties to create a strong and continuing level of scientific and technical cooperation.

The foregoing accounting of the AQA activities suggests more than simply a pattern of inattention and incomplete implementation. The governments themselves have had little or nothing to say about these deficiencies. Officials have been reluctant to admit a mutual lack of implementation and, almost without exception, have not only failed to acknowledge commitments they themselves have not met but also avoided criticising the other side for its failures. Hence, cooperation has become collusion.

Why? It would likely require another chapter to explain fully the collusion that has occurred. A brief explanation would start with a basic fact: states do not like to acknowledge they are failing to meet their international commitments, and certainly do not appreciate other states pointing out such failures. Both the United States and Canada are thus inclined to make the best they can of the record and to avoid throwing stones in a neighborhood of glass houses. The United States, for its part, wants to portray the considerable achievements of the 1990 Clean Air Act acid rain program as an unblemished success. The EPA is rather proud of its innovative SO_2 allowance program, and recent converts, one might note, tend to be the most zealous of disciples. Canada, for its part, has moved away from the position it took in the 1980s that acid rain was its major environmental problem; other priorities have now taken pride of place. A new group of managers also took positions of power in Environment Canada in the 1990s, and after 1994 they faced severe budget cutbacks imposed by a government determined to reduce the federal deficit. Among the most vulnerable to these cuts were the expensive monitoring programs related to acid rain. Moreover, the influence of the scientists declined after the Air Quality Agreement was negotiated. Positions for both countries on the bilateral Air Quality Committee are held largely by air quality program mangers, most of whom have a vested interest in declaring the programs for which they are responsible to be successful.

While there may be some merit to the argument that other environmental problems are more pressing than acid rain, the fact is that key pollutants now recognized to be part of such currently high priority problems as ground-level ozone and airborne particulates are also precursors to acid rain (NO_x and SO_2). As such, the maintenance of existing, long-term monitoring programs is of vital importance to proper assessment of these priority problems, as well as to assessment of controls designed to reduce acid deposition.

9. Conclusion

In North America, the transboundary politics of acid rain have gone through three distinct periods. From the time it appeared on the bilateral agenda in the late 1970s until 1990, it was a matter of much *conflict.* Canada's environmental sensitivity to transboundary flows of pollution from the United States made acid rain the priority issue in the bilateral relationship. Canada was the *demandeur* and the United States was very much the reluctant party. The acid rain issue thus did not reflect what K. J. Holsti has described as the traditional "problem-solving" approach adopted in Canadian-American relations.

From the 1990 passage of the U.S. Clean Air Act through the negotiation of the Canada-United States Air Quality Agreement in 1991 and into the mid 1990s or so, acid rain was a subject of significant bilateral *cooperation.* Control programs were mandated and inaugurated and then showed remarkable early success, at least in terms of emission reductions. In the United

States in particular, acid rain controls have been cheap, quick and administratively clean but not entirely successful. Phase I of the Clean Air Act saw utilities complying early, complying often and complying more than fully. That early momentum has not been maintained, and emission levels in the United States have actually increased relative to 1995.

While a cooperative spirit has prevailed since the signing of the Air Quality Agreement, the level of actual cooperation and coordination has arguably declined. In short, and stated more bluntly, both Canada and the United States have neglected and violated the terms of a legally binding international accord. While both countries have met some but not all Agreement obligations with respect to reducing key pollutants, they have failed to meet self-assumed obligations to coordinate scientific research and other activities. Bilateral harmony has become only barely disguised *collusion.* A close analysis of the biennial progress reports under the Agreement shows the two governments have each overlooked weaknesses and failings in their own and in the other's implementation of the Agreement. Given that both sides acknowledge further emission reductions will be necessary in the future to solve the acid rain problem, both the control program commitments in Annex 1 and the cooperation/coordination commitments in Annex 2 remain important.

Contrary to widespread assumption and popular belief, the acid rain problem has not been solved. Substantial SO_2 emissions continue. Acid rain from transboundary flows continues to affect ecosystems. The cuts mandated by existing control programs, after all, were of the order of 50% not 90% or 100%, and they were viewed by experts at the time to be interim steps. Moreover, given that compliance costs under Title IV of the Clean Air Act have been much lower than expected, and given that the 50% target in the United States was based on what can now be seen as inflated assumptions about control costs, the required emissions reductions set in 1990, arguably, should have been much more stringent. The acid rain provisions of the 1991 Agreement are by no means passé.

It is surely not in the interest of either Canada or the United States to sign and ratify international environmental agreements and then not implement the terms of these agreements. Nor is it in either country's interest, in terms of managing transboundary air quality problems, to forgo the advantages of close and sustained scientific cooperation under the aegis of the Air Quality Agreement. The record of the past ten years is therefore both disturbing and in need of further and closer examination.

Appendix 9.1. Relevant Provisions of the AQA

ANNEX 1: SPECIFIC OBJECTIVES CONCERNING SULFUR DIOXIDE AND NITROGEN OXIDES

1. Sulphur Dioxide
   A. *For the United States:*
      1. Reduction of annual sulfur dioxide emissions by approximately 10 million tons from 1980 levels in accordance with Title IV of the Clean Air Act i.e., reduction of annual sulfur dioxide emissions to approximately 10 million tons below 1980 levels by 2000 (with the exception of sources repowering with qualifying clean coal technology in accordance with section 409 of the Clean Air Act, and sources receiving bonus allowances in accordance with section 405(a)(2) and (3) of the Clean Air Act.).
      2. Achievement of a permanent national emission cap of 8.95 million tons of sulfur dioxide per year for electric utilities by 2010, to the extent required by Title IV of the Clean Air Act. [...]

   B. *For Canada:*
      1. Reduction of sulfur dioxide emissions in the seven easternmost Provinces to 2.3 million tonnes per year by 1994 and the achievement of a cap on sulfur dioxide emissions in the seven easternmost Provinces at 2.3 million tonnes per year from 1995 through December 31, 1999.
      2. Achievement of a permanent national emissions cap of 3.2 million tonnes per year by 2000.

ANNEX 2: SCIENTIFIC AND TECHNICAL ACTIVITIES AND ECONOMIC RESEARCH

1. For the purpose of determining and reporting on air pollutant concentrations and deposition, the Parties agree to coordinate their air pollutant monitoring activities through:
   a. coordination of existing networks;
   b. additions to monitoring tasks of existing networks of those air pollutants that the Parties agree should be monitored for the purposes of this Agreement;
   c. addition of stations or networks where no existing monitoring facility can perform a necessary function for purposes of this Agreement;
   d. the use of compatible data management procedures, formats, and methods; and
   e. the exchange of monitoring data.
2. For the purpose of determining and reporting air emissions levels, historical trends, and projections with respect to the achievement of

the general and specific objectives set forth in this Agreement, the Parties agree to coordinate their activities through:

   a. identification of such air emissions information that the Parties agree should be exchanged for the purposes of this Agreement;
   b. the use of measurement and estimation procedures of comparable effectiveness;
   c. the use of compatible data management procedures, formats, and methods; and
   d. the exchange of air emission information.

3. The Parties agree to cooperate and exchange information with respect to:
   a. their monitoring of the effects of changes in air pollutant concentrations and deposition with respect to changes in various effects categories, e.g. aquatic ecosystems, visibility, and forests;
   b. their determination of any effects of atmospheric pollution on human health and ecosystems, e.g. research on health effects of acid aerosols, research on the long-term effects of low concentrations of air pollutants on ecosystems, possibly in a critical loads framework;
   c. their development and refinement of atmospheric models for purposes of determining source receptor relationships and transboundary transport and deposition of air pollutants;
   d. their development and demonstration of technologies and measures for controlling emissions of air pollutants, in particular acidic deposition precursors, subject to their respective laws, regulations and policies;
   e. their analysis of market-based mechanisms, including emissions trading; and
   f. any other scientific and technical activities or economic research that the Parties may agree upon for purposes of supporting the general and specific objectives of this Agreement.
4. The Parties further agree to consult on approaches to, and share information and results of research on, methods to mitigate the impacts of acidic deposition, including the environmental effects and economic aspects of such methods.

Appendix 9.2. Annex 2 Commitments and Actions Reported (1992–2004) Under the 1991 Canada-United States Air Quality Agreement

Commitment	Actions Reported
For the purpose of determining and reporting on air pollutant concentrations and deposition, the Parties agreed (in paragraph 1 of Annex 2):	
1. to coordinate existing deposition networks	No coordination has been reported (beyond co-location of a small number of instruments). Results from Canadian SO_2 deposition networks reported recently have been incomplete, thus making integrated deposition maps impossible. No reporting of coordinated results on nitrogen deposition since 1994. No coordinated deposition networks for fog or dry sulphate deposition.
2. to add monitoring through existing networks of those air pollutants the Parties agree to monitor	No additional monitoring has been reported for acid deposition. Some existing monitoring networks have been closed or are no longer reporting.
3. to add stations or networks as necessary	No additional stations have been added for acid deposition. Monitoring of ozone and particulate matter is being reported for some areas only.
4. to use compatible data management procedures, formats and methods	No adoption of compatible procedures has been reported recently. Canada and the United States continue to use different measurement procedures for dry deposition of sulphates.
5. to exchange monitoring data	Monitoring data of various sorts have been exchanged.
For the purpose of determining and reporting air emission levels and projections, the Parties agreed (in paragraph 2 of Annex 2):	
6. to coordinate the identification of relevant air emission information	The Parties have developed coordinated data sets on SO_2 and NO_x emissions. The Parties have identified precursors of ozone and particulate matter.
7. to coordinate the use of comparable measurement and estimation procedures	No coordination of measurement and estimation procedures has been reported. Canada and the United States are using different measurement and estimation procedures for ozone, particulates and dry sulphate deposition.Some comparison of measurement of deposition is being done through the use of co-located monitors.

Appendix 9.2. (Cont.)

Commitment	Actions Reported
8. to coordinate the use of compatible data management methods	No coordination of compatible data management methods has been reported. Some efforts have been made to improve consistency of SO_2 and NO_x data. Other pollutants are still subject to different data collection procedures.
9. to coordinate the exchange of air emission information	The Parties have coordinated the exchange of air emission information for SO_2, and NO_x. The Parties have exchanged information regarding ozone precursors, including NO_x. A particulates inventory is underway.

With respect to other aspects of transboundary air quality, the Parties agreed (in paragraph 3 of Annex 2):

10. to cooperate and exchange information relating to the monitoring of the effects of changes in air pollutant concentration and deposition with respect to changes in aquatic ecosystems	No cooperative monitoring of the effects of changes in air pollutant concentration and deposition on changes in aquatic ecosystems has been reported. U.S. studies done but no comparable general work in Canada. Information on separate national monitoring is exchanged. (Outside of the AQA, some cooperation is facilitated by joint participation in the LRTAP program and by state and provincial governments.) No cooperation on "critical loads" between the Parties.
11. to cooperate and exchange information relating to the monitoring of the effects of changes in air pollutant concentration and deposition on changes in visibility	No cooperative monitoring of the effects of changes in air pollutant concentration and deposition on changes in visibility has been reported. The Canadian monitoring program is very limited. There has been no recent reporting on the exchange of information on visibility.
12. to cooperate and exchange information relating to the monitoring of the effects of changes in air pollutant concentration and deposition with respect to changes in forests	No cooperative monitoring of the effects of changes in air pollutant concentration and deposition on changes in forests has been reported since 1994. The Parties have exchanged information.

Appendix 9.2. (Cont.)

Commitment	Actions Reported
13. to cooperate and exchange information relating to the monitoring of the effects of changes in air pollutant concentration and deposition with respect to changes of other kinds	Cooperative monitoring of ozone has begun, but no cooperation on monitoring its effects has been reported.
14. to cooperate and exchange information relating to the determination of any effects of atmospheric pollution on human health and ecosystems	No projects relating to the determination of any effects of atmospheric pollution on human health and ecosystems have been carried out under the AQA. Some scientists cooperate in a non-official capacity. The Parties have exchanged information.
15. to cooperate and exchange information relating to the development and refinement of atmospheric models for purposes of determining source-receptor relationships and transboundary transport and deposition of air pollutants	Cooperation and the exchange information relating to the development and refinement of atmospheric models have been reported. The models being used, however, are still particular to each country.
16. to cooperate and exchange information relating to the development and demonstration of technologies and measures for controlling emissions of air pollutants	The Parties have exchanged information. No cooperation on the development and demonstration of technologies and measures for air pollution emission control has been reported.
17. to cooperate and exchange information relating to the analysis of market-based mechanisms	The Parties have exchanged information. No cooperation relating to the analysis of market-based mechanisms has been reported recently. Some information has been exchanged.

Appendix 9.2. (Cont.)

Commitment	Actions Reported
18. to cooperate and exchange information relating to other scientific and technical activities that the Parties may agree upon	No cooperation relating to the other scientific and technical activities or economic research has been reported. No information exchanged.
The Parties also agreed (in paragraph 4 of Annex 2):	
19. to consult on approaches to mitigating the impacts of acidic deposition, including environmental effects and economic aspects, and to share information and results of research on these methods	No consultation relating to approaches to mitigating the impacts of acidic deposition has been reported. No information exchanges reported.

Notes: This table uses the wording of the Air Quality Agreement, but the articles of the Agreement are disaggregated into distinct commitments. "Actions reported" are those reported in the Progress Reports under the Canada-United States Air Quality Agreement. It should be noted that the commitments made in the accord are commitments assumed by the Governments of Canada and the United States.

Glossary: "Exchange" of information refers to a verbal discussion in which information held by both parties is provided to the other, or the physical transfer of information between the two in both directions. "Cooperation" involves parallel or joint actions by the two parties, beyond the mere exchange of information, such as the undertaking and implementation of studies by each party with some common elements as well as the sharing of the results of those studies. (A study done by one side and communicated to the other is a one-way provision of information, not evidence of scientific cooperation. Two studies done in isolation even if communicated to others do not by themselves constitute cooperation.) "Coordination" involves the collaborative design and execution of a single, joint scientific project or integrated projects. The two parties share in the work and its conclusions.

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Notes

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2. The history of the science of acid rain is discussed in Cowling (1982) and Munton (1981). Among the important early studies in the 1970s on acid rain

in North America were Beamish and Harvey (1972), Beamish (1976), Likens, Bormann and Johnson (1972), and Likens and Bormann (1974).

3. The Ontario studies that provided evidence of aquatic acidification in southern Ontario in the 1970s were not the first evidence of acidification of Ontario ecosystems. Prior to the 1970s there was in fact a lengthy history of scientific discovery and of not-so benign neglect of the problems associated with smelter emissions. Ontario government researchers had observed lake acidification and fish loss in a wide area surrounding Sudbury in the latter 1960s and forest damage as early as the 1950s (Munton 2002, 2004). Similar conclusions about SO_2 damage had emerged in the 1930s in the scientific investigations of the famous Trail smelter case (National Research Council 1939; Read 1963).
4. While the 1970 and 1977 U.S. Clean Air Acts did lead to a gradual overall reduction in SO_2 emissions across the United States (from 32 to 27 million tons during the 1970s), one unintended consequence may have been to increase long-range transport and thus acidic deposition through encouraging "tall stacks." *See* Gschwandtner et al (1986) and Wetstone and Rosencranz (1983), p. 101.
5. The two major exceptions to the pattern of negative rhetoric and impasse during the 1980s were an ill-fated 1983 proposal for a limited, first step, acid rain control program championed by EPA head William Ruckelshaus and the ultimately unsuccessful efforts of Canadian prime minister Brian Mulroney in 1985–87 to use his personal relationship with Ronald Reagan and summit diplomacy to move the United States toward acid rain controls. The "atmospherics" of Canada-United States relations improved during the Mulroney years, but the impasse was not broken and reducing acid rain remained Canada's objective.
6. "The Urgency of Controlling Acid Rain," speech by the Hon. John Roberts, Minister of the Environment Canada, to the Air Pollution Control Association, Montreal, June 12, 1980, Statements and Speeches, No. 80/8, Department of External Affairs, Ottawa.
7. On these protocols, *see* the chapter by Schreurs (this volume) and Soroos (1997), chap. 5.
8. The impact of the LRTAP process during the 1980s was lessened by the fact that the United States did not sign the sulfur protocols under this convention and also by the fact that the first LRTAP nitrogen protocol, which both Canada and the United States signed, did not require the United States to adopt significant new measures.
9. On the concept of environmental dependence, *see* Munton (1980–81).
10. United States-Canada, Memorandum of Intent on Transboundary Air Pollution, "Executive Summaries, Work Group Reports", February 1983; National Academy of Sciences (1981, 1983); United States Office of Science and Technology Policy (1984); United States Congress (1984); and United States NAPAP (1987).

11. The electric utilities were particularly opposed to legislation mandating expensive scrubbers to reduce sulfur dioxide emissions. The coal producers and miners wanted to ensure power plants continued to burn coal but had little concern about the costs imposed on the utilities. The United Mine Workers and some coal companies broke with the utilities in 1987 to support control measures that would have required scrubbers.
12. Another important political development was the replacement of the Senate majority leader, Senator Byrd, a long-time opponent of such controls, with George Mitchell of Maine, a strong proponent.
13. The western provinces refused to take part. Their argument was that their own emissions had little impact on either the eastern part of the country or on the highly alkaline soils of western Canada, and that acid rain was therefore an eastern problem.
14. The Canadian view was that the accord at its heart ought to comprise an exchange of *commitments* to reduce *transboundary* air pollution. Each country would also promise the other to implement the (domestically) established reductions. The American position, on the other hand, was that the agreement ought merely to contain statements by each country of what its domestic law required in terms of controls. In the U.S. version, then, there would be no mutual commitments; neither the United States nor Canada would make explicit and solemn promises *to the other* and thus each would be free to alter its domestic acid rain laws and policies without directly violating the bilateral accord. For the Canadians, the difference was critical. An international agreement that the United States could change unilaterally and at will was simply not acceptable. The compromise eventually reached involved a mutual exchange of commitments on reducing national emissions (though *not* on reducing *transboundary* pollution flows) as well as commitments on cooperating and coordinating scientific and other research.
15. The AQA only provides for the Commission to collate public responses to the AQC reports. It has no "watch-dog" role. The governments gave themselves the responsibility of deciding, in their wisdom, whether or not they are meeting their commitments to reduce emissions and engage in scientific cooperation. Although the AQC suggested in the 1996 five-year review of the Agreement that a third party might assist in the implementation process, it reversed itself on this issue in 2002.
16. NO_x emissions were a somewhat different story. In both countries, despite new regulations on motor vehicles, the overall NO_x levels have remained roughly stable or declined only slightly. As emissions per vehicle go down, the numbers of vehicles continue to go up, particularly those with relatively high gas consumption and emissions such as SUVs. Programs to reduce haze in wilderness areas have also met with little success.
17. Previously emission standards applied only to new plants and existing plants had been exempted. The number of plants was later expanded. Phase I set a limit of 2.5 lbs of SO_2 per million BTU as of 1995. Phase II required the

affected units to meet a more stringent 1.2 lbs standard. 1980's emissions from targeted plants averaged 4.2 pounds and ranged from 2.5 to more than 10 pounds.

18. The 1990 CAAA was thus more like the original 1970 version of the Act in not specifying how emission reductions had to be carried out.
19. Although touted as a complete departure from the "old-style" "command and control" type of pollution legislation, the 1990 Amendments were in fact riddled with detailed and specific "commands," including the national cap of 8.95 million tons, the 2.5 and 1.2 lbs limits for SO_2 emissions, the (fixed) number of allowances provided each year, non-compliance fines of $2,000 per ton of excess emissions, etc.
20. Environmental Defense, a promoter of the allowance system, itself admitted that the low cost of compliance during Phase I was the result of "the flexibility [the CAAA] afforded to plant operators, particularly to choose and blend fuels" (Environmental Defense 2000, p. 20). The unit cost of allowances was thus much lower in the 1990s than had been expected earlier, due to the low prices for low-sulfur coal and reduced transportation costs.
21. Canada-United States Air Quality Agreement 2004 Progress Report
22. Environmental Defense (2000), p. 22; Canada-United States, *Air Quality Agreement, Progress Report,* Ottawa and Washington, 1996.
23. All these reports are available on the web (http://www.ec.gc.ca/pdb/can_us/canus_links_e.cfm (March 2005)). They are however difficult to find from the Environment Canada home page since they are slotted under the heading "CAC," which, in department jargon, stands for "criteria air contaminants." The 2004 progress report is on the U.S. EPA website (www.epa.gov/airmarkets/usca/2004report.html) but EPA apparently does not keep earlier versions of the reports on the web.
24. Canada-United States Air Quality Agreement, 2002 Progress Report, Section VI, p 6.
25. The first five-year *Review* in 1996 was a perfunctory 5 pages; the second, released a year late in 2002 (without comment on the date), came in at 5½ pages. Emphasizing the lack of independence between implementation and assessment, both Reviews were tacked on to the end of biennial progress reports, comprising "Section VI" of the respective reports.
26. It must be noted that the reductions did not come about *because* of the 1991 international agreement. Both governments were required by domestic arrangements or legislation to do what they did, to reduce the precursors of acid rain, and these federal-provincial arrangements (in the case of Canada) and legislation (in the case of the United States.) were already in place at the time that international agreement was concluded.
27. *AQA Progress Report,* 2004, p. 3. The considerable difference of 1.2 million tons between the target and the actual emissions would not seem to lie within the normal realm of the term "approximately." Higher emission level

figures for 1998 through 2000 are provided in the 2004 progress report than in the 2002 progress report (2002, p. 2; 2004, p. 3).

28. The 2004 biennial report, however, does note in passing that total U.S. SO_2 emissions increased in 2003, to 10.6 million tons. As a result, the EPA acknowledged, U.S. power plant emissions in 2003 were only slightly below what they were in 1995, and only 38% below 1990 levels (*Air Quality Agreement Progress Report,* 2004, Section 1, p. 2).
29. One major reason why U.S. SO_2 emissions were higher in 2003 was that natural gas price increases led to a greater use of coal to produce electric power.
30. The fact that Annex 1 of the AQA omits reference to the possibility (now realized) of the use of banked allowances to offset higher than allowed emissions was likely an oversight by the American negotiators. The AQA was negotiated, in a fairly short period of time, only months after passage of the CAAA.
31. The 2002 progress report mistakenly states that the SO_2 commitment by the United States in the AQA was a reduction of 10 million tons (1980 to 2000) "taking into account credits ('allowances') earned for reductions from 1995 to 1999." In fact, the quoted phrase is not the wording of Annex 1 and thus not exactly the American commitment in the Agreement.
32. 2004, p. 2. This chapter does not focus on the complicated question of nitrogen deposition and the two governments' problematic efforts to reduce it. Both "recognize that nitrogen emissions are an important part of the acidification issue" (2002 review, p. 1) and both have called for nitrogen critical loads. At the same time, while claiming to have met targets for emissions from stationary sources, the official reports have little to say about mobile sources, the most troublesome aspect of this problem, and have not explicitly called for further control programs beyond those specified in the AQA. Other articles in this collection provide a valuable assessment of the key role nitrogen plays in acidification.
33. The fact that the United States has pointed to this failure on Canada's part makes it the one exception to the rule that neither Canada nor the United States have taken each other to task for AQA commitments not met. *See*, for example, *Canada-United States Air Quality Agreement, 2002 Progress Report,* Section VI, Second Five-Year Review and Assessment, p 5.
34. *See* earlier note.
35. The AQC did address transboundary ozone and led the effort to incorporate a new ozone annex into the Agreement. It has also begun to address the problem of airborne particulates. While these successful efforts at cooperation are to be applauded they relate only indirectly to scientific and technical cooperation on acid rain. The ozone annex is at: http://www.ec.gc.ca/cleanair-airpur/CAOL/air/can_usa_e.html

10. Air Quality and Power Production in the United States: Emissions Trading and State-Level Initiatives in the Control of Acid-Producing Emissions, Mercury, and Carbon Dioxide

Daniel Sosland[1]

Recently, the U.S. federal government has pursued a determined strategy toward increased energy production while paying little heed to the impact of this strategy on air quality and failing to take effective measures to reduce emissions of pollutants from the fossil-fueled power plants that dominate U.S. energy generation. While the evolution of the Clean Air Act and its important amendments—particularly the 1990 Acid Rain Program amendment—demonstrated a strong popular and political commitment to improved air quality in previous decades, the excessive focus on fossil-fuel based energy production and absence of a coherent national energy strategy that have characterized U.S. policy since 2000 represent a troubling reversal. Thus it is critical to take a closer look at how U.S. energy policy is impacting on air quality, and review the steps taken prior to 2000 that might provide useful lessons for the formulation of more effective energy and air quality policies in the future.

In this chapter, I will examine two key aspects of the air quality regime that evolved in the United States (1960s–90s) and the lessons they may hold for addressing future challenges: (1) the adoption of a market-based approach to SO_2 in 1990; and (2) the role of state and local initiatives in the formulation of federal air quality legislation. First, the 1990 adoption of a federal market-based incentive system for controlling acid-producing emissions will be reviewed; while this program has been fairly successful in addressing acid-producing emissions from coal-fired power plants, whether or not it provides a sound method for reducing emissions of other unwanted pollutants needs to be carefully considered. I will argue that recent steps toward instituting a cap-and-trade regime to control mercury emissions from power plants represent an inappropriate approach to the reduction of this potent toxin and are likely to result in an unjust and dangerous regional distribution of this pollutant. On the other hand, carbon dioxide emissions from power plants—which are a major factor in climate change—provide a much more appropriate candidate for successful application of a market-based approach.

