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BY THE COMPTROLLER GENERAL

Report To The Congress

OF THE UNITED STATES

An Analysis Of Issues Concerning "Acid Rain"

Although science has largely determined that man-made emissions cause acid rain, there is uncertainty concerning the extent and timing of its anticipated effects. Thus, at the present time scientific information alone does not lead unequivocally to a conclusion on whether it is appropriate to begin control actions now or to await better understanding. Given this uncertainty, decisionmakers must weigh the risks of further, potentially avoidable environmental damage against the risks of economic impacts from acid rain control actions which may ultimately prove to be unwarranted.

GAO examines the implications of current scientific knowledge for policy decisions on acid rain and offers a series of observations on the following issues involved in the debate:

- To what extent has it been scientifically demonstrated that acid rain is resulting in damage to the environment?
- What are the causes of acid rain and where is it most prevalent?
- What alternatives exist for controlling acid rain and what are their economic effects?

UNITED STATES GENERAL ACCOUNTING OFFICE

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COMPTROLLER GENERAL OF THE UNITED STATES
WASHINGTON D.C. 20548

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The President of the Senate and the
Speaker of the House of Representatives

This report examines the issues involved in the decision about controlling acid deposition, commonly referred to as "acid rain." We undertook this study to examine whether available scientific information could indicate clearly either that it was appropriate to begin control action now, or that control actions should await better understanding. Finding that the scientific information alone did not lead unequivocally to either decision, we have analyzed the current state of knowledge on the subject and presented a series of observations on what that knowledge does indicate about actions to control acid deposition.

Copies of this report are being sent to appropriate House and Senate committees, the Secretaries of Commerce, Energy, and State; the Administrators of the Environmental Protection Agency and National Oceanic and Atmospheric Administration, the Chairman, Council on Environmental Quality, and other interested parties.

Charles A. Bowsher

Comptroller General
of the United States

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How far does current knowlege lead?

APPENDIX

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TABLES AND ILLUSTRATIONS

MOI	Memorandum of Intent: A memorandum signed, in this case, by U.S. and Canadian Government officials in August 1980, announcing their intention to undertake efforts to reach a bilateral agreement on limiting long-range transboundary air pollution
mg/m^3	milligrams per cubic meter, a metric unit of concentration usually used for measuring gases or pollutants present in small amounts compared to the free oxygen and nitrogen that dominate the atmosphere
Mt	metric tonne: a unit of mass equal to 1000 kilograms or 1.1023 short tons (English tons)
MW	Megawatt: one million watts or one thousand kilowatts, a unit used to quantify power level or electric generating capacity--the average large nuclear generating unit coming into use in the U.S. is about 1000 MW (one GW)
MW(e)	Megawatts (electric): a unit used to quantify the electric generating capacity or power level of a facility, expressed in terms of the *electricity output from* the facility
MW(t)	Megawatts (thermal): a unit used to quantify the power level of a facility, expressed in terms of the *heat input to* the facility. Conventional thermal power plants produce one unit of electric energy output for every 2.5 to 3 units of heat energy input, so the thermal power rating of a plant will be 2.5 to 3 times the electric power rating.
NAAQS	National Ambient Air Quality Standard: a limit, determined and promulgated by EPA, on the *concentration* of a pollutant allowed in the atmosphere at or near ground level
NAS	U.S. National Academy of Sciences: sometimes also a shorthand for NRC/NAS, the National Research Council, the body established by the NAS to do studies as requested usually by some part of the federal government
NO_3^-	nitrate ion: a major part of nitric acid and all nitrate compounds, also the chemical state in which nitrogen is chemically stable after fully reacting with oxygen in the atmosphere
NO_x	a generic term for either or both of two gaseous compounds of nitrogen and oxygen: nitric oxide (NO) and nitrogen dioxide (NO_2)

NSPS	New Source Performance Standard: an upper limit, established either in law or EPA regulation, on the amount of an air pollutant that may be released by a source which began construction or substantial modification after a specified date.
NWF	The National Wildlife Federation
OECD	Organization for Economic Cooperation and Development: an international organization whose membership includes the United States, Canada, Japan, Western European nations, and several others
OTA	The Office of Technology Assessment: a Congressional staff organization
pH	a widely used measure of acidity: defined as the negative of the logarithm (to the base 10) of the H^+ (hydrogen ion) concentration. Acidity increases with decreasing pH; pH7 is neutral
R&D	Research and Development
RHC	reactive hydrocarbons: that part of the total amount of emitted hydrocarbon gases which is involved in atmospheric reactions that produces ozone and other gaseous oxidant secondary pollutants
SIP	State Implementation Plan: a plan, by a state government, for controlling emissions from older (pre-NSPS) sources so as to meet NAAQS's
SO_2	sulfur dioxide: a gaseous compound of sulfur and oxygen
$SO_4^=$	sulfate ion: a major part of sulfuric acid and all sulfate compounds, also the chemical state in which sulfur is chemically stable after fully reacting with oxygen in the atmosphere
SURE	The Sulfate Regional Experiment: a major cooperative study of air pollutants and their physicial and chemical behavior in the air over the eastern United States, sponsored and administered by EPRI
TBS	Temple, Barker and Sloane, Inc.
TVA	The Tennessee Valley Authority

ABBREVIATIONS

Btu	British thermal unit: the quantity of energy required to heat one pound of liquid water by one Fahrenheit degree
CBO	Congressional Budget Office
CRS	Congressional Research Service of the Library of Congress
DOE	The U.S Department of Energy
EEI	The Edison Electric Institute: an investor-owned utility trade association
EIA	Energy Information Administration
EPA	Environmental Protection Agency
EPRI	The Electric Power Research Institute
FBC	Fluidized Bed Combustion
FGD	Flue Gas Desulfurization ("Scrubbing"): a technique for removing sulfur combustion products from the exhaust gas of a fuel-burning facility

GAO	The General Accounting Office
gpm	grams per mile (traveled): a unit used to quantify emissions of air pollutants from vehicles
GW	Gigawatt: one billion watts or one million kilowatts, a unit used to quantify power level or electric generating capacity--about the size of the average large nuclear generating unit coming into use in the U.S.
H^+	The hydrogen ion, a free proton: a portion of some chemical compounds. Compounds which contain or release this entity are defined as acids (This is usually a chemical shorthand since, in water-containing systems, H^+ is actually found attached to a water molecule, forming H_3O^+, the hydronium ion.) The concentration of H^+ (or of H_3O^+) is the chemist's measure of the amount of free acid in a system.
ha	hectare: a metric unit of area, equal to 10,000 square meters or 2.471 acres
kg	kilogram: a metric unit of mass, equal to 1000 grams or 2.2046 pounds
kwh	kilowatt-hour, one thousand watt-hours: a unit of electric energy, the amount produced from a 100 watt generator in 10 hours, a 1000 watt (1kw) generator in 1 hour, or any other product of power x time equalling 1000 watt-hours
lb/mmbtu	pounds per million Btus: a unit for quantifying rates of emission of air pollutants from fuel combustion--the number of pounds of the pollutant released to the atmosphere from the combustion of an amount of fuel providing 1 million Btu of energy.
LED	Least Emissions Dispatch: a strategy to lower emissions from electricity generation, by deciding which plant(s) a utility will operate so that it chooses the plant(s) to use most which will release the smallest amount of pollutant, rather than to minimize its costs (least-cost, or economic dispatch)
LIMB	Limestone Injection with Multi-staged Burners: A technique being developed to lower both SO_2 and NO_x emissions from coal-fired furnaces, using limestone injected into the furnace to capture SO_2 and staged burners to control air-fuel ratios to lessen NO_x formation

COMPTROLLER GENERAL'S REPORT
TO THE CONGRESS

AN ANALYSIS OF ISSUES
CONCERNING "ACID RAIN"

DIGEST

Intense debate continues over whether it is time for the United States to take control actions against the environmental effects of acid deposition. Often referred to as "acid rain," this phenomenon occurs when oxides of sulfur and nitrogen emitted by coal-fired power plants, smelters, vehicles, and other sources, both man-made and natural, are transported in the atmosphere and return to earth as acid compounds. This has become a national and international issue because these substances can often be transported beyond the jurisdictions in which they are emitted, and possibly damage the environment across state and even national boundaries.

In response to an inquiry from Senator Wendell Ford of Kentucky, GAO issued a report entitled The Debate Over Acid Precipitation--Opposing Views--Status of Research (EMD-81-131, Sept. 11, 1981). That report described the views of supporters and opponents of control actions and identified points of agreement on technical questions in the debate. At that time, GAO noted a number of areas concerning the effects and causes of acid deposition which required closer examination to help reach decisions about controlling acid deposition.

Recognizing the extent of congressional concern with the issue, GAO reviewed and analyzed the extensive scientific literature on the subject, and discussed it with many of the scientists conducting acid deposition research, as well as with representatives of a number of concerned organizations.

This report describes the present state of knowledge on the acid deposition issue. It also presents a series of GAO observations based on its analysis of the major questions involved in the debate.

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EFFECTS OF ACID DEPOSITION

Investigation of the effects of acid deposition has concentrated on lakes and fish, forests and agriculture, materials, and human health.

GAO found that, to date, the only thorough and convincing documentation of damage in North America concerns the acidification of some lakes and rivers and the consequent reduction or elimination of populations of certain fish species in three areas of the northeastern United States and southeastern Canada. Significant biological damage caused by acid deposition in the United States has thus far been identified only in the Adirondack Mountain lakes of New York, but there is evidence of acidification of lakes and streams in wider areas which may continue and result in further loss of fish populations. Observations in Norway, which experienced the complete loss or substantial reduction of fish populations in thousands of lakes because of acidification, suggest its potential seriousness. However, attempts to forecast the future extent of the problem in North America remain inconclusive because of the wide range in scientific predictions, which range from no further damage to greater and more wide-spread damage at presently expected levels of deposition. (See pp. 9-17.)

Effects on agriculture and forests are less well documented than those on lakes and fish. GAO could find very little solid evidence that acid deposition is causing damage directly to crops, while many studies show no damage. Moreover, the treatment of soils with alkaline substances (liming) represents a relatively inexpensive way to neutralize acids that might harm crop plants indirectly by affecting their soils. (See pp. 17-19.)

The possibility that acid deposition damages forests is a far more serious concern than it is for crops. Slowed growth in a number of species of forest trees, with severe damage and even death in some, has been observed in a number of areas of the eastern United States. GAO found that, although existing data are not definitive, emerging scientific evidence suggests that acid deposition may be one of the pollutants that plays some part in these forest effects, either directly or through effects on soils. Forest soils, in contrast to soils used in agriculture, are less nutrient-rich and more susceptible to damages from acidification; in addition, if such damage does occur, the size of forested areas is so great that liming could not be an economic solution. Recent reports by the West German government of rapidly increasing damage to forests there gives further cause for concern, but does not yet offer a basis for forecasting the extent of damage in North America.

Research literature makes it clear that air pollutants are responsible for damage to man-made materials and structures involving substantial economic value, and that acidic pollutants, such as sulfur dioxide (SO_2) play a significant role in this damage. However, neither the share of the damage attributable to acidic pollutants, nor the relationship between amounts of pollutant and of damage, are yet quantified. Furthermore, since most man-made materials and structures are located in developed areas, in and near the locations where most pollutants are emitted, much of the materials damage is attributable to local, as opposed to transported, pollutants. (See pp. 24-26.)

GAO could find no reported direct effects of acid deposition on human health; however, instances have been reported of indirect effects involving potentially toxic heavy metals made soluble by acid deposition. Some evidence suggests that acidified water may play a role in increasing the mercury content of fresh water fish. Mercury in food can cause nervous system disorders which can be fatal. However, without an understanding of the mechanism linking acid deposition to the mercury-contaminated fish, it cannot be determined whether the correlation is evidence of a causal relationship.

Reports of harm from metals dissolved in acidified drinking water are rare to date in the United States but this question needs wider examination since lead causes brain damage. If individual cisterns and/or wells are acidified, as has occurred in Scandinavia, protective treatment could be inconvenient and relatively expensive. (See pp. 26-28.)

CAUSES OF ACID DEPOSITION

To understand the causes of acid deposition, it is necessary to examine the sources of the pollutants involved, their transformation into acids, and their transport over various distances.

Identifying the sources of acid deposition had been subject to question, but scientific work has now confirmed that it arises predominantly from man-made emissions of SO_2 and oxides of nitrogen (NO_x). In the United States, coal-burning electric power plants and industrial facilities are the largest emitters of SO_2. While these sources also emit large quantities of NO_x, the transportation sector is the leading source of NO_x emissions. The primary source of NO_x in Canada is also the transportation sector, whereas non-ferrous smelters are the principal source of SO_2 emissions. Geological evidence, as well as research findings, are able to link acid deposition far above naturally occurring levels directly to these emissions. (See pp. 32-47.)

Transformation of SO_2 and NO_x emissions to acids may occur either before or after they are deposited, and they can be deposited either "wet" (in precipitation) or "dry" (as gases, fog and cloud droplets, or particles). Knowledge about transport and deposition indicates that the majority of sulfur deposition in the United States occurs in the dry form, as SO_2 gas, with conversion to sulfate happening after deposition. The amount of sulfur deposited in this form will be directly proportional to the amount emitted. Recent scientific studies have now shown that, on a regionwide basis, the other main deposited form--wet sulfate deposition--will also be essentially proportional to SO_2 emissions. Thus, scientific work confirms that overall reductions of SO_2 emissions will lead to proportional reductions of acidifying sulfur deposition. (See pp. 47-57.)

Analysis of wind patterns enables researchers to trace the paths of deposited materials toward their sources with acceptable accuracy but can only indicate direction, not distance. Such evidence, plus the use of models, has shown that SO_2 is deposited out gradually as air masses move, with no sharp break in the amount that a source contributes at increasing distances. In general, significant contributions to deposition at any location can come from sources as far as 500 miles in the direction of prevailing winds, although at greater distances the acidity contributions will tend to become less significant. (See pp. 57-68.)

PROPOSED SOLUTIONS TO ACID DEPOSITION

The main approach proposed for controlling acid deposition is a reduction of total SO_2 emissions, because acid damage to ecosystems is largely caused by deposition of sulfur compounds rather than nitrogen compounds. However, the proportion of nitric acid to sulfuric acid is higher in recent precipitation than in precipitation 1 or 2 decades earlier. If this trend continues, then NO_x emissions could become an important contributor to acid deposition in the next century.