Secondly, the air quality regime in the United States did not begin with action at the federal level; rather, it was the steps of cities, states, and regions that eventually catapulted the issue to the federal level and resulted in the adoption of the Clean Air Act. In looking toward the future and the absolute need for the U.S. to reduce emissions of climate change gases—particularly, carbon dioxide—the lessons of this earlier period may be important. While the federal government has failed to take any action to reduce CO_2 emissions, all the while steadily increasing our fossil fuel based power production, cities, states, and regions are now taking it upon themselves to adopt policies aimed at reducing their emissions of climate change gases. For example, northeastern states are currently engaged in a serious first-in-the-nation effort to develop a regional program to cap power plant emissions of carbon dioxide, using the method of cap-and-trade that has proven so successful in reducing acid-producing emissions of sulfur dioxide – and at lower than predicted cost. In this chapter, I will review the myriad of sub-federal level initiatives underway across the United States and argue that in the absence of federal leadership on this issue, states and regions are stepping in to fill the gap. It must be hoped that their actions will spur more responsible U.S. energy and air quality policies at the federal level.

1. Background: Clean Air Act

Federal clean air policy has a long history of bipartisan support, starting with the original Clean Air Act (CAA) in 1970.[2] The CAA was championed by Democrats such as Ed Muskie of Maine and signed into law by President Richard Nixon.[3] The major amendments to the Clean Air Act in 1990 were also championed by Democrats like Representative Henry Waxman (D-CA) and signed into law by a Republican President, George H. W. Bush. The first air pollution regulations, however, were passed at state level. For example, the California Motor Vehicle Pollution Control Act of 1960 established the Motor Vehicle Pollution Control Board (MVPCB), which was instructed to certify emissions control technology and require installation of certified technology. The prospect that other states would adopt stringent emissions standards sparked movement at the federal level. When Pennsylvania and New York introduced their own emissions regulation bills, industry representatives grew concerned that these new state bills might represent the first of a host of diverse standards. This led to support for the creation of a national standard with the condition that national standards be no more stringent than California's existing standards.[4] Similar events occurred with power plant emissions. Many observers conclude that statutes such as the CAA of 1970 were the outgrowth of laws which began a decade earlier at the state level.[5]

There is also a 20-year history of using emissions trading as a means of reducing pollutants while seeking to lower compliance costs. Clean air tradable emissions policies were first instituted in the early 1980s. In 1982 the U. S. Environmental Protection Agency (EPA) established a lead trading program to allow gasoline refiners greater flexibility during a period when the amount of lead in gasoline was being significantly reduced due to public health concerns. This program ended in 1997 after lead was phased out of gasoline products.[6] EPA also used trading systems to phase-down use of ozone depleting substances starting in 1988 by granting tradable permits based on 1986 production levels to U.S. producers. The annual supply of permits declined in compliance with the Montreal Protocol on ozone depleting emissions. All U.S. major producers and consumers of controlled substances were allocated baseline production or consumption allowances using 1986 levels as a basis; 100% of baseline was initially allowed, with smaller allowances being granted after deadlines; permits were transferable internationally to producers in other signatory nations; and inter-pollutant trading was possible within categories of pollutants.[7]

Title IV of the 1990 Clean Air Act Amendments (CAAA) established the Acid Rain Program. The program sought to cut acid rain by reducing sulfur dioxide emissions from electric generating plants to about half their 1980 levels beginning in 1995. This was a broad, market-based approach intended to help government and industry produce emissions reductions with greater flexibility at lower cost. The Acid Rain Program was the first large-scale, long-term environmental program relying on tradable emissions permits or *allowances*. There are four distinct elements to the acid rain policy:

- *Netting or internal trading:* Applies to new emissions sources in an individual firm. Allows firms that create a new source of emissions in a plant to avoid stringent emission limits by reducing emissions from another source in the plant. Administered at state level.
- *Offsets:* Applies to new emission sources in non-attainment areas. Specifies that new sources be allowed to locate in non-attainment areas if they offset their new emissions by reducing emissions from existing sources by even larger amounts. Offsets can be obtained through internal trading but also from other firms through *external trading*. Administered at federal level.
- *Bubbles:* Allows existing sources to aggregate the emission limits from individual sources of a pollutant in a plant and adjust the levels of

control applied to different sources as long as the aggregate limit is not exceeded. Administered at state and/or federal level (varies from state to state.)
- *Banking:* Allows existing sources to save emission reductions above and beyond permit requirements for future use in emissions trading. Administered at state level.

The Acid Rain Program was structured in phases. In Phase I, from 1995 to 1999, aggregate annual emissions from the 263 dirtiest large generating units were to be no greater than approximately 2.5 pounds of sulfur dioxide (SO_2) per million Btu of heat input. In Phase II, for 2000 and beyond, all existing and new fossil-fueled electric generating units in the continental U.S. become subject to a tighter cap on aggregate annual emissions. The cap is approximately nine million tons, or average emissions rate of less than 1.2 pounds of SO_2 per million Btu.[8]

By most measures, the Acid Rain Program has been a model for successful emission trading systems. Allowances in Phase I sold for approximately $100 per ton of SO_2, well under half of what had been forecast (about $250/ton.). Since 1994, allowances have cost $65 to $210 per ton as scrubber technology has become cheaper, and it has become less expensive to switch to low-sulfur coal. Sulfur dioxide emissions declined faster than anticipated and the market has now reached a value of $2 billion/year. Twenty-seven units added scrubbers accounting for 45% of the reductions in 1995–96. Seven large units accounted for two-thirds of this amount. Many units also switched fuels – almost all from high to lower sulfur coal.[9]

2. Recent Federal Clean Air and Energy Policies

In contrast to the history of federal policies to improve air quality, recent federal efforts have promoted traditional energy supply, such as additional power plant construction, particularly coal plants, without the framework of a comprehensive energy policy that could support a wider range of available tools, many with lower air emission problems, such as energy efficiency. The centerpiece of the Bush Administration's air and energy policies is highlighted in their National Energy Policy, and the Clear Skies Act (CSA) of 2005, which died in Congress in March after failing to pass out of committee. Both these policy efforts would have increased air quality problems compared to other approaches. The CSA would have covered emissions of Nitrogen Oxide (NO_x), SO_2 and mercury at 1,300 power plants in the country. The Clean Air Task Force in Boston (CATF) commissioned a comparison of these proposals using EPA's own power system cost modeling, emission dispersion modeling and cost-benefit methods, and employed consultants routinely retained by EPA to conduct the analysis. In all cases, the model assumptions were calibrated to run apples to apples comparisons with EPA's 2003 modeling of CSA. A comparison of CSA with other pending proposals in Congress shows its weaknesses (Table 10.1).

As the CATF testimony to Congress states:

> The results are instructive: *Each one of the competing proposals provides significantly greater health benefits than CSA and those additional benefits far outweigh the additional costs.* This analysis is very conservative because it completely ignores the added environmental benefits from the added acid rain reduction, added visibility gains, reduced nitrogen saturation, additional reduced mercury deposition and constraint of global warming pollution that CSA lacks.

Importantly, CSA would have reversed several major provisions of the Clean Air Act – including the New Source Review program (see NSR review on p.15) and the right of one state to sue another over windblown pollution. As the CATF concluded:

> CSA repeals or significantly weakens many provisions of existing law that have protected health and the environment since the enactment of the 1970, 1977, and 1990 Clean Air Act (CAA) amendments, including: Interstate air pollution protections; New Source Review requirements; Air Toxics controls applicable to the electric power industry; Provisions designed to bring air quality into attainment with national standards and to protect areas from air quality degradation; The deadlines by which states must attain national air quality standards; and Visibility protections for National Parks.[10]

A more effective approach to clean air policy, and one example of a sensible program being implemented by the federal government, is the Clean Air Interstate Rule or CAIR, intended to reduce emissions of particulate matter and NO_x in the eastern United States. EPA finalized the CAIR rules on March 10, 2005 in response to a finding of non-attainment of National Ambient Air Quality Standards (NAAQS) for fine particulate matter and ozone formation. CAIR requires reductions in SO_2 and NO_x emissions from coal-fired power plants; the emissions limits spelled out in the CAIR plan affect electric utilities in 28 eastern states and the District of Columbia. Emissions reductions are to be achieved through a market-based cap-and-trade system similar to other air pollution programs EPA has put into practice over the last few decades.

Under CAIR, states can achieve the required CAA emissions reductions (same emission reduction targets) using one of two options for compliance: (1) require power plants to participate in an EPA-administered interstate

Table 10.1. Comparison of major provisions of S.366 (Jeffords/Lieberman/Collins), S.843 (Carper, Gregg, Chafee), S. 485 (Bush Administration), EPA 2001 Proposal, and EPA Proposed Interstate Air Quality, Regional Haze, and Mercury MACT Rules

	Nitrogen Oxide (NO_x)	Sulfur Dioxide (SO_2)	Mercury (Hg)	Carbon Dioxide (CO_2)	New Source Review	Effect on other CAA Programs
Jeffords/ Lieberman/ Collins Clean Power Act S.366	1.51 million ton cap by 2009	2.255 million ton cap by 2009	5 ton cap by 2009. Each plant limited to 2.48 grams of mercury per 1000 megawatt hours, or less as determined by EPA	2.05 billion ton cap by 2009	No change to existing law	No changes to visibility or air toxics sections of existing law
Bush Clear Skies Act S.485	2.1 million ton cap by 2008 1.7 million ton cap by 2018	4.5 million ton cap by 2010 3 million ton cap by 2018	34 tons per year by 2010 (trading allowed) 15 tons per year by 2018. Sources can avoid emission reductions through mercury emissions credit trades	No limit on CO_2 emissions	Would practically eliminate new source review for new and existing power plants	Would eliminate visibility and interstate air pollution protections, delay attainment of NAAQS and repeal power plant air toxics controls
Carper/Greg Chafee Clean Air Planning Act S.843	1.87 million ton cap by 2009 1.7 million ton cap by 2013	4.5 million ton cap by 2009 3.5 million ton cap by 2013 2.25 million tons by 2016	24 tons by 2009 10 tons by 2013 Each unit must cut emissions to 50% of the mercury in delivered coal by 2009 and 70% of Hg in coal by 2013, or meet an alternative output emission rate. Limited mercury emission trading and banking is allowed	Power plant emissions capped at year 2006 level for calendar years 2009–2012 Power plant emissions capped at year 2001 level by 2013 and beyond	Retains NER for new plants, but eliminates offsets for new sources with reductions from other sources; also limits cost of new source controls Performance standards for all plants in 2020 of 4.5 lbs/mWh SO_2 and 2.5 lbs/mWh NO_X	Would eliminate the requirements for a Mercury MACT standard for power plants. Would grant a 20 year exemption from BART requirements in the visibility provisions of existing S.169A

table continues on next page

Table 10.1. (Cont.)

EPA 2001 Proposal	1.87 million ton cap by 2008 1.25 million ton cap by 2012	2 million ton cap by 2010	24 ton cap by 2008 7.5 ton cap and a 70% facility-specific reduction requirement by 2012	No limit on CO_2 emissions	Would repeal new source review for existing power plants	Would replace nearly every CAA program applicable to power plants except NAAQS
Proposed Interstate Air Quality Rule, Regional Haze/ Best Available Retrofit Technology (BART) Rule, and Mercury Rule	2.4 million ton cap by 2010 2.1 million ton cap by 2015	4.6 million ton cap by 2010 3.5 million ton cap by 2015	MACT proposal: 34 tons by 2008 Section 112(n) trading alternative: 34 tons by 2010 Section 111 cap and trade alternative: 34 tons by 2010 and 15 tons by 2018	No limit on CO_2 emissions	Final rule may include NSR rollbacks	N/A

cap-and-trade system that caps emissions in two stages, or (2) meet an individual state air emission limits through measures of the state's choosing.[11]

According to EPA's Integrated Planning Model (IPM) Modeling results:

- In 2010, CAIR will reduce SO_2 emissions by 4.3 million tons – 45% lower than 2003 levels, across states covered by the rule. By 2015, CAIR will reduce SO_2 emissions by 5.4 million tons, or 57%, from 2003 levels in these states. At full implementation, CAIR will reduce power plant SO_2 emissions in affected states to just 2.5 million tons, 73% below 2003 emissions levels.
- CAIR also will achieve significant NO_x reductions across states covered by the rule. In 2009, CAIR will reduce NOx emissions by 1.7 million tons or 53% from 2003 levels. In 2015, CAIR will reduce power plant NO_x emissions by 2 million tons, achieving a regional emissions level of 1.3 million tons, a 61% reduction from 2003 levels.
- In 1990, national SO_2 emissions from power plants were 15.7 million tons compared to 3.5 million tons that will be achieved with CAIR. In 1990, national NO_x emissions from power plants were 6.7 million tons, compared to 2.2 million tons that will be achieved with CAIR.
- CAIR is also projected by 2015 to prevent 17,000 deaths, 700,000 acute bronchitis and asthma attacks and 2.2 million absences from work and school; and produce more than $20 in benefits for every dollar spent.[12]

Although CAIR is a favorable development in clean air regulation, EPA's recently announced mercury regulations represent a step backwards on existing air toxic provisions of the Clean Air Act. Each year, uncontrolled coal-fired power plants in the United States emit nearly 50 tons of mercury to the air in addition to an estimated 33 tons disposed of in the waste left over after power plants burn coal. In 1997, the U.S. EPA estimated that coal-fired power plants accounted for about 33% of all U.S. emissions, with municipal, medical and hazardous waste combustors accounting for another 33% combined.

The EPA has already issued regulatory requirements for municipal waste combustors, medical waste incinerators and hazardous waste combustors. Yet coal fired power plants not only remain uncontrolled, they account for a larger and larger share of mercury emissions as other source categories meet their obligations to reduce their mercury releases.[13] Under the Clean Air Act, EPA was required to finalize a "maximum achievable control technology" (MACT) rule for mercury emissions from coal-fired power plants by December 20, 2002. Clean Air Act § 7412(n)1)(A) provides that EPA shall issue MACT standards for power plants if the Administrator determines, after a study, that such standards are "necessary and appropriate." The Administrator made this determination for power plant emissions of mercury in 2000.[14] Pursuant to a consent decree in *NRDC v. EPA, et al.*, Case No. 92-14[15] (D.C. Circuit), EPA was required to propose a MACT standard for mercury by the end of 2003, and to promulgate the standard by the end of 2004.

In March 2005, the Bush Administration released its Clean Air Mercury Rule. The agency maintains that controls mandated under CAIR will have the *co-benefit of* reducing mercury emissions to the levels set for 2010. The rule will require coal-burning plants producing more than 25 megawatts of electricity to reduce mercury emissions. The rule will be implemented in two phases: 20% reductions by 2010 and 70% by 2018, based on 1999 levels. Under a cap-and-trade program the rule allows some plants or utilities to reduce mercury emissions by more than 70% and others to do less as long as the overall mandates are met.15 In effect, the Clean Air Mercury Rule will result in weaker controls and longer periods of emissions of one of the most toxic metals in the environment. The plan will delay even modest mercury reductions by a decade or more; in fact, the Administration's own modeling shows that its goal of a 70% reduction will not be met by the 2018 deadline, but may be delayed to 2030. The rule will rescind a prior determination that power plants must be regulated according to the MACT standards and instead proposes a far weaker standard. Under this approach, power plants would no longer be required to limit emissions from each and every plant to the maximum extent possible. Instead, some plants would be able to avoid making reductions by buying credits from other plants, significantly increasing the likelihood and severity of *hot spots,* or communities where mercury deposition is more prevalent. The rule will delay reductions by a decade and set targets so weak that the industry will be allowed to emit six to seven times more mercury between 2010 and 2018 and three times more mercury after 2018.[16]

3. Federal Energy Policy

The choices we make on energy policy will affect the country for many years to come. These choices will affect what kinds of power plants are built; whether energy efficiency and clean power choices will have the opportunity to demonstrate their potential; and, of course, air quality. For those concerned with setting the country on a course of sustainable, more balanced energy policies, the last four years have seen proposals that fail to proceed with policies that will wean the United States from its dependency on foreign oil, undermining efforts to make the United States more energy independent and energy secure. Soon after taking office, the Bush Administration led by Vice President Dick Cheney presided over a task force charged with setting a new national energy policy. For months high-ranking administration officials met with lobbyists and executives from utility companies and the oil, gas, coal and nuclear energy industries. Not surprisingly, the final task force recommendations released in May 2001 include increased reliance on fossil fuels and nuclear power.

4. Content of Bush Energy Plan[17]:

4.1. Electric Power

The National Energy Plan calls for a reversal of the Clean Air Act's New Source Review program (targeted in Clear Skies); assumes the need to build at least 1,300 new power plants, mostly fueled by coal and natural gas, over the next 20 years; excludes carbon dioxide from a "three-pollutant" power plant emissions policy; and interferes with pending Clean Air Act enforcement cases by ordering a Justice Department "review."

4.2. Oil Drilling in Pristine Wilderness Areas

The Plan calls for drilling in pristine areas such as the Arctic National Wildlife Refuge and the Outer Continental Shelf (OCS) despite research from the U.S. Geological Survey (USGS) determining that the refuge's coastal plain likely contains only 3.2-billion barrels of oil that could be economically recovered and brought to market, assuming a price of $20 per barrel. Even at $40 per barrel USGS estimates there would be only 6.7 billion barrels that could be profitably brought to market, still less than the 7.3 billion barrels we consume every year.[18] Senator Olympia Snowe notes that it could take up to 50 years to extract it all, and during that time, the oil would satisfy only 1% of projected U.S. demand.[19] Additionally, the Plan calls for a review of statutes, regulations and executive orders pertaining to Outer Continental Shelf activities, and recommends that the interior secretary examine impediments to federal oil and gas leasing on public lands, which includes offshore areas. This recommendation sets the stage for lifting the OCS moratoria that protects the East and West Coasts, Alaska's Bristol Bay, and most of the Eastern Gulf of Mexico off Florida from new leasing, and threatens the ability of states to object to offshore oil and gas development off their shores and to control the siting of energy facilities on their coasts. On-shore the Plan also calls for accelerated leasing and development of oil, gas, and coal reserves on public lands, and drilling for oil in roadless areas.

4.3. Nuclear Energy

The Plan recommends the expansion of nuclear energy; proposes an initiative to reexamine reprocessing of nuclear fuel; and calls for renewal of the Price-Anderson Act limiting the nuclear industry's liability for accidents.

4.4. Motor Vehicles

The Plan delays any action to close the *SUV loophole* or raise fuel efficiency standards; proposes a tax credit for hybrid-electric and fuel cell vehicles; and takes no action to improve public transit or support *smart growth* development that would reduce the need to drive.

4.5. Renewable Energy

The Plan proposes extending the existing production tax credit for wind, expanding the tax credit for biomass, and creating an investment tax credit for solar systems; directs the secretary of energy to review the renewable energy research and development budget, following proposed budget cuts of nearly 50% for most renewable technologies; fails to propose a renewable portfolio standard that would ensure a steady expansion of the share of electricity generated by renewable energy resources; and fails to break down market barriers to distributed renewable energy systems through interconnection standards and net metering.[20]

The energy bills introduced in the U.S. Congress in recent sessions adopt many of these policies. The Energy Policy Act of 2004 (H.R. 4503, based on the energy bill conference report H.R.6) passed in the House in June 2004, and S. 2095 (which includes revisions to H.R. 6) was reintroduced in the Senate in February 2004. S. 2095 exempts oil and gas construction activities from having to obtain permits for polluted storm water runoff required by the Clean Water Act; includes large incentives for coal, oil, and gas (Titles III, IV, VII); does not provide for standards for renewable energy sources (Title XII); has no global warming provisions; creates incentives for expanded offshore oil and gas drilling (Title XIV Subtitle B); provides millions in taxpayer funds to uranium companies supporting polluting mining practices that threaten drinking water aquifers (Title VI Sec. 631); and provides tax breaks and subsidies to big energy companies. The Congressional

Budget Office estimates that the price tag (based on the 2003 version of the bill) exceeds $50 billion over the next decade. The bill would extend the Price Anderson Act's limits on liability for nuclear plant operators for 20 years.[21]

In addition to legislative and policy proposals, the Bush Administration has taken agency action against efforts to implement common-sense energy efficiency measures. For example, one of the first actions that the Administration took when it came into office was to turn back an effort to establish a new efficiency standard for air conditioners. The SEER 13 standard (Seasonal Energy Efficiency Ratio) was first promulgated in 2001 near the end of the Clinton Administration after a seven-year review. SEER 13 has the effect of decreasing energy use by new air conditioners by 23%. According to an analysis by the American Council for an Energy Efficient Economy (ACEEE), this will reduce the peak demand for electric power by 41,500 Megawatts by 2020 (equivalent to 138 typical new power plants of 300 MW each) and save consumers approximately $5 billion over the 2006–2030 period. It will also reduce air pollutant and greenhouse gas emissions, saving 7.2 million metric tons of carbon in 2020, which is equivalent to taking more than 3 million vehicles off the road. Notwithstanding these benefits, the Administration began discussing a roll back in the SEER 13 standard as early as April 2001, and in 2002 reduced the SEER 13 standard to SEER 12. A lawsuit led to the restoration of the SEER 13 standard in January 2004 in a decision by the U.S. Court of Appeals for the Second Circuit brought by northeastern state Attorneys General and environmental groups. In 2004, the Air Conditioning and Refrigeration Institute (ARI – the industry trade association) announced it was withdrawing its appeal of the SEER 13 standard (filed in 2001 in the 4th Circuit of the U.S. Court of Appeals), finalizing the SEER 13 standard.[22]

5. State Policy: Aggressive Activity in Lieu of Federal Action

In contrast to activity on the federal level, states have stepped up their efforts to utilize both federal and state law to address air pollution.

5.1. Air Quality

States have forged ahead on their own to limit emissions from power plants. Many of the following laws were passed during the last four years as part of a regional power plant emissions reduction campaign.[23]

5.1.1. Massachusetts

Massachusetts (2001) passed four-pollutant "Emissions Standards for Power Plants" (310 CMR 7.29) to reduce emissions of SO_2, NO_x, Hg and CO_2 from older fossil-fuel fired power plants in Massachusetts. The standards will require six power plants to make 10% reduction from 1997–1999 CO_2 levels. Power plants have the choice between switching fuels, changing generation technologies or trading emissions-reductions credits with other plants. They can also invest in certified offset projects.[24]

5.1.2. Connecticut

In 2002, PA 02-64, "An Act Concerning Reducing Sulfur Dioxide Emissions at Power Plants" was enacted requiring the six dirtiest power plants and others to adhere to the same pollution restrictions for sulfur dioxide emissions as newer power plants.[25]

5.1.3. New Hampshire

The "Clean Power Act" of 2002, RSA 125-0, was the first multi-pollutant legislation passed in the country. This bill establishes caps for emissions of sulfur dioxide, oxides of nitrogen, and carbon dioxide from existing fossil fuel-burning steam electric power plants. It permits the banking and trading of emissions reductions to achieve compliance with the caps. Compliance is not required of a plant that installs qualifying repowering technology or an eligible replacement unit. The Act is part of "New Hampshire's Clean Power Strategy" wherein the state's three fossil-fueled power plants will have a five-year window to reduce emissions of sulfur dioxide by 75%; nitrogen oxides by 70%; mercury by 75%; and carbon dioxide by 7% below 1990 levels.[26]

5.1.4. Rhode Island

H. 5201, passed in 2003, urges the federal government to block the implementation of EPA rules which will weaken the New Source Review provision of the Clean Air Act; the bill also takes action to further restrict emissions from power plants.[27]

5.1.5. North Carolina

The 2002 General Assembly passed the "Clean Smokestacks Bill" (SB 1078) a multi-pollutant strategy that requires a 77% reduction in utility NO_x emissions by 2009, year-round. The bill also would require substantial reductions in SO_2 emissions, the primary cause of haze, acid rain and fine particles. Power plants would have to cut their SO_2 emissions about 50% by 2009 and 73% by 2013. Finally, the bill calls for an examination of mercury and carbon dioxide emissions in North Carolina.[28]

States have also been active in supplementing EPA enforcement actions under the Clean Air Act by joining in or initiating a range of lawsuits. For example, the New Source Review program (NSR) in the Clean Air Act, was designed to reduce air pollution from industrial facilities by requiring them to install up-to-date pollution controls when they make *major* physical or operational changes that increase air pollution. The provision applies to an estimated 17,000 power plants, refineries, and other industrial facilities. Ever since the NSR program went into effect industry has fought tooth and nail against the requirements, filing lawsuits to challenge the program and refusing to comply. At one point EPA estimated that approximately 80% of refineries were out of compliance with the law.

During the 1990s, EPA launched a series of enforcement lawsuits against violators for pollution increases that resulted in millions of tons of air pollution. A few of these cases were successful yet most of the legal challenges were stalled when EPA revised its NSR program in 2003. The revisions create a loophole that would allow more than 20,000 power plants, refineries and other industrial facilities to replace existing equipment with "functionally equivalent" equipment without undergoing the clean air reviews. This exemption would apply if the cost of the replacement does not exceed 20% of that of the entire "process unit" even if a facilities' air pollution increased by tens of thousands of tons as a result of the replacement. The weakened rules also allow sources to evade review by claiming that emissions increases result from growth in market demand.[29] In October 2003, a coalition of 12 states, the District of Columbia, local governments, and environmental groups sued EPA to block proposed changes to the New Source Review provisions. The new rules would have gone into effect in December 2004; however, in a case brought solely by environmental groups the U.S. Court of Appeals granted a motion to stay implementation of the new rule. The court will not lift the stay unless it ultimately decides the administration's rule change is legal.[30]

Attorneys General have brought NSR suits directly against power generators. New York AG Eliot Spitzer began a set of lawsuits in September 1999. His office sued eight utilities that operate 17 Midwest power plants and achieved settlements with several lawsuits affecting both out of state (Ohio Edison ruling in 2003) and in-state power plants.[31] In the New York settlements announced on January 11, 2005, Governor Pataki and AG Spitzer announced two NSR major settlements that will reduce emissions from six upstate coal-fired power plants, including the state's two largest coal plants. These settlements with four power plant owners, NRG, AES, New York State Electric and Gas Co., and Niagara Mohawk, will reduce NO_x emissions by more than 18,000 tons annually and SO_2 emissions by more than 123,000 tons/year. The original case was filed in 2002 charging that plants had made major modifications as defined in the CAA's NSR requirements.[32]

6. Global Warming and Energy Policy: State Progress and Innovation

The lack of federal action is most notable on the issue of global warming, where U.S. policy is to deny the clear evidence supported by a consensus of the scientific community and most nations around the world that human sources are greatly contributing to well-documented warming trends. A growing number of states have moved forward with legislation or other means of addressing greenhouse gas (GHG) emissions. According to the Pew Climate Center, 28 states have developed programs for reducing GHG emissions and 39 states have established some form of GHG inventories.