Although the Clean Air Act has led to substantial reductions of SO_2 and NO_x emissions, it still allows large emissions of both pollutants--about 22 million tons of SO_2 and 14 million tons of NO_x in the eastern United States in 1980. Under present policy, these emissions are currently anticipated to increase over the balance of this century in the United States while some scientists have suggested that reduction of SO_2 emissions by as much as 50 to 75 percent may be required to prevent future damage. The Clean Air Act has not been effective in dealing with the problems of interstate and transboundary air pollution. It is primarily aimed at controlling the concentrations of pollutants in the air and therefore has its main effect in the local areas around pollution sources. (See pp. 71-75.)

Many older coal-fired power plants not stringently regulated by the Act will be in operation in the eastern United States through the end of the century, at which time they are expected to still account for nearly 85 percent of all eastern utility SO_2 emissions. Recent efforts to extend the usable lifetimes of these plants could maintain these high emissions for an additional decade or two into the next century. The primary targets for SO_2 emission reductions in the eastern United States are therefore older coal-burning power plants, although significant SO_2 control opportunities can be obtained by steps which would restrain the anticipated growth of coal-fired industrial emissions. (See pp. 75-77.)

Emission reduction techniques

The three emission reduction techniques most extensively considered are washing coal, "scrubbing" the flue gases from older power plants, and switching to lower sulfur fuels. GAO examined each of these alternatives for its effectiveness and economic as well as social consequences. (See pp. 77-83.)

Coal washing, already used extensively, can remove about 20 to 35 percent of the sulfur from high-sulfur coals at low capital costs. It can also reduce operating costs and increase plant reliability--advantages in addition to sulfur removal which compensate, in part, for its costs. Its use on all high-sulfur coals could remove up to an added 1.5 million tons of SO_2 annually in the eastern United States at costs which have been estimated to range upwards from a low of $200 per ton of SO_2 removed. However, washing would be much less cost-effective for lower-sulfur coals, costing up to thousands of dollars per ton of SO_2 removed.

Switching to lower sulfur coals or scrubbing are the two methods asserted capable of making larger reductions in eastern U.S. SO_2 emissions. Each method can, under favorable circumstances, reduce SO_2 emissions for costs that have been estimated as low as a few hundred dollars per ton of SO_2, with the lowest cost estimate for switching being less expensive than the lowest cost estimate for scrubbing. The cost and feasibility of both methods are strongly influenced, however, by a number of factors that are very specific to individual plants.

In GAO's view, the divergent estimates of the prospects for widespread switching have yet to be reconciled. There remains a range of estimates of coal switching's capacity to reduce annual eastern U.S. SO_2 emissions more cost effectively than scrubbing. These estimates range from as little as 1.5 million tons to over 6 million tons.

In contrast, estimated eastern U.S. emission reductions possible through scrubbing are quite large--as much as 8 million annual tons of SO_2 or more. However, there remains a substantial range in the costs calculated for scrubbing. At some plants, where installing scrubbers may be easy, costs are estimated to range between about $250 and $500 per ton of SO_2. Scrubber installations at smaller or older plants, which would have to be scrubbed to reach the 8 million ton level of SO_2 control, would be more expensive, with costs estimated to exceed $1,500 per ton in some cases.

GAO found that the combination of uncertainties in costs and capabilities for SO_2 control leaves unclear the comparative costs and effects of two different approaches to SO_2 control. One approach would allow emission sources to choose their own control method and the other would mandate scrubbing to avoid adverse effects on employment and the economies in high-sulfur coal-producing regions that would likely result if extensive coal switching occurred.

The cost per ton of SO_2 controlled is widely recognized to increase at greater levels of control, since the least cost opportunities will generally be chosen first, except if more expensive methods are required. Regardless of which control policy is chosen, however, a range of studies shows that costs for a major reduction of SO_2 emissions would amount to billions of dollars annually. Specifically, proposed programs which aim to reduce annual eastern U.S. SO_2 emissions by 10 million tons, from the 1980 level of about 22 million tons, are estimated to have annual costs of between $3 billion and $7 billion in 1982 dollars. The ultimate benefit of such a program, measured as the amount of acid-caused damage that it would avoid, is so uncertain that it could easily be much larger or much smaller than its cost.

A potentially important control technique, the limestone injection multistage burner (LIMB), is currently in early development in the United States, but it is undergoing commercial demonstrations in West Germany. It is aimed at achieving 50 to 70 percent SO_2 removal as well as low NO_x emissions when installed in stationary coal-burning facilities. LIMB is expected to be a great deal less complex, and therefore have much lower capital costs and be somewhat more cost-effective overall, than present scrubbing methods. However, it is not expected to be commercially available in the United States before the 1990's.
(See pp. 83-86.)

Because it provides less acidity than SO_2, and also because much of the nitrate it generates is neutralized by living plants, NO_x currently ranks as a lower priority than SO_2 for emission reductions to control acid deposition. Most methods of NO_x control involve different steps than can be used for SO_2 control. NO_x can currently be controlled by combustion modification steps in stationary coal-burning sources at modest costs--in the range of $60 to $95 per ton of NO_x removed--but only to the extent of 20 to 30 percent removal. More complex modifications, expected to be ready in a few years, are estimated to allow 50 percent NO_x removal, while still higher removal rates may be obtained with flue gas treatment.
(See pp. 104-107.)

CONTROLLING ACID DEPOSITION

Congressional decisions about acid deposition revolve around whether to begin control actions promptly or to wait until better scientific data are available. Closely following on that question, however, are two issues sufficiently important to influence decisions on this main issue--what kind of control actions would be best, and how, and by whom, should they be paid for?

GAO's principal observation is that, although science has largely determined the causes of acid deposition, there is uncertainty concerning the amount and the timing of the effects which can be anticipated from it. Thus, scientific information alone cannot determine whether it would be better to begin control actions now or wait until estimates of effects can be made more accurate. Instead, the issue must be approached by weighing the relative risks of alternative decisions: the risks of adverse economic impacts in some regions of the country, caused by immediate control actions whose benefits cannot be accurately evaluated, versus the risks of further, potentially avoidable harm to the environment and possibly to public health in other regions of the United States and/or Canada if actions are delayed. (See pp. 112-114.)

In examining these alternatives, the information GAO presents in this report can give guidance on the more specific issues that must be analyzed in reaching an overall decision. (See pp. 115-124.)

DAMAGE DUE TO ACID DEPOSITION

--Limited damage caused by transported acid deposition has been confirmed in only a few areas of North America to date. However, the major reason for concern is the possibility of much greater damage, which may occur in the same or other regions and be of the same or other types, as that which has already been confirmed.

--Uncertainties about anticipated damage rates, more than any other of the scientific uncertainties about acid deposition, make it difficult to assess the need for and effectiveness of control actions.

CAUSES OF ACID DEPOSITION AND DAMAGE

--The substances from which acid deposition is produced, in and near major industrial and populated areas such as eastern North America and western and central Europe, are predominantly the man-made pollutants, SO_2 and NO_x.

--In eastern North America, sulfur compounds, predominantly emitted as SO_2, are the main damaging acidic pollutant.

--Precise estimates of the shares of sulfur compounds deposited at any particular location which are contributed by individual sources or source areas are not yet available. However, approximate estimates indicate that most areas, except possibly those within a few tens of miles of large sources, receive partial contributions to their deposition from many sources spread over large areas up to hundreds of miles away, with the majority coming from within about 500 miles in the average upwind direction, and from shorter distances in other directions.

TECHNIQUES FOR CONTROLLING THE EFFECTS OF ACID DEPOSITION

--Mitigation actions taken where deposition occurs, such as liming of lakes, can prevent damage in some cases. However, they have limited capabilities both because they cannot control all kinds of damage, and also because they could not be applied economically to large unmanaged areas such as forests.

--If deposition reduction is desired, to control the risk of damage stemming from acid deposition, the greatest reduction in risk would come from lessening the deposition of acidic sulfur compounds, which could be accomplished best by reducing SO_2 emissions.

--Because deposition at almost any location includes significant contributions from sources spread over a wide area, emission controls intended to produce substantial reductions of acid deposition, even at one location, would be needed over a wide area rather than at one source or a narrowly localized set of sources.

ATTEMPTS TO DETERMINE THE PROPER LEVEL OF CONTROL ACTIONS

--While cost/benefit analysis can often aid in identifying a range of economically efficient pollution control policies, current scientific uncertainty about the value of the benefits expected from proposed levels of acid deposition control is so great that cost/benefit analysis is of limited value in deciding whether additional controls on SO_2 emissions would or would not have benefits that justified their costs.

--Marginal costs of emission reductions increase at greater levels of reduction, so that if reductions were chosen seeking to eliminate damage completely, the last increments of emission reduction would be very costly compared with the last increments of damage prevented.

--The control method of switching to low-sulfur coal may or may not offer lower overall costs. However, it would have indirect costs, in the form of employment shifts between regions, which would disproportionately affect limited areas where high-sulfur coal is mined.

--Agreement on an approach to the acid deposition problem is likely to be aided by separating the question of when and in which areas of the country control actions should occur, from the question of how the control actions are to be financed.

GENERAL OBSERVATIONS ON THE ACID DEPOSITION ISSUE

--Because the Clean Air Act currently focuses on concentrations of pollutants near their sources, any air pollution control approach to deal with acid deposition in this century would necessitate additions to, or a basic reorientation of, the ambient air quality standard approach in the present act.

--The dispute persists over whether it would be advisable to establish emission controls promptly to reduce acid deposition or to wait further. However, at a minimum, having control plans ready could save time, and therefore spare resources, if/when a need for rapid action becomes evident.

- Further scientific work on acid deposition will be needed for a number of years, no matter what decisions are made on control actions in the short run.

AGENCY COMMENTS

Because this study does not review or evaluate executive agency activities, a draft was not sent to the cognizant agencies for their formal review and comment. GAO did, however, discuss the general observations in the report with Environmental Protection Agency officials.

CHAPTER 1

INTRODUCTION

Acid deposition continues to be one of the most controversial energy/environment issues of the 1980's with potentially profound implications for environmental quality in the eastern United States and Canada. The issue may also have significant impacts on the economies of some high-sulfur coal-producing regions in the eastern United States, as well as on electric utility costs in areas where high-sulfur fuels are used extensively for electricity generation.

Acid deposition, often referred to as "acid rain,"[1] occurs when sulfur and nitrogen oxides emitted by coal-fueled power plants, smelters, vehicles, and other sources, both man-made and natural, are transported in the atmosphere and return to earth as acid compounds. It is an issue requiring federal government attention because air pollutants are often transported in the atmosphere beyond the jurisdictions in which they are emitted, and contribute to acid deposition, and therefore to the occurrence or potential for damage to the environment, across state and even national boundaries. Existing air pollution control policies under the Clean Air Act have been successful neither in resolving the interstate disputes over acid deposition nor in making it possible to reach agreement with Canada over how our two nations should deal with the exchange of acidifying air pollutants across our common border.

BACKGROUND AND PERSPECTIVE ON ACID DEPOSITION

Acid deposition was first brought to prominent public attention by the Scandinavian countries, and particularly Sweden, at the 1972 United Nations Conference on the Human Environment in Stockholm. This was done because Sweden and Norway had begun to recognize the effects of acid precipitation in their own countries, in the form of acidification of freshwater lakes and streams, and the decline or loss of fish populations in these waters.

The fact that the problem was first seen in Scandinavia turns out, in retrospect, to be largely due to a combination of three geographic factors. First, air movement patterns bring a good deal of the air pollution from industrialized areas in the northern part of Europe toward southern Sweden and Norway. Second, parts of the southern areas of these countries are at

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[1] The term "acid deposition" refers to the depositing of these acid compounds from the atmosphere to the earth in both wet and dry forms. The terms "acid rain" and "acid precipitation" include only those acid compounds deposited in wet form, or other precipitation, omitting dry deposition of some sulfur and nitrogen oxide gases and particulates.

significant elevations above sea level and, because of a well-known meterological process, receive more precipitation than low-lying areas. Third, because these areas are at high elevation and high latitudes, they have been subject to quite severe climates, which make growing conditions for plants difficult. This has resulted in the formation of less soil there since the last ice age, compared with areas with more moderate climates. Hence, material deposited from the atmosphere has less chance of being absorbed by chemical or physical processes in these soils, and it will pass on more readily to affect the waters of streams and lakes.

Intensive research in Sweden and Norway, work in other European countries, and joint European studies followed in the next few years. By 1977, it led to recognition of the international nature of the problem--at least in the European context of relatively small countries--with a number of countries actually estimated to receive more of their acid precipitation from foreign than domestic sources.

Research focused on acid deposition in North America started later, in the middle 1970's, although fishery losses had been observed earlier in lakes in the La Cloche Mountain area of Ontario. However, these lakes were relatively near the site of the world's largest single sulfur-emitting source--a smelting complex at Sudbury--so they did not necessarily give reason for concern about damage occurring because of emissions transported over long distances, as had been observed in Europe. Still, the amounts of the man-made pollutants, sulfur dioxide (SO_2) and oxides of nitrogen (nitrogen dioxide (NO_2) and nitric oxide (NO), jointly abbreviated as NO_x), emitted in eastern North America were as large as or larger than those emitted in Europe so there was reason to seek for similar effects here.

Acidification of lakes and associated losses of fisheries in about 180 of the higher altitude lakes in the Adirondack Mountains in northern New York were in fact detected, as was a correlation between water acidity and the decline of Atlantic salmon fisheries in some of the small rivers of Nova Scotia. In addition, studies were made to attempt to estimate the susceptibility to acidification of fresh waters in other areas of North America. These led to growing concern about the possibility of wider damage in a number of areas including northern Minnesota, much of New England, some mountain areas in the southern Appalachians, and large parts of southern Ontario and Quebec in Canada.

By 1978 the United States and Canada had established a Bilateral Research Consultation Group on the subject of transboundary air pollution. On July 26, 1979, the two governments released a joint statement announcing their intention to develop a cooperative air quality agreement, and in August 1980, a Memorandum of Intent (MOI) was signed to launch the process leading to negotiation of such an agreement. This process centered on establishment of a set of joint Work Groups to assemble and analyze information and help propose measures for possible inclusion in an agreement.