In June 2003, Maine became the first state in the country to pass a law that sets a statewide target for reducing greenhouse gas emissions. The law, called "An Act to Provide Leadership in Addressing the Threat of Climate Change," requires the state to develop a climate action plan to reduce greenhouse gas emissions to 1990 levels by 2010, 10% below 1990 levels by 2020, and a long-term target of 75 to 80% below 2003 levels. These goals reflect the targets of the Climate Action Plan endorsed by the Conference of New England Governors and Eastern Canadian Premiers (NEG-ECP). The NEG-ECP plan, released in Mystic, CT in 2001, calls on the six New England states and five eastern Canadian provinces to work towards the three sets of emission reduction goals summarized above.[33] Connecticut has also incorporated these targets into state law in 2004 in "An Act Concerning Climate Change," PA 04-252. The Connecticut law included a provision setting 2050 as the target date for the deep reduction goal if there is no other consensus achieved in the region.

Maine, Connecticut, Massachusetts and Rhode Island each have developed state climate change action plans. California, Oregon and Washington are working together in the West Coast Governor's Global Warming Initiative of November 2004 on a set of specific actions to reduce GHG emissions from diesel engines, remove barriers to renewable energy development, promote energy efficiency and establish a purchase pool for high efficient vehicles. Following the New England states model, Arizona, New Mexico and Wisconsin are gearing up to embark on state climate action planning processes.

California passed legislation in 2002 (AB 1493, or sometimes referred to as the "Pavley" bill after CA Representative Fran Pavely who championed the bill) making it the first state in the nation to regulate CO_2 emissions from motor vehicles. CARB adopted regulations that achieve maximum feasible and cost-effective reduction of GHG emissions from passenger cars and light trucks sold in CA. Regulations apply to 2009 and later model year vehicles and require an average of 30% reductions of GHG emissions by 2016.

On the regional level, building off the success of other clean air cap-and-trade programs, eleven states are participating in the Regional Greenhouse Gas Initiative (RGGI) to design a cap-and-trade program for CO_2 emissions from power plants in the northeastern states. This rule would be the first in the country regional cap-and-trade program for CO_2 and is being monitored by other regions in the country, most notably the West Coast states. Potentially, the RGGI rule could set the stage for a national program[34].

States have also sought to use the courts to spur federal action on global warming. In October 2003, 12 states, 14 environmental groups, and two cities filed suit in federal court to appeal EPA's decision to reject a petition from environmentalists that sought to have the federal government regulate GHG emissions from new motor vehicles. In rejecting the petition, EPA concluded that the CAA does not give it authority to regulate GHGs. This suit could set a precedent for the federal government's role in regulating greenhouse gases under current laws.[35]

States are choosing to pursue a more sustainable energy policy – one that provides incentives, market mechanisms and mandates to develop greater percentages of renewable energy and obtain the benefits of cost-effective energy efficiency programs. For example, states are moving forward in setting energy efficiency standards for certain appliances and equipment that are not federally regulated. In 2004, Connecticut joined California and Maryland to become the first New England state to pass legislation—with unanimous bipartisan support—setting minimum energy efficiency standards for eight different appliances and products sold in state. Over a five-year period, Connecticut consumers and businesses will save $40 million in reduced electricity bills – while cutting annual greenhouse gas emissions by an amount equivalent to removing 50,000 cars from the road. New Jersey has since adopted similar legislation and other states in the northeast are now following Connecticut's lead.

Some 18 states, including Texas, and the District of Columbia have Renewable Portfolio Standard requirements in state law that require minimum purchases of qualifying clean energy to be obtained by electric suppliers. Indeed the Texas law, enacted in 1999, is one of the first and most successful in the country. It requires 2,000 megawatts (MW) of new renewable energy capacity by 2009. The state is ahead of its annual requirement schedule with nearly 1,200 MW of new renewable energy already installed. Colorado held the first statewide referendum on the "The Renewable Energy Initiative," which passed by a vote of 53% yes to 47% no. The initiative requires Colorado's top utility companies to procure a percentage of their retail electricity sales from renewable resources beginning with 3% by 2007, 6% by 2011 and increasing to 10% by 2015.[36]

Many states such as Connecticut, Massachusetts, Rhode Island, Maine, New York, Wisconsin and California support active energy conservation funds that provide incentives for homeowners and businesses to invest in more efficient equipment and reduce consumer energy bills while avoiding powerplant emissions. Fifteen states have funds that invest in renewable energy, funds that will total $4 billion by 2017.[37] Other states such as Oregon, Massachusetts, Rhode Island and New York, field green power options to utility customers; Connecticut's new Clean Energy Option will be offered soon in the state.

These examples demonstrate that as the federal government is largely preoccupied with expanding use of fossil fuels and nuclear power and ignoring the threat and challenges posed by global warming, states—both red and blue—are choosing a different path, incorporating clean power and energy efficiency and electing to proceed with climate change action plans. Innovation in sound policy development is clearly at the state level, policies that will hopefully set the stage for a reinvigorated federal approach in the not too distant future.

Notes

1. This summary relies on research conducted by Heather Kaplan, policy analyst at Environment Northeast, and materials prepared by Conrad Schneider and David Marshall of the Clean Air Task Force.
2. President Eisenhower was the first president to call for a review of the public health impacts of motor vehicle emissions in the early 1960s by signing the Schneck Act into law.
3. EPA Oral Interview-1: William D. Ruckelshaus, interview conducted by Michael Gorn (Washington, D.C.: GPO), pp. 10ff.
4. Krier, J. and Ursin, E., *Pollution and Policy,* (Berkley: University of California Press, 1977) 53.
5. "The story behind the Air Quality Act of 1967 is complicated, but here too the threat of state and local regulation provided the impetus for a crucial

industry to acquiesce in federal legislation ... During the mid-1960s, the soft coal industry faced increasingly strict air pollution regulations in the Northeast, which eventually threatened it with the loss of a major market." Elliott, Donald E. et al. "Toward a Theory of Statutory Evolution: The Federalization of Environmental Law," *Journal of Law, Economics, and Organization,* **1:2** (1985), 326.

6. *See* Stavins, R.N. 2000. "What Can We Learn from the Grand Policy Experiment? Lessons from SO2 Allowance Trading," in *Economics of the Environment,* 435.
7. Note: simultaneous application of permit and tax systems: EPA allocated allowances to seven major domestic producers of CFCs and halons, but imposed tax on production to *soak up* the rents created by the regulation induced scarcity (*Source:* Tietenberg, T. 2000. *Environmental and Natural Resource Economics,* 5th ed).
8. Environmental Protection Agency, *The Plain English Guide to the Clean Air Act. April 1993,* EPA-400-K-93-001.
9. For more information see Ellerman, D.A. Harrison, D. May 2003. "Emissions Trading in the U.S.: Experience, Lessons, and Considerations for Greenhouse Gases," Pew Center on Global Climate Change; Percival, R. et al. 2000. *Environmental Regulation: Law, Science, and Policy,* Third Edition.
10. *See* Testimony of Conrad G. Schneider, Advocacy Director, Clean Air Task Force, Senate Subcommittee on Clean Air, Climate Change, and Nuclear Safety, January 26, 2005. Hearings on the Need for Multi-Pollutant Legislation; www.catf.us.
11. *See* 40 CFR Parts 51, 72, 73, 74, 77, 78, and 96. Clean Air Interstate Rule. March 10, 2005.
12. *See* http://www.epa.gov/air/interstateairquality/pdfs/cair_final_reg.pdf.
13. *See* Keating, M. February 2003. "Mercury and Midwest Power Plants." *Clean Air Task Force.* http://www.catf.us/publications/reports/Midwest_Mercury.pdf.
14. 65 Fed. Reg. 79, 825 (December 20, 2000).
15. 40 CFR Parts 60, 63, 72, and 75. Clean Air Mercury Rule. March 15, 2005.
16. *See:* Testimony of David G. Hawkins, Director, NRDC Climate Center, Hearings on S. 385, "Clear Skies Act of 2003", U.S. Senate Committee on Environment & Public Works, Subcommittee on Clean Air, Climate Change, and Nuclear Safety, April 8, 2003; National Environmental Trust: http://www.cleartheair.org/mercury/mercuryhurts.
17. Report of the National Energy Policy Development Group. May 2001. "National Energy Policy." ISBN 0-16-050814-2.
18. U. S. Department of the Interior, U. S. Geological Survey. 2001. Arctic National Wildlife Refuge, 1002 Area, petroleum assessment, 1998, including economic analysis. USGS Fact Sheet FS-028-01 (Supersedes FS-040-98) April 2001.

19. The Free Press (Rockland, ME), March 24, 2005 at 2.
20. See "Slower, Costlier, and Dirtier: A Critique of the Bush Energy Plan." NRDC. May 2001. http://www.nrdc.org/air/energy/scd/scd.pdf.
21. *See* "Summary of provisions that are harmful to public health and the environment, public lands and coastal areas, consumers and taxpayers, and energy and national security." *Earthjustice,* last updated November 18, 2003. http://www.earthjustice.org/backgrounder/documents/Energy_bill_problems.pdf.
22. *See* www.aceee.org/preee/0403.htm.
23. *See* Pew Climate Center, "State and Local Net Greenhouse Gas Emission Reduction Programs," http://www.pewclimate.org/states.cfm. Site visited 03/14/05.
24. *See* 310 CMR 7.29, http://www.mass.gov/dep/bwp/daqc/files/regs/729final.doc.
25. *See* Public Act No.02-64, http://www.cga.ct.gov/2002/act/Pa/2002PA-00064-R00HB-05209-PA.htm.
26. *See* Chapter 125-0, Multiple Pollutant Reduction Program, http://www.gencourt.state.nh.us/rsa/html/indexes/125-O.html.
27. *See* H.5201, http://www.rilin.state.ri.us/billtext/billtext03/housetext03/h5201.htm.
28. *See* SB 1078, Clean Smokestacks Act, http://daq.state.nc.us/news/leg/stackfacts.shtml.
29. Turner, T. November 2003. "Background: New Source Review." www.earthjustice.org/background.
30. http://www.oag.state.ny.us/press/2003/oct/rmrr_rule_petition_for_review.pdf.
31. For more information see http://www.oag.state.ny.us/press/2005/jan/jan11c_05.html.
32. Text of settlements: http://www.oag.state.ny.us/press/2005/jan/jan11c_05_attach1.pdf. http://www.oag.state.ny.us/press/2005/jan/jan11c_05_attach2.pdf. http://www.oag.state.ny.us/press/2005/jan/jan11c_05_ attach3.pdf.
33. *See* New England Governors and Easter Canadian Premiers "Climate Change Action Plan." August 2001. http://www.negc.org/documents/NEG-ECP%20CCAP.PDF.
34. The world's first regional cap and trade program for greenhouse gases went into effect in January 2005, the EU's Emissions Trading Scheme (ETS) to help facilitate compliance with Kyoto targets. For more information on RGGI *see* http://www.rggi.org/docs/rggi_flowchart.pdf.
35. Commonwealth of Massachusetts, et al. vs. Environmental Protection Agency. United States Court of Appeals for the District of Columbia Circuit, October 2003.
36. Source: Pew Climate Center, "State and Local Net Greenhouse Gas Emission Reduction Programs," http://www.pewclimate.org/states.cfm.
37. *See* summary information by the Union of Concerned Scientists, http://www.ucsusa.org/clean_energy/index.cfm.

Part III

Sulfur Dioxide and the Market

11. Market-Based Approaches to Environmental Policy: A "Refresher" Course*

Paul R. Portney

Upon hearing the term market-based approaches to (or economic incentives for) environmental protection, some people assume this means letting unfettered competition between unregulated private firms determine how clean our air or water will be, how much open space we will have, or how many fish stocks will be driven to collapse.

Nothing of the sort is intended. In fact, market-based approaches to environmental protection are a clever form of government regulation. They are premised on the recognition that while competitive markets are a wonderfully efficient means of deciding what types and quantities of consumer goods should be produced, they generally fail with respect to environmental quality, the provision of *public goods* like open space and common-property resources like fisheries. Every undergraduate and graduate economics textbook discusses this notion of *market failure,* and the environment is always the first illustration that is used.

Given the very necessary government role in protecting the environment, the real question becomes how best to do this. Market-based approaches to environmental protection are premised on the idea that it is possible to confront private firms, individuals, and even other levels of government with the same kinds of incentives they face in markets for labor, capital, and raw materials—that is, prices that force them to economize. The rationale for market-based approaches, in other words, is to try to put the powerful advantages of markets to work in service to the environment.

1. Command-and-Control Era

To paint a quick picture of traditional regulation, consider the case of air and water pollution control. Prior to the early 1970s, the regulation of air and water pollution was almost exclusively the responsibility of state and

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local governments. In fact, the Clean Air Act amendments of 1970 and the Federal Water Pollution Control Act Amendments of 1972 marked the first really substantial federal involvement in environmental protection.

Under the Clean Air Act, the federal government (in the form of the then-new U.S. Environmental Protection Agency, or EPA) began specifying the pollution-control equipment that any new plant had to embody. In addition, EPA required local areas to formulate plans to reduce pollution from existing sources so that the air quality standards that EPA began issuing would be met. These plans typically required large, privately owned industrial facilities to reduce their pollution the most, and often required other sources to roll back their pollution by uniform amounts. Both new and old facilities had to apply for and receive operating permits from EPA that specified allowable emissions. In addition, the federal government also began limiting for the first time the tailpipe emissions of new cars rolling off the assembly lines of both domestic and foreign manufacturers. While the emerging water pollution regulations differed somewhat, at their heart, too, were a series of technological requirements for both newly constructed and existing plants, coupled with mandatory permits that specified allowable emissions.

Despite protests to the contrary, both programs have had significant successes, most notably in the case of the Clean Air Act. Since 1970, air quality around the United States has improved dramatically in almost every metropolitan area and for almost every air pollutant. For one notable example, airborne concentrations of lead, an especially insidious threat to health, were 93% lower in 2000 than they were in 1980. Success under the Clean Water Act has been less dramatic, though quite obvious in many places. Rivers that 30 years ago had almost ceased to support aquatic life have seen fish strongly rebound (even if it is still inadvisable to eat the fish one catches in some places).

Despite these successes, by the late 1980s dissatisfaction with the technology-based standards approach had become rampant. First, by requiring sources of air and water pollution control to meet emissions standards keyed to a particular type of technology, many regulations had effectively frozen pollution control technology in place. No one had an incentive to invent a more effective and/or less expensive pollution control technology as long as some other technology had received EPA's blessing. Second, by requiring regulated firms to have specific types of pollution control in place, they were denied the flexibility to modify their production process or reformulate their product(s) in such a way as to reduce their emissions because they would still be required to use whatever technology was applicable. Finally, it was becoming clear that the technology-based, command-and-control system was overly expensive. Study after study showed that it would be possible to meet the same environmental goals—either in terms of ambient air quality or in terms of emissions from affected sources—for much less money than the current approach was costing.

2. Cap and Trade vs. Pollution Taxes

There are two principal market-based approaches to environmental protection, both of which owe much of their popularity today to a small group of economists, most notably the late Allen Kneese of Resources for the Future (RFF). While mirror images of one another in many important respects, one market-based approach looks not unlike the current regulatory system while the other appears to be a more radical departure. The more familiar-looking approach to air or water pollution control would still be based on a system of required emissions permits. Under this approach—generally referred to as a cap-and-trade system—each pollution source is given an initial emissions limitation. It can elect to meet this limit any way it sees fit: rather than being required to install specific types of control technology, the source can reduce its pollution through energy conservation, product or process reformation (including substitution of cleaner fuels), end-of-pipe pollution control, or any other means. Importantly, and not surprisingly, each source will elect to reduce its pollution using the least expensive approach available to it.

More surprisingly, a source has one additional option under the cap-and-trade system: it can elect to discharge *more* than it is required so long as it buys at least equivalent emissions reductions from one or more of the other sources of that pollutant. All that matters is that the total amount of emissions reductions that take place from all sources are equal to the initial cap established by EPA (or another regulatory authority). Those sources that will elect to make significant emissions reductions under this system are precisely those that can do so inexpensively; likewise, those that elect to buy emissions reductions from other sources rather than cut back themselves will be those that find it very expensive to reduce. (This is the analogue to Adam Smith's famous *invisible hand* that steers producers and consumers to the most efficient allocation of resources.) Moreover, all sources have a continuing incentive to reduce their pollution—the more a source's emissions fall short of its limitation, the more emissions permits it will have to sell to other sources.

The flip side of this approach is one in which *no* limits are placed on each ton of pollution that a source emits, but in which each ton is taxed. Pollution taxes are paid to the government, which is then free to use the revenues as it sees fit—to reduce other taxes, spend on pollution control R&D, reduce the national debt, etc. While appearing very different than the cap-and-trade approach, this system creates the very same set of incentives. That is, the firms that can reduce their pollution inexpensively will invest in doing so because each unit of pollution reduced is that much less paid in pollution taxes. Firms that find it very expensive to reduce their pollution will continue to discharge and pay the taxes; note, however, the strong and continuing incentive the latter have to find ways to cut their emissions—and the higher the taxes on pollution, the stronger that incentive. Also, both a cap-and-trade system and a pollution tax create the same incentive to reduce pollution that the wage rate creates for firms to minimize the amount of labor they use or that the interest rate has in disciplining firms' borrowing.

The cap-and-trade approach began to be implemented in a small-scale way in the late 1970s and early 1980s in both Democratic and Republican administrations. But the first really large-scale application of cap-and-trade-which resulted in the most significant environmental policy success since 1970—came in the 1990 amendments to the Clean Air Act. In order to reduce emissions of sulfur dioxide by 50% in the eastern half of the United States, an ambitious cap-and-trade system was created under which more than 100 large coal-fired power plants were given initial emissions reductions. These plants could meet their emissions reductions targets themselves, through any means they selected, including shifting from high- to low-sulfur coal. However, the affected plants were also given the ability to purchase excess emissions reductions generated by other plants that found it easy to reduce their sulfur dioxide.

This approach has resulted in reductions in sulfur dioxide emissions that have been both larger and faster than required by the law. Moreover, the annual savings to electricity ratepayers nationally (compared to the previous command-and-control approach) range from 50–80% and these savings amount to $1–6 billion annually, depending on whose estimates one wants to use. As a result of this success, cap-and-trade approaches are now being proposed for additional reductions of sulfur dioxide, nitrogen oxides and mercury under the Bush administration's Clear Skies Initiative. They have also been put forward by former EPA Administrator Christie Todd Whitman for reducing water pollution in certain watersheds, by state and local governments seeking smog reductions, and by foreign governments exploring lower-cost approaches to a variety of environmental problems. The European Union has just announced that it will use a cap-and trade system to control carbon dioxide as it struggles to comply with the terms of the Kyoto Protocol, which is still alive in Europe.

3. Uncertainties Created by Each System

Large-scale experiments with pollution taxes are harder to find in the United States. Under the 1987 Montreal Protocol to phase-out worldwide use of chlorofluorocarbons (CFCs) and other ozone-depleting substances, a tax was levied on CFC production during the time mandatory phase-out was taking place, although this is clearly a hybrid system under which command-and-control regulation was augmented by a pollution tax. The evidence to date suggests that this hybrid approach is working well—CFC emissions have fallen and early evidence is that the stratospheric ozone hole has stopped growing.

Interestingly, perhaps the most ambitious application of pollution taxes is occurring not at the federal or even state levels of government, but at the local level. Hundreds of communities around the United States have adopted pay-as-you-throw systems for household garbage collection. Rather than charge every household the same amount for refuse collection, these communities are charging households a fixed amount per bag of garbage collected at curbside. This has had the effect of reducing the amount of yard wastes that end up in municipal landfills (households are composting more) and possibly even changing households' purchasing decisions toward products which come with less packaging.

Why have cap-and-trade policies flourished in comparison to pollution taxes in the United States? Perhaps most obviously, a system in which discharge permits are issued, but made saleable, looks rather like the regulatory system currently in place in the United States, with the added twist of marketability. Another reason has to do with the uncertainty each system creates. Specifically, under a cap-and-trade system, the total amount of pollution is firmly fixed-that is the purpose of the cap. What is uncertain are exactly where the emissions will occur (this depends upon who trades with whom), and how much an emissions permit (the right to emit one ton in a given year, say) will cost—the latter is determined in a competitive market.

Under a pollution tax, sources are allowed to discharge as much as they want, as long as they pay the per unit charge for each ton emitted. Thus, there is uncertainty about the total amount of pollution discharged (though we can be sure that the higher the tax, the lower the amount of pollution discharged). There is no uncertainty under the latter system about the maximum amount it will cost to reduce a ton of pollution, though, because that will not exceed the per-ton tax. The total amount of revenue raised by such a system is not predictable, because if sources can reduce their emissions less expensively than is believed to be the case, they will discharge less to avoid the tax. In years past, environmentalists objected to pollution taxes on the grounds that sources faced no pollution limits at all and could continue to pollute as long as they paid the corresponding taxes. Note, however, that this approach makes sources pay for every single unit of pollution that they discharge—unlike the command-and-control system in which firms are given considerable amounts of *free* emissions in the form of any discharges they may make so long as they are beneath their permitted levels.

The choice between cap-and-trade systems and pollution taxes rests at least in part on the pollutant in question. For pollutants like sulfur dioxide, CFCs, or carbon dioxide that mix equally in the atmosphere and that pose few or no local health effects, cap-and-trade works well because we are unconcerned about where emissions take place. On the other hand, if we are concerned that limiting emissions might impose too big a burden on the economy, the pollution tax approach is best because sources know that they will never have to pay more for a ton of pollution discharged than the tax. Effluent charges also raise revenue—not a trivial issue in many places, including developing countries.

One thing is for sure. Market-based approaches to environmental protection have become the default option in much of modern environmental policy, both in the United States and abroad. But it would be a mistake to claim that command-and-control regulation is dead. First, there are some cases where market-like solutions won't do the job. If an imminent and very serious hazard to human health and the environment is discovered, an outright ban is likely to be the appropriate policy response. Second, some still seem to prefer that companies be punished for their emissions by making them pay as much as possible to alleviate them. But this is premised on the misguided notion that firms pollute because they are malevolent, rather than because pollution is one consequence of making things that society demands. Moreover, such an approach really only punishes the customers, employees and/or shareholders of the firm, for these persons are the ones who will end up bearing the costs.

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12. Economic Incentives Versus Command and Control: What's the Best Approach for Solving Environmental Problems?*

Winston Harrington and Richard D. Morgenstern

Now, decades after the first environmental laws were passed in this country, policymakers face many choices when seeking to solve environmental problems. Will taxing polluters for their discharges be more effective than fining them for not meeting certain emission standards? Will a regulatory agency find it less costly to enforce a ban or oversee a system of tradable permits? Which strategy will reduce a pollutant the quickest?

Clearly, there are no one-size-fits-all answers. Many factors enter into the decision to favor either policies that lean more toward economic incentives (EI) or toward direct regulation, or what is commonly referred to as command-and-control (CAC) policy. Underlying determinants include a country's governmental and regulatory infrastructure, along with the nature of the environmental problem itself.

Even with these contextual factors to consider, we thought it would be useful to compare EI and CAC policies and their outcomes in a real-world setting. To do this, we looked at six environmental problems that the United States and at least one European country dealt with differently (Box 12.1) For each problem, one approach was more of an EI measure, while the other relied more on CAC. For example, to reduce point-source industrial water pollution, the Netherlands implemented a system of fees for organic pollutants (EI), while the United States established a system of guidelines and permits (CAC). It turned out, in fact, that most policies had at least some elements of both approaches, but we categorized them as EI or CAC based on their dominant features.

We then asked researchers who had previously studied these policies on either side of the Atlantic to update or prepare new case studies. We analyzed the 12 case studies (two for each of the six environmental problems) against a list of hypotheses frequently made for or against EI and CAC, such as which instrument is more effective or imposes less administrative burden.

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1. The Evolution of Command-and-Control and Economic Incentives

Only recently has it been possible to find enough EI policies to carry out a project such as this. Until about 15 years ago the environmental policies actually chosen were heavily dominated by CAC approaches. In the United States, the 1970s saw a great volume of new federal regulation to promote environmental quality, none of which could be characterized as relying heavily upon economic incentives. Since then, however, there has been a remarkable surge of interest in EI approaches in environmental policy. Since the late 1980s, whenever new environmental policies are proposed, it is almost inevitable that economic incentive instruments will be considered and will receive a respectful hearing.

The reasons for the newfound popularity of EI policies are unclear. Perhaps it is due to the growth in awareness of economic incentive approaches among policy-makers and policy analysts the 20 or so years between 1970 and 1990. In the 1970s these approaches were generally unfamiliar to those outside the economics profession. Another possibility is the emergence of tradable emission permits in the late 1970s. Before then, the main EI

Box 12.1. The six environmental problems we studied

In our analysis, we selected six environmental problems, summarized below, to serve as a control in order to compare EI and CAC approaches in trying to solve each one. We paired a policy from the United States with one implemented in one or more European countries.