At the same time, the United States government was establishing a coordinating body for acid precipitation research, first under presidential order in August 1979, and then through the Acid Precipitation Act of 1980 (Title VII of the Energy Security Act of 1980, P.L. 96-294, June 30, 1980.) This body, now known as the Interagency Task Force on Acid Precipitation, has issued annual reports for 1981, 1982, and 1983, and a National Acid Precipitation Assessment Plan in June 1982. Much of the work of the Task Force has been done in parallel with the work of the U.S. contributors in the U.S.-Canada Work Groups.

Intense political disputes about the need for and cost of control actions, and how their costs should be paid, have surrounded the acid deposition issue, tending to divide along geographic rather than party or ideological lines. This is because, to a substantial extent as the result of accidents of geography, the areas experiencing or most vulnerable to damage are different from the areas from which the greatest share of man-made air pollutants are emitted. Leading examples of this controversy in Europe, for a decade or more, have seen Norway and Sweden seek reductions of utility and industrial emissions particularly from the United Kingdom, West Germany, France, the Netherlands, and several eastern European nations. Now, in more recent years, the eastern Canadian provinces and the northeastern states have been seeking similar action from major emission sources in the Midwest and Ohio River Valley.

Change on this front has not come easily, because emission controls are expensive[2] and the concept of long-range transboundary pollution is quite new and has been accepted slowly. Thus, while substantial emission reductions were made in the United States and Canada since SO_2 emissions reached their peaks 1 to 2 decades ago, these were generally done in response to policies aimed at protecting public health by meeting ambient air quality standards, with relatively few further reductions expected without further changes in emission policies. In fact, since the late 1970's, the only important changes in emission control policies in nations involved in the acid deposition controversy have occurred in Canada and West Germany, which recognized that they themselves were at risk of or beginning to experience substantial damage to their own resources--particularly the large number of lakes at risk in eastern Canada and rapidly developing damage to forests in West Germany.

The major focus of concern on the part of those seeking emission reductions because of acid deposition has been on sulfur compounds, mainly the oxide SO_2, which is the predominant form in which sulfur is emitted from combustion and other industrial processes. In most of western Europe a great deal of the SO_2 is

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[2] Except possibly in earlier years in the Netherlands, where SO_2 emissions dropped markedly as newly developed domestic natural gas replaced imported oil, at the same time lowering the average sulfur content of fuel and, therefore, lowering SO_2 emissions.

omitted by combustion of heavy oils, with coal playing a lesser role than in the United States, both because Europe has not used as great a proportion of coal as the United States in the post-World War II period, and also because some European coals, in West Germany for example, have lower sulfur contents than much of that used in the eastern United States.

In the United States and Canada, the call for reduction of SO_2 emissions is ironic because, as the 1970's ended, both countries had largely achieved the goals of SO_2 emission control programs that had started years earlier in response to concerns about health effects. As of 1980, total U.S. SO_2 emissions were estimated to be down 17 percent from their peak in 1973, and Canadian SO_2 emissions were down 28 percent from their peak, which occurred somewhat earlier, in about 1965.[3]

According to projections made by the respective governmental air pollution control experts,[4] however, these decreases are not expected to continue much longer in either country unless further controls are applied in response to new policies. In particular, by the year 2000, U.S. SO_2 emissions are projected to be more than 10 percent above 1980 levels, despite modest declines anticipated up to 1990, while western Canadian SO_2 emissions are projected to increase by one-quarter from 1980 to 1990 and by another 8 percent from then until 2000. It is only for eastern Canada, where policies to lessen acid deposition by reducing total SO_2 and NO_x emissions have recently been put into effect,[5] that further reductions of SO_2 emissions are projected. The decrease is already estimated to total 14 percent between 1980 and 2000, and the reduction could be significantly greater and could occur earlier if further emission reductions being planned by Canadian federal and provincial governments are actually ordered and implemented. Still, these reductions do not match those already achieved and further ones planned in Sweden, the country which has been most aggressive in its efforts to reduce SO_2 emissions.

Swedish emission control policies have focused on reducing acid deposition for a longer period than any other nation whose policies we have reviewed. According to the Swedish Environmental

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[3] Emission data for the United States and Canada are presented in appendix I, which gives detailed data on their sources, and an analysis of trends and controlling factors, as well as emission projections to and beyond the end of the century.

[4] See appendix I, table A-10, summarizing data from: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982.

[5] Some final emission reduction orders have been issued in Ontario, while others are under consideration there and in other eastern Canadian provinces.

Protection Board,[6] this had already resulted in a reduction of SO_2 emissions in Sweden of 46 percent between 1970 and 1978. In addition, the Board's assessment was that emissions by 1990 would be reduced about another 31 percent from the 1978 amount, so that total SO_2 emissions in Sweden by 1990 would be down to only 37 percent of the 1970 amount,[7] representing a 63-percent reduction from that peak level.

Canada has been making extensive efforts through many kinds of contacts, both governmental and nongovernmental, to urge the United States to join in carrying out major reductions of acidifying emissions, particularly of SO_2. The first formal Canadian emission reduction proposal made to the United States, in negotiations held in February 1982 under the MOI, was for the two nations each to lower their SO_2 emissions 50 percent below the 1980 levels. If such reductions were carried out, then United States and Canadian emissions would, in fact, drop about as much as the planned 1990 level in Sweden--for the United States down about 58 percent to around 42 percent of the peak level, and for Canada, down about 64 percent, to around 36 percent of the peak level. To date, however, the nearest that any other nations we know of have come to the Swedish plans occurred in a March 1984 meeting in Ottawa, where Canada and eight other European governments joined Sweden in agreeing to each reduce SO_2 emissions 30 percent below 1980 levels by 1993.

As we will show below, it is not clear whether emission reductions as severe as those now planned in Canada, or the even greater reductions planned in Sweden and proposed by Canada, will be necessary to prevent or sufficiently limit damage from acid deposition in North America. However, the scale of these actions does show how genuine and intense the concern about the problem is in countries that are experiencing acid deposition or are at risk of damage, and are also seeking emission reductions from their neighbors to aid in their domestic efforts to reduce the risk and amount of damage they suffer from acid deposition.

The central decision for the United States is whether (and if so where, when, and how much) it should pursue further reductions in total emissions of SO_2, and possibly NO_x, to lessen acid deposition caused by these emissions both in the United States and in Canada.

PREVIOUS GAO WORK

In late 1980, Senator Wendell Ford of Kentucky asked us to examine the issue, with a main focus on the question of whether it

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[6] Cited in Acidification Today and Tomorrow, Swedish Ministry of Agriculture, Environment '82 Committee, prepared for the 1982 Stockholm Conference on the Acidification of the Environment, English translation by S. Harper, p. 39.

[7] Ibid.

was appropriate to begin action promptly to control acid deposition or to wait until better scientific information was available. In response to that request, we issued a preliminary report, to clarify the viewpoints of the parties debating the question and the extent to which there was agreement on the scientific and technical issues underlying the debate. In that report,[8] we also identified a number of uncertain and disputed matters in the areas of effects, causes, and possibilities for control which have a major bearing on policy decisions about acid deposition. That work also contributed to testimony before the Senate Energy Committee in August 1982 and to an analysis of a major legislative proposal issued in September 1982.[9]

In view of the intense congressional interest in finding a solution to the acid deposition problem, we have carried out an indepth study of the issue since the release of our report to Senator Ford.

OBJECTIVE, SCOPE, AND METHODOLOGY

The objective of this report has been to provide the Congress and others with an explanation of what is known about acid deposition and the implications that knowledge has for decisions on the policy questions on the subject. This required (1) detailed scrutiny and analysis of many of the uncertain and disputed matters about the causes and effects of acid deposition which were discussed in our earlier report, particularly in light of the increasing amounts of scientific data that have begun emerging on the subject, and also (2) an extensive examination of studies and views on methods of controlling acid deposition and the results and effects of proposed controls.

A major point to be examined was whether cost/benefit analysis could be used to reach a decision on controlling acid deposition by identifying that level or range of control actions which would result in the greatest net benefits. If cost/benefit analysis proved incapable of indicating a range of control decisions that was narrow enough to allow a firm decision on control policy, then we would ascertain what implications current scientific knowledge does have for the individual questions which must be answered in the process of reaching a control decision.

Where the answers to scientific or technical questions were found still to be uncertain, we sought to learn, from those in the research community, what the range of present uncertainty is, what

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[8] U.S. General Accounting Office, The Debate Over Acid Precipitation--Opposing Views--Status of Research, EMD-81-131, Sept. 11, 1981.

[9] U.S. General Accounting Office, Analysis of the Acid Rain Proposal Approved by the United States Senate Committee on Environment and Public Works on July 22, 1982, B-209020, Sept. 24, 1982.

approaches are being taken to reduce the uncertainty, how long this might take, and what their views were on whether or how much further the uncertainty may be reduced. Finally, we have used the knowledge of scientific findings and their implications obtained in this effort to clarify wherever possible the considerations and trade-offs involved in reaching a decision on whether to act now to control acid deposition or wait for further scientific information.

Our report is based primarily on review and analysis of much of the scientific and technical literature in the field, including both primary research journals and many secondary sources, technical studies of control technologies, and technical and economic studies of various control proposals. This was supplemented by detailed discussions with many of the scientists involved in carrying out acid deposition research, both in and out of government. Where appropriate, we analyzed and supplemented data in the literature with our own calculations, for example, in estimating current levels of sulfur emissions from oil combustion, and in computing possible ranges of unemployment resulting from switching to lower sulfur coal.

Our treatment of deposition and emission information for the United States and Canada is based on 1980 data. These were developed by the Joint U.S.-Canada Work Group and are the most recent data developed in a consistent fashion for both countries. We examined more recent data on emissions and deposition for the United States alone and found them generally consistent with the 1980 data.

Particularly in the area of technical and economic analyses of control proposals and their costs and effects, many of the available analyses were performed by or for parties whose interests would be affected by policy decisions on acid deposition. Because of this, we attempted, whenever feasible, to examine the methods and assumptions of such studies by comparing different studies of the same subject presented by those of neutral or opposing views. In some instances, where we could not sufficiently scrutinize certain studies, we note that a study we mention should be considered with some caution.

Review of subject matter, rather than agency activity, was the main focus of our work. However, we did contact, and did receive excellent cooperation from officials at the Departments of Agriculture, Energy, and State; the National Oceanic and Atmospheric Administration of the Department of Commerce; and the Environmental Protection Agency (EPA). We also interviewed personnel from a number of agencies who were involved in the U.S. Interagency Task Force on Acid Precipitation and the U.S.-Canada Work Groups operating pursuant to the August 1980 MOI on Transboundary Air Pollution. Other sources of information included industry and environmental groups; state governments; interstate organizations; Canadian, western European, and Japanese officials; and European as well as North American scientists. In addition to interviews and review and analysis of published research, we sent inquiries to many experts and interest group representatives, asking for their recommended course of action to deal with acid deposition, and the reasons and supporting data for their views. Texts of the letters and the names of those who responded are shown in appendix II.

Since this study examines the facts of a technical issue and their implications for congressional policy decisions, and does not review or evaluate executive agency activities, a draft was not sent to the agencies for their formal review and comment. We did, however, discuss the general observations in the report with EPA officials.

REPORT PRESENTATION

The following three chapters examine in detail the scientific and technical issues that underlie the acid deposition debate in the following order:

--How extensive are the past and anticipated damages associated with acid deposition? (ch. 2)

--How well are the causes of acid deposition understood? (ch. 3)

--What would be the costs and other impacts of proposed strategies to control acid deposition? (ch. 4)

Finally, in the concluding chapter, we summarize our findings and examine the implications this information has for reaching decisions on the acid deposition problem.

CHAPTER 2

HOW EXTENSIVE ARE THE PAST AND ANTICIPATED EFFECTS ASSOCIATED WITH ACID DEPOSITION?

The major concerns about acid deposition's effects deal with present and potential damage to freshwater aquatic ecosystems, terrestrial ecosystems (forests, crops, and soils), man-made materials, and human health. Some of these effects, particularly aquatic impacts, are currently better understood than others; some are difficult to measure, although scientific evidence suggests that damaging effects may be occurring. Defining and reaching scientific understanding of all these effects could be a very long-term enterprise. For example, some scientific studies suggest that acid deposition's effects on soils may take many decades or even centuries to reach completion.

The following sections focus on the observed and potential damages that arise from acid deposition, but also take note of other recognized or possible nondamaging effects where information on them exists. In addition, we briefly discuss the impacts associated with other pollutants derived from acid deposition's precursor emissions such as (1) sulfate aerosol derived from SO_2 emissions and (2) ozone, which is produced by reactions involving NO_x and hydrocarbon emissions. While this study focuses on acid deposition, we mention these other impacts because some control actions proposed to reduce acid deposition would also alter the amounts of these related pollutants, and thereby change the extent of their impacts. The chapter ends with a brief discussion on estimating the cost of damage from acid deposition.

EFFECTS ON AQUATIC ECOSYSTEMS

Acid deposition's effects on aquatic ecosystems are better understood and more amenable to quantification than impacts on other kinds of ecosystems. Current damage in North America, in the form of elimination of populations of fish, has been confined to some of the freshwater lakes and streams in certain limited areas in the northeast United States and southeast Canada. The major concern, however, is the possibility that large numbers of lakes and streams which have already shown some nondamaging acidification, or which appear particularly susceptible to acidification, could be acidified further until damage did occur. These affected or susceptible waters are located both in areas that have already shown aquatic damage, and over much wider areas which are currently receiving substantial levels of man-made acid deposition.

Deposition of acid in greater quantities than are provided in natural precipitation can affect freshwater lakes and streams and possibly ground water, by shifting the acid-base balance of the waters, thus making them more acidic (i.e., of lower pH).[1] Since most of the deposition passes over or through land before entering the water, the type of vegetation present and the nature and depth of soils play important roles in determining effects of acid deposition on waters because they interact with materials present in precipitation before it reaches the waters.

Water acidity is a major factor in determining what lives in a lake or stream, since many aquatic organisms can only survive or reproduce in waters with pHs near or above neutral. For example, crayfish are reported only rarely in waters below pH 6; shellfish species drop sharply at pHs below 7, and only a few are found at pHs below 5.7; the numbers of water plant, snail, and insect species present in a body of water decrease as pH goes below 7; and different fish species will not reproduce, or will even be killed directly, at pHs ranging downward from about 6 to 4.