For clarity's sake, although almost all contain some blend of EI and CAC elements, those that are more closely associated with EI instruments are listed first:

1. SO_2 emissions from utility boilers: Permit market (United States) vs. sulfur emission standards (Germany)
2. NO_x emissions from utility boilers: Emission taxes (Sweden and France) vs. NO_x New Source Performance Standards (United States)
3. Industrial water pollution: Effluent fees (Netherlands) vs. Effluent Guidelines and National Pollutant Discharge Elimination System permits (United States)
4. Leaded gasoline: Marketable permits for leaded fuel production (United States) vs. mandatory lead phase-outs plus differential taxes to prevent misfueling (most European countries)
5. Chlorofluorocarbons (CFC): Permit market (United States) vs. mandatory phase-outs (other industrial countries)
6. Chlorinated solvents: Source regulation (United States) vs. three distinct policy approaches (Germany, Sweden, Norway)

alternative to the regulatory policies being implemented was a per-unit tax on pollution (sometimes referred to as an effluent fee). By the 1980s the policy community was generally aware of a quantity-based EI alternative—tradable emission permits—that seemed to provide the same assurances of the achievement of environmental goals that were offered by CAC approaches.

A third possible cause is the widespread disappointment with outcomes of the CAC regulations adopted in the 1970s. The nearly limitless variety of American industries and industrial processes required the EPA to write very detailed and complex regulations, but despite these efforts, the Agency faced a raft of legal challenges. Regulatory complexity combined with litigiousness delayed the implementation of most regulations far beyond the schedules envisioned by Congress. In other words, much of the enthusiasm for EI could be attributed to disenchantment with CAC.

2. The Two Sides of the Pond

It is first worth underscoring some differences between the United States and Europe that serve as a backdrop to policy decisions and implementation.

First, of course, we are comparing a single federal system in the United States with the many countries of the European Union (EU). Beginning in the late 1960s, environmental policymaking became centralized in the United States. In Europe, each country has adopted policies according to its own timetable, generally beginning in the late 1960s in the wealthiest nations and sweeping south and east to the former Soviet empire by the 1990s. Environmental policy in Europe is now a mix of country-specific and EU-wide measures, which these cases reflect.

Second, there are major differences between the United States and Europe in the extent of pre-regulatory studies undertaken. Because of the U.S. requirement on agencies like the U.S. Environmental Protection Agency to conduct a Regulatory Impact Analysis before taking action, substantially more information was available about the hoped-for benefits of U.S. policies. A further issue concerns the greater reliance on taxes for regulatory purposes in Europe compared to the United States. A number of European nations use such taxes—sometimes combined with incentive compatible rebate schemes—to achieve environmental objectives.

In the United States, environmental taxes are virtually nonexistent. Overall, however, and despite these various differences in approaches, we were not able to discern clear differences in regulatory outcomes across the Atlantic: in some cases one or more European nations acted sooner or more aggressively to address environmental problems while in other cases the United States acted sooner or more aggressively.

3. Testing the Hypotheses

Since the 1970s, when western countries began forming comprehensive environmental policies, there has been a good deal of speculation and disputation over the differences between EI and CAC instruments in practice. These discussions boil down to assertions or hypotheses about comparative advantages of each instrument. We compiled a list of the 12 most commonly stated hypotheses, recognizing that different observers might develop very different lists.

Below we discuss the five hypotheses that we consider most important in evaluating a policy instrument. For each, we state the hypothesis, review the original rationale in making it, and then test whether the hypothesis holds up in light of one or more of our case studies.[1]

1. EI instruments are more efficient than CAC instruments: that is, they result in a lower unit cost of abatement.

   Rationale: It is commonly believed that EI instruments have an efficiency advantage over CAC instruments, although the case is not airtight. EI instruments are more cost-effective at achieving a given emission reduction. But to get from cost-effectiveness to efficiency requires additional assumptions, including that the system is one of perfect competition and that the emissions are not location-specific. A theoretical counter to this hypothesis is that a CAC instrument can be as efficient if the emission standard for each plant is chosen so that the marginal costs of abatement equal the marginal social costs of pollutant damage.

   Performance: The cases we analyzed show that EI is generally more efficient. For example, in looking at the U.S. program of marketable permits to lower SO_2 emissions, realized costs are only about one-half what was expected back in 1990 and about one-quarter of the estimated cost of various CAC standards. EI also achieved substantial cost savings in the elimination of CFCs and lead in gasoline, in part because of cost heterogeneities that could be exploited during the phase-down period. However, in instances where the regulations are so stringent that practically all-available abatement measures must be taken, there is little scope for choosing the most cost-effective ones, and EI instruments do not achieve significant cost savings over CAC. EI also enjoys little advantage if all plants face similar abatement costs. Both these conditions limited, for example, the efficiency losses of using CAC for the German SO_2 emissions.

2. The real advantages of EI Instruments are only realized over time, because they provide a continual incentive to reduce emissions, thus promoting new technology, and permit maximum flexibility in achieving emission reductions.

   Rationale: The effects of CAC on technology are potentially complex. On the one hand, costly regulations provide a spur to find less costly ways of compliance. On the other, the requirement to install a specific technology conceivably discourages research, since discovering new ways to reduce emissions can lead to more stringent regulations. More stringent performance standards for new plants have the stated objective of promoting technology, but they can also have the pernicious effect of postponing retirements of older, dirtier plants and discouraging entry by outside firms.

   Performance: EI provides greater incentives than CAC for continuing innovation over time in many, but not all, cases studied. For example, the Swedish nitrogen oxides (NO_x) tax induced experimentation in boiler operations that led to substantial reductions in NOx emissions. Because NO_x emissions from boilers are idiosyncratic, it was unknown beforehand what would work in each boiler. Achieving these reductions from CAC would therefore have been impossible. Similarly, the U.S. SO_2 trading policy induced many nonpatentable boiler-specific innovations on utility boilers. Elsewhere, the Netherlands became a world leader in water purification technologies and its industries adopted more advanced, process-integrated measures to reduce pollutants.

   Innovation also occurs under CAC, but the results are often different. For example, the lead phase down induced emissions reductions in all plants during the period when a CAC policy was employed, but when the policy allowed permit trading and banking, the reductions were concentrated in newer plants with longer expected lifetimes, where the improvements were most cost-effective. In the United States, SO_2 policy examination of patents suggests that in a CAC regime only cost-reducing innovations are encouraged, while under EI both cost-reducing and emission-reducing innovations are encouraged.

3. CAC policies achieve their objectives quicker and with greater certainty than EI policies.

   Rationale: In the early 1970s, CAC was seen as the way to expedite compliance, even if the approach was not the least costly. It appeared then that EI instruments, particularly emission fees, would not achieve the same objectives.

Performance: The evidence from the cases is mixed. Supporting the relative effectiveness of CAC is the U.S. effort to phase out the solvent trichloroethylene (TCE), in which EPA ultimately mandated limits. The EI aspects of the rule did not attract significant industry participation. In phasing out leaded gasoline in Europe, progress would have significantly slowed without mandating catalytic converters and maximum lead content in addition to tax differentials.

On the other hand, several cases argue that EI policies are more effective. In the Dutch water case, for example, the influence of effluent fees on organic waste load reductions was prompt and large. Similarly, by eliciting industry cooperation, the trading and banking program probably achieved a much more rapid phase-down of lead in gasoline than would have been possible with a CAC program that industry would have opposed.

A final point on effectiveness is that two cases show that both approaches can result in significant environmental gains, but with undesirable longer-term side effects. In the United States, NO_x emissions from coal-fired power plants were reduced, but the standards, which only affected new plants, caused firms to extend the life of older, more polluting plants to avoid the costs associated with newer ones. In Sweden, TCE users persuaded the public and authorities that complete implementation of a ban would cause them undue harm. They received numerous waivers and exceptions, thus undermining the authority of the environmental agency and perhaps emboldening other firms to oppose other regulations.

4. Regulated firms are more likely to oppose EI regulations than CAC because they fear they will face higher costs, despite the greater efficiency of EI instruments.

   Rationale: Although EI instruments may have lower social costs overall, firms pay higher costs under EI than CAC. Under CAC, the argument goes, the polluting firm pays to abate pollution; under many EI instruments, the firm pays the cost of abatement plus a fee for the remaining pollution it discharges. The firm is better off only if the abatement cost is lower by an amount at least as great as the fee payments.

   Performance: Experience on both sides of the Atlantic suggests that no government has put this hypothesis to the test, which, in a way, is strong support for it. In nearly all cases, governments eliminate the burden of EI instruments by returning fees to the firms. For example, in France, revenues collected through NO_x discharge fees subsidized the firms' abatement investments, while in Sweden the fees were returned to the firms on the basis of the energy they produced. In the United States, where the EI instrument of choice is a tradable permit, the permits have always been given away rather than auctioned off.

5. CAC policies have higher administrative costs.

   Rationale: Administrative costs are determined by the amount of interaction between the regulator and regulated source. Supporters of this hypothesis note that the complexity of setting and enforcing specific requirements is higher than implementing fee-based EI policies. In addition, fees for increased emissions tend to rise gradually, whereas with CAC, a line separates compliance from violation. The potentially high incremental cost at the point of violation gives regulated sources an incentive to defend themselves legally rather than accept sanctions, thus adding to the regulators' burden.

   Performance: The cases show no clear pattern. While the CAC-oriented effluent guidelines program in the United States imposed high administrative costs on EPA, so did the EI instruments of the lead phase-down program. Looking at SO_2 reduction, the EI-oriented U.S. trading program gained a reputation for low administrative costs, but the SO_2 reduction program in Germany does not show evidence of higher administrative costs than a comparable EI program. Overall, because the evidence on this hypothesis is mixed, we could not form a firm conclusion about whether policy outcomes supported or refuted it.

4. Apples and Oranges?

Questions of effectiveness and efficiency were at the core of the initial selection of policy instruments in the 1970s and 1980s. As these cases show, EI instruments appear to produce cost savings in pollution abatement, as well as innovations that reduce the overall cost. The concern that EI instruments are not as effective is not borne out in our analysis. However, the finding about EI's economic efficiency is tempered by evidence that polluting firms prefer a CAC instrument because of its perceived lower costs to them. In all but one of the case studies, the actual or potential revenue raised by EI instruments had to be reimbursed in some way to the firms. This, of course, means the revenues cannot be used for other purposes.

In the 1970s, almost all environmental policies relied on direct regulation, with very rare instances of EI instruments. Since the late 1980s, on the other hand, whenever a new policy is proposed, policymakers at least consider, and often select, an EI instrument. That said, almost all the policies that we studied are a blend of both, beginning as a CAC policy and then having EI elements added or substituted. In the 12 cases we studied, in fact, only a few (reduction of SO_2 emissions in Germany; TCE in Germany and Sweden) had no EI elements in their design. Moreover, we can report significant environmental results from the cases we studied. Averaged across all twelve, emissions fell by about two-thirds when compared to baseline estimates. Most outcomes either met or exceeded policymakers' original expectations. This is encouraging news for those seeking environmental improvements in the future.

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Note

1. *See* Winston Harrington, Richard D. Morgenstern and Thomas Sterner (Eds.), Choosing Environmental Policy: Comparing Instruments and Outcomes in the United States and Europe, Washington, DC: RFF Press, 2004.

13. Benefits and Costs From Sulfur Dioxide Trading: A Distributional Analysis

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1. Introduction

Prior to the passage of Title IV of the 1990 Clean Air Act Amendments (CAAA), there had been a lively debate involving Congress, the U.S. Environmental Protection Agency (EPA), and academics, about the need for reducing sulfur dioxide (SO_2) emissions due to the problem of acid rain. In addition to domestic pressure, Canada was putting political pressure on the United States to decrease acid rain. Just after the passage of the CAAA the United States and Canada signed the Canada-United States Air Quality Agreement, aimed at controlling transboundary acid rain. How damaging is acid rain? The National Acid Rain Precipitation Assessment Program found that acid rain causes minor damage to crops and modest damage to aquatic life in acidified lakes and streams. Burtraw et al. (1998) estimate the expected environmental damages from acid rain on recreational activities, residential visibility, and morbidity to be about $13 per capita in 1990.

On the other hand, SO_2 also combines in the atmosphere with ammonia to form sulfates—fine particulates ($PM_{2.5}$)—which have been shown in several studies to contribute significantly to pre-mature mortality. Thus, even if acid rain has only a marginal environmental impact, reductions in SO_2 emissions have additional (and potentially much larger) health benefits, through reduced pre-mature mortality. EPA (2003) estimates that the human health benefits of the Acid Rain Program will be roughly $50 billion annually, due to decreased mortality, fewer hospital admissions and fewer emergency room visits, by the year 2010.

Coal from fossil-fuel fired electric utilities accounts for most of SO_2 emissions in the United States. Title IV of the 1990 CAAA set an annual 9 million ton cap on SO_2 emissions from all fossil fuel fired electric utilities. This cap, which is to be fully achieved by 2010, requires the affected electric utilities to reduce their aggregate SO_2 emissions by 10 million tons below their 1980 levels. Along with requiring substantial SO_2 reductions, Title IV also abandoned the command-and-control approach to the regulation of utilities, where utilities were required to meet individual emission standards set by regulators, in favor of a more flexible, cost-efficient tradable permit approach. This more flexible approach made the substantial SO_2 reduction politically feasible and is widely believed to have led to tremendous cost savings relative to the command-and-control approach. Keohane (2003) estimated that the system of allowance trading resulted in cost savings between \$150 million and \$270 million annually, compared to a uniform emissions-rate standard.

Title IV allows permits to be bought and sold freely anywhere in the continental United States.[2] Allowing permits to be bought and sold freely may inadvertently create a divergence between the people who are paying for the SO_2 reductions and those that are benefiting from the reductions. Morgan and Shadbegian (2003) find that the SO_2 trading program may have inadvertently resulted in some environmental injustices—mainly higher levels of emissions in disproportionately poor and minority areas.[3]

In this chapter we extend the work of Morgan and Shadbegian (2003) by examining the spatial distribution of the costs and benefits associated with air quality improvements that occurred during the first year under Title IV of the CAAA. The air quality improvements are measured relative to the level of emissions under the former command-and-control regime, which allowed a greater level of emissions. We examine the spatial distribution of the costs and benefits both in terms of the states and regions being affected and the socio-economic composition of the affected population.

The vast majority of dollar-valued benefits from air pollution abatement arise from the impact of airborne particulates ($PM_{2.5}$) on premature mortality. A 1997 EPA study reports that of the estimated \$22.2 trillion worth of benefits derived from the Clean Air Act of 1970, reductions in particulate-related mortality contributed more than \$20 trillion. We use a spatially-detailed air pollution dispersion model (the Source-Receptor Matrix) to evaluate the impact of SO_2 emission reductions from each plant on county-level concentrations of particulates during Phase I of Title IV. Using existing evidence on the connection between particulate exposures and mortality, we translate the reductions in secondary particulate concentrations in each county in the United States into the dollar benefits from reductions in premature mortality.

Who pays for the improvements in air quality? One possible answer is "nobody," if efficiency improvements resulting from the new emissions trading system (e.g., more flexible production switching, less uncertainty about regulatory requirements) outweigh the additional abatement costs on a plant-by-plant basis. A more likely scenario is that some plants face higher costs of abatement, which are passed along to their customers. If some plants increase their emissions and buy additional allowances, the population affected by the worsening air quality will be "paying" some of the costs of the greater air quality improvements near other plants that reduced their emissions in order to sell the allowances.

Arrow et al. (1996) argue that along with a cost-benefit analysis measuring the aggregate net benefits from a regulation, a good analysis will also examine the distributional consequences. In this chapter we compare the overall net health benefits that were achieved under Title IV along with the spatial distribution of those net benefits to test whether there were unforeseen consequences of the regulatory change in terms of adverse impacts on particular regions or socio-economic groups. The findings will indicate whether these distributional impacts are of only second-order importance compared to the overall net benefits, or whether they are sufficiently large for policy-makers to take them into account when considering future market-oriented regulatory reforms.

Using data for the 148 dirtiest coal-fired utilities we find, as expected, that the aggregate benefits in 1995 caused by reductions in SO_2 emissions under Title IV greatly exceed their costs: we estimate benefits of $56 billion (a bit larger than EPA's estimates of total benefits of $50 billion by 2010) and costs of only $558 million. Therefore, the net benefits from the SO_2 reduction are roughly $55 billion or $100 in benefits for every $1 in abatement costs. The net benefits are positive in every EPA region, but are highly concentrated. We find that nearly 90% of the benefits and costs of the overall reductions under Title IV are concentrated in four regions—the northeast, north central, mid-Atlantic, and southeast. In terms of the socio-economic distribution of net benefits, we find that minority groups (African-Americans and Hispanics) receive a greater share of the benefits than of the costs. The poor are the only group raising any environmental justice concerns, receiving a slightly higher share of the costs than of the benefits. However this calculation assumes that the poor purchase as much electricity as the rich, so we most likely overstated the share of abatement costs borne by the poor.

The rest of the chapter is organized as follows. In section 2 we present background information on Title IV of the CAAA of 1990. Section 3 contains a brief survey of the literature on studies examining various aspects of the Title IV trading program and various aspects of environmental justice. Section 4 describes the methodology we use to estimate both the health benefits and the costs of SO_2 abatement under Title IV and section 5 describes our sample of plants. In section 6 we discuss our findings and we end with some concluding remarks in Section 7.

2. Title IV: Background Information

Title IV of the CAAA completely changed the way coal-fired utilities were regulated in the United States. Prior to Title IV utilities were regulated by a command-and-control regime that targeted the sulfur content of the coal used at each individual plant. Title IV established a cap-and-trade program that set a cap on total SO_2 emissions, distributed allowances among generating units equal to that cap, and allowed plants to freely trade these allowances among their own units, to sell them to other plants, or to bank them for future use. The only requirement faced by a plant under the trading program is that it must have enough allowances at the end of the year to cover each ton of SO_2 emitted that year. Thus, the allowance trading program instituted by Title IV provides much greater flexibility to achieve any given emission standard because utilities which face high marginal abatement cost may purchase SO_2 permits from utilities which face lower marginal abatement costs.

The goal of Title IV was to reduce aggregate SO_2 emission levels to approximately 9 million tons by 2010, roughly half of the 1980 level. The reduction was to be achieved in two phases. Phase I (1995–1999) targeted the dirtiest 110 power plants (with 263 generating units). These generating units, called the Table A units, were required to reduce their emissions to 7.2 million tons per year starting in 1995, 6.9 million tons per year in 1996, and then 5.8 million tons per year from 1997–1999. The Table A units emitted 8.7 million tons of SO_2 in 1990 and only emitted 4.5 million tons in 1995 (roughly 50% less). The number of allowances a unit received was based on its average 1985–1987 heat input times an average emission rate of 2.5 lbs of SO_2 per million BTUs of heat input. Each allowance gave a unit the right to emit one ton of SO_2, and the unit could only emit an amount of SO_2 equal to the number of allowances held.[4]

Phase II, which began in the year 2000, brought the smaller generators—generators that have an output capacity of 25 megawatts or greater—under the cap-and-trade system.[5] In addition to imposing constraints on the smaller and cleaner units, the Table A units were required to make additional reductions in their SO_2 emissions—reducing their overall emissions by another 3.4 million tons, down to 2.4 million tons by 2010. Annual allowance allocations to each unit were based on an average emission rate of 1.2 lbs of SO_2 per million BTUs of heat input, a much stricter standard than the 2.5 lbs during Phase I.

In 1995 SO_2 emissions dropped dramatically. Phase I units emitted a total of only 4.9 million tons, a reduction of 4.6 million tons—3.2 million tons more than was required.[6] In fact, SO_2 emissions started to decrease right after the passage of Title IV, even before the trading system was in place. Several explanations have been offered for the pre-1995 reduction. Plants may have complied early in order to pass on to consumers the additional cost of low-sulfur coal or the cost of installing scrubbers. Some states amended their State Implementation Plans (SIPs) requiring utilities to reduce their emissions before the first year of Phase I. The most likely explanation is that railroad deregulation made it cheaper to transport low-sulfur coal to Midwest electric power plants, the geographic area that experienced the most reductions in SO_2 emissions between 1985 and 1993 (Ellerman and Montero 1998).

Another important feature of the SO_2 allowance market is that allowances that are not used in one year may be banked and used in any subsequent year. That is, a plant may reduce emissions below its annual allocation and deposit the extra allowances in an emissions bank. These *banked* allowances are perfect substitutes for future year allowances, and may be used or sold. Banking during Phase I could help plants adapt to the more stringent limits imposed under Phase II by smoothing the required reductions over time. This explanation is borne out by experience: plants banked over 11.5 million allowances during Phase I (1995–1999), then used 1.2 million of these banked allowances in the first year of Phase II (2000), followed by 1.08 million allowances in 2001 and another 650,000 million allowances in 2002. This suggests that the extra abatement during Phase I was intentional (rather than being an unexpected result of lower than expected prices for low-sulfur coal).

3. Literature Review

3.1. Sulfur Dioxide Trading Program

Long before the advent of emissions trading, Gollop and Roberts (1985) estimated that a cost-effective allocation of pollution abatement across electric utilities would result in a nearly 50% reduction in pollution abatement costs, suggesting potentially large savings from emissions trading. Since the passage of the 1990 CAAA, many papers, including Joskow et al. (1998), Schmalensee et al. (1998), Carlson et al. (2000), Keohane (2003), and Shadbegian and Morgan (2003) have examined various aspects of the actual SO_2 allowance trading program including its cost savings, environmental effectiveness, spatial patterns of abatement, pollution control innovations, and the efficiency of the banking of permits. The potential success of any pollution permit-trading program depends on the efficiency of the market of the tradable permits. Joskow et al. (1998) assess the efficiency of the market for

SO_2 permits by comparing the price of permits auctioned by EPA between 1993 and 1997 with private market indices. Joskow et al. (1998) find that by the end of 1994 these prices were virtually identical and thereby conclude that the private market for tradable permits was relatively efficient. Schmalensee et al. (1998) also conclude that the private market for tradable permits was relatively efficient by noting the growth in the level of the trading volume in the market: 1.6 million, 4.9 million, and 5.1 million allowances were traded in 1995, 1996, and 1997, respectively.

Keohane (2003) estimates that using a system of tradable allowances resulted in annual cost savings between $150 million and $270 million compared to a uniform emissions-rate standard. However, Carlson et al. (2000) conclude that the large decrease in abatement costs during the beginning of Title IV relative to the original estimates resulted more from a technological change that reduced the cost to switch to low sulfur coal and the decrease in the price of low sulfur coal rather than the ability to trade permits per se. Shadbegian and Morgan (2003) examine the impact of the stringency of SO_2 regulations on the productivity of electric utilities. They find that regulatory stringency had a significantly negative effect on productivity prior to Title IV, but that during Title IV, regulatory stringency had only a small and insignificant negative impact on productivity.

3.2. Distribution of Pollution

During the past decade there has been an increasing number of studies that examine various aspects of environmental justice—polluting plants' location decisions, expansion decisions of hazardous waste facilities, fees paid to communities to host facilities, plant emissions, and regulator decisions—in a formal multiple regression framework. Previous anecdotal evidence (see Government Accounting Office 1983 and United Church of Christ 1987) suggests that firms tend to locate their polluting plants in areas with a greater percentage of poor people and minorities. However, Been and Gupta (1997) examining the location decisions of commercial hazardous waste treatment storage and disposal facilities (TSDFs) find mixed evidence of environmental injustice. In particular, they find no statistical evidence that TSDFs were more likely to be sited in neighborhoods that were disproportionately African American at the time of siting and that poor neighborhoods are actually negatively correlated with TSDF sitings, but they do find evidence that TSDFs were more likely to be sited in disproportionately Hispanic areas. Contrary to the anecdotal evidence, Wolverton (2002a) shows that if one considers the socioeconomic characteristics of the community at the time the plant is sited, the location decision of toxic waste emitting plants in Texas do *not* depend on race, and poor communities actually attract disproportionately *fewer* polluting plants—a finding similar to Been and Gupta (1997).

Hamilton (1993, 1995) examines whether exposure to environmental risk is related to socioeconomic characteristics of a neighborhood and political activism. Specifically, Hamilton examines the relationship between the net capacity expansion decisions of commercial hazardous waste facilities and race, income, education, and voter turnout (level of political activity). Hamilton finds that the decision to expand net capacity is not significantly related to any of the socioeconomic variables, but is significantly negatively correlated with voter turnout. On the other hand, Jenkins et al. (2004) show that counties with greater percentages of minority residents receive lower host fees for the siting of landfills, while richer counties receive higher host fees, results consistent with the idea of environmental injustice.

Three additional studies examine the relationship between pollution emissions and the socioeconomic characteristics of communities to assess the validity of the claim of environmental injustice: Arora and Cason (1999), Wolverton (2002b), and Gray and Shadbegian (2004). Arora and Cason (1999) examine 1993 Toxic Release Inventory (TRI) emissions for the entire United States, finding evidence of racial injustice only in non-urban areas of the south. Wolverton (2002b) examines the relationship between TRI releases and socioeconomic characteristics of communities in Texas and finds that plants tend to reduce TRI releases *more* in minority neighborhoods than in non-minority neighborhoods, exactly the opposite of the claim of environmental racism. Gray and Shadbegian (2004) examine the relationship between SO_2, PM_{10}, BOD, and TSS emissions of pulp and paper mills and socioeconomic variables finding mixed results.[7] For all four pollutants Gray and Shadbegian find that plants with a greater percentage of poor nearby emit more pollution, a result consistent with environmental injustice, but that plants with more minorities nearby actually emit less pollution, a result inconsistent with environmental injustice.

Finally Becker (2004), using establishment-level data on manufacturing plants from the U.S. Census Bureau's Pollution Abatement Costs and Expenditures (PACE) survey, examines the relationship between air pollution abatement expenditures and community demographics. Becker (2004) finds that, after controlling for a number of plant-level characteristics and levels of federal, state, and local regulation, communities with higher home ownership rates and higher per capita income enjoy greater pollution abatement activity from their nearby plants.

4. The Benefits and Costs of Cleaner Air

4.1. Benefits from Cleaner Air

We identify the benefits of reducing SO_2 emissions (SO_2BEN) from a given source with the change in mortality risk from exposure to ambient particulate concentrations caused by those SO_2 emissions. These health benefits are measured using a simplified linear damage function, based on estimated parameters from the appropriate literature:

$$SO_2BEN = SO_2DIFF * AIR\_QUAL\_TC * HEALTH\_CHG * POP * VSL.$$

AIR_QUAL_TC is the transfer coefficient—the change in air quality (ambient particulates) per unit change in SO_2 emissions (SO_2DIFF). HEALTH_CHG is the change in mortality risk to the affected population due to the changes in air quality. POP is the size of the affected population, and VSL is the dollar value placed on reducing pre-mature mortality.

We measure the changes in air quality at any given location using the Source-Receptor (S-R) Matrix Model, as described in Latimer (1996) and Abt (2000). The S-R Matrix model was originally calculated using the Climatological Regional Dispersion Model (CRDM). The model incorporates data on pollution emissions from 5,905 distinct sources in the United States, along with additional sources from Mexico and Canada.[8] The S-R Matrix relates emissions of specific pollutants from each source to the resulting ambient concentrations of each pollutant in every county in the United States. Specifically, the S-R Matrix provides a set of transfer coefficients which yield county-by-county changes in annual average pollutant concentrations for each one ton change in emissions of a particular pollutant from a particular source. The S-R Matrix transfer coefficients are a function of many factors including wet and dry deposition of gases and particles, chemical conversion of SO_2 and nitrogen oxide (NO_X) into secondary particulates, effective stack height, and several atmospheric variables (wind speed, wind direction, stability, and mixing heights). We use the impact of SO_2 emissions on ambient concentration of $PM_{2.5}$ in each county to measure AIR_QUAL_TC.

Our measure of HEALTH_CHG concentrates on the long-term mortality effects of particulate matter ($PM_{2.5}$)—an assumption consistent with past studies (Rowe et al. 1995; Levy et al. 1999). Since our study focuses on the benefits of reduced SO_2 emissions we concentrate on the health benefits from lower concentrations of secondary particulates that result from SO_2 emissions. We use the findings from the American Cancer Society study, the most comprehensive analysis of long-term mortality effects from air pollution to date (Pope et al. 2002). They find approximately 4% higher mortality rates in people exposed to a 10 μg/m3 increase in $PM_{2.5}$ concentrations (95% confidence interval: 1%, 8%). We assume that the point estimate is applicable to the secondary particulates formed from SO_2 (Pope et al. 2002 found similar numbers for sulfate particles in their study).[9]

Our estimate of the exposed population, POP, is based on county-level data from the 1990 Census of Population. This data identifies the total number of people living in each county (and hence the number affected by the average ambient pollution concentrations in that county). In addition, it provides information on the socio-economic characteristics of each county's population (e.g., income, age, race), which helps us examine issues of environmental justice.

Finally, to place a dollar value on pre-mature mortality, we use a recent EPA (1997) benefit-cost analysis that estimated the value of a statistical life (VSL). The EPA study pooled contingent valuation and wage-risk studies to produce a central estimate of $5.4 million (in 1995 dollars) per life saved. Note that our calculations assign constant values of the VSL and HEALTH_CHG terms for the entire population. Each exposed person faces the same average dollar harm from exposure to a given concentration of particulates, allowing for neither differences in sensitivities for different populations nor differences in valuation.[10] The very large estimates we obtain for the benefits of reducing SO_2 emissions could be reduced by assuming either smaller health effects or a lower VSL, although with benefits that exceed costs by a factor of 100:1, the conclusion that benefits exceed costs is likely to remain the same. In addition, the central conclusions of this chapter would not be affected by changes in VSL or HEALTH_CHG, since such changes would affect everyone's exposure valuation proportionately, leaving the ratios in valuation between population groups unchanged.

4.2. Costs of Cleaner Air

There are three options (or combinations of options) available to plants to comply with Title IV: installing a scrubber, switching to low sulfur coal, or buying allowances. Our measure of SO_2 abatement cost (COST) is based on the method each plant actually used to comply with Title IV. Based on Ellerman et al. (1997) we have the total cost of abatement for each of the 374 Phase I units (plant-boiler observations) affected by Title IV. In 1995, the average cost per ton of *switching* and *scrubbing* is $153 and $265 respectively, while the average cost of a permit is $128.50.[11]

We assume that all of the additional costs of abatement are passed along to the utility's customers, and further assume that all customers live within the state where the utility is located.[12] We use the 1990 Census of Population to allocate each plant's abatement costs equally to all people living within that state, with the different socio-economic groups receiving benefits and costs proportional to their share in the overall population.

5. Sample Coverage

Phase I of Title IV regulated the emissions of 263 generating units (the Table A generating units) owned by 110 plants. An additional 38 *substitution and compensation* plants (111 generating units) "opted into" Phase I, bringing the final total to 374 generating units. Our sample consists of all 148 plants and their 374 generating units. The geographic distribution of these plants—heavily concentrated in the Midwest—is shown in Figure 13.1.

In Table 13.1 we present information on SO_2 emissions and the allocation of SO_2 allowances obtained from the EPA's Allowance Tracking System (ATS).[13] The 148 plants in our sample emitted a total of 9.5 million tons of SO_2 during 1990, the year Title IV was passed. By 1995, our 148 plants had reduced their SO_2 emissions by 4.6 million tons from their 1990 levels, cutting them almost in half, although Title IV had only required them to reduce emissions by 15%, to 8.1 million tons.

6. Distribution of Benefit and Costs

In Table 13.2 we present the health benefits and abatement costs associated with the actual 1995 SO_2 emissions reductions: counterfactual SO_2 emis-

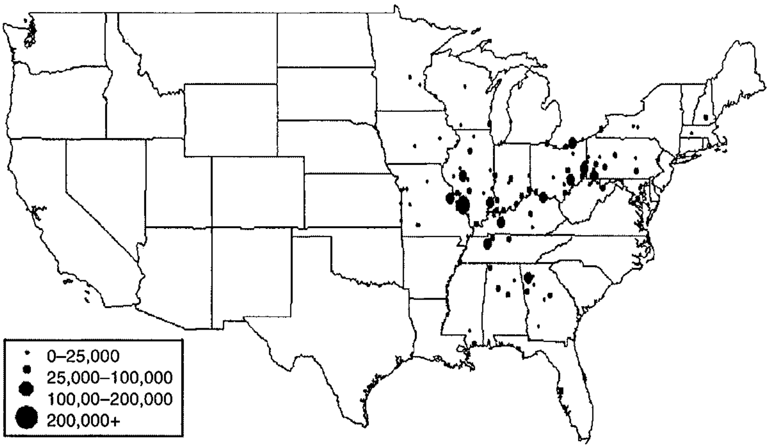

Figure 13.1. Distribution of plants in database (148 plants; scale=1995 SO_2 emissions in tons)

Table 13.1. Phase I units*

SO_2 Emissions in 1990 (tons)	9,468,183
SO_2 Emissions in 1995 (tons)	4,902,778
Allowances in 1995	8,076,472
Boilers	374
Plants	148

*Includes all Phase I units—the 110 Table A plants (263 units) plus the 38 "Substitution and Compensation" plants (111 units)

sions minus actual emissions. The counterfactual emissions in 1995 are those we would have observed in the absence of the 1990 CAAA and are the same as those presented in Ellerman et al. (1997). As expected, the aggregate benefits in 1995 resulting from reductions in SO_2 emissions from the 1995 counterfactual levels far outweigh their costs: we estimate benefits of nearly $56 billion and costs of only $558 million. An alternative assumption on abatement costs, that the actual cost of a ton of abatement is equal to the permit price ($128.5 in 1995), results in total abatement costs of only $496 million. In either case these increased abatement costs are dwarfed by the increased benefits from the SO_2 reduction, which are roughly 100 times as large.

The net benefits are positive in every region, however they are highly concentrated across regions. Not surprisingly, given the concentration of the plants in the Midwest and the pattern of airflow from west to east, the benefits that result from the large reductions in emissions are highly concentrated geographically in the east. Table 13.3a contains the distribution of benefits and costs across the 10 different EPA regions. As shown in Figure 13.2, the overwhelming majority of the net benefits (89%) are concentrated in four regions (2, 3, 4, and 5). In addition, three of these regions (3, 4, and 5) pay a very large percentage of the overall costs (90%). Regions 4, 5, and 7 all pay a higher percentage of the costs than they receive in terms of health benefits. Region 5 (the North Central states) is the biggest relative loser, paying 45% of the costs while only receiving 26% of the benefits. On the other hand, Regions 1 (New England) and 2 (NY and NJ) are the biggest

Table 13.2. Benefits and costs

Benefits	$55.94 billion
Costs	$0.56 billion
Net Benefits	$55.38 billion

Table 13.3a. Percentage distribution of benefits and costs across regions

Region	States	Benefit	Cost
1	CT, MA, ME, NH, RI, VT	6.21%	0.19%
2	NJ, NY	16.84%	1.24%
3	DC, DE, MD, PA, VA, WV	23.69%	15.36%
4	AL, FL, GA, KY, MS, NC, SC, TN	22.05%	30.33%
5	IL, IN, MI, MN, OH, WI	26.19%	44.74%
6	AR, LA, NM, OK, TX	2.82%	0.00%
7	IA, KS, MO, NE	2.07%	8.14%
8	CO, MT, ND, SD, UT, WY	0.11%	0.00%
9	AZ, CA, NV	0.02%	0.00%
10	ID, OR, WA	0.00%	0.00%

relative winners, only paying 0.2% and 1.2% of the costs while receiving 6% and 17% of the benefits, respectively.

In Table 13.3b we compare the net benefits per capita in each region and this leads to a somewhat different ranking of relative winners and losers than we observed with the shares of benefits and costs. Regions 1–5 each derive more than $249 per capita net benefits. Region 3 (the mid-Atlantic

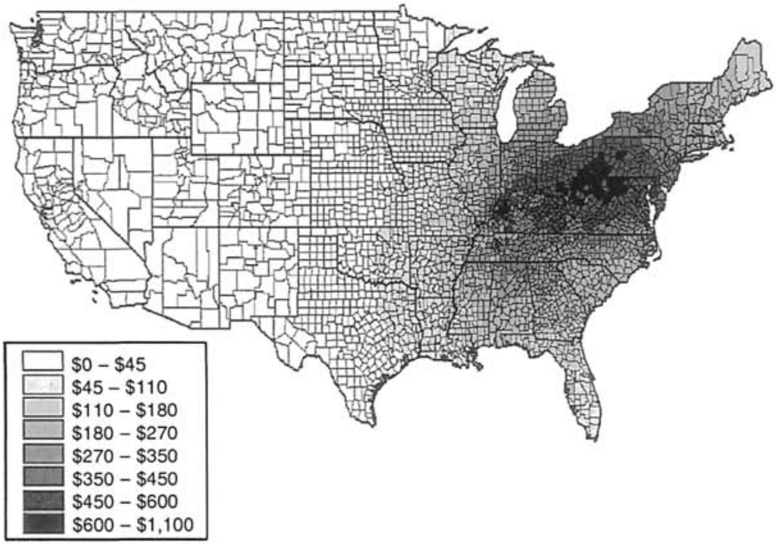

Figure 13.2. Geographic distribution of net benefits across U.S. counties

Table 13.3b. Average dollar per capita distribution of benefits and costs across regions

Region	Average Benefit	Average Cost	Average Net Benefit
1	256.2	0.1	256.1
2	354.7	0.2	354.4
3	505.5	3.3	502.2
4	252.7	3.5	249.2
5	303.7	5.2	298.5
6	51.3	0	51.3
7	93.2	3.7	89.5
8	7.5	0	7.5
9	0.3	0	0.3
10	0.3	0	0.3

states) receives the highest level of net benefits, $502 per capita, followed by Regions 2, 5, 1, 4. Interestingly Region 5, which was the biggest relative loser in terms of shares of benefits versus shares of costs, does reasonably well in terms of net benefits (nearly $300 per capita), because benefits are so much larger than costs in absolute magnitude.

To examine whether or not there are any environmental justice concerns surrounding the SO_2 trading program we consider the distribution of benefits and costs received by different demographic groups. To do this, we used the demographic composition of every county in the United States, assuming that everyone in the county was equally affected by changes in pollution and by changes in electricity prices, to calculate the fraction of national benefits and national costs received by each group. Table 13.4a shows the per capita benefits, costs, and net benefits for the total population and for five different demographic groups: African-Americans, Hispanics, poor (the population living below the poverty line), kids (the population under the age of 6), and elders (the population over the age of 65). Table 13.4b then shows the ratio of benefits to costs for the different groups. The results show that both the Hispanic and African-American communities received a much larger share of the benefits than the costs, although this arises for different reasons. The African-American community pays costs similar to the overall population yet receives 20% higher benefits, while the Hispanic community receives roughly half the amount of the average per capita benefits, but pays

Table 13.4a. Benefits and costs across different populations (average per capita $1995)

Demographic Group	Benefits	Costs	Net Benefits
Total	213.1	2.1	211.0
African-Americans	253.6	2.1	251.5
Hispanics	102.0	0.6	101.4
Poor	202.8	2.2	200.6
Kids	204.9	2.0	202.9
Elders	220.8	2.2	218.6

only 30% of the average costs. Kids and elders received roughly the same share of benefits and costs as the overall population. On the other hand, the poor received slightly less of the benefits than of the costs from SO_2 reductions, which could raise some environmental justice concerns if the poor purchase as much electricity as the rich.

To further examine the distribution of benefits and costs along demographic lines, we calculated them separately for each plant in our sample, asking whether that plant's changes in emissions led to a disproportionately large increase in costs (relative to benefits) for any of these groups. For each group we then calculated the fraction of plants that had disproportionately large costs relative to benefits. These numbers are presented in Table 13.5. A number greater than 50% indicates that changes in emissions had negative effects more often than positive ones on that demographic group. Since these calculations are not weighted by plant size, they need not give the same results as those in Table 13.4. The results are, on the whole, reasonably similar to those in Table 13.4, although we do not see the poor being

Table 13.4b. Benefit/cost ratio across different populations

Demographic Group	Benefits/Costs
Total	100
African-Americans	121
Hispanics	180
Poor	93
Kids	100
Elders	99

Table 13.5. Distribution of benefits and costs across different populations (percent of plants with cost share>benefit share)

Demographic Group	Cost Share>Benefit Share
African-American	25%
Hispanic	10%
Poor	48%
Kids (6 and under)	52%
Elders (65 and older)	43%

disadvantaged here: only kids show a (very slightly) disproportionately negative effect. As in Table 13.4, the African-American and Hispanic communities do quite well—only 25% and 10% of the plants have a negative effect on these communities respectively. Therefore we conclude that there are no significant environmental justice concerns raised by Title IV, except, as noted above, the poor may have received slightly less of the benefits than of the costs from SO_2 reductions.

7. Concluding Remarks

In this chapter we analyze plant-level information on fossil-fuel fired electric utilities to examine the distribution of costs and health benefits associated with the air quality improvement achieved by Title IV of the 1990 CAAA. We examine the distribution of benefits and costs both in terms of the regions being affected and the socio-economic composition of the affected population.

Our results suggest that, as expected, the aggregate health benefits in 1995 caused by reductions in SO_2 emissions under Title IV greatly exceeded their costs. We estimate benefits of $56 billion and costs of only $558 million leading to $55 billion dollars of net benefits from the SO_2 reductions. The net benefits are positive in every region of the country, but are highly concentrated across regions. In particular, nearly 90% of the benefits and costs are concentrated in Regions 2–5 representing the northeast, north central, mid-Atlantic, and southeast. Maryland, Ohio, Pennsylvania, Washington DC, and West Virginia are the biggest winners in terms of per capita net benefits—all have per capita net benefits of $500 or above. Six other states have net benefits greater than $350 per capita: Delaware, Indiana, Kentucky, New Jersey, Tennessee, and Virginia.

In terms of the socio-economic distribution of net benefits, we find very little if any evidence for environmental justice concerns. The African-

American and Hispanic communities receive a substantially greater share of the benefits associated with SO_2 abatement under Title IV than they do of the costs (higher benefits for the African-American community, lower costs for the Hispanic community). The poor do have a slightly higher share of costs than benefits, the only (weak) evidence supporting any environmental justice concerns, but this result assumes that the poor purchase the same amount of electricity as the rich.

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Notes

1. The authors wish to thank Douglas Latimer, currently with EPA Region 8, for providing us with the Source-Receptor Matrix Model and for insightful conversations about how to use the model. We are grateful to seminar participants at Harvard and the 2004 AERE Summer Workshop for helpful comments and to Denny Ellerman and Nathaniel Keohane for providing us with cost data. Gray gratefully acknowledges financial support for this research from the Environmental Protection Agency (grant RD-83215501-0). The opinions and conclusions expressed are those of the authors and not the United States Environmental Protection Agency.
2. The only time a plant would be prevented from buying allowances to emit more SO_2 would be if that plant was located in a county which was in violation of the National Ambient Air Quality Standard (NAAQS) for SO_2, which were set at levels to prevent local adverse health outcomes. However, this has rarely posed a problem for permit trading since the Title IV cap requires a significantly greater reduction of aggregate SO_2 emissions than what is required to meet the NAAQS for SO_2.
3. According to the Office of Environmental Justice at EPA, environmental justice exists when "no group of people, including racial, ethnic, or socioeconomic group, ... bear[s] a disproportionate share of the negative environmental consequences resulting from industrial, municipal, and commercial operations."
4. Generating units face a fine of $2000 for each ton of SO_2 emitted for which they do not have an allowance.
5. Some of these smaller generators 'opted' into Phase I, under the "substitution" and "compensation" provisions, and are included in this analysis.

6. Phase I units include all 263 Table A units plus 111 units that 'opted' into Phase I—*see* section 5 Sample Coverage for details.
7. BOD (biological oxygen demand) and TSS (total suspended solids) are two commonly used measures of water pollution.
8. Emissions sources in the United States combine ground-level sources, county-level sources and individual sources. Ground-level sources were estimated for each of the 3,080 contiguous counties, while elevated sources were grouped according to effective stack height. Point sources with an effective stack height greater than 500 meters were modeled as individual sources of emissions. All the sources in the same county that had an effective stack height less than 250 meters were grouped together into a single county-level source, as were those with effective stack heights between 250 meters and 500 meters. In total there were 5,905 U.S. sources modeled in the S-R matrix (ground-level sources were also aggregated at the county level).
9. Chay and Greenstone (2003a, 2003b) examine the effect of particulate exposures on infant mortality, and obtain impacts of a similar magnitude, measured in terms of increased mortality rates.
10. Our data would readily permit the calculation to differ in sensitivity and valuation for different subpopulations—if one could generate a consensus on how to quantify such differences, a politically charged issue that we avoid here.
11. We would like to thank Denny Ellerman for providing us with these data.
12. If we had data on cross-state electricity sales, we could adjust our cost calculations to reflect this.
13. We would like to thank Denny Ellerman for providing us with these data.

14. From Sulfur Dioxide to Greenhouse Gases: Trends and Events Shaping Future Emissions Trading Programs in the United States

Joseph Kruger[1]

1. Introduction

The success of the United States sulfur dioxide (SO_2) trading program has led to worldwide interest in emissions trading. The program has become a model for policymakers in the United States and in other countries that are considering cap-and-trade programs to reduce emissions. Once a theoretical option discussed only by economists, emissions trading is now considered a mainstream policy instrument in the United States with bipartisan political support. Internationally, emissions trading is no longer considered a *crazy American idea.* It is now a fundamental component of the international framework to address climate change. Even developing countries from Chile to China are beginning to consider emissions trading programs to control conventional pollutants (U.S. EPA 2004).

Many articles and studies have examined the lessons learned from the SO_2 trading program for greenhouse gas (GHG) trading. At the same time, fundamental differences between the climate change issue and conventional air pollution problems have led a number of authors to argue that certain features of the SO_2 program should be modified to address climate change. Analysis of both the similarities and differences between SO_2 and greenhouse gases has also informed U.S. legislative and policy proposals.

While debate on the optimal design of greenhouse gas trading programs continues, there is still no consensus at the federal policymaking level on the need for a mandatory program. However, the successful experience with SO_2 trading has inspired other domestic and international initiatives. These initiatives are incorporating lessons from SO_2 trading as they anticipate and try to shape the future of greenhouse gas trading. For example, private companies are adopting voluntary targets and other initiatives, some of which are designed to influence future mandatory approaches. State governments in seven New England and Mid-Atlantic states are developing a regional greenhouse gas cap-and-trade program, which is designed to shape certain elements of a national program. States are also developing emissions registries, which may have implications for the design of a future national trading program. Finally, the European Union (EU) has recently begun operation of the world's largest emissions trading program. As a first mover on emissions trading, the EU program could set precedents and develop new features that may have an impact on the ultimate design of both the international trading regime and any potential U.S. domestic program.

In this chapter I will discuss the path from SO_2 to greenhouse gas trading from both a design and a political economy perspective. First, I will briefly summarize the literature on the lessons learned from the SO_2 trading program for greenhouse gas trading. This will include discussion of potential differences in design for a national greenhouse gas trading program that may be necessary because of the different sources, science, mitigation options, and economics of climate change. Next, I will discuss how the three initiatives mentioned above have been shaped by lessons from past trading programs at the same time they are making changes to the SO_2 model. Finally, I will conclude with an assessment of the most likely impacts from these initiatives on a future U.S. federal greenhouse gas trading program.

2. Sulfur Dioxide Trading: Lessons Learned

The SO_2 trading program has been widely studied, and there is an extensive literature examining various aspects of the program.[2] A subset of this literature has looked specifically at lessons that would apply to greenhouse gas trading.[3] In general, most authors view the SO_2 program as a successful experiment, proving that emissions trading can be an effective and efficient policy instrument. Many authors have noted that emissions trading is particularly well suited for greenhouse gases because they are uniformly mixed and do not require limitations on trading to limit *hot spots*.[4]

Analysis of the SO_2 program also sheds light on specific design elements that would be desirable to apply to a future greenhouse gas trading program. For example, experience with the SO_2 trading program has shown significant benefits from the temporal flexibility provided by banking provisions (Burtraw and Palmer 2004; Stavins 2003; Ellerman et al. 2000).[5] Second, analysts have noted that allowance distribution, particularly the allocation of allowances at no cost to affected facilities, has been critical in gaining political acceptance for the emissions trading concept (Stavins 1998; Ellerman 2005).[6] Third, authors have emphasized the importance of strong monitoring and enforcement provisions, including reasonably accurate emissions measurement, automatic excess emissions penalties that are not subject to appeal or waivers (Ellerman 2003; Swift 2001), and public access to emissions and trading data through the use of information technology and the Internet (Tietenberg 2003; Kruger et al. 2000). Finally, observers have lauded the hands-off design of the cap-and-trade model, in which regulators track emission results but don't interfere in company decisions on emissions reduction options or conduct case-by-case reviews of trades (Ellerman 2005).

While there is general agreement that the SO_2 program "proves the concept" of emissions trading, there is also considerable discussion in the literature about modifications to the basic SO_2 model that would be needed for a greenhouse gas program. Although a full discussion of these differences is beyond the scope of this chapter, these differences boil down into five types. First, analysts have noted that the ideal program for greenhouse gas trading would be economy-wide, rather than in specific sectors (e.g., electric power). This is because of the prevalence of CO_2 in virtually every economic sector and the efficiencies that arise by equalizing marginal abatement costs across the entire economy. To facilitate an economy-wide system, these analysts have argued that CO_2 emissions should be regulated "upstream" (i.e., by producers or processors of fuel) rather than "downstream" (i.e., direct emitters such as power plants and industrial facilities) to capture the largest percentage of emissions and to encompass the fewest number of sources (Keeler 2002; Morgenstern 2005).[7]

Second, a trading system should consider multiple greenhouse gases to capture the widest array of cost-effective sources.[8] For example, Reilly et al. (2003) find that inclusion of all six greenhouse gases regulated under the Kyoto agreements could provide increased emissions reduction at a lower cost.[9] Inclusion of multiple gases in Kyoto and other trading regimes is possible because of a system of global warming potential (GWP) factors, which have been adopted by the Intergovernmental Panel on Climate Change (IPCC). These GWP factors serve as an "exchange rate" to set equivalencies for the six gases regulated under the Kyoto agreement.

Third, because of the magnitude of the emission allowance assets involved, a greenhouse gas trading program should place more emphasis on the equity implications of how allowances are distributed. For example, economists often advocate auctioning, rather than free distribution of allowances for a greenhouse gas program. Cramton and Kerr (2002) describe a number of equity benefits from the auctioning of allowances, including providing a source of revenue that could potentially address inequities brought about by a carbon policy, creation of an equal opportunity for new entrants in the allowance market, and avoiding the potential for "windfall profits" that might accrue to emissions sources if allowances are allocated at no charge.[10] Bovenberg and Goulder 2001, and Burtraw et al. 2002 find that allocating only a small portion of allowances at no cost can compensate industry for losses due to a carbon policy. This finding has led some observers to conclude that a portion of allowances might be distributed to non-regulated entities, who would then sell them to regulated entities. Revenues from allowance sales would ease the cost burden from a greenhouse gas program for these non-regulated entities.[11]

Fourth, some analysts have advocated additional mechanisms to limit price risks from a greenhouse gas trading system.[12] For example, Pizer (2002) proposes a "safety valve" mechanism that would mitigate price risks by allowing sources to purchase additional allowances at a set price if allowance prices rise to that level. Kolstad (2005) finds that intensity targets, i.e., targets that index emissions to GDP or production, can reduce uncertainties associated with the cost of emission reduction under uncertain economic growth levels. Advocates of these types of mechanisms argue that CO_2 prices may be more unpredictable than SO_2 prices because there are relatively few mitigation options for CO_2, and there are currently no cost-effective post-combustion controls.[13] Moreover, CO_2 is a "stock pollutant," which accumulates in the atmosphere over an extended period. There is therefore less concern over short-term increase of CO_2 as long as the overall trajectory of CO_2 emissions is downward over an extended period.