The ways in which acidified water affects organisms can be very diverse because pH is a very important influence on many kinds of chemical and biological systems and processes. In fish, it has been observed that elevated concentrations of aluminum can be toxic by interfering with gill functioning. Aluminum is very common in soil minerals and becomes increasingly more soluble in water at pHs below 6. In addition, calcium is less common in low pH water, and calcium deficiency can give rise to imbalances of ionic substances in fish, which can also be fatal.

Two main negatively charged substances, sulfate and nitrate, usually accompany the positively charged acid ion in precipitation. This is illustrated in figure 1, which shows the annual average pH of wet deposition (panel a), and the total amounts of acid (H^+: panel b), sulfate ($SO_4^=$: panel c) and nitrate (NO_3^-: panel d) deposited in precipitation on North America

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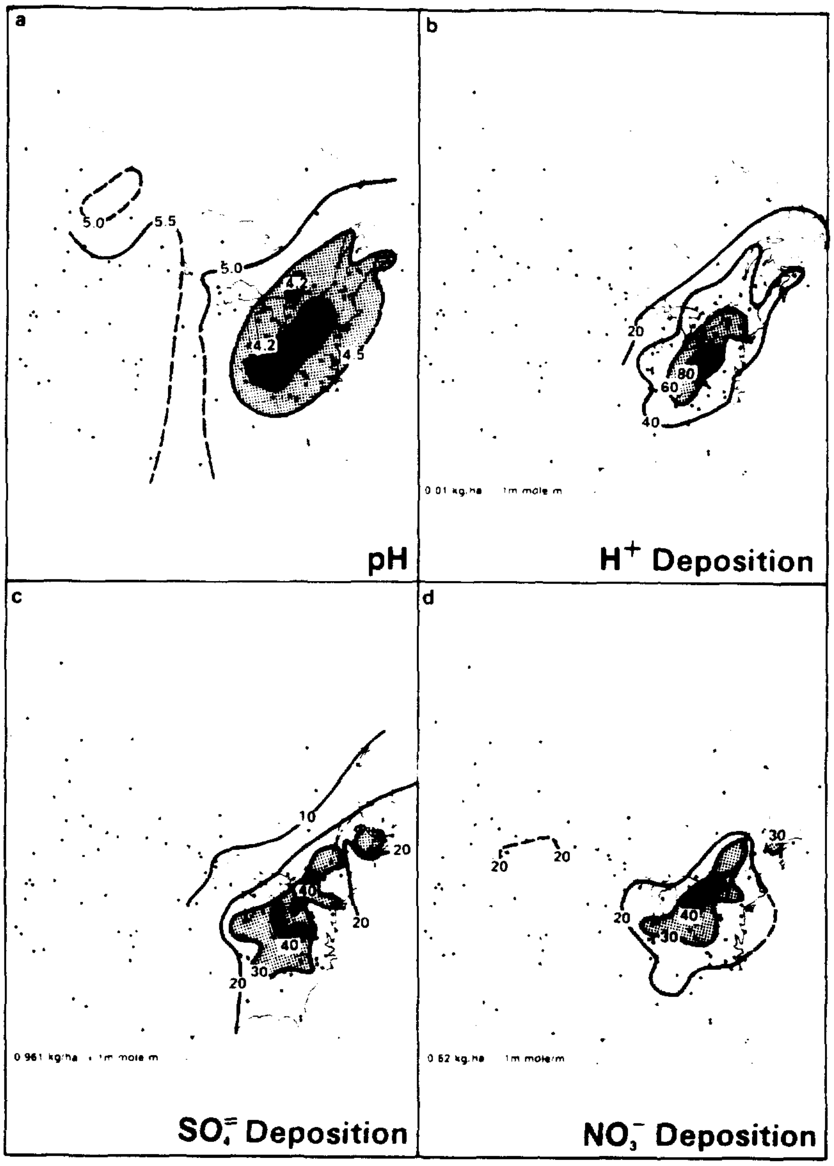
[1] pH is the scientific measure of the acidic or basic (alkaline) state of a liquid. A neutral solution--neither acidic or basic--has a pH of 7. Higher pHs are basic, and lower ones are acidic. The pH scale is logarithmic, so that a change of 1 pH unit means a 10-fold change in the concentration of acid. Rain from an atmosphere containing CO_2 as its only acid would be of pH 5.6. Actual rain in unpolluted areas should average about pH 5 because of a background of acidity from natural emissions. (There can be substantial variations around this average, sometimes as much as 0.5 pH units, and windblown soil particles can sometimes raise pH to neutral or slightly basic.) Rain at pH 4.1 is about 8 times as acidic as this level. For comparison, vinegar has a pH of about 3.0.

Figure I



Maps of Annual Average pH and Total Quantities of Acid, Sulfate, and Nitrate in Wet Deposition on North America, 1980

Dots show the locations of precipitation/wet deposition monitoring stations. The lines connect points computed to show approximately the same pH or the same amount of wet deposition of a substance, as given by the number on each line. The area inside any closed curve has lower pH or larger deposition than that shown on the line. Panel (a): pH, panel (b): acid (H^+), panel (c): sulfate (SO_4=), panel (d): nitrate (NO_3^-).

Source: Work Group 2, Atmospheric Sciences and Analysis, Final Report No. 2F, under U.S.-Canada Meomorandum of Intent on Transboundary Air Pollution, Oct. 15, 1982, figs. 6.1, 6.3, 6.4, and 6.5.

in 1980.[2] The relationship of the three deposited materials can be seen in the similarity of deposition patterns in figures 1b, 1c, and 1d; this similarity is the reason why deposition is referred to as sulfuric and nitric acid. Neither sulfate nor nitrate has been reported to be directly harmful in itself. However, of the two, the sulfate of sulfuric acid is less likely to be absorbed or affected en route from precipitation to the aquatic systems; therefore, sulfuric acid is the main cause of aquatic acidification. In contrast, the nitrate ion is often absorbed by vegetation in the ecosystems on which it falls, usually resulting in the release of a hydroxide (base) ion which can neutralize the acid ion that accompanied the nitrate. Thus, nitric acid is generally regarded as less damaging to aquatic ecosystems than sulfuric acid, at least during the growing season. In winter, however, when plant absorption is lessened or absent, nitrate can accumulate in the snowpack. Also, since sulfate deposition is less in winter, nitrate will represent a larger proportion of the total acidity in the snowpack than during the rest of the year. Therefore, nitrate may make a more significant contribution to the short-term "acid shock" sometimes experienced at spring snowmelts, than it does to long-term acidification, for which sulfate is seen as mainly responsible.

While any changes or losses of species might be considered damaging to a natural ecosystem, the usual view taken has been that economic damage is measured in the reduction or loss of gamefish populations and the resultant loss of both recreational opportunities and income to operators of recreational facilities. In some cases, lowered pH leads to extensive growth of organisms which produce unpleasant masses of floating material or obnoxious odors. In other cases, lakes become clearer at lowered pH because of reduced numbers of microscopic plants, which may make them more attractive.

Acid deposition's current aquatic damages are well documented but thus far geographically limited

Three areas in the northeast United States and southeast Canada have already suffered significant losses of fish

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[2] Deposition patterns have remained generally similar over the last few years, since measurements have begun to be taken by networks of wet deposition monitoring stations in the United States and Canada. Data for the pH of wet deposition in 1982, which are quite similar to those for 1980, have recently been released in: Annual Report, 1983, National Acid Precipitation Assessment Program, U.S. Interagency Task Force on Acid Precipitation, June 1984, p. 36. See app. V.

populations from acid deposition.[3] This has resulted from high levels of acid deposited on areas that do not contain sufficient amounts of basic material which can neutralize it. In addition, many more lakes and streams in these and other areas, show evidence of having been made more acidic, but without significant biological damage to date.

Fishery losses in North America due to acid deposition have been confined to three areas:

--About 180 lakes in the Adirondack Mountains in New York State, mostly at higher altitudes, which had supported natural or stocked brook trout populations in the, 1930's, no longer supported these populations by the 1970's and, in some cases, had lost entire communities of six or more fish species.

--A few dozen lakes in the La Cloche Mountains in southern Ontario, Canada, about 40 miles from major SO_2-emitting smelters at Sudbury, which were studied intensively in the 1960's and 1970's, showed progressive decline and loss of various fish species in parallel with decreases of pH.

--Of 22 rivers studied in southeastern Nova Scotia which had Atlantic salmon fisheries in the 1930's, the 12 with recent pHs above 5.0 showed no mean change in salmon catch through the 1970's. In contrast, the 10 rivers with current pHs at or below 5.0 showed a progressive decline of salmon catch starting about 1950, to a mean in the late 1970's around one-tenth of their 1930's level.

All of these waters have relatively little capacity to neutralize deposited acid because they are in areas with little or no limestone or other basic rock, and many of the watersheds have relatively thin soils. Furthermore, an examination of the acidity of deposition in these three areas shows that all are receiving substantial amounts of deposited acid but with some interesting differences. This can be seen by looking at figure 1a, (p. 11) in which the damaged Ontario lakes lie relatively close to the small pH 4.2 peak on the deposition map which is associated with the metal smelters in the Sudbury area. Thus, they have probably been disproportionately affected by relatively local emissions.

In contrast, the Adirondack area, while within the large pH 4.2 peak on the deposition map, is in the downwind tail of that peak, an area which is some distance east of the most intense SO_2 emission source areas. (See fig. 2, p. 38.) Also, the fact that it is a mountain area makes it have more precipitation, which can result in greater total acid deposited from precipitation of the same average acidity as is received on lower-lying land nearby.

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[3] By loss of a population, we mean complete absence of a species in a single body of water. Thus, one lake may have a number of populations equal to its number of species.

Finally, the Nova Scotia rivers, while still in an area of substantial deposition, are not in the central peak area (pH 4.2 or lower) but appear to be receiving deposition at about pH 4.5, about half or less the level of acidity deposited in the most acidic areas.

Potential damage may be substantial but predictions are uncertain

In addition to lakes and rivers where fisheries have been reduced or eliminated, larger numbers of lakes and streams in these and other areas show evidence of having been acidified to some extent by acid deposition. These and many similar waters in sensitive regions may be at risk of damage if current amounts of acid deposition continue.

Damaging acidification of waters in North America has been of serious concern because of the precedent of very severe progressive damage in Scandinavia. As an example, in the most heavily affected area of Norway, of a total of over 2,800 lakes that had had brown trout populations, more than half of the populations were entirely lost by the late 1970's, at a rate that accelerated since the 1940's, and the majority of the remaining populations were in decline.[4] Such precedents have led to concern about the possibility that the limited effects experienced in North America to date are just the start of much greater and more widespread damage.

In order to determine whether similar damage can be expected in North America, scientists are studying the sensitivity of aquatic systems to acidification. As might well be expected, they are finding that different waters show a wide and essentially continuous range of potential sensitivities, from extremely vulnerable to quite well protected. Estimating the sensivitity of waters to acidification is not solely a question of pH. One criterion widely used is "alkalinity," an approximate measure of the bicarbonate concentration in the water, which has been taken as an index of the ability of the watershed to neutralize acid.

Surveys in the United States have reported extreme sensitivity (defined as alkalinity below 40 microequivalents per liter) in varying percentages of lakes and streams in different areas. The percentage of waters found to have extreme sensitivity ranged from 2 percent to 40 percent in surveys in New England, New York, the upper Midwest, and other areas in the Appalachian Mountains from Pennsylvania to North Carolina and Tennessee, depending on local conditions. Combining both extreme and moderate sensitivity

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[4] I.H. Sevaldrud, I.P. Muniz, and S. Kalvenes, Loss of Fish Populations in Southern Norway. Dynamics and Magnitude of the Problem, proceedings of International Conference on Ecological Impact of Acid Precipitation, Sandefjord, Norway (1980), pp. 350-351.

(alkalinity below 200 microequivalents per liter), the percentages ranged from 13 percent to 81 percent in these same areas.

In Canada, surveys of lake alkalinity showed extreme sensitivity in a number of provinces. It was estimated to extend to more than 10 percent of the lakes in Ontario, with moderate sensitivity in an additional 34 percent of lakes (these percentages were estimated excluding districts near the major smelters). In Quebec, using a related criterion--the ratio of sulfate to bicarbonate--extreme sensitivity was estimated for 28 percent and moderate sensitivity for an added 19 percent of lakes in the part of the province north of the St. Lawrence River. These sensitivity estimates, applied to the hundreds of thousands of lakes in each of these two provinces, have been a central focus of Canadian concern about acid deposition.

Mathematical analyses of the acidification process, largely developed in Scandinavia, have been used to estimate the probable future extent of lake and stream damage in North America, and have suggested it could be very substantial. However, scientists studying acidification of freshwaters here have raised some questions about the applicability of these analyses. In particular, North American scientists have suggested that the mathematical analyses do not take account of the ways that increases in precipitation acidity can be neutralized, possibly to a great extent, by "weathering," the process in which acids dissolve neutralizing material out of rock or from insoluble particles in soils. Current research is seeking to refine the analytic methods to take weathering into account.

Currently, views on the prospects of future aquatic damage in North America among scientists studying the process range substantially, with two questions recognized as central to being able to reach agreement on this subject.

First, while all the scientists we contacted agreed that acidification is a cumulative process, some suggest that a lake's adjustment to a change in acid deposition rate is relatively fast--possibly taking only a few years--while others suggest the process could take a number of decades. This is a very significant difference, since a short response time could mean that most North American waters are currently at or near equilibrium with present rates of acid sulfur deposition--rates which have not changed much in the last decade and, under current policies, are likely to change relatively little over the balance of the century. In contrast, if those who believe lake response is slow are correct, then many areas may still have some time during which their waters will continue to degrade, progressively showing more biological damage.

Second, disagreement exists over effects in soils, which are increasingly recognized as very important to waters, since most of the precipitation that ends up in lakes or streams travels through soil. The question here is again about rates of response to acid deposition. In particular, soil supplies of materials such as calcium and magnesium ions (which can be exchanged to neutralize acids in the weathering process) or the soil's ability to absorb sulfate ions may be diminished enough by many years of processing deposited acids so that the acid-neutralizing capacity of the soil would be exhausted, or at least lowered to a point where it can no longer protect the surface waters from acidification. In this case, as with waters themselves, the uncertainty has to do with the time scale over which this exhaustion has occurred or would occur, with different estimates ranging from a few decades to centuries.