Finally, many analysts have advocated international trading of greenhouse gas allowances, given that the atmosphere is indifferent to the location of emission reductions. International trading provides opportunities to incorporate reductions from developing countries like China and India, where analysis shows some of the lowest cost emission reduction opportunities (Ellerman and Decaux 1998). This concept is fundamental to the Kyoto Protocol, which provides for international trading among parties and for the development of project-level offsets in developing countries (Tietenberg et al. 1999).

2.1. Legislative and Policy Proposals for Greenhouse Gas Trading

Many of the features discussed above have been incorporated into legislative and policy proposals for greenhouse gas trading programs. For example, the McCain-Lieberman Climate Stewardship Act (S. 139) would cover much of the economy and would address the transportation sector "upstream" by requiring oil refiners to hold allowances. The McCain-Lieberman bill allows for emissions offsets (including non-CO_2 and biologic sequestration projects) and contains provisions for international trading, as does a bill introduced by Senator Carper (S. 843) that addresses CO_2 and other emissions in the electric power sector. Finally, an economy-wide, upstream proposal by the National Commission on Energy Policy (NCEP) includes a safety valve mechanism to limit price uncertainties (NCEP 2004).[14]

While debate continues on these policy proposals, there is still no consensus in Congress on the need for a mandatory program, and few believe that new legislation will be adopted quickly.[15] Meanwhile, the Bush administration has rejected a mandatory approach to climate change. Instead, the administration has emphasized the long-term development of "climate-friendly" technology and the improvement of energy intensity through the implementation of a suite of voluntary measures.[16]

Thus, the most direct road to a national program, i.e., new national legislation, appears to be blocked for now. Instead, a variety of other initiatives, all of which were influenced by past U.S. trading programs, are establishing "facts on the ground" that may set precedents for a future U.S. program. The following sections will explore these initiatives.

3. Voluntary Corporate Initiatives

Some companies have implemented strategies to prepare for what they believe is an inevitable carbon constraint. In part, these strategies are designed to mitigate the risks of future carbon legislation. However, some companies have also staked out positions that they hope will influence potential future mandatory programs. This section will:

- Summarize the literature on why companies may voluntarily take environmental actions;
- Review company efforts on climate change; and
- Discuss the implications of voluntary corporate actions on the design of a future greenhouse gas trading program.

3.1. Why Do Companies Take Voluntary Action?

There are a variety of reasons that corporations adopt voluntary environmental actions to address greenhouse gas emissions or other environmental issues. Some companies have attributed these actions to sustainable development goals or environmental stewardship policies (Margolick and Russell 2001). However, it is often difficult to separate these goals from economic motives (Kolk and Pinske 2004).

Less controversial is the notion that companies adopt voluntary initiatives to create financial value in one form or another. Lyon and Maxwell

(2004) propose a political economy framework for understanding corporate voluntary environmental action. In particular, they note that corporations may act in a desire to pre-empt or influence future regulation.[17] For example, adopting environmental technologies or other strategies ahead of regulatory mandates can signal to regulators that these alternatives are practical or relatively cost effective (Lyon and Maxwell 2004). Reinhardt (1999) describes how this strategy was pursued in the chlorofluorocarbon (CFC) manufacturing industry before the adoption of the Montreal Protocol. In other cases, trade associations have sponsored codes of management practices, which are partly intended to forestall the imposition of government mandates. For example, Nash and Ehrenfeld (1996) describe the Responsible Care Program, a voluntary code of management practices set up by the U.S. chemical industry that was designed as a voluntary effort to forestall the imposition of mandatory regulations on chemical production.

3.2. Voluntary Corporate Targets

A recent study found that as many as 60 U.S. corporations have adopted corporate greenhouse gas emissions reduction targets (Hoffman 2005). Some of these companies have participated in one of several partnership programs run by government agencies or non-governmental organizations (see Box 14.1). Under many of these programs, companies develop a corporate greenhouse gas inventory and adopt an emission target.[18] Some of the partnership programs allow additional flexibility in meeting a target through the use of emissions trading or the purchase of greenhouse gas offsets.

Corporate voluntary targets and participation in initiatives run by NGOs or government may have several impacts on future trading programs. First, companies may gain expertise that may be useful in assessing the impact of various regulatory policies on their companies. Hoffman (2005) finds that some companies have adopted internal emissions trading schemes or greenhouse gas measurement programs to gain expertise that will help them influence future national or international policies. Both British Petroleum (Akhurst et al. 2003) and Shell (Margolick and Russell 2001) have cited this experience as helping them gain influence in the design of the UK and EU trading systems.

Second, some aspects of voluntary corporate targets have implications for how a company will be affected by a future greenhouse gas cap-and-trade program. For example, fast-growing companies may be favored by rate-based or intensity targets as opposed to absolute targets based on mass emissions. This type of target may also favor companies with low carbon intensities. The type of corporate obligation chosen could also be designed to establish a precedent for a specific allocation methodology, such as a methodology based on historic emissions, heat input, or production output. Similarly, the "baseline protection" offered by registries may imply that a future allowance distribution will be done based on historic emissions or an output based methodology rather than through an auction.

Box 14.1. Government and private programs for emission reduction and reporting

- *Chicago Climate Exchange:* The Chicago Climate Exchange is a greenhouse gas emissions reduction and trading pilot program for emissions sources and offset projects in the United States, Canada, Mexico and Brazil. It is a self-regulatory, rules-based exchange designed and governed by the members, which include businesses, state and local governments, and other organizations. Members have made a voluntary, legally binding commitment to reduce their emissions of greenhouse gases by 4% below the average of their 1998-2001 baselines by 2006. They include around 60 businesses and around 10 other organizations.
- *WWF Climate Savers:* The NGO World Wildlife Fund of Nature (WWF) has built partnerships with individual leading corporations that pledge to reduce their global warming emissions worldwide to 7% below 1990 levels by the year 2010.
- *Environmental Defense Partnership for Climate Action:* Under this program, companies partner with the NGO Environmental Defense and declare a GHG emissions target and the management actions, policies, and incentives necessary to achieve that goal. They measure, track, and publicly report net GHG emissions.
- *Pew Center on Global Climate Change, Business Environmental Leadership Council:* Under this partnership, companies "demonstrate leadership in addressing climate change by establishing and meeting emissions reduction objectives; investing in new, more efficient products, practices, and technologies; and supporting action to achieve cost-effective emissions reductions."
- *Climate Leaders:* Under this government-sponsored program, companies develop comprehensive greenhouse gas inventories, set corporate emission reduction targets, and report annually their emissions and progress towards reaching their targets to the U.S. Environmental Protection Agency.
- *Climate VISION:* The U.S. Climate VISION program encourages industry efforts to reduce, capture, or sequester greenhouse gases. Climate VISION links these objectives with technology development, commercialization, and commercial utilization activities supported by the private sector and the government.

Finally, to the extent that corporate targets or baselines are measured from specific years, companies become invested in allocation methodologies or future targets that include these years. For example, in describing its voluntary commitment of a 5% reduction below 2000 levels, the electric power company Cinergy notes:

> "Besides the obvious commodity price risk implicit in our target, we also are bearing some risk that our efforts will not be counted within a future regulatory regime. If the baseline is established far in the future, many years beyond the year 2000, with a specific provision to disallow early reductions, we would find that we had implemented our program too early. We believe this is unlikely, though it remains a risk." (Cinergy 2004)[19]

3.2.1. Chicago Climate Exchange

The corporate initiative most directly influenced by the SO_2 trading experience and most open about influencing a future climate regime is the Chicago Climate Exchange (CCX). For more information on CCX and other programs see Box 14.1. Richard Sandor, the president of the CCX, has invoked the SO_2 experience as a model "for the design of key elements" of the program (Sandor 2002). One of the key program goals of CCX is to give companies experience with the mechanics and methodologies that will be necessary for the greenhouse gas market. Sandor (2002) also emphasizes the goal of influencing future legislation, noting that "a variety of legislative proposals have provided further indication that participation in CCX will help position participants to intelligently influence and benefit from possible future regulations." Similarly, an official from American Electric Power (AEP), a participant in CCX, has noted that one of the motivations for participation in the voluntary program is the company's interest in the development and use of greenhouse gas offsets or "off-system" reductions (Braine 2004). A report by independent directors of AEP notes that the company's participation in CCX and other voluntary efforts has given it important experience in the design of potential future regulatory programs. The report finds that

> "the company is particularly well positioned to build on this experience to advocate effectively in policy and regulatory forums for the most efficient program designs, not only for the environmental benefits, but also for the benefits to its customers over the long-term." (American Electric Power (AEP) 2004)

4. State Initiatives

Over the past few years, there has been an explosion in state programs that address emissions from greenhouse gases (Rabe 2004; McKinstry 2004).

These programs run the gamut from renewable energy portfolio standards in more than a dozen states to a proposed CO_2 emissions standard for automobiles in California. One of the most widespread categories of state activities are programs to measure, report, register, and, in some cases, trade greenhouse gas emissions. These programs raise questions about whether they will build institutions and set precedents that are helpful or harmful to future national efforts. This section will examine the motivations behind these state programs and explore the challenges and opportunities they pose.

4.1. Why Do States Take Voluntary Action?

There is extensive literature debating the appropriate level of government to address environmental problems. Much of the debate on this issue-usually referred to as Environmental Federalism-centers on weighing consistency and competitiveness issues that might arise from different environmental standards in different jurisdictions against the benefits of more localized decision making on environmental issues (Oates 2001; Revesz 2001). Authors have also highlighted the influence that state governments in the United States have had on national policy by experimenting with innovative initiatives (Oates 2001; McKinstry 2004; Vogel et al. 2005).

Additional issues arise for state climate change policies because of the global nature of the problem. First, although states or regional governments may adopt emission reduction programs to address “their share” of the global problem, their efforts may be futile if other states or national governments avoid emission reductions. Similarly, the public within these subnational entities may be concerned that non-participating regions are “free riders” who benefit from the actions of the participating areas without paying the costs (Kousky and Schneider 2003). Second, state actions raise the potential problem of “leakage” if mandatory requirements in one jurisdiction cause a shift in economic activity and emissions to another jurisdiction without mandatory requirements (Keeler 2004; Kruger and Pizer 2005). For example, limits on power plants emissions in one state or region may simply shift power generation to other regions of the country. Finally, Keeler (2004) notes that solutions developed at the state or local level may not create the institutions necessary for a robust national trading program.

Nevertheless, states have cited a number of reasons for developing greenhouse gas programs. Many of these reasons are relevant to the registry or emissions trading initiatives that have proliferated. First, some states have adopted programs to address public or other stakeholder concerns about the impacts of climate change (Rabe 2004). Second, states have expressed a desire to influence national policy or regulations in ways that benefit their companies or other stakeholders. For example, Rabe (2004) argues that some U.S. states enacted greenhouse gas policies to facilitate recognition of emission reductions by companies in the event of future national regulations. Finally, some states have developed GHG programs to spur innovative technologies, encourage economic development benefits, or create environmental co-benefits (Peterson 2004).

4.2. State Greenhouse Gas Trading Programs

4.2.1. Regional Greenhouse Gas Initiative (RGGI)

One of the most significant state climate programs is an effort by states in the northeastern and Mid-Atlantic United States that have joined together to develop a regional cap-and-trade program for CO_2 in the electric power sector. This program, known as the Regional Greenhouse Gas Initiative (RGGI), was launched in April 2003 when New York Governor George Pataki sent a letter proposing a regional emissions trading program to fellow governors. In December 2005, governors from seven states signed a Memorandum of Understanding (MOU) outlining the design elements of the program and the process by which states would individually implement their commitments under the proposal.[20] The MOU calls for the RGGI program to begin in 2009.

Organizers of RGGI have noted that the program builds upon past models of successful trading programs.[21] Although this includes the SO_2 program, RGGI is perhaps more similar to the OTC NO_x budget program, which was itself highly influenced by the national SO_2 trading program (Kruger and Pizer 2004; World Resources Institute 2005).[22] Both the OTC and RGGI programs depend upon an innovative collaboration between states that voluntarily adopt model rules that allow the states to have the compatible features necessary for an emissions trading program. The fundamental structure of RGGI is similar to past trading programs in the electric power sector. However, the RGGI program will experiment with some additional features that may be useful for a future greenhouse gas emissions trading program. Some of these features are discussed below.

Emissions Allowance Distribution: RGGI offers the opportunity to try out innovative allowance distribution features at the state or regional level, such as auctioning portions of the annual allocations or allocating some allowances directly to groups that are disproportionately affected by the costs of a cap-and-trade program. These types of approaches could set precedents for a future national program. Under the December 2005 MOU, RGGI states would be required to allocate 25 percent of allowances for a "public benefit" purpose (Regional Greenhouse Gas Initiative 2005). This could include mitigating the impacts of the program on ratepayers or promoting low- or no-carbon technologies.

Development of an Effective Offset Program: Greenhouse gas offsets could be a particularly cost-effective way to reduce the costs of a mandatory greenhouse gas program. Unfortunately, there are no effective models for offset programs to draw upon. Although project-based emissions offset programs for conventional pollutants have been around for many years, many of these programs have had limited effectiveness because of high transaction costs and uncertain environmental integrity (Swift 2002). More recently, the process set up to implement the Clean Development Mechanism, the Kyoto Protocols project-level offset provision, has been costly and cumbersome (Jepma 2005a). If states could advance an environmentally credible model with low transaction costs, it would be a significant contribution to a future national program. To this end, there have been discussions by the RGGI staff working group and their stakeholders about developing performance standards and other objective criteria that would provide clear signals to investors about the types of projects that would be acceptable (Sherry 2005). The December 2005 MOU would allow experimentation with a limited set of offset project types, including landfill methane, sulfur hexafluoride (SF6) from electricity transmission and distribution systems, afforestation, leak detection in natural gas distribution systems, methane capture for animal operations, and natural gas, oil, or propane home-heating efficiency projects. Additional project types would be added to the program over time (Regional Greenhouse Gas Initiative 2005).

International Linkages: The launch of RGGI has sparked great interest in Europe, where, as will be discussed later, an even larger experiment with greenhouse gas trading began on January 1, 2005. There have been informal contacts between state officials and officials of the European Commission and European member states to share information on how the new European Union Emissions Trading System (EU ETS) is developing. Contacts between the EU and states may provide opportunities to explore a number of "linking issues" that will be useful for any future GHG program (Fontaine 2005). Although the December 2005 MOU does not address these linking issues directly, it does include a provision that would allow the use of allowances or credits from "internationally recognized trading regimes" if RGGI allowances hit a $10/ton trigger price for two years in a row.

4.2.2. West Coast Trading Initiatives

Three western states—California, Washington, and Oregon—have launched an initiative to develop a coordinated greenhouse gas reporting system

(Rabe 2004; Pew Center on Global Climate Change 2004). These states have also indicated that they may consider a trading system. A staff report released in November 2004 recommended that the governors of the three states consider a regional market-based carbon allowance program (West Coast Global Warming Initiative 2004).

West Coast states may develop a different model for a cap-and-trade program than has been developed by RGGI. Concerns about addressing imports of power from outside the state have led these states to consider design approaches that focus on the distribution of electric power rather than generation. For example, California has adopted a proposal to allocate allowances to load-serving entities, which would be required to hold allowances to cover the emissions of the electric power they distribute (California Public Utilities Commission 2006).

Similarly, an advisory group to the governor of Oregon has recommended a tradable carbon content standard for power consumed in the state, which would take power imports into account. (Governors Advisory Group on Global Warming 2004).

4.2.3. Registries as a Building Block for Trading?

More than ten states have adopted or are in the process of adopting voluntary registries for greenhouse gas emissions (Progressive Policy Institute 2003).[23] Registries are electronic databases that track emission reductions by companies or other organizations. Under some of these programs, companies report their corporate-wide emissions and establish a baseline against which future corporate emission reductions can be counted. In other registry programs, companies simply report specific emission reduction actions without reporting their overall corporate-wide emissions. Voluntary registries have a number of benefits, including helping corporations understand the scope of their emissions and possible mitigation measures that they might take. Registries may also raise awareness of the climate change issue and highlight the actions of companies that are leaders in reducing their emissions.

Some analysts have touted voluntary registries as an important building block for a future mandatory emissions trading program (Progressive Policy Institute 2003). However, several aspects of voluntary registries may complicate the development of a future emissions trading program. First, there is an inherent contradiction in voluntary registries because these programs must balance the desire to encourage participation with the costs associated with a rigorous emissions reporting program. If measurement and reporting requirements are too rigorous and costly, there will be few participants. Conversely, if program reporting restrictions are too lenient, the resulting data may not be an appropriate foundation for a future mandatory program. Second, most voluntary registries require reporting at the company-wide level rather than the facility level. While this is appropriate for a voluntary program that tracks a corporate emissions goal, it is less useful for a sector-wide or economy-wide mandatory trading program, where it is important to carefully track emissions at the facility level. Third, the promise of "credit" for early reductions, while potentially desirable for encouraging early action and building support for an eventual mandatory program, also raises a variety of complicated issues. These types of programs raise questions about whether they are awarding credits to actions that would have happened anyway as a result of business-as-usual activities (Parry and Toman 2002). Moreover, as discussed earlier, "baseline protection" programs may imply certain methodologies for future distribution of allowances under a mandatory program.

5. The European Union Emissions Trading System

Undoubtedly, the most important development in emissions trading since the landmark SO_2 trading program has been the launch of the European Union Emissions Trading System (EU ETS) on January 1, 2005. EU officials and others have noted the influence of the SO_2 program on the EU ETS (Dimas 2005; Delbeke 2003; Zapfel and Vainio 2002; Christiansen and Wettestad 2003). Nevertheless, the EU ETS dwarfs existing U.S. trading programs in size and complexity (see Table 14.1), and the EU views its trading program as "shaping the future debate" over the use of emissions trading for climate policy (European Commission 2004a). The EU program encompasses a variety of new features and will experiment with a more decentralized approach to linking programs in the different EU member states. It is also the flagship program in Europe's efforts to meet requirements of the Kyoto protocol. As such, it has been both driven forward by the Kyoto mandate as well as burdened by some of the less desirable features of this international agreement. Potentially influential features of the EU ETS are discussed below.

5.1. Participation by New Sectors

The EU program incorporates a broader variety of industrial sectors than previous trading programs. For example, the program includes cement, lime, ceramics, and glass facilities, sources in the pulp and paper sector, and

Table 14.1. Comparison of key features of the EU ETS and U.S. programs

Features	U.S. SO_2 Program	U.S. NO_2 Program	EU ETS
Sectors	Electric Power Voluntary opt-in of industrial combustion sources	Electric Power Large Industrial Combustion Sources	Energy (including electric power, oil refineries, coke ovens) Metal ore, iron-and-steel production Minerals (including cement, lime, glass, ceramics) Pulp and paper
Number of Regulated Sources	3,000 units[a]	2,400	11,000–12,000 installations[b]
Number of Political Jurisdictions	1 (U.S. federal government)	22 (21 states and the District of Columbia)	25 member states
Emissions Covered	SO_2	NO_x	CO_2, some or all of five other "Kyoto Gases" may be added later
Project-Level Offsets?	No	No	Yes
Value of Annual Allocation	$2.25 billion[c]	$1.2 billion[d]	$37 billion[e]

[a] A "unit" is defined in U.S. trading programs as a combustion boiler. Thus, a power plant with five distinct boilers would be considered five units under the U.S. SO_2 and NO_x programs.

[b] The classification of a regulated source of emissions is different in the EU ETS than it is in the U.S. programs. An installation could consist of multiple sources of emissions that have a technical connection with the activities carried out at a site. For example, a power plant would be considered one installation, even though there are multiple boilers.

[c] Assumes an annual allocation of 8.9 million tons and an allowance price of $250/ton. (Note: SO_2 allowance prices have recently increased dramatically from this level in anticipation of the significantly tighter cap that will be required under the Clean Air Interstate Rule (CAIR) to reduce fine particulate matter. See http://www.epa.gov/cair/ for information on CAIR.

[d] Assumes an annual allocation of 500,000 tons and an allowance price of $2,400/ton.

[e] Although the size of the EU ETS cap won't be known until the National Allocation Plans for Phase II are final, Harrison and Radov (2002) cite an EU study that estimates an annual value of 30 billion ($37.5 billion) for allowances in the EU ETS.

sources in the metal ore and iron and steel industry. Although U.S. NOx programs have included some industrial sources, the scope and number of sources outside of the electric power sectors in the EU ETS are far greater. Inclusion of these sources has required the development of new emissions measurement protocols and may foster the development of additional options to reduce emissions. Moreover, the participation by officials from companies outside of the electric power sector, which is relatively experienced with energy trading in many companies, will shed light on how different corporate cultures adapt to the new organizational issues posed by emissions trading programs (Kruger 2005).

5.2. Flexible Monitoring, Reporting, and Verification

The European Union's emissions monitoring, reporting, and verification system is less prescriptive than systems used in U.S. trading programs (Kruger and Engenhofer 2006). The guidelines spell out different "tiers" of methodologies with different degrees of assumed accuracy. Firms propose installation-specific methodologies to the relevant authority in each member state. Installations are assumed to use the top tiers, but they may petition to use lower-tiered methods with lower assumed accuracy if they show that a methodology is impractical or cannot be achieved at reasonable cost. Each member state has the autonomy to grant waivers from use of the top-tier methods (European Commission 2004b).

In contrast to the U.S. trading systems, member state authorities may require companies to use private, third-party verifiers. Third-party verification may reduce government costs while providing valuable technical expertise to some member state authorities. Nevertheless, the use of third-party verification for a cap-and-trade program raises several questions, including whether third-party verifiers will provide sufficient consistency in their interpretations of the monitoring guidelines.[24]

5.2.1. Decentralized Approach

The preceding discussion of monitoring and verification illustrates how some of the features that have been decided centrally within the federal U.S. SO_2 trading program and the multi-jurisdictional U.S. NO_x programs are decentralized within the EU system. While absolute standardization is not feasible or necessary, it is an open question whether the EU ETS model strikes the right balance between consistency and national sovereignty. For example, if firms in different member states face significantly different compliance and enforcement regimes, will there be different gaming responses that undermine both the environmental credibility and the efficiency of the trading system? This may be a particularly critical question in Member States with historically weak environmental institutions, such as new EU members from former Soviet-bloc countries or some member states in Southern Europe.[25]

Ultimately, questions of standardization in the design and operation of emissions trading systems are applicable beyond the EU. To the extent that future climate regimes link different domestic trading systems, similar issues are likely to arise. Thus, the balance between centralized and decentralized features in the EU system should be closely evaluated during the pilot phase of the program.

5.3. Uncertainties Within the European Union Emissions Trading System

While the EU system blazes new territory in many areas and provides important experience, it also faces a number of challenges. Many of these challenges are byproducts of uncertainties and flaws in the Kyoto agreement. For example, banking between the first and second phases of the program is not mandatory, and member states have generally restricted banking out of concern that use of banked allowances may make it more difficult to meet the target in the first Kyoto compliance period. The lack of banking may undermine longer-term mitigation plans because firms have little incentive to implement strategies that create extra emissions reductions beyond their allocated levels.

A more fundamental difficulty raised by the Kyoto process is uncertainty about the form and level of international commitment beyond 2012. This will constrain EU member states in planning for the next phase of the EU ETS. It also makes it difficult for European industry to take a long-term approach to investing in climate-friendly technologies and to planning a least-cost, longer-term strategy for GHG abatement. Moreover, although banking will be available between the second period and subsequent periods, uncertainty over the structure of a future international regime could make member states and their industries reluctant to make the investment decisions that would enable them to take advantage of a banking provision.

There is also uncertainty associated with EU ETS linkage to project-level offset provisions of the Kyoto agreement. As noted above, utilizing low-cost emission reductions in developing countries could be a critical component of cost-effective greenhouse gas trading program. In fact, analysis of EU ETS Member State National Allocation Plans shows that there will be considerable reliance on project-level offsets in many EU member states (Zetterberg et al. 2004). Unfortunately, the provision for project-level offsets in developing countries, the Clean Development Mechanism (CDM), has been marked by problematic implementation and delays (Jepma 2005a). It is not clear that the process to review and approve CDM projects will be capable of handling the necessary number of projects to meet worldwide demand for CDM credits (Kruger and Pizer 2004). There are, however, new efforts to reform the CDM process and make it more efficient (Jepma 2005b). It is still uncertain whether these efforts will prove successful.

6. Conclusions

Few would have predicted the impact of the U.S. Acid Rain Program's experiment with emissions trading. The SO_2 trading model has influenced the development of the international emissions trading regime under the Kyoto Protocol, and emissions trading has become the primary policy instrument for addressing greenhouse gases. At the same time, the SO_2 program has provided a useful benchmark as policy makers have considered what different or additional features might be necessary for a greenhouse gas trading program. As discussed earlier, many of these features have now been incorporated into legislative and policy proposals. However, with little likelihood of quick federal action on greenhouse gas trading in the United States, the actors discussed in this chapter are filling the vacuum by testing approaches and staking out positions that will likely have an important influence on future efforts on greenhouse gas trading.