These uncertainties leave a wide range open in estimates of the prospects for further damage in North American waters. On the one hand, Canadian and Swedish government scientists have estimated, through the use of acidification models, that annual sulfate wet deposition rates should be decreased from their present levels to no more than 20 kg/ha (kilograms per hectare), or even as low as 9 kg/ha, to avoid further acidification of most of the vulnerable waters currently experiencing heavier acid deposition.[5] When compared to current levels of wet sulfate deposition in eastern North America, shown by the contours in figure 1c (p. 11) to be near or above 40 kg/ha annually over a significant area in the United States and Canada, this would require sulfate deposition reductions of up to 50 percent or even 75 percent. While the applicability of the acidification models used to make these estimates has been questioned, as noted above, on the basis that they do not take account of neutralization by weathering, the issue of whether the models are correct for North America (fully or even just partially) can only be resolved by empirical study, which is still in its early stages.

At the other extreme is a view which effectively anticipates no further damage from continuation of present rates of deposition.[6] According to this view, in areas which have experienced long histories (in the range of many decades) of sulfate deposition, the response to a prospective reduction of sulfate deposition could be a relatively rapid improvement in surface water quality, in the form of an increase in alkalinity. This difference is a striking illustration of how uncertainty about the time scale of acidification processes makes for uncertainty about the prospects of future aquatic damage.

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[5] Canadian estimate of 20 kg/ha: Impact Assessment Final Report, Work Group 1 under U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Canadian members summary, p. 1-7, Jan. 1983; Swedish estimate of 9 kg/ha: Water Acidification-Effects and Countermeasures, by W. Dickson, National Swedish Environmental Protection Board, in "Ecological Effects of Acid Deposition," Board Report # PM 1636, Jan. 1983.

[6] This view was presented as a hypothesis meriting further study in a discussion with O. Loucks, Science Director, The Institute of Ecology, Indianapolis, Indiana.

In summary, evidence of severe damage to aquatic systems from acid deposition in Scandinavia gives reason for concern that greater damage than has been observed to date may be in process in North America. However, scientists have yet to agree on the prospects, with views covering a spectrum from one that we may experience little further damage if deposition does not increase, to one that anticipates much more damage at current deposition rates. There is agreement that further research is needed to have a better understanding of this question, and discussions with a number of the scientists in this field suggest that the question should be better answerable with relatively few years of continued research.

EFFECTS ON TERRESTRIAL ECOSYSTEMS

Scientific knowledge about acid deposition's effects on terrestrial ecosystems--crops, forests, and soils--was more limited than that about aquatic effects at the end of the 1970's, when concern about the subject first began to spread in the United States and Canada. Except for damage occurring because of high concentrations of pollutants close to emission sources, European studies had shown no consistent pattern of air pollution effects on forests. In contrast, preliminary experiments on crop plants in the United States, in 1979, had shown some cases where plant foliage was damaged and/or yields reduced by simulated acid precipitation.

In the past 5 years, this situation has changed considerably. Increasingly realistic controlled experiments on field-grown crops have been conducted, and have shown very little effect from simulated acid deposition, with indications that even the modest damaging effects observed could be avoidable. On the other hand, serious and spreading forest declines have developed rapidly over the last several years in West Germany and other European countries. Furthermore, growth decline, serious damage, and even death have been observed in one species of tree in the northeastern United States. And more growth declines, affecting a number of tree species during the past 2 to 3 decades in a number of different areas of the eastern United States, have been detected recently.

There are several forms of air pollution that could be responsible for damage to the forests: acid deposition is one of these. While scientists working in the field in both Europe and North America generally agree that air pollution plays a significant role in these forest declines, the mechanism of the damage and, indeed, whether the problems are the same in Europe and North America is not yet known. Therefore, the extent that each of three types of transported pollutants--acid deposition, oxidants (including ozone), or trace metals--are responsible for these forest declines is not known. Hence, control actions aimed at only one type of pollutant are not sure to be effective in slowing or stopping the damage. Still, the magnitude of the damage to forests and the rapidity with which it is developing in Germany and other central European countries have made eastern United States and Canadian forests the leading area for concern about the potential effects of acid deposition in North America.

Agricultural impacts from acid deposition do not appear to be cause for concern at this time

The first study on acid deposition effects on crop plants in the United States, done on potted plants in controlled field chambers during the 1979 growing season, showed some damage to the foliage of most kinds of plants tested with synthetic acid rain at pHs 3.0 and 3.5, but only 5 out of 35 varieties showed any kind of foliage damage with pH 4.0 deposition. Of the 28 crops grown to harvest, yields were not affected in the majority of cases; yields decreased in 5 cases and increased in 6.[7]

Over the intervening years, the experiments on crop plants being done in the United States have become increasingly more realistic and sophisticated. Field plantings of crops have been studied under controlled precipitation and, most recently, with controlled atmospheres as well, so that the effects of combinations of pollutants could be examined. This is to determine whether the effects of combinations of pollutants simply are the sum of their separate effects, whether they may be causing either greater crop damage together than separately (a situation called synergism) or the opposite result, that is, one substance somehow protecting the plants against the effect of the other (antagonism).

As stated in the annual report of the National Acid Precipitation Assessment Program for 1983, in the section on agricultural crops:

> ". . . the effects of photooxidants have been studied for many years and serious damage documented. Evidence shows little if any impact of acid deposition on most crops in pH ranges typical of current conditions."[8]

In specific studies cited in this report, potatoes and corn were unaffected in cases with simulated rain pH as low as 3.4. Other work cited showed some short-term effects on soybean foliage, and one minor variety of soybeans showed an 8-percent yield decrease in response to a change in precipitation, from a near-neutral pH of 5.6 to a more acidic pH of 4.2. However, no effects

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[7] N.R. Glass et. al., "Effects of Acid Precipitation," Environmental Science and Technology, Vol. 16, No. 3, pp. 162A-169A, Mar. 1982.

[8] Interagency Task Force on Acid Precipitation, Annual Report, 1983, Wash. D.C., June 1984, p. 49.

were seen in a number of other more widely used varieties of soybeans under similar conditions.

These studies will have to continue over additional growing seasons to confirm and extend the results so far obtained for other crops and in other locations, and to learn clearly whether, and to what degree, the effects of oxidants are altered by the simultaneous presence of acid deposition. However, the results to date give little reason for concern about the effects on crops of acid deposition alone. Since many crop plants come in different varieties, it is likely that choice among the varieties will be capable of avoiding crop damage from acid deposition in cases where it might occur.

Impacts on agricultural soils might be possible since some soils can resist nutrient depletion when acidified, while others are more vulnerable. The threat of indirect crop damage through acidification of soils can be substantially mitigated, however, through the ongoing practice of liming the land used for many marketable crops. While the cost of increased liming to offset the effects of acid deposition is not trivial, it does not approach the magnitude of potential emission control costs. The New England River Basin Commission estimated, for instance, that acid deposition increases farmers' liming costs in New England by about $800,000 per year.[9]

In summary, the sparse evidence of direct impact of acid deposition on crop plants, the likelihood that alternative unaffected varieties of the same crop could be chosen, and readily available methods to offset impacts on agricultural soils, suggest that acid deposition's impact on agriculture is not cause for concern. Still, more extensive field experiments do appear to be indicted, and the possibility of synergistic effects with oxidants remains to be clarified.

Acid deposition's effects on forests are a matter of serious concern

Acid deposition's effects on forests are not understood at this time, but recent field observations in Europe and the eastern United States show widespread declines in tree growth and substantial damage to and death of trees, which are not explainable by natural causes alone. Scientists working in the field agree that air pollutants play some role in these effects on forests, which gives cause for concern that acid deposition may be contributing to widespread forest damage in North America.

As with agricultural crop research, a good deal of early scientific work in the United States on acid deposition's effects

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[9] New England River Basin Commission, The Economic and Social Significance of Acid Deposition in the New England/New York Region, Boston, May 1981, p. 11.

on trees involved experiments on single plants under controlled conditions. Depending on the species studied, soil type, and other environmental factors, this research demonstrated adverse, beneficial, or negligible impacts on tree growth. Combined with field studies in Scandinavia in the 1970's, which found no consistent pattern of effects on forests that correlated with acid deposition, this led to little concern about forest damage.

Field studies in the United States since 1980, however, have focused on two types of adverse developments in eastern forests, seeking to learn whether acid deposition has had a role in causing them. One is a decrease over the past 2 or 3 decades in the growth rates of several species of trees in many areas of the eastern United States. The other is a reduction in the growth of red spruce in Vermont and New Hampshire over the past 20 years and, more recently, extensive decline and death in red spruce in these two states plus some in New York and North Carolina.

Decreases in tree growth in the United States were first detected in 1980, in three species of pines in the southern New Jersey Pine Barrens region, where they were found to have started in the period 1954-65.[10] The next problem, also recognized in the early 1980's, was the red spruce decline and death in higher elevation forests in New England, with growth decreases again found to have started earlier, as far back as the mid-1960's.[11]

More recent surveys of tree growth rates, started in 1982 to systematically determine whether there were any overall effects occurring in eastern U.S. forests, have measured and compared the sizes of annual growth rings or annual increases in overall tree diameter. They have shown "substantial and easily visible changes in growth"[12] in a number of kinds of coniferous (cone-bearing, usually evergreen) trees, including pitch pine, short leaf pine, red spruce, slash pine and loblolly pine, in areas of 11 states from Maine to the Piedmont region of South Carolina, Georgia, and Alabama. Because scientists "have been unable to explain this . . . effect solely on the basis of natural growth-controlling factors," it was found ". . . reasonable to suspect that airborne pollutants alone or in combination with natural stresses could be involved." However it was emphasized

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[10] A.H. Johnson, et al., "Recent Changes in Patterns of Tree Growth Rate in the New Jersey Pinelands: A Possible Effect of Acid Rain," Journal of Environment Quality, Vol. 10, pp. 427-430, 1981.

[11] T.G. Siccama, M. Bliss & H.W. Vogelman, "Decline of Red Spruce in the Green Mountains of Vermont," Bulletin of the Torrey Botanical Club, Vol. 109, pp. 162-168, 1982.

[12] This quotation and others following in the paragraph are from Prof. A.H. Johnson, University of Pennsylvania, Testimony before Senate Environment & Public Works Committee, Feb. 7, 1984.

> ". . . that there is no clear evidence that acid rain is involved, and there is no clear evidence that acid rain is not involved."

A much worse forest damage problem, which has resulted in growth declines, serious damage, and deaths of a number of species of trees is occurring in West Germany, with the worst effects again seen in coniferous species. In 1982, West Germany's Federal Minister of Food, Agriculture, and Forestry reported damage in 60 percent of fir trees, 9 percent of spruce, 5 percent of pine, and 4 percent of deciduous species in a sample which covered about 60 percent of all forest areas, with about 8 percent of all trees showing some damage.[13] This aroused great concern in West Germany and, while attribution of shares of the damage to particular influences was not yet accomplished, the Minister's report concluded that atmospheric pollutants and their conversion products were a major cause, with SO_2 probably the most important. This has been a major reason for the establishment, as of July 1983, of greatly strengthened controls on SO_2 and other emissions in West Germany, which are, in many cases, a good deal more stringent than those here. (See app. IV.) German concern has since grown still more intense, especially when a survey conducted a year later, in 1983, found that the 8 percent of trees showing damage in 1982 had risen dramatically to 34 percent. Furthermore, other European countries including Switzerland, Austria, and Czechoslovakia are also experiencing similar forest declines.

This damage in West German forests has become apparent and developed to its present severe level in relatively few years, since first being noticed on a small scale in fir trees in the early 1970's. Such a pattern might suggest, by a parallel to the timing sequence of development of fishery damage in Norway, that rapidly spreading forest decline and destruction could be a threat in the near future in eastern North America as well. Caution must be expressed, however, regarding both the nature and the causes of these tree growth declines, damages, and deaths for three reasons.

First, the symptoms, while partly similar, are not all the same in Europe and North America, so it is not clear whether the declines on the two continents are the same phenomenon. For this reason, the future course of the problem in North America may not follow that in Germany, where growth declines for a decade or 2 often preceded the first appearance of external damage, which has then been followed by rapid decline and death.

A second reason for caution is that the complex mixtures of air pollutants, including acid deposition, oxidants, heavy metals,

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[13] Federal Minister of Food, Agriculture and Forestry, Forest Damage Due to Air Pollution: The Situation in the Federal Republic of Germany, English version provided by FRG Embassy, Washington, Datelined: Bonn, Nov. 1982.

particulates, SO_2, and NO_x, which reach forests on the two continents, are not the same. Therefore, it is possible that any one of them or synergistic combinations of two or more could be responsible for the damage instead of, or in addition to, acid deposition.

Finally, even if acid deposition is the only pollutant involved in the forest effects on both continents, the quantities involved are not exactly comparable. In particular, the amounts of sulfur deposited in the German forests (even outside the regions of very high emission density in the industrial Ruhr Valley and in some metropolitan areas) are estimated to be higher than those in North America.

The German deposition estimates are counted as sulfur, while those for North America (fig. 1c, p. 11) are in slightly different units and counted as sulfate. To make the estimates comparable, the German values have to be multiplied by three, to convert from sulfur to sulfate, and the values in figure 1c must be reduced by 4 percent to account for the conversion from the units in the figure to kilograms per hectare. When these conversions are done, the German estimates of total sulfate deposition nationwide, outside the heavy emission areas, are between 120 and 180 kg/ha. Out of these totals, fractions between about one-quarter and one-half, some 30 to 90 kg/ha, are attributed to wet deposition while the larger shares, averaging nearly two-thirds, are attributed to dry deposition. In contrast, figure 1c shows only two small regions in North America where wet sulfate deposition reaches as high as 40 units, equivalent to 38.4 kg/ha, which is close to the lower end of the German range. Furthermore, dry deposition in North America, outside high-emission regions, is estimated to range between equal to, and somewhat less than, wet deposition, so that total sulfate deposition in North America will generally be less than 77 kg/ha outside the two small high-deposition areas or between 43 percent and 64 percent of the German range.[14]

These calculations show that only a relatively small share of the land of eastern North America, well under a tenth by a rough eye estimate of figure 1c, is receiving sulfate deposition at rates comparable to the minimum level being experienced throughout West Germany. This does not represent a margin of safety which justifies ignoring the risks to North American forests, however, since it could imply that the areas with deposition levels reaching those in Germany may soon be facing accelerating declines and tree deaths. Furthermore, damage may still be possible over longer exposure times at lower deposition levels. This might mean that areas several times larger, for example, those with deposition more than half the German level (much of the areas inside the boundaries marked 30 in figure 1c), could also be at risk.

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[14] Some mountain slope areas, which are often covered by clouds or fog, would have higher values for total sulfate deposition, which would be in the German range. (See pp. 49-50.)

Currently available scientific information can neither give good estimates of the dose-response relationship (the amount of damage that would occur at different levels, or "doses," of acid deposition) for forest damage, nor even sort out the contributions to such damage from different components of the mixtures of pollutants present either in Europe or North America. Thus, a more definite prediction of the likelihood, scale, and timing of potential forest damage in North America is not yet possible.

Earlier experimental studies on effects of acid deposition on forest species had demonstrated potential direct effects, under controlled conditions, including physical damage to leaf tissue, leaching of material from leaf surfaces, alteration of microbial populations on exposed tree surfaces, alteration of the species composition of forest ground plants, death of leaf tissue, and synergistic interaction with other environmental stresses. While some scientists had cautioned that these results were hard to detect outside a controlled environment, some had noted that many direct impacts reported in controlled tests could occur as precipitation pH reached down to 4.0 or lower on occasions.[15]

In addition to direct effects of acid deposition on trees, a particular concern to many scientists is the possible indirect effects on forests through soil acidification. Forest soils usually have limited nutrient supplies and may therefore be more vulnerable to damage from further nutrient depletion than agricultural soils. Among the impacts of acid deposition on forest soils are accelerated loss of nutrients through leaching, possible slowing of organic matter decomposition, and mobilization of metal ions. Metal ions are of particular concern, especially aluminum. Aluminum, a relatively abundant element in soils, can be released and absorbed by plants when the soil is acidified. This phenomenon has been observed in field studies as well as controlled experiments.

One of the theories being considered to explain forest damage in West Germany attributes the death of small roots on spruce and beech trees to aluminum release caused by acid deposition. This may be the theory that is best known in the United States, because it is based on many years of work in a well-known experimental forest in Germany.[16] However, other possible explanations have included the gaseous pollutants, ozone and SO_2, combined; a general stress theory combining gaseous pollutants, drought, nutrient deficiencies, and biological disease agents; and a theory that emphasizes soil nutrient losses through leaching.

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[15] G. Abrahamsen and G.J. Dollard, "Effects of Acid Precipitation on Forest Vegetation and Soil," in Ecological Effects of Acid Precipitation, Electric Power Research Institute, Palo Alto, CA (Dec. 1979), p. 3.

[16] B. Ulrich, article pp. 1-29, in Effects of Accumulation of Air Pollutants in Forest Ecosystems, Ulrich & Pankrath, Eds., E. Reidel Pub. Co., Hingham, MA., 1983.

Regarding nutrient leaching, the U.S.-Canada Impact assessment work group noted that:

> "Studies of forest soils (Lea et. al. 1979) indicate that [calcium and magnesium] levels can be leached following applications of acid deposition stimulants. Leaching of these elements from forest soils, as a result of high [sulfate] mobility (Mellitor and Raynal 1981), may lead to a chronic decrease in nutrient status of certain soils."[17]

To be clear on whether acid deposition is affecting North American forests, further research is needed to separate the effects of nitric and/or sulfuric acid from other influential factors that may affect forest health. As was noted in the testimony cited earlier:

> "Convincing evidence regarding the involvement of emissions could emerge from ongoing and planned research, but that evidence is not presently in hand."[18]

In summary, conclusive evidence does not exist to prove that acid deposition is causing forest damage in North America. However, there is damage occurring in forests which is quite likely to be a consequence of air pollution, one form of which is acid deposition. Further, severe damage is being experienced in Europe at sulfate deposition levels which are reached in some areas of North America, and experimental studies have shown that acid deposition alone can damage trees.

EFFECTS ON MAN-MADE MATERIALS

Damage to materials caused by airborne pollutants, including acid deposition, has received much attention because large dollar values are attributed to these impacts. While scientific and economic uncertainties preclude accurate estimates, currently available information indicates that (1) while these effects are caused by a combination of pollutants, acid deposition (particularly dry deposited SO_2) is a substantial contributor to this damage and (2) because most man-made materials are located in developed areas, where most emission sources are also located, the sources of this damage are primarily local rather than transported pollutants.

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[17] U.S.-Canada Work Group I, Final Report under Memorandum of Intent on Transboundary Air Pollution, Jan. 1983, p. 4-44.

[18] A.H. Johnson, op. cit. (footnote 12).

Acid deposition is one of several key pollutants causing materials damage

Acid deposition affects certain materials, although further information is needed to clarify the dose-response relationships and damage rates for acids and specific surfaces. For example, sulfuric acid in water reacts with carbonates (e.g., calcium carbonate) in limestone, marble, and sandstone structures to form gypsum or other sulfates. These sulfates are relatively soluble in water and dissolve quickly. The result is a deterioration of masonry surfaces and, in some cases, a weakening of building structures. Acids also affect copper, zinc, aluminum, and other metals, although the dose-response relationships are not clear.

A major route by which acids affect materials is through dry deposition. Sulfur dioxide directly deposited on buildings can then be oxidized to sulfate and sulfuric acid by humidity or precipitation. Nitric acid can be particularly damaging to metals because the compounds formed when nitric acid reacts with metal surfaces are generally soluble and are easily removed, exposing underlying layers to deterioration.

These acid-induced effects are difficult to separate quantitatively from those of different, although often related, pollutants. Among these other pollutants is ozone, which is reported to damage paints, plastics, and reduce the elasticity and durability of rubber products. Atmospheric ammonia and bacteria are also suspected as indirect agents in accelerating material decay. Furthermore, exposure to a natural unpolluted atmosphere and precipitation can cause significant damage. While this weathering is an obvious source of damage, most studies have not included air quality and meteorological measurements in field experiments. These data are needed to accurately distinguish between pollution-related impacts and damage caused by normal weathering.

Materials damage is related more to local than to transported pollution

Materials deterioration is more closely identified with local pollution sources than are aquatic or terrestrial damage. Sulfur dioxide emissions--the sources of which are often co-located with most man-made materials in large cities and towns--can be dry deposited directly on buildings and other materials and act to damage materials either directly or after conversion to acid when it is wet by precipitation. A number of studies suggest that this local SO_2 represents the single largest source of materials damage. Combined with other local emissions, it therefore appears that local sources are substantially more important in materials damage than distant sources. Indeed, the U.S.-Canada Impact Assessment Work Group notes that "In most cases, the atmospheric load from local sources tends to dominate the low concentrations arriving from remote sources upwind."[19]

This would suggest that secondary ambient air quality standards, which can be established under the Clean Air Act as it is presently structured to protect against such harm to public welfare as economic losses through damage to materials, could be capable of dealing with acid deposition-related materials damage, without modification of the act to take account of long-range transport of pollutants. Indeed, not only does dealing with the major part of materials damage not seem to require modification of the act's basic structure, but also it is quite possible that consideration of damage to materials could give economic justification for SO_2 and/or NO_x emission limits that are more stringent than those presently in place. Such action would, at the same time, lessen the amounts of these pollutants available to be transported to distant areas, and thus, in addition to its benefits in reduced local materials damage, it might also result in reduction of the risk or amount of other classes of damage which derive more from transported pollutants. However, to take full account of the effects of acid deposition, it would still be necessary for the Clean Air Act to treat pollutant effects distant from their release sites, as well as locally.

EFFECTS ON HUMAN HEALTH

Concern about acid deposition's impacts on human health center largely on reports of indirect effects--through contamination of edible fish and drinking water supplies. The possibility of direct health effects, through inhalation of ambient sulfate particulates, is discussed briefly in the following section (p. 28).

Regarding edible fish, reports have been made showing correlations between high mercury concentrations in freshwater fish and high acidity (low pH) in the lakes from which they came in Scandinavia, the northeast United States, and southeast Canada.[20,21] In some cases, as our earlier report[22] noted, the mercury content of these fish has exceeded levels at which U.S. and Canadian governments require actions such as avoidance or reduction of use as food. Importantly, many of these instances

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[19] U.S.-Canada Work Group I, op. cit. (footnote 17) p. 6-1.

[20] Ibid, pp. 5-1 to 5-6.

[21] Committee on the Atmosphere and the Biosphere, U.S. National Research Council, Atmosphere-Biosphere Interactions: Toward a Better Understanding of the Ecological Consequences of Fossil Fuel Combustion, Washington, D.C., Sept. 1981, p. 157.

[22] U.S. General Accounting Office, The Debate Over Acid Precipitation: Opposing Views,--Status of Research, EMD-81-131, Sept. 11, 1981, p. 12.

occurred in locations remote from any point source of mercury discharge. This information suggests that acidified water may play a role in increasing the mercury content of freshwater fish. However, without an understanding of the mechanism linking acid deposition to the mercury-contaminated fish, it cannot be determined whether the correlation is evidence of a causal relationship.

Evidence from an episode in Japan, not linked to acid deposition, shows that mercury from fish can cause nervous system damage in humans, but such cases have been traced to direct industrial releases of mercury-containing material into water. We have seen only one reported instance--in a population of Indians in Northwest Quebec--where there was possible evidence of health impacts (in this case, mild neurological abnormalities) that appeared linked to mercury intake from fish for which there was no evidence of direct mercury releases into the affected waters. Whether this is related to acid deposition remains only hypothetical at this time. Scientists examining the issue, however, have urged that further research be done to determine whether acid deposition does play a role in mercury accumulation in fish and what the scope of the problem may be.

Evidence directly linking acid deposition to heavy metal contamination of drinking water has been found in New York and Pennsylvania, but reports of adverse health effects from such contaminated drinking water have been limited to date. The New York State Department of Health reported that acidified groundwater was responsible for ". . . two cases of elevated lead levels in the blood of children on water supplies which had pH values of 4.2 to 5.0 and had long lead service lines."[23]

Acidic water poses a risk of heavy metal contamination by dissolving the metals from watershed deposits or water storage and distribution systems. Such situations have been observed in a number of places in the Northeast, requiring treatment to raise the pH of the water. Swedish studies have corroborated the American experience and have also demonstrated the occurrence of acidified groundwater which has led to substantial corrosion of household plumbing, resulting in high drinking water concentrations of copper, zinc, cadmium, and aluminum.[24]

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[23] Letters from Dr. G. Wolfgang Fuhs, Director, Environmental Health Center, New York Department of Health, to the Honorable Toby Moffett, former Chairman, Subcommittee on Environment, Energy and Natural Resources, House Committee on Government Operations, dated July 16, 1980, and to Dr. Michael H. Surgan, Environmental Scientist, New York Department of Law, dated July 15, 1981.

[24] Hans Hultberg and Axel Wenblad, Acid Groundwater In Southern Sweden in Drablos and Tollan, Ecological Impact of Acid Precipitation (Proceedings of an international conference) Sandefjord, Norway (Mar. 1980), p. 220.

The risk from heavy metal contamination of drinking water because of acidification, if uncontrolled, could be quite serious, since a number of these metals, including lead, mercury, and cadmium, are toxic. However, in public water supply systems, this is avoided at relatively modest cost by adjusting water acidity before distribution. For example, the cost for water supply systems in New England has been estimated at the order of $1 million annually,[25] essentially the cost of lime used. Mitigation measures would probably not be as cheap and convenient for areas where individual rather than community water supply systems are used, particularly in cases where cisterns are used to gather precipitation or if the groundwater feeding wells were to become acidic. The scope of this possible problem needs to be better assessed.

OTHER AIR POLLUTION ISSUES ALSO INVOLVE THE EMISSIONS ASSOCIATED WITH ACID DEPOSITION

The sulfur and nitrogen oxide emissions associated with acid deposition are also contributors to other types of air pollution by way of several different mechanisms in the atmosphere. Among these pollutants are: ozone and other oxidants produced by reactions involving NO_x, ambient sulfate particulates produced by atmospheric oxidation of SO_2, and ambient concentrations of SO_2 and NO_x. While this study focuses on acid deposition, we briefly mention these related pollutants since their effects could also be reduced by some acid deposition control measures.

Ozone, a secondary product of NO_x emissions, formed by complex reactions in air involving NO_x, hydrocarbons, and sunlight, is an oxidant gas with recognized damaging effects. It has been demonstrated to cause significant crop damage; has had field-documented effects on forests; corrodes man-made materials such as paints, rubber, and plastics; and is strongly suspected of harming human health. In 1979 the entire Northeast and many other sections of the country exceeded the 1-hour national ambient ozone standard--0.12 parts per million--at least several times per year, and a number of metropolitan areas in these regions are still likely not to attain the standard in the late 1980's.

Ambient sulfates and nitrates, the secondary products of SO_2 and NO_x, can have harmful effects without necessarily being deposited as wet or dry acid deposition. They are considered part of the sulfur-particulate mix which is regulated under the Clean Air Act. Excessive concentrations of these particulates have been demonstrated to affect human health when inhaled. Sulfates are found predominantly in the fine-size fraction of particulates for which a separate standard is being considered because they are the only ones which can be breathed deep into the lungs. In addition,

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[25] New England River Basins Commission, The Economic and Social Significance of Acid Deposition in the New England/New York Region, Boston, May 1981, p. 9.

there is good evidence that ambient sulfates play a major role in lessening visibility over large areas of the country.

Ambient SO_2 and NO_x can also affect human health, but relatively few areas in the United States currently exceed primary (health-based) ambient air quality standards for these pollutants.

In addition to these pollutants' physical effects on health and the environment, emissions from upwind regions can have direct economic consequences on downwind areas by contributing to their violating ambient air quality standards under the Clean Air Act. Areas with pollutant concentrations exceeding these standards are designated "non-attainment" areas. This designation can restrict new industrial development or require expensive pollution control measures if new emitting sources are built.

ESTIMATING THE COST OF DAMAGE RESULTING FROM ACID DEPOSITION

In deciding on whether, or particularly to what extent, to control acid deposition, a comparison should be made of the costs and benefits of various possible control policies. By making this comparison, a range of policies could be chosen which would include those that give the greatest net benefits, that is, the largest value of benefits minus costs. In such a comparison, the value of the benefits for each policy option would be taken as the difference betwen costs that would be incurred from damage without taking control action, and the (lessened) cost of damage that would occur if the particular control action were taken.