What are the implications of these initiatives for a future U.S. national program? First, these programs offer opportunities to experiment with some of the mechanisms that will be needed as we advance from the relatively simple SO_2 trading model to a much more complex greenhouse gas model. For example, experiments with new offset and emissions reporting methodologies for non-CO_2 gases under state and company initiatives could provide valuable insights. Similarly, exploration of new allowance distribution methodologies under the RGGI initiative offers the opportunity to improve understanding of the distributive issues associated with allowance distribution. Finally, experience with linking domestic programs in the European Union and with linking the EU ETS to project-level offsets under the CDM could be fruitful for exploring certain aspects of the architecture of future international agreements.

The three initiatives discussed have been less trailblazing in addressing some of the other design features that are unique to greenhouse gas trading. For example, although the EU ETS covers a broader array of sectors, it still covers less than half of Europe's CO_2 emissions. In particular, it does not include emissions from the fast-growing transportation sector, and it is reportedly unlikely that the transportation sector will be added in the second phase of the program (Carbon Market Europe 2005). There has also been no experimentation with cost-limiting mechanisms such as the safety valve.[26]

Second, efforts to stake out potential design elements such as target types and allocation methodologies will likely grow as more companies begin to believe that a carbon constraint is inevitable. Whereas allowance allocation was a relatively new phenomenon when the Clean Air Act Amendments were passed in 1990, U.S. companies now have more understanding of the financial implications of these methodologies. The desire to influence future allowance allocations by establishing early reductions or signaling a preference for future allocation methodologies could build political support for an eventual mandatory program. On the other hand, the establishment of competing claims on allowances could also complicate the eventual development of such a program and could make it less likely that mechanisms such as auctions will be utilized.

Third, some have argued that the proliferation of state approaches to registries or trading may hasten the advent of a national program by creating a "patchwork" of state programs that cries out for federal harmonization.[27] However, with an issue as controversial as climate change policy, there is also a danger with this approach. Just as an early experiment with electricity restructuring in California may be one of the factors that has derailed momentum for national restructuring legislation (Joskow 2003), unsuccessful attempts by states on greenhouse gas trading could give ammunition to opponents of a national program. While experimentation at the state level is healthy and useful, these experiments must be successful to prove the concept for a national approach.

Given its size and visibility, the perceived success or failure of the EU ETS is even more significant. The degree to which it succeeds or fails is likely to influence deeply any future international attempt to reduce greenhouse gas emissions and the climate change that they cause. Moreover, if the EU trading program can demonstrate relatively predictable allowance prices and effective institutions for ensuring credible emission reductions, it will be more likely that the United States will adopt a program sooner rather than later. Conversely, if the EU program is viewed as excessively costly or ineffective, it may be a longer road from SO_2 to greenhouse gas trading in the United States. On the other hand, unpredictable or higher than expected allowance prices in the EU system could increase the likelihood that a future U.S. greenhouse gas trading system would contain a safety valve mechanism to limit price uncertainty. Ultimately, this shift to a price-based policy could be the single largest change from the SO_2 cap-and-trade model.

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Notes

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2. For economic assessments of the SO_2 program *see* Ellerman et al. (2000), Carlson et al. (2000), and Burtraw and Palmer (2004). For an independent assessment of the benefits of the program, *see* Burtraw et al. (1998). U.S. EPA 2005 summarizes the environmental impacts of the program.
3. *See*, for example, Stavins (1998), ELI (1997), Tietenberg (2003), and Ellerman (2005), and Burtraw et al. (2005).
4. In other words, climate change is affected by the total amount of emissions into the atmosphere, rather than the location of the emissions source. For example, the atmosphere is indifferent as to whether a ton is emitted in Chicago or Beijing.
5. In contrast, the lack of an adequate banking provision in the RECLAIM trading program in Southern California may have been at least partially responsible for extreme price volatility following high electricity demand in 2000. *See* Ellerman et al. (2003).
6. In the U.S. SO_2 program, an *allowance* is the right to emit one ton of SO_2. The term is also used for the right to emit one ton of CO_2 in the EU ETS. Economists sometimes refer to allowances as *tradable permits.*
7. A hybrid system with both upstream and downstream elements is also possible (Environmental Law Institute 1997, Hargrave 2000). *See* Baron and Bygrave (2002) for a discussion of the advantages and disadvantages of different points of obligation for holding allowances.
8. Stavins and Richards (2005) find that biologic carbon sequestration is also a cost-effective strategy that could be part of a climate mitigation regime.
9. In cases where it may be difficult to measure total mass emissions from these sources but relatively easy to measure emission reductions (e.g., reductions of methane from a landfill) these sources might be captured with project-level offset provisions rather than through inclusion in the cap-and-trade program (U.S. EPA 2003).
10. Although much of the discussion about allowance distribution focuses on the equity implications of different approaches, Goulder et al. (1999) and Dinan and Rogers (2002) found that recycling revenues from auctioned allowances could also have economy-wide efficiency benefits if they are used to reduce certain types of taxes.
11. For example, in an upstream program, electric power companies or energy intensive industries might not be required to hold allowances. However, they might face increased costs that they would not be able to pass on fully to their customers. For a discussion of these issues, *see* Domenici and Bingaman (2006).
12. Allowance banking can also serve this function by creating a cushion that will prevent price spikes and hedge uncertainty in allowance prices (Jacoby

and Ellerman 2004). Jacoby and Ellerman (2004) also note that some environmental groups have opposed a safety valve because it creates less certainty about the quantity that will be emitted.

13. In the longer term, there may be cost-effective technologies to remove and sequester CO_2 from combustion. *See* Newell and Anderson (2004).
14. Senator Jeff Bingaman of New Mexico developed draft legislation based on the NCEP proposal (Eilperin 2005).
15. The Senate has adopted a non-binding resolution calling for a mandatory market-based program to address greenhouse gases (Blum 2005). In May 2006, the House Appropriations Committee, but not the full House, adopted a similar non-binding resolution.
16. For a more detailed description of the Bush administration climate change plan, *see* http://www.whitehouse.gov/news/releases/2002/02/climatchange.html.
17. There are also non-political motivations for voluntary corporate actions. Drivers of voluntary action described in the literature include: (a) the desire to limit future regulatory risk (Margolick and Russell 2001); (b) the desire to reduce costs through practices that also have environmental benefits (Esty and Porter 1998); (c) desire to differentiate a company or its products on an environmental basis (Reinhardt 1998); and (d) the desire to enhance employee morale and motivation (Reinhardt 2000).
18. These targets take different forms. Companies such as U.S. electric power generators American Electric Power and Energy have adopted absolute targets based on mass emissions corresponding to a base year (King et al. 2004). Other companies, such as Rio Tinto and IBM, have adopted intensity targets, under which they commit to surpass a benchmark of emissions or energy use per unit of production or sales (Margolick and Russell 2001).
19. Cinergy has recently announced a plan to merge with Duke Energy, which declared its support for a carbon tax in April 2005 (Borska 2005).
20. The seven RGGI states are New York, New Jersey, Connecticut, Vermont, New Hampshire, Delaware, and Maine. Two additional states that had been participating in the RGGI development process, Massachusetts and Rhode Island, declined to sign the MOU. In April 2006, Maryland adopted a law requiring the state to join RGGI.
21. RGGI organizers have made building on past successful trading programs one of their “guiding principles.” *See* http://www.rggi.org/goals.htm.
22. The NO_x budget program is a multi-jurisdictional partnership between federal and state governments. It has evolved in geographic scope over time, first encompassing nine northeastern states in the late 1990s. In 2004, it was expanded to include 19 states and the District of Columbia with two additional states added in 2005. The initial 9 state program is referred to as the Ozone Transport Commission (OTC) program and the expanded 21 state program is sometimes referred to as the NO_x SIP Call Program.
23. There is also a national registry set up under section 1605(b) of the Energy Policy Act. *See* http://www.eia.doe.gov/oiaf/1605/frntvrgg.html.

24. A more extensive discussion of these issues appears in Kruger and Pizer (2004) and Kruger (2005).
25. For example, Blackman and Harrington (2000) have described some of the difficulties Poland has had with enforcing its emissions fee system. Tabara (2003) argues that Spain's environmental administrative capacity has not always been adequate to face complex problems such as climate change.
26. The one exception is Canada, which has proposed a safety-valve mechanism for its domestic emissions trading program. *See* Government of Canada (2002).
27. For a discussion of this political dynamic, *see* Swanson (2004) and Lee (2003).

Part IV

Lessons Learned and Future Prospects

15. Atmospheric Deposition and Conservation: What is the Role for Conservation Organizations?

Timothy H. Tear

1. Introduction

The world of conservation is rapidly changing. The increasing severity and scope of widespread anthropogenic threats to biological diversity can now be seen to impact even local conservation efforts. As these impacts become more visible in our backyards, a new reality has begun to shake the status quo. Brave actions to reverse these disturbing trends are being called for that fall outside traditional operating procedures. One need only consider the issue of global climate change to understand the pattern. In less than a decade, discussion of the potential impact of global climate change has progressed from a largely theoretical debate in halls of academia to a fact being reported in local newspapers. Meanwhile, on-the-ground conservation efforts are awakening to the reality that they have to play catch-up.

This chapter describes the history of another anthropogenic threat to biodiversity conservation,—the atmospheric deposition of air-borne pollutants—in relation to one conservation organization, The Nature Conservancy. As a single case study, it is symbolic of a growing trend in conservation. An ever-increasing body of science more accurately and directly links the consequences of human society's actions to the very natural resources upon which we depend for survival. As this evidence mounts, it calls into question the past actions and future roles and responsibilities that conservation organizations must play to conserve not only biodiversity for its intrinsic value, but also the essential ecological services provided by natural resources that are central to our very survival.

2. Past Performance—Three Monkeys and a Voice?

Acid rain and its impacts are not new to science or society. As is well known, acid rain was first discovered in England over 130 years ago. It took nearly a century for it to be recognized in the United States. Much of this recognition is credited to Gene Likens, founder and current Director of the Institute of Ecosystem Studies, and father of the term *acid rain.* Gene

Likens recently told the story of how he personally approached The Nature Conservancy over 20 years ago in an attempt to get the Conservancy interested in the acid rain problem. To him, it made perfect sense. The Conservancy as a large land owner, was being negatively impacted by acid rain, and therefore should be speaking out against these impacts to its investments. Nearly a quarter century later, we are just beginning to understand the significance of this lone voice and to heed his sage advice.

The obvious question is "why didn't the Conservancy act earlier?" There are a host of possible political and social issues that could explain this hesitancy. Let's consider four factors that contribute to answering this question.

2.1. See No Evil?

The impacts of atmospheric deposition are becoming more obvious in day-to-day life and reinforce local to global links. In many land trusts and other property-owning conservation organizations, the people on the ground have historically addressed those threats that can be seen or felt, and that hit closest to home. In many cases, these conservation practitioners have mobilized to fend off habitat fragmentation and habitat loss to important areas by setting aside land in perpetuity. They have even mounted impressive efforts to keep out many invasive weeds, and have replanted forests and prairies that have been lost. The goal has been clear and obvious; preserve the land from the shopping mall or subdivision.

For many years, the threat of atmospheric deposition has not had a significant and wide-spread enough visual impact to evoke the necessary visceral response. As one astute scientist recently pointed out at a conference on this very issue, "I drove here past miles and miles of beautiful green forest—most people don't see a problem!" There has not been an obvious enough impact to evoke the crisis response that would induce many conservation organizations to react. This lack of visual impact stems from the fact that many of the most significant ecological impacts have been underwater. These are well-documented in the Northeast (e.g., Driscoll et al. 2001a, b) and Southeast (e.g., Sullivan et al. 2002). From a terrestrial perspective, much of our knowledge has concerned the alteration of biogeochemical cycles, not in the loss of plant or animal species that society has come to know and value. Without being able to easily see, touch, or taste the threat or its impacts, the response of conservation groups large and small has waxed and waned while society remains apathetic.

But when fish continue to disappear from our waters from chronic and episodic acidification, fishing restrictions from mercury warn us of the hazards we can't see, and dominant trees such as economically important sugar maples and mountain-top spruce trees are dying or show signs of damage, conservationists begin to see and feel these impacts in their own backyards. These atmospheric threats that come from afar now affect our backyards, bringing this issue to the forefront of day-to-day life and work.

2.2. Hear No Evil?

Justifiable or not, a head in the sand strategy for addressing atmospheric deposition is no longer defensible. Understanding the nitrogen cycle, mercury methylation, ground-level ozone production, aluminum absorption, and food web dynamics are just a few of the key components of this issue that leave many concerned conservationists hoping that someone else can be responsible for understanding these complex biogeochemical interactions. Such complexity inhibits the relevance of this issue to much of society. It is easy to avoid trying to comprehend the complexities of how atmospheric deposition impacts our environment, especially when much of this information was relegated to relatively obscure scientific publications and was not reported in the popular press. In recent years, much more easily digestible information has become available to conservation practitioners and the general public that distills these complex chemical equations and intricate biological feedback loops. As pointed out by Janetos, Driscoll and Dillon separately in this volume, the ecological impacts of atmospheric deposition are well known, and making this knowledge more readily understandable and digestible is critical in raising awareness and urging action.

The Hubbard Brook Research Foundation produced one of the more influential publications, *Acid Rain Revisited* (Driscoll et al. 2001a), which was derived from Driscoll et al. (2001b). As described by Driscoll et al. (this volume), the impacts of acid rain in the Northeast are profound. First, we have learned that the impacts of acidification cause even greater damage than originally expected. Soils have been altered, trees stressed, and lakes and streams have been impaired. The terrestrial and aquatic ecosystems that conservationists try to conserve are still at risk. Second, although the Clean Air Act has helped reduce this threat, emissions and deposition remain at high, detrimental levels, and pollution abatement has not gone far enough to sustain ecosystem recovery. The third, and perhaps most important message, is that recovery is attainable, and is directly related to the timing and degree of emissions reductions.

These messages are immediately useful for conservationists, and are instrumental in motivating change. With credible, scientifically defensible information readily available, it is easier to build the case for action and harder to keep our collective heads in the sand.

2.3. Speak No Evil?

Billions of dollars of conservation investments are not secure. This is perhaps the strongest motivation to spur conservation organizations to act. A rough estimate of the land value alone of a century's conservation work by private, state and federal agencies in the Northeast is now easily billions of dollars. These areas were set aside to secure lands and waters for conservation in permanent conservation protection. Yet even in some of the most remote and well protected lands, these investments are suffering from significant impacts that threaten the very reasons they were established (Figure 15.1). The clear links between atmospheric deposition and degraded ecological health has enabled the Nature Conservancy to now see and hear the message delivered by Gene Likens nearly a quarter century ago. The time has come to speak out, and play a much more active role in reducing this threat.

One of the more notable examples of this new approach came in 2001 in the Catskill Mountains of New York, where the Eastern New York Chapter

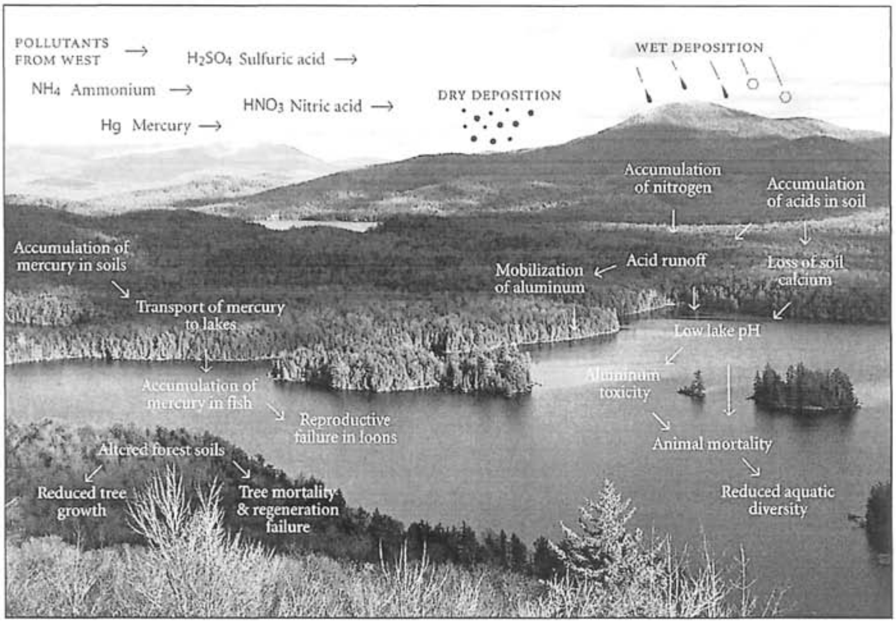

Figure 15.1. A conceptual illustration of the ecological impacts of acid deposition to protected areas in the Adirondacks. Conservation organizations and the general public have invested millions of dollars for over a century to conserve these areas. (Courtesy of the Adirondack Nature Conservancy and Adirondack Land Trust – photograph by Bill Brown; conceptual diagram by Jerry Jenkins)

of the Nature Conservancy organized a meeting of 20 conservation groups to chart out the most essential conservation work left in this large, relatively well-protected area. The Catskill Park, which encompasses 705,500 acres, celebrated its one hundredth anniversary in 2005. Within the Park, some 290,000 acres of forest have been designated as forest preserve, receiving the highest possible conservation status. Yet closer examination reveals a forest in decline, with red spruce trees showing visible signs of damaged from the cold on Hunter Mountain, beech bark disease prevalent at lower elevations, insect outbreaks occurring in the remnant hemlock forests, and severely leached soils and degraded streams. Should we be satisfied with the fate of this forest after investing in total protection of hundreds of thousands of acres as *forever wild* for a century?

One of the top conservation strategies identified collectively by this group of conservation organizations was the need to address atmospheric deposition as a threat to the entire Catskills ecosystem. As the forests of the Catskill Mountains provide significant ecosystem services, not least of which is the water supply for 9 million people in New York City, this is no small concern. To the planning participants, the links to the sources of ecosystem stress were obvious, and the significant, century-old conservation investments were not secure.

The Catskills are just one of many places in the Northeast where conservationists are reaching similar conclusions. Long-term monitoring in the Adirondack Park shows that significant detrimental impacts from atmospheric pollutants continue to degrade one of our nation's treasured conservation jewels (Jenkins et al. 2005). In addition, a pilot study released by the Northeast Governors and Canadian Premiers (NEG/ECP 2003) suggests that under current emissions levels, 31% of the forest lands in Vermont and 23% in Newfoundland are at risk of cation (calcium, magnesium, and potassium) depletion. In Quebec, a 19-year study of northern hardwood and boreal coniferous forest stands showed significantly lower forest growth rates where deposition levels exceeded a critical load. Approximately a third of forest productivity was lost due to atmospheric deposition (Ouimet et al. 2001 as reported in NEG/ECP 2003). Many of the impacts to watersheds from atmospheric deposition, including acidification and nitrogen saturation, ultimately end up in our coastal marine ecosystems, where additional degradation occurs, such as eutrophication and algal blooms (Driscoll et al. 2003a, b). In short, many of the sources of pollution and their negative impacts are known. The simple result is that our collective natural resource investments, whether for preservation or consumption, are not secure.

2.4. The Time is Right

Since Gene Likens' voice was first heard two decades ago, the capacity of conservation organizations to address this issue has increased dramatically. Within the Conservancy, an order of magnitude growth in personnel has brought about a strong investment in science that provides the necessary capacity to better address this problem. For example, when Gene Likens first approached the Conservancy, the local chapter where the Institute of Ecosystem Studies is located had a staff of only two, and neither were scientists. Now the Eastern New York Chapter of The Nature Conservancy has a staff of approximately 40, with at least five staff dedicated to science and ecological management issues. Similarly, the Conservancy as a whole has expanded from several hundred employees to several thousand, with a strong investment in science staff capacity that has made it possible to finally hear the concerned voice of Gene Likens and place it in the appropriate context for action. The Conservancy recently decided to officially sanction work on public policies concerned with atmospheric deposition. As a non-advocacy, not-for-profit organization, this is a significant decision. It marked an end to an era of See No Evil, Hear No Evil, and Speak No Evil.

3. Shifting the Emphasis to Include Ecosystem Health—a Viable Role for Conservation?

We know that reductions in air pollution set in place with the 1990 Clean Air Act Amendments are not enough to sustain ecosystem recovery (Driscoll et al. 2001a, b; and this volume). As Driscoll et al. (2001a) predicted, while chemical recovery may start within years to decades following reductions, the delay in biological recovery could take decades and more likely centuries. Of critical importance is that a host of other global issues have arisen—most notably global climate change—that effect our perception of the *recovery clock.* The longer the delay before ecosystem recovery starts, the greater the likelihood that this recovery may not produce what we are hoping for. For example, in some areas like the Catskills, leaching of calcium from calcium poor soils has been going on for many years. It is possible that at some point, the loss of calcium may be so significant that it may be lost as an essential element from this forested ecosystem. Similarly, it is possible that excess nitrogen deposition combined with other human-induced nitrogen inputs may tip the balance so much that nitrogen will no longer be the limiting factor in some ecosystems. At that point it is unclear what ecosystem recovery would look like. As fundamental ecosystem building blocks change and alter complex biogeochemical cycles, recovery under these new conditions is not well understood.

It is in this context that a stronger role for conservation is emerging. The recent National Research Council report on "Air Quality Management in the United States" called for a greater understanding of ecosystem impacts (NRC 2004). In particular, it recommended enhancing the protection of ecosystems and public welfare, arguing that "many of the programs and actions undertaken in response to the [Clean Air Act] CAA have focused almost entirely on the protection of human health. Further efforts are needed to protect ecosystems and other aspects of public health...Although mandated by the CAA, the protection of ecosystems affected by air pollution has not received appropriate attention in the implementation of the act. A research and monitoring program is needed that can quantify the effects of air pollution on the structure and functions of ecosystems. That information can be used to establish realistic and protective goals, standards, and implementation strategies for ecosystem protection."

In order for such recommendations to succeed, a stronger link is needed between science, ecosystems and their services, and society. Greater awareness is needed in the general public of the close ties between healthy ecosystems and healthy human communities, as we have begun to do in the United States with wetland protection. Such an emphasis would seem appropriate given the growing debate about conservation, ecology, and sustainability and the role of ecosystem services (e.g., Kremen 2005).

The NRC (2004) report calls for better monitoring of impacts, and the lands and waters under the ownership of conservation organizations could become the laboratories of the future. There is a need to fund such monitoring efforts and conduct them in a coordinated way so that regional analyses that rely on multi-jurisdictional databases are possible. Conservation organizations should support these efforts because the results of existing long-term monitoring helped make us aware of the problem, moved us to action, and provided some insight into the effectiveness of those actions.

Finally, conservation organizations need to speak out in support of public policies that will achieve ecosystem recovery. Such actions must start soon if we are to see signs of ecological recovery in our lifetimes. In this context, there are several potential roles for conservation organizations in turning this threat from another doom and gloom prediction into a conservation success story. Success will require that conservation organizations, large and small, share these roles.

3.1. Raise Public Awareness About Ecosystem Impacts

Conservation groups could take a more active role in informing their members about the impacts of atmospheric deposition to the ecosystems that support society. Although mountains of research exist on the specific impacts of particular pollutants on a few well studied places, there has been relatively little consolidation of this information into a format that is readily available, and understandable, to the general public. The membership of conservation organizations represents an important and potentially vocal constituency that requires a much more concerted effort if they are to be effectively mobilized.

The Conservancy is well-positioned to make an important contribution to this cause. As part of their large-scale ecoregional assessment process, the Conservancy and many partner organizations have identified lands and waters critical to the conservation of biological diversity in ecoregions all over the world. Ecoregions are large areas of land and water defined by geographically distinct assemblages of natural communities, sharing a large proportion of species, environmental conditions, and ecological processes (Groves et al. 2002; Groves 2003). By assessing the degree of threat that atmospheric deposition (i.e., nitrogen, sulfur, mercury, and ground-level ozone) poses to this extensive collection of sites across the eastern United States, the Conservancy hopes to demonstrate for the first time the severity and the scope of this issue to biodiversity (Tear et al., in prep).

3.2. Promote Stronger Collaboration Between Academia, Non-Governmental Organizations, and Public Policy Decision Makers

It often takes many years of research, analysis, and writing to produce credible scientific articles in peer-reviewed journals. By comparison, critical policy decisions can sometimes be made over the span of months, weeks, or even days. The scientific literature on the impact of atmospheric deposition on biological diversity is vast, dispersed, often compartmentalized into various scientific specialties, and usually difficult to understand for non-scientists. Policy work requires synthesis and a solution oriented approach that is widely understood. Non-governmental organizations may be able to play an important role in improving the flow of information from science to policy.

3.3. Maintain and Expand Long-Term Monitoring

There are few long-term monitoring stations tracking atmospheric deposition levels, and funding for these stations is constantly at risk. There is a need to sustain lobbying pressure to insure that state and federal support for monitoring stations continues and expands. In areas where large holes in the existing monitoring network are identified (such as mercury), expansion to cover these areas should be considered. Furthermore, the identification of critical conservation questions should help to guide additional information that could be gathered by these monitoring networks. Finally, we must sustain these monitoring stations as they provide society with the critical information necessary to evaluate whether broad public policies are effective enough to achieve society's goals for cleaner air and healthier human and natural communities.

3.4. Support Critical Research

Support of ecological research into the impacts of atmospheric deposition on biological diversity has waned in recent years. Yet the synergistic relationships expected between acid rain, mercury deposition, ground-level ozone concentrations, pest and pathogen outbreaks, and global climate change are poorly understood. There is a clear need to promote continued research into these and other areas if we are to truly understand the magnitude and intensity of the problem that atmospheric deposition presents to society and the ecosystems that support it.

4. Reading the Tea Leaves: Is Poor Past Performance a Good Predictor of Future Involvement? Overcoming Major Challenges in a New Era for Conservation

Another possible reason that a see no evil, hear no evil, speak no evil period persisted was the lack of a clear role for conservation. Scientists were busy quantifying the impacts to biogeochemical processes and impacts on species wherever possible, and policy was aided, and some would say driven, largely by society's response to the negative human impacts that were occurring. The Clean Air Act and its amendments were passed largely without the aid of the conservation community. Apathy in the conservation world could persist at minimal cost when there was no obvious role or need.

Yet even if there is a clearer role for conservation groups to play, what indications do we have that poor past performance will not be the predictor of future actions? I propose that there are three compelling reasons why it is possible to bet on a stronger, more positive contribution from conservation organizations in the years to come. Each one has its origins in earlier doom and gloom conservation messages, but instead of hopelessness, they evoke a strong call to action.

4.1. Conserved as a Concept is Dead ... but Long Live Conservation

For many years, there has been a simple assumption that conservation organizations can invest in one place, and once it is *conserved,* move on to the next place that is in need of help. This model has worked well for the Conservancy, which has built a core competency of responding quickly to land acquisition opportunities. In this model, once lands were purchased and *protected* by the Conservancy, they could be transferred to another appropriate conservation organization, freeing up the funds to be used for another acquisition. However, the time may have come when we must view conservation work differently. The harsh reality is that conservation work to insure that any place is effectively conserved is never done. Instead, the risks of increasing globalization and the associated host of threats (including global climate change, habitat loss and fragmentation, atmospheric deposition, and increased pest, pathogen, and invasive species movement and outbreaks, to name a few) have forced us to concede that no place on earth can be labeled as permanently conserved in the past tense. Instead, conservation work must be viewed as the continuous and unrelenting process—in the present tense—to conserve those places and the biodiversity they support over the long haul. The key question is whether effective conservation continues to occur in these priority places. In this new context, effective conservation is defined by management capable of adapting to new information, particularly about new or changing threats, and responding accordingly by mounting conservation strategies that ultimately result in the long-term persistence of biodiversity.

There are indicators that such a transition is already underway. For many years, the conservation world has been emphasizing bigger and broader ecological boundaries over the political boundaries of the past. For example, ecoregions have been embraced by a number of conservation organizations as useful constructs for ecological assessment at continental and global scales (Groves et al. 2000, 2002; Groves 2003). This move toward ecological, rather than political, boundaries was partially motivated by a need to respond more appropriately to the ecological processes and threats that are shaping the natural world. An increasing awareness of the magnitude of these global issues is beginning to sink in, helping to drive more conversations toward addressing these daunting issues, as opposed to moving on to the next new place.

Similarly, there is growing recognition that in order to be truly effective in conservation, we must participate in stronger partnerships, as no single organization can go it alone and make substantive progress on many of these issues. The conservation vision for The Nature Conservancy states explicitly that we must work through partnerships to achieve success. How to best accomplish this is an ongoing experiment. The challenges of cross-cutting or multi-site strategies pepper the Conservancy's jargon as we learn how to reach out beyond our own borders. In this volume, the same concepts are being worked on in the policy arena, as Munton, Schreurs and Andonova each describe in detail the transboundary issue and the lessons that have already been learned. In short, the challenges are known and the opportunities for greater collaboration are apparent. We simply need to reach out and make them happen.

4.2. Global is Local

The old expression *think globally, act locally* has a new twist. In today's increasingly global economies, the lines between global and local are being progressively blurred. In many ways, when thinking about the threats that conservation organizations face to the lands and waters they strive to conserve, the laundry list is now easier to see and understand, as the visual signs of these threats are beginning to appear in our back yards. Altered biogeochemical cycles resulting from atmospheric changes favor many invasive and exotic species, and increase the opportunity for pest and pathogens to take hold. Technological advances in transportation increases global trade, resulting in more and more exotics landing on the doorsteps of our most cherished protected areas. The spread of the generalized, opportunistic species and the homogenization of our ecosystems is one of the most dangerous and challenging threats we face. The conservation of biological diversity, in any place, is now inextricably linked due to the global economies we now sustain. Global *is* local, and local *is* global.

The conceptual blurring of local and global boundaries may be partially responsible for smaller conservation projects overcoming past apathy and increasing their willingness to invest in broader strategies to reduce large scale threats. For example, within the Conservancy local project teams have conducted formalized assessments of threats to the biological diversity at their sites. In the past, the vast majority of site conservation planning processes ignored large-scale threats. I believe this occurred for at least three reasons. First, impacts like atmospheric deposition and global climate change were not immediately visible, and hence did not evoke a visceral response to land managers and planners that something must be done. Second, there was little organizational knowledge about the complex biogeochemical processes or advanced climate and transport modeling to enable effective use of available information. Similarly, most of these studies lacked a direct link to on-the-ground conservation activities. Third, and perhaps more importantly, individuals and even teams of people—both at specific sites and even for entire ecoregions—could not see how they could really have an impact on these threats. Being action-oriented conservationists, they opted to leave it out due to a sense of helplessness that there was really "nothing they could do."

Only in the past few years are these more global threats commonly reported. The internet and the rapid increase in information sharing have dramatically improved the information available to project teams in the field. There is greater acceptance that these threats are real. For example, global climate change is now accepted as a reality by the vast majority of scientists, and the Kyoto protocol is recognition that actions must be taken to address its impacts on humans and ecosystems. Finally, science continues to provide more information to further reduce the uncertainty surrounding some of these larger-scale threats. Initial skepticism about climate change fades with each passing day.

But perhaps most importantly, the feeling of helplessness is gradually dissipating. Because of the pursuit of greater accountability and credibility in conservation assessments, the availability of more information and the prevalence of real, visible impacts on these sites, more project teams are now reporting these broad threats at their sites even if they are not sure what to do about them. For example, at a recent workshop series on forest conservation in the Northeast focusing on seven Conservancy-lead large landscape projects, atmospheric deposition was one of the top four major threats identified (in addition to development pressure, global climate change, and invasive pest and pathogens). Until recently, it was the only one of these critical threats that the Conservancy had taken no action to address. Five years ago, it is unlikely that a collection of forest conservation project teams would have identified atmospheric deposition as a critical threat. Information that clearly demonstrates direct impacts to projects on the ground, where we make substantial investments to accomplish conservation, dramatically increases the level of awareness that a problem has on our work.

4.3. Measurable Progress is Evident

Perhaps the single, most important contribution the current atmospheric deposition science and policy work discussed in this volume can have is on the conservation community. First, we must reject the approach of doom and gloom predictions in the past, which has lead to a backlash of criticism that ecologists are *crying wolf.* (e.g., Mann 1991; Mann and Plummer 1992; see also Redford and Sanjayan 2003). Early predictions of dramatic and catastrophic ecosystem collapses and species extinctions were intended to heighten the urgency of the message, grab society's attention, and focus on the problems at hand. Instead, as the work in this volume suggests, the overall status of our ecosystems are declining through the less obvious yet perhaps more insidious problem of soil and cation depletion, combined with multiple stresses from global climate change and globalization, that may ultimately deprive us of the opportunity—which still exists—for ecosystem recovery. As the conservation world struggles to show conclusive successes in the face of these daunting threats, the work on atmospheric deposition gives us not just a reason for hope, but the data to prove the effectiveness of our actions. As Driscoll et al. (2001a, b) have shown, if emissions are reduced enough, ecosystem recovery is possible, as chemical recovery in some locations is already evident. We know how well the 1990 CAA worked, and how well it didn't work. We have the information to plot out our recovery, and it is clear from the evidence that recovery is possible. In a time when it would be easy to slip back into the doom and gloom mentality, this work shows us unequivocally that recovery can happen as long as society has the will to make it happen. By making the right choices, and continuing to curb emissions, we can show success. This could become a critical precedent to set for society as we move towards the larger and related challenges ahead of fending off global climate change.

The conservation community can help communicate such highlights and indicators of success. We can help to continue raising awareness that acid rain, and atmospheric deposition of many pollutants, is a global problem. While not yet the severe problem that it is in the eastern United States, episodic acidification of high elevation headwater streams and ephemeral ponds, tied to high acidity pulses during the spring when sulfate and nitrates are released from the melting snow pack, is a growing concern in the western United States. The issue appears to be mostly limited to alpine and subalpine ecosystems in areas with poorly buffered soils, where nitrogen deposition is of increasing concern (Fenn et al. 2003a, b). Officials from the Rocky Mountain National Park in Colorado recently concluded that air quality passed a critical threshold point, saturating plants and soil with nitrogen compounds and creating an unacceptable risk of ecosystem damage. Finally, acidification of the world's oceans is now a significant concern, as it could alter many important ocean ecosystem processes, including the ability of shellfish to produce shells, and coral to grow and regenerate (Doney 2006).

Beginning in the 1970s, acid rain was identified as a serious issue in Europe, most prominently in Scandinavia and in the Black Triangle, a large swath of Poland, the Czech Republic, and southeastern Germany that suffered from acute forest damage induced by acid rain. Acid rain is a rapidly growing concern in Asia, particularly in China. As of 2001, China accounted for 25% of the world's coal combustion, an amount that is projected to continue to rise. China now is the largest sulfur-emitting country in the world. By the mid-1990s, 30% of China's land area was affected by acid deposition. Recent studies have documented significant acid deposition impacts on forests and aquatic systems, and on the productivity of agricultural lands across broad regions of southern China. China adopted its first acid rain policy in 1998, setting a goal of reducing sulfur emissions by 20% from 2000 levels. Before 1998, sulfur emissions had been essentially unregulated. Acid rain is also a large and growing problem in many other Asian and Pacific region countries, including India, Thailand, Korea, Vietnam, and Japan. Sulfur dioxide emissions in Asia are now greater than those of North America and Europe combined, and these emissions are expected to as much as triple by 2010 (compared to 1990 levels) if current trends continue. There is little question; atmospheric deposition is a global problem.

We have the opportunity here in the northeastern United States to use our extensive information to document that ecosystem recovery is not only possible, but that through conscious action, we made it happen. For those areas in the world that may not be able to invest in such extensive monitoring networks to detect change, this effort could provide exceptionally important information to guide decision making in other regions that may suffer similar impacts but lack the resources to assess the effectiveness of their actions.

In this context, we not only blur the lines between local and global, we blur the lines between public and private, science and policy, for-profit and not-for-profit. In a world of growing global threats, society desperately needs a success story. Perhaps no other issue is as well poised for success as the reduction of atmospheric deposition. Yet the only way we will achieve this goal is through stronger partnerships that continue to advance science and policy through the support of an informed public. We need to show that society ultimately values ecosystem recovery enough to bear the costs it will take to get there.

5. Conclusion

The period of see no evil, hear no evil, speak no evil for conservation organizations is over. The local to global links are clear and understandable, *head in the sand* or apathetic approaches to regional and global threats are no longer justifiable, and the conservation community's investments, and society's future, are clearly at risk. A new era in conservation is upon us, where a persistent, unrelenting struggle against threats from human globalization is today's reality. In this context, conservation organizations must play an integral role alongside science, technology, and policy to insure that our society leaves behind a better planet for future generations. It is possible to envision a future in the northeastern United States in which reductions in atmospheric deposition result in dramatic ecosystem recovery. Given this possibility, we have a great responsibility to present such a success story to the world.

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Tear, T. and others. *In prep.* It is time to consider biodiversity conservation in air quality regulation (to be submitted to *Science*).

16. Achieving a Solution to Acid Deposition and Other International Environmental Problems

Robert A. Askins

Acid deposition was first recognized as a large-scale ecological threat about forty years ago when Swedish scientists tied the environmental decline of lakes to acid rain. Solving this problem presented a special challenge because the sources of the pollution were hundreds of miles away, in other countries that had little at stake in the water quality and biological diversity of Swedish lakes. Today we recognize that our most important environmental problems can only be solved through international cooperation, often at a global scale. Efforts to deal with acid deposition across political boundaries serve as a model for dealing with other international environmental problems such as greenhouse gas emissions, ozone depletion, and protection of migratory birds and marine mammals.

The chapters in this volume derive from a two-day conference held at Connecticut College in 2005. The goal was to review what we have learned about acid deposition and to determine whether we have developed an effective strategy to solve the problem. The organizers decided that this was an important time to hold such a conference because interest in the threats of acid deposition had waned both in public discussions and scientific research programs. Also, control of sulfur dioxide, one of the main pollutants contributing to acid deposition, has become a model for other emission control programs, including proposals to control emissions of greenhouse gases. Does this mean that the acid deposition problem has been solved so that we can now move on to other environmental threats?

The 2005 conference on acid deposition was the fifth in a series of conferences sponsored by the Goodwin-Niering Center for Conservation Biology and Environmental Studies at Connecticut College that pulled together scientists, economists and policy makers to address a particular environmental issue. These conferences included representatives of non-profit organizations, government agencies, and private companies as well as researchers from academic institutions. In the case of the acid deposition conference, we viewed the history of the problem from the perspective of biologists, political scientists, economists, policy makers and conservationists. In previous conferences (on the future of commercial fishing in New England, for example), we had heated debates about environmental policies. Although there were some disagreements among the participants in the acid deposition conference, the general conclusions of the different speakers were remarkably consistent. The chapters in this book reflect this consensus, offering complementary accounts of our successes and failures in trying to understand and control acid deposition.

It is generally accepted that the efforts to reduce sulfur dioxide emissions in North America and Europe have been a technical success and can serve as a model for other parts of the world and for other pollution problems. The success of this program, in fact, has led to a popular impression that we have largely dealt with the acid deposition problem. None of the participants at the conference agreed that the problem has been solved. One reason is that, although sulfur dioxide emissions have declined substantially, the emissions of other major components of acid deposition, nitrogen oxides and ammonia, have shown only minor declines. Much of the nitrogen oxide comes from vehicles and ammonia primarily comes from farming, so these emissions are more difficult to control than the sulfur dioxides derived from large power plants. Another problem is that even in regions where acid deposition has declined substantially, there may be little or no recovery of ecosystems. A few aquatic systems responded relatively quickly to the reduction in acid deposition, but these are exceptions to the general pattern. It may require even lower levels of emissions and a longer period of time before the nutrient balance of lakes and forests is fully restored.

1. Ecosystem Recovery

In Chapter 2, Anthony Janetos argues that we would have a better understanding of the acid deposition problem if research had focused more on the pattern of acid deposition and the effects of this deposition on natural ecosystems. Instead, funding agencies favored research on understanding and controlling emissions. Janetos' argument resonates with the conclusions in the next four chapters, which emphasize the bewildering web of changes in complex ecosystems caused by the direct and indirect effects of the deposition of large amounts of sulfur dioxide and nitrogen oxides. If our primary concern were the effect of acid rain on marble statues, then perhaps a simple reduction in emissions would result in an immediate cessation in degradation. With ecosystems, however, the effects play out with the intricacies of soil chemistry (which varies with the type of soil), and with the interactions of soil bacteria, mycorrhizae (symbiotic fungi that absorb nutrients from the soil), and different species of plants (which respond differently to changes in soil nutrients). After years of acid deposition, the soil loses its buffering capacity; the concentration of cations such as calcium, potassium, and magnesium that are needed for plant growth is reduced; and sulfate and nitrogen build up in the soil. Nitrogen, which is quickly taken up by plants in a healthy ecosystem, may reach a high enough concentration so that it flows into streams and lakes, and eventually reaches coastal waters where it leads to eutrophication. Acid deposition may also change the dynamics of forests, causing a shift from conifers to hardwoods (which are favored by high nitrogen concentrations). These changes in forest composition may occur slowly, however, because many tree species are long-lived.

Lake ecosystems are also complex, but they sometimes respond more quickly than do forest ecosystems to reductions in sulfur dioxide emissions. In many lakes, however, the decline in sulfate concentrations is less than expected given the drop in atmospheric sulfur dioxide deposition, and the pH shows surprisingly little change. As Peter Dillon et al. describe in Chapter 4, water quality in lakes in Ontario improved very little despite a major reduction in atmospheric acid deposition because sulfates are carried into the lakes from the surrounding watershed in stream water, especially following drought years. Sulfur dioxide emissions will need to be lowered even more before accumulated sulfates are removed from this system.

Many of these complex interactions among soil, organisms and surface water are not well understood, and both funding for and interest in research on acid deposition have declined. As Anthony Janetos (Chapter 2) and Don Munton (Chapter 9) both emphasize, we even lack an adequate monitoring program in North America to track the ecological effects of acid deposition. To a remarkable extent, our knowledge of long-term ecological effects of acid deposition comes from one site: the Hubbard Brook Experimental Forest in New Hampshire. Clearly we need to monitor how ecosystems respond to emissions reductions at more sites and in more habitats, particularly in sensitive habitats such as high-elevation forests and boreal lakes.

In Chapter 6, Robert Howarth describes one of the most neglected issues concerning acid deposition, the potential negative effects on coastal waters. Acid deposition was not considered a problem for coastal marine ecosystems because they are well buffered against acids and already contain high concentrations of sulfates. They are subject to the indirect effects of acid deposition, however, particularly the release of nitrogen into freshwater streams due to the saturation of upland forest soils with nitrogen. Howarth estimates that 30% of the nitrogen input into coastal ecosystems is ultimately due to atmospheric acid deposition, with most of this input derived from runoff from adjacent uplands rather than direct deposition on coastal waters. Increased nitrogen concentrations lead to eutrophication of coastal waters, resulting in algal blooms, lower oxygen concentrations, and loss of biological diversity. The link between acid deposition and degradation of coastal ecosystems requires more research, however.

Other high priorities for scientific research recommended by authors in this volume are: 1) better data on retention of nitrogen by the soil (with special attention to whether forest ecosystems are approaching a threshold at which more nitrogen will be carried with runoff into streams because it is not taken up by plants or retained in the soil), 2) the effects of year-to-year variation in weather and long-term climate change on runoff of sulfates into lakes, and 3) the potential for recovery of the health of trees as calcium, magnesium and other needed cations begin to accumulate in the soil after acid deposition ends. A common theme of several of the chapters is that decades of acid deposition have changed soil chemistry in major ways and it may take a long time for the soil to recover its pre-industrial characteristics. An open question is whether long-lasting changes in plant species composition will occur before the soil recovers.

2. Policy Lessons

Although reduction of sulfur dioxide emissions has not been sufficient to solve the problem of acidification of lakes and forests, it is a necessary step. The rapid progress in reducing these emissions in both North America and Europe is therefore promising, and provides a good model for regions in eastern Asia where sulfur dioxide deposition is a growing problem.

The European and North American approaches to reducing acid deposition have been distinctly different, with the European Union emphasizing government regulations to set limits on emissions and the United States emphasizing a "cap-and-trade" program in which companies may use, bank, or trade emission permits. Both programs were successful, but the consensus among economists is that the program in the United States was more cost effective. Winston Harrington and Richard Morgenstern (Chapter 12) estimated that the costs of the sulfur dioxide cap-and-trade program were one quarter of the costs of a command-and-control program that emphasizes rigid government regulations. The cap-and-trade system potentially may also lead to greater innovation in pollution abatement methods because it does not specify a particular technology. In contrast, some command-and-control regulations mandate particular types of technology (scrubbers for smokestacks, for example). Paul Portney (Chapter 11) argues that such regulations tend to inhibit technological innovation because only previously approved methods are legal. In the case of the sulfur dioxide cap-and-trade program, the program began at a time when a combination of new technology for using low-sulfur coal and lower prices for this coal permitted power plants to reduce emissions far below the mandated limits, resulting in little incentive for further innovation. However, Harrington and Morgenstern list some other examples of technological innovations in pollution control induced by economic incentive programs (either cap-and-trade or taxes on emissions).

Several authors discuss the potential risks of cap-and-trade programs. Ronald Shadbegian et al. (Chapter 13) discuss the possibility that a trading program in emissions allowances might result in plants in poor or minority areas producing more emissions than plants in wealthier or more influential regions. Poor and minority customers would then effectively pay for environmental benefits received by people in other communities. Their analysis of the distribution of costs and benefits from sulfur dioxide emission abatement from the sulfur dioxide cap-and-trade program in the United States indicated that this is at most a minor problem, however. African Americans and Hispanics actually benefited more than the general population from emission reductions. Compared to the general population, the poor had a slightly higher share of the cost of the program than their share of the benefits, but even this difference would probably disappear if the unrealistic assumption that poor people purchase the same amount of energy as wealthier people were relaxed. In any event, all groups considered in this analysis derived substantial health benefits from the program that far outweighed their costs, so environmental justice does not appear to be a major concern.

Under the current cap-and-trade program for sulfur dioxide, plants can bank allowances they do not sell for use in the future. This may be useful in buffering the economic impact of a transition period to even lower emissions limits. On the other hand, it could lead to a large pulse of sulfur dioxide emissions during a short period if plants have difficulty meeting new limits. This could reverse progress in restoring ecosystems and could have adverse effects on human health.

As Paul Portney points out in Chapter 11, cap-and-trade programs are not appropriate for extremely serious hazards. For example, Daniel Sosland (Chapter 10) argues that a cap-and-trade program in mercury could result in "hot spots" for this highly toxic substance near plants that purchase allowances. Cap-and-trade systems work best for pollutants that become generally distributed and affect everyone. Sulfur dioxide, nitrogen oxides, chlorofluorocarbons and greenhouse gases meet these criteria.

The economic benefits of the cap-and-trade approach in the United States were convincing enough so that Europeans have adopted a similar approach in their efforts to reduce the production of greenhouse gases. There are elements to the European approach to acid deposition problems that could serve as models to North Americans, however. The most important of these is the "critical loads approach", which Miranda Schreurs describes in Chapter 7. With this approach, the goal is to reduce the levels of pollutants or a mix of pollutants to the point where they do not harm human health or sensitive natural ecosystems. This goal fits with the conclusions of ecological analyses in this volume (Part I, Chapters 3–6), which emphasize that it is not sufficient to control a single pollutant (sulfur dioxide) and that we can only gauge the effectiveness of pollution abatement by monitoring the environment. To implement the critical loads approach, Europeans have established extensive systems to monitor both emissions and their ecological impacts, another method recommended as a high priority by North American ecologists. Schreurs emphasizes that Europeans now focus on acid deposition not as a separate problem, but as part of a larger air quality problem.

Liliana Andonova (Chapter 8) shows how the European Union (EU) approach to sulfur dioxide abatement was successfully applied in eastern and central European countries that were required to comply with EU environmental regulations before applying for membership in the European Union. Early compliance with emissions limits resulted from a severe economic downturn during the transition from a communist to a free market economy, but some countries have been able to control sulfur dioxide levels even as they build a new industrial economy.

3. Where Do We Go From Here?

There are several consistent themes that run through this volume that indicate what we still need to do. One of the highest priorities is to find a more effective way to reduce major sources of acid deposition other than sulfur dioxide. This means reducing emissions of nitrous oxides and ammonia that are derived from dispersed sources (vehicles and farming activities, respectively) and so are more difficult to control. Another theme is that we need to understand and monitor the ecological effects of acid deposition in a wide range of ecosystems, and determine if there are ways to speed their recovery from decades of acid deposition.

Another, more general theme is that it is important to pursue sound environmental programs even when consensus with the main contributors to the problems is not possible. Schreurs (Chapter 7) describes how Sweden campaigned for reduction of pollutants from Central Europe and the United

Kingdom for years without results. East and West Germany and the United Kingdom did not act to reduce hydrogen sulfide during the 1960s and 1970s, but instead called for further research on the issue. Sweden proceeded with domestic policies of pollution control, so it was well placed to push for international agreements once the United Kingdom and Germany became convinced that acid deposition was affecting their own natural environments.

Similarly, hydrogen sulfide emissions crossing the U.S.-Canadian border caused extensive damage to boreal lakes in Canada. As Don Munton describes in Chapter 9, Canada gained little cooperation from the United States in reducing these emissions in the 1970s and early 1980s, but pursued its own program to effectively reduce sulfur dioxide emissions. When the U.S. government acknowledged that acid deposition was a serious problem for both countries in the late 1980s, a way was open to agreeing on an air quality treaty. Although not all of the provisions of this treaty have been carried out, it did establish a foundation for cooperation on reducing acid deposition across temperate North America.

In Chapter 14, Joseph Kruger explores how lessons from sulfur dioxide control might be applied to the control of greenhouse gases, emphasizing how a modified version of the cap-and-trade policy used for sulfur dioxide emissions might be effective. Although the European Union is already moving in this direction, there is little hope for federal regulations to control greenhouse gas emissions in the United States under the current administration. Kruger describes how states, either acting separately or in groups, are moving to set up cap-and-trade systems and other programs to reduce the emission of greenhouse gases. As Kruger emphasizes, efforts by states can establish "facts on the ground" that would provide a model of what works if the United States adopts a national policy to reduce greenhouse gases in the future. Thus, programs such as the Regional Greenhouse Gas Initiative, which is sponsored by seven northeastern and middle Atlantic states, could play much the same role in the control of greenhouse gases as Sweden and Canada played in the reduction of acid deposition.

The story that we can piece together from the numerous perspectives in this volume tells us not only about acid deposition, but also how to effectively institute policies for environmental protection. The acid deposition problem served as a testing ground for how to apply scientific insights to public policy, and how to forge international agreements and create economically effective mechanisms to solve environmental problems. Not all scientific insights have been applied successfully, and not all of the international agreements have worked as well as intended for controlling acid deposition. Also, the economic methods used to reduce acid deposition still need to be assessed and improved. Both the successes and shortcomings of these efforts point the way to eventually restoring the ecosystems that have been damaged by acid deposition, however, and show us how we can deal more effectively with other international environmental problems.

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