In the situation that now exists as described in this chapter, however, only very imprecise estimates can be made of the value of damage that can be expected at currently anticipated levels of acid deposition, because of the scientific uncertainty about damage predictions. Due to this great uncertainty, cost/benefit analysis is of limited value in determining appropriate levels of control because the benefits that would result from the expenditures are unknown.

This is not equivalent to saying that control actions would not be worth taking, however. The discussion in this chapter has shown it may be possible that substantial amounts of damage will occur in North America if control actions are not taken--for instance widespread harm to freshwater fisheries or forest damage of the kind developing in West Germany. In such a case, it might well be that strong control measures would be desirable because they could prevent much of the damage at costs less than those that would be sustained from the damage. On the other hand, it may be that very little or no further damage will occur from anticipated levels of acid deposition. In that case, even the risk of damage will diminish greatly as older high-emitting facilities are eventually retired and replaced by new units with much lower emissions, so that major control expenditures could prove to have been unnecessary, and therefore wasteful.

Thus, because of the uncertainty about the extent of damage currently expected, and about the dose-response relationship, there remains great uncertainty about the level of acid deposition which can be allowed without sustaining damage that could be avoided by control actions that would cost less than the avoided damage.

This uncertainty belies the impression given by a widely quoted estimate that the dollar value of damages stemming from acid deposition is $5 billion annually,[26] and reinforces the view expressed by the author of the estimate himself, that the estimate is ". . . no more than an educated guess."[27] A more accurate characterization of the value of acid deposition damage, which could be consistent with the scientific evidence we have reviewed, is the following from a recent study of the impacts of possible acid deposition controls by the Energy Information Administration:

> "Available evidence on the dollar value of benefits that may be obtained from reducing emissions suggests three very broad ranges of benefits: (1) improvements of zero to $5 billion to U.S. water and wildlife, forestry and agriculture, and man-made products; (2) improvements from zero up to some billions of dollars in health improvements and avoided premature deaths in the United States; and (3) a zero-to-near-infinite benefit of avoiding the uncertain possibility of irreversible and serious environmental effects."[28]

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[26] Thomas D. Crocker, et. al., Methods Development For Assessing Acid Deposition Control Benefits (Laramie, WY) 1981.

[27] Thomas D. Crocker, Perspectives on the Economic Benefits of Controlling Acid Depositions (Laramie, Wyo.) June 1981, p. 11.

[28] Energy Information Administration, Impacts of the Proposed Clean Air Act Amendments of 1982 on the Coal and Electric Utility Industries, DOE/EIA-0407, June 1983, p. 43.

CHAPTER 3

HOW WELL ARE THE CAUSES OF ACID DEPOSITION UNDERSTOOD?

The reasons for understanding how acid deposition is caused center on the issue of controlling it. The question can be formulated as: "If controlling acid deposition is desired, as opposed to mitigating its consequences, which steps would be effective in accomplishing this?"

To answer this question, it would be necessary to understand what acid deposition is composed of, where it comes from, and how it gets to where it is deposited. At the qualitative level, the answers to these three questions have been known for a decade or more. Acid deposition is composed predominantly of sulfuric acid and nitric acid. It comes from compounds of sulfur and nitrogen, mostly gases, which are released and spread into the atmosphere by a variety of sources. These compounds, or the acids into which they are transformed, are carried to their destinations by the movements of the air into which they are released, with some drifting back down to the earth on their own and others being carried down by precipitation.

These qualitative answers, however, do not provide sufficient guidance to design control policies or programs. Rather, they indicate how the questions must be refocused to point up the information which would actually be needed to support the design of acid deposition controls. Basically, the necessary information is quantitative:

--How much of the deposited material is sulfuric acid and how much is nitric acid?

--What are the amounts and identities of the emitted compounds of sulfur and nitrogen, where are they released geographically, and by what sources?

--Where and how are the compounds transformed into acids, how quickly do the transformations occur, and what controls the rates of transformations?

--How far, and in what directions, are the emitted compounds and their acid products transported?

--How do transformation, transport, and deposition combine quantitatively to link emissions to deposition?

Finally, in light of the information and questions in chapter 2 about the relationships between amounts and rates of deposition and of damage, it would be very useful to know how long and how intensely acid deposition has been occurring, both to examine the role of man-made emissions as sources, and to help predict the prospects for future damage.

This chapter describes the current state of knowledge about acid deposition, with the presentation designed to focus on the quantitative questions just stated. The information is discussed in the following order:

--Deposition and emissions: What are the sources of the substances which give rise to acid deposition?

--Time trends: How have sources and deposition changed over time?

--Transformations: How do emitted compounds contribute to acid deposition?

--Transport: In which directions are acids and acidifying substances carried by the atmosphere, and how far and how much is transported?

DEPOSITION AND EMISSIONS: WHAT ARE THE SOURCES?

Acid deposition occurs both in precipitation ("wet deposition") and in the dry form (as gases and particles). Increasing concern about the issue has led to a much greater intensity and scope of monitoring wet deposition in recent years. It is now routinely monitored by several networks in North America and western Europe, and at a number of other points around the world. Dry deposition is not measured routinely because monitoring techniques are not yet developed, though some estimates of its overall nature, extent, and distribution are available from indirect measurements and basic physical principles applied through mathematical modeling. While wet acid deposition occurs essentially entirely as sulfuric and nitric acids, dry deposition of acidic sulfur compounds usually occurs predominantly as SO_2, which is transformed to sulfuric acid after deposition. For this reason, as well as the fact that some acidic sulfur compounds can be deposited in the form of other sulfates besides sulfuric acid, the deposited material is often referred to loosely as "sulfate," or just "sulfur."

At the regional level, precipitation acidity estimates for the northern hemisphere show three strongly acidic regions--eastern North America, western Europe, and Japan--with average precipitation pH in substantial areas of these regions reaching below pH 4.5, and two large regions--the plains of western North America and central Asia--showing relatively alkaline precipitation with pH above 6.0 in substantial areas.

In the most studied regions--western Europe and eastern North America--wet deposited acid is currently about two-thirds sulfuric acid and one-third nitric acid. Deposition maps for North America for 1980 were shown in figure 1 (page 11). Deposition patterns have generally been similar from year to year, since systematic measurements were started in the late 1970's. For example, the current annual report of the Federal Interagency Task Force on Acid Precipitation[1] shows a pH pattern for 1982 (see app. V) that is quite similar to that shown in figure 1a for 1980. Most studies and comparisons have focused on 1980 data, since that was a year for which detailed measurements and estimates were made of emissions and deposition in both the United States and Canada using consistent methods. Some areas in the western United States show higher proportions of nitric than sulfuric acid in precipitation. Acid deposition, almost entirely as sulfuric acid, is also seen in some remote areas, such as on isolated islands in the Pacific, where it can show pHs substantially below the 5.6 value calculated for theoretically "clean" rain. Research indicates that natural sources alone should result in a range of precipitation acidities in unpolluted locations centered around pH 5.0, but occasionally going a few tenths of a unit below 5, as well as up to 6 or higher.[2]

The three strongly acidic regions--eastern North America, western Europe, and Japan--are the three places on the earth where man-made SO_2 and NO_x emissions are most concentrated. Research discussed in our previous report had already shown that man-made SO_2 emissions substantially exceeded natural sulfur compound emissions in eastern North America. In the interval, the same has been found true for NO_x, with an estimate that only about 10 percent of the NO_x emissions over eastern North America are from natural sources.[3] Scientists have known for some time that essentially all sulfur and nitrogen emissions do get deposited, rather than accumulating in the atmosphere. The central implication of these two facts is the conclusion that man-made SO_2 and NO_x emissions are the predominant source of the acid precipitation problem in eastern North America and other high-emissions areas. Also, western United States areas with high proportions of nitric acid in precipitation tend to have proportionately larger NO_x emissions (mainly from mobile sources) and smaller SO_2 emissions, thus further supporting the association of acid deposition with man-made emissions.

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[1] Interagency Task Force on Acid Precipitation, Annual Report, 1983, National Acid Precipitation Assessment Program, June 1984, p. 36.

[2] R.J. Charlson & H. Rodhe, "Factors Controlling the Acidity of Natural Rainwater," NATURE, Vol. 295, pp. 683-5, Feb. 25, 1982.

[3] J. Logan, "Nitrogen Oxides in the Troposphere: Global and Regional Budgets," Journal of Geophysical Research, Vol. 88, pp. 10785-10807, Dec. 1983. Logan explains that earlier high estimates of natural NO_x emissions were based on erroneously high older readings of NO_x concentrations in the atmosphere in remote regions rather than on actual emission estimates. Lower concentration readings obtained with modern instruments lead to lower estimates of natural NO_x emissions.

The mid-continental regions of relatively alkaline precipitation in North America and Asia--with pH even higher than the 5.6 expected from a theoretical "clean" atmosphere--are dry or semi-arid. In these regions, alkaline material from wind-blown soil dust is regarded to be the dominant influence on the pH of precipitation. The fact that precipitation in some places such as remote Pacific islands, far from major emitting regions, can still be more acidic than pH 5 is explained by a combination of patchiness in the geographic distribution of natural sources and variability in the efficiency with which clouds absorb natural acid material from the air[4], plus some possible very long-range transport from industrial regions.

Man-made emissions in North America

The emissions which are the source materials of acid deposition are certain oxides of sulfur and nitrogen. While some amounts of each of these are emitted by natural sources, and some sulfur oxides can be formed in the atmosphere from other natural sulfur compound emissions, the great preponderance of the oxides involved in acid deposition in major industrial areas like eastern North America or central and western Europe are emitted by man-made sources. Since emission and deposition take no notice of political boundaries, we will look at Canadian and U.S. emissions together.

Sulfur oxides, predominantly in the form of SO_2, with a very small portion already fully oxidized to some form of sulfate, are formed by combustion from sulfur contained in fuels or from sulfur involved in certain industrial processes or contained in metal ores being extracted.

Some fuels contain nitrogen which can also react with oxygen from the air to form nitrogen oxides ("fuel NO_x") when the fuel is burned. In addition, nitrogen from the air can combine with oxygen to form nitrogen oxides ("thermal NO_x"), especially at the very high temperatures reached in many of the more efficient fossil fuel combustion systems, even if the fuel has little or no nitrogen. Two different nitrogen oxides formed in these processes are reactive in the atmosphere and so can contribute to acid deposition: NO (nitric oxide) and NO_2 (nitrogen dioxide). NO in the atmosphere is generally transformed rapidly to NO_2; therefore, little distinction is made between the two and they are referred to together as NO_x.

Estimates of sulfur oxide and NO_x emissions from the United States and Canada are shown for 1980 in table 1. This is the last year for which estimates were made for both countries on the same basis. While U.S. emissions decreased between 1980 and 1982, the decrease was not very large and projections of future emissions suggest that these decreases will be overridden by increases

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[4] R.J. Charlson and H. Rodhe, op. cit.

expected over the next several years. Further data on present, past, and anticipated man-made emissions in North America are described and analyzed in detail in appendix I. Table 1 shows that U.S. emission estimates are substantially greater than those for Canada--more than fivefold greater for SO_2, more than threefold greater for sulfates, and more than tenfold greater for NO_x. Comparison of totals in the table also shows that sulfur emitted in the form of sulfates is estimated to make up only a very small part of total sulfur oxide emissions in each country--somewhat over 2 percent in the United States and under 4 percent in Canada.

Table 1
Estimated Man-Made Emissions of Sulfur Oxides and NO_x in 1980

	United States			Canada		
	SO_2	Sulfates	NO_x[a]	SO_2	Sulfates[b]	NO_x[a]
	(millions of metric tonnes)			(millions of metric tonnes)		
Electric utilities	15.8	0.26	5.6	0.74	0.02	0.25
Industry	5.3	0.16	4.2	1.54	0.07	0.37
Residential and commercial	0.8		0.7	0.21		0.09
Nonferrous smelters	1.4	0.02	-	2.13	0.03	0.01
Transportation	0.8	0.03	8.5	0.16	-	1.11
Other	-	0.11	0.3	-	0.07	-
Total	24.1	0.58	19.3	4.77[c]	0.19	1.83

[a] Counted as NO_2.

[b] Canadian sulfate emissions estimated for 1978.

[c] Totals do not add exactly because of rounding.

SOURCE: Work group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Final Report June 1982, tables A.2.1 to A.2.5.

Other data in appendix I, on the sources which contribute the sulfur in SO_2, show that the great majority in the United States came from the sulfur in combustion fuels--some 67 percent from coal burning and 16 percent from oil burning--while the remaining parts came from the sulfur in metal ores being smelted and from a mixture of other industrial processes and minor fuels. The main difference in Canada is that smelters are the leading SO_2 source.

The distribution of NO_x sources in the United States is much different from that for SO_2, largely owing to the production of thermal NO_x as well as fuel NO_x, and the much greater role of transportation vehicles. Taking all transportation and stationary source combustion emissions into account, petroleum fuel burning contributed 51 percent of total U.S. NO_x emissions, of which more than nine-tenths came from transportation. Coal burning followed, producing 27 percent of the NO_x total, more than nine-tenths of which was from utilities. Natural gas combustion was responsible for 16 percent of NO_x emissions (about two-thirds from industry and the rest mostly from utilities), while the remaining 6 percent was emitted by various industrial processes and minor fuels. The division of NO_x sources in Canada is roughly similar, except that transportation has an even larger share and utilities a smaller one.

Eastern North American emissions

Table 2 shows the sources of SO_2 and NO_x emissions in eastern Canada and the United States.

Table 2

1980 Emissions of SO_2 and NO_x in the Eastern United States and Eastern Canada[a]

	SO_2				NO_x[b]			
	U.S.		Canada		U.S.		Canada	
Source type	Mt x 10^6	Percent	Mt x 10^6	Percent	Mt x 10^6	Percent	Mt x 10^6	Percent
Electricity generation	14.58	73.5	0.67	16.8	4.34	34.1	0.17	14.9
Nonutility combustion	2.81	14.2	0.74	18.6	2.19	17.2	0.24	21.1
Transportation	0.49	2.5	0.12	3.0	5.63	44.2	0.70	61.4
Nonferrous smelters	0.16	0.8	2.09	52.5	0.57	4.5	0.03	2.6
Oil and gas processing	0.50	2.5	0.08	2.0				
All others	1.29[c]	6.5	0.28	7.1				
Total	19.83	100.0	3.98	100.0	12.73	100.0	1.14	100.0

[a] Eastern United States includes the District of Columbia and 31 states east of or bordering the Mississippi River; eastern Canada includes Manitoba, Ontario, Quebec, and the Maritime Provinces.

[b] Counted as NO_2.

[c] Out of 6.5 percent of SO_2 emissions in the "all others" category in the eastern United States, about 1 percent comes from sulfuric acid plants and about 2 percent each from the cement and iron and steel industries. No breakdown of this category of SO_2 emissions was found for eastern Canada.

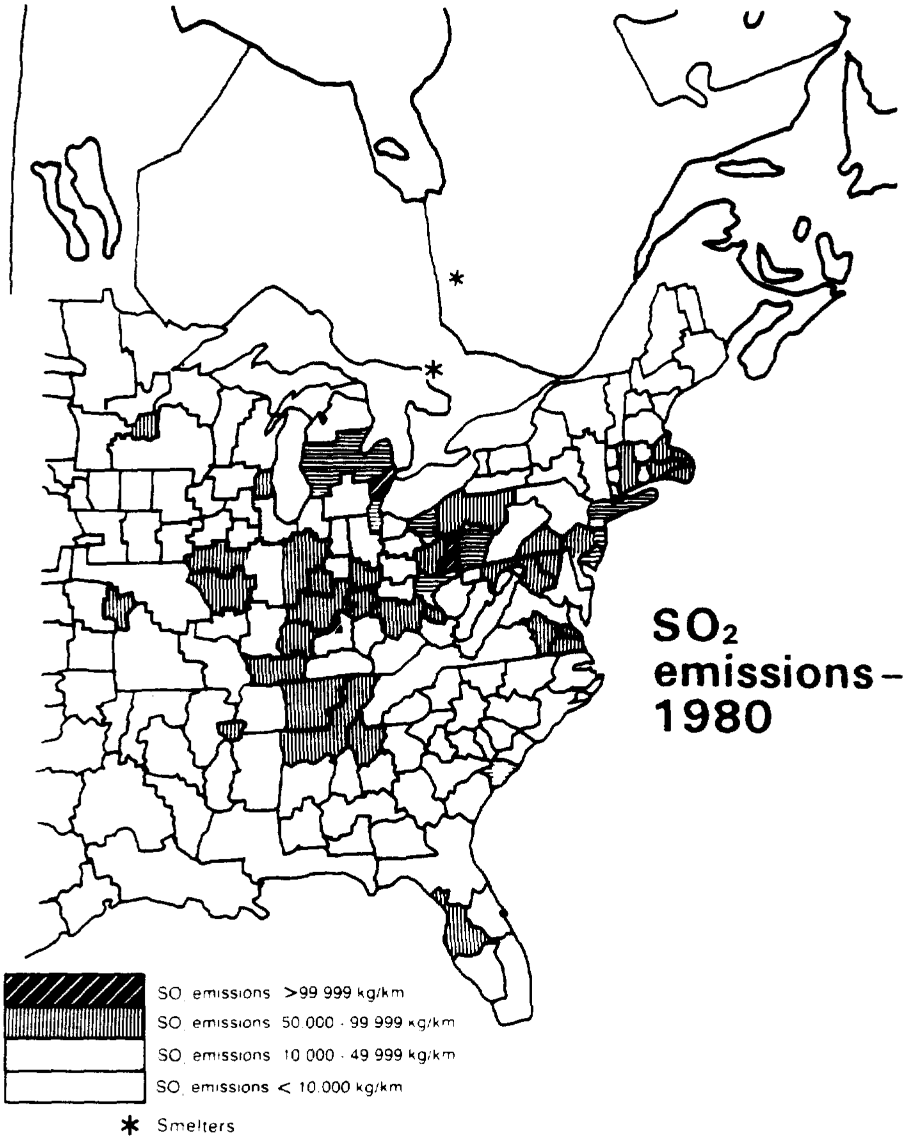
SOURCE: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B Final, June 15, 1982, tables B.2.1 - B.2.5.

Comparisons with table 1 show that in both countries, most of the SO_2 and NO_x is emitted in the more heavily populated and industrialized eastern areas. However, this imbalance is greater for SO_2 than for NO_x, because the leading class of SO_2 emitters is located overwhelmingly in the eastern part of each country. In 1980, electric utilities contributed about two-thirds of all SO_2 emissions in the United States, with over nine-tenths of these utility emissions released in the 31 eastern States. In Canada, utilities are a much less significant source of SO_2 than in the United States, while the leading source--nonferrous smelters--emitted nearly half of all SO_2 in 1980. Essentially all of this was emitted in Manitoba and provinces to the east. Largely as a result of these imbalances, 82 percent of all U.S. SO_2 and 83 percent of all Canadian SO_2 were emitted in the eastern part of the continent. In comparison for NO_x (for which transportation and nonutility combustion are more important sources and are more evenly distributed geographically than they are for SO_2), 66 percent of U.S. emissions and 62 percent of Canadian emissions were released in the eastern part of the continent in 1980.

For SO_2, the great majority of emissions in the East in each country come from a limited set of stationary sources. In the eastern United States, electric utilities are dominant, producing 74 percent of all SO_2, followed by nonutility stationary source combustion at 14 percent, of which industry provided the main share. In eastern Canada, nonferrous smelters alone emit slightly over half of all SO_2, while electric utilities and nonutility combustion are close to equal, together accounting for about another one-third of eastern SO_2, most of which is from utilities and industrial combustion alone.

A map of SO_2 emission densities (emissions per unit area) for the eastern United States is presented in figure 2, which also includes the two large Canadian smelters near the Ontario-Quebec border, to show the geographic distribution of eastern North American SO_2 emissions. A large pattern of high emission density is found in the industrial Midwest and Appalachians, stretching from the Ohio River Valley and areas on Lakes Erie and Michigan as far as northern Alabama and eastern Missouri. Several separated smaller areas of high emission density are also found at the Canadian smelters and scattered along coastal states from southwestern Florida to southern New England.

Figure 2



Density of Man-made SO_2 Emissions From Eastern North America, 1980

The borders in the United States are those of standard Air Quality Control Regions, which in some cases follow and in other cases cross state borders. A similar map was not available for Canada, but the two largest SO_2 sources in eastern Canada, smelters at Sudbury, Ontario and Noranda, Quebec, are shown by asterisks.

Sources: U.S. emission densities: paper given March 29, 1982, at American Chemical Society meeting by B. Niemann, USEPA Acid Deposition Staff and joint U.S.-Canada Work Group on Atmospheric Sciences and Analysis; Smelter locations from atlas map of Canada.

In summary, of all eastern North American SO_2 emissions, at least 83 percent in Canada and 84 percent in the United States come from utilities, industrial combustion and smelters, with the utilities being predominant in the United States and the smelters in a majority, but less dominant position in Canada.

Nitrogen oxide emissions are different from sulfur emissions in one major regard--44 percent of eastern U.S. NO_x emissions and 61 percent of eastern Canadian NO_x emissions were estimated in 1980 to come from transportation sources. These shares represent the same percentage of eastern NO_x as transportation's share is of total national NO_x emissions in each country. Utilities and non-utility combustion contribute most of the balance of eastern NO_x in both countries, with utilities second (at 34 percent) in the eastern United States and third (at 15 percent) in Canada. Again, industrial combustion dominates the nonutility stationary-source NO_x emissions in the eastern parts of both countries. In short, mobile sources are the leading NO_x emitters throughout North America and, combined with utility and industrial combustion, are responsible for 90 percent or more of all NO_x emissions in the eastern parts of both the United States and Canada.

TIME TRENDS IN ACID DEPOSITION

Scientific studies show that current levels of acid deposition in heavily affected areas, such as eastern North America, are much greater than those that occurred from natural sources before man-made emissions increased sharply with the Industrial Revolution. Also, it has been found that some lakes which were still near neutral acidity, and therefore able to support fish over recent millennia, have been acidified sharply in this century. This evidence is seen in two types of studies. First, actual samples of past acid deposition have been found in preserved snow in glaciers and ice caps. Second, changes in lake acidity are detected by examining the remains of certain preserved organisms found in lake bottom sediments. The following sections discuss these findings, and then present the evidence regarding recent decades and present trends of acid deposition in the area of concern in eastern North America.

Acid deposition history in preserved snow

Measurements of acidity in preserved snow indicate that contemporary levels of acid deposition did not occur before the Industrial Revolution. However, this finding has been obscured by inaccurate descriptions of the data that demonstrate it.

In a statement to a congressional subcommittee, an industry association asserted that

> ". . . precipitation which fell before the Industrial Revolution and is preserved in glaciers or ice

> caps in Antarctica and Greenland has been found in some cases to be highly acidic."[5]

Examining the two articles on which this remark was based,[6] we found that the snow and ice samples do show sulfuric acid, but generally at low levels which would correspond to precipitation pHs in the range of 5.2 to 5.5. Brief 1- or 2-year intervals of higher acidity noted in the Greenland study are clearly attributed by the authors of the study to volcanic eruptions. Moreover, even in the two cases in which major volcanic eruptions on Iceland produced the highest sulfuric acid levels deposited on relatively nearby Greenland, the ice only reached acidity levels corresponding to about pH 5.

Other measurements performed by scientists from the University of New Hampshire, which found acidity preserved in snow in the Himalayas, have also been cited in the press, although not in scientific publications, to support arguments that natural sources can explain most observed acid deposition. In discussing this work with one of the scientists who did it, however, we were told that they found very little sulfate or nitrate, so the acidity they measured most likely came from carbon dioxide from the atmosphere but clearly did not arise from deposition of sulfuric or nitric acid.

Thus, the significance of the acidity in these ice and snow samples is quite the opposite of that suggested in the statement quoted above. Rather, it shows that precipitation in previous ages has (1) generally been even less acidic than the current background average in remote areas and (2) only about matched this current background for 1 or 2 years at a time, after a few volcanic eruptions. This contrasts strongly with the present situation, which shows that the annual average precipitation in the most acidic areas of eastern North America is some 6 to 8 times as acidic as the background level, and has persisted at least since the mid-1960's.

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[5] Statement by the American Mining Congress, submitted to the Subcommittee on Health and Environment, House Committee on Energy and Commerce, Oct. 5, 1981.

[6] We were referred to the following two articles when we asked American Mining Congress officials for the basis of the quoted statement: (a) R.J. Delmas, A. Aristarain and M. Legrand, "Acidity of Antarctic Snow: A Natural Reference Level for Acid Rains" in Ecological Impact of Acid Precipitation, D. Drablos and A. Tollan, Eds., (Conference Proceedings, Sandefjord, Norway, March 11-14, 1980), pp. 104-105. (b) C.U. Hammer, H.B. Clausen and W. Dansgaard, "Greenland Ice Sheet Evidence of Post-Glacial Vulcanism and Its Climate Impact," Nature, Vol. 288, pp. 230-235 (Nov. 20, 1980).

Prehistoric and recent lake acidification evidence in sediments

Historical lake acidity data, measured in lake sediments, show that certain North American and European lakes, which had stabilized at near-neutral pHs capable of supporting fish at some time after glaciers receded thousands of years ago, have been rapidly acidified in recent decades.

The method used to examine long-term, as well as recent, lake acidification is based on studies of the remains of small organisms called diatoms in lake bottom sediments. Many species of diatoms have been classed according to the pH of water in which they can live. On the basis of these classes, diatom remains in sediments can be used to estimate pH values of the lake waters from which the sediments formed at earlier times.

In some Swedish[7] and Adirondack[8] lakes, it was found from deep sediments that the lakes slowly acidified moderately in the first few thousand years after the glaciers receded, but remained at acidities still able to support fish. Then, in the next 4,000 to 6,000 years, either no changes of pH took place or only small changes associated with changes in dominant tree types in the watersheds.

In the last several decades, however, starting about in the late 1950's, steep drops occurred in the Swedish lake pHs, from values at or above pH 6 to levels as low as about pH 4. The Adirondack lakes also show diatom evidence of very recent acidification, and another Adirondack lake study[9] showed a sharp pH drop from diatom analysis of the most recent sediment layers, corresponding to a known recent history of acidification. A similar

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[7] I. Renberg and T. Hellberg, "The pH History of Lakes in Southwestern Sweden, as Calculated from the Subfossil Diatom Flora of the Sediments," Ambio, Vol. 11, No. 1, pp. 30-32 (1982).

[8] D.R. Whitehead, D.F. Charles, S.E. Reed, S.T. Jackson, and M.C. Sheehan, "Late Glacial and Holocene Acidification History of Adirondack (N.Y.) Lakes." Manuscript submitted for publication, August 1982.

[9] A. Del Prete and C. Schofield, "The Utility of Diatom Analysis of Lake Sediments for Evaluating Acid Precipitation Effects on Dilute Lakes," Arch. Hydrobiol., Vol. 91, No. 3, pp. 332-340, (1981).

pattern was reported in southern Norwegian lakes,[10] where diatom sequences show sharp drops of pH starting between about 1930 and 1945. Other research has shown that this rapid lake acidification in Scandinavia is not a consequence of other human influences besides acid deposition. However, the sediment study technique has not been in use in North America as long or as much as in Scandanavia; therefore, some further studies will be needed here before it can be fully proven that human activities such as farming or logging did not play some role in lake acidification here.[11]

In summary, evidence in sediments shows that lake acidification, which had apparently occurred only slowly (on a timescale of milennia) and had then stabilized for many centuries, has been accelerated sharply in this century in sensitive areas receiving heavy acid deposition. Scandinavian studies show that other human influences were not responsible for acidification there, but they have not yet been excluded in all cases in eastern North America.

Recent acid deposition trends in the eastern United States

Trends of acid deposition in the United States since about the 1950's have been disputed, as we noted in our prior report. This occurred in part because clear data on which to base the findings are very scarce. Our review of the limited data and analyses of this data done recently indicate that the following appear to be consistent with available information:

--Total precipitation acidity in the northeastern United States has remained relatively unchanged, at least since the 1960's;

--Indications are that precipitation acidity in the southeastern United States has increased in the past 2 to 3 decades;

--The proportion of sulfuric acid has dropped somewhat, while the proportion of nitric acid has risen, in precipitation in the eastern United States in recent decades;

--Partial correlations exist between trends in deposition and trends in emissions in recent decades;

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[10] R.B. Davis, S.A. Norton, D.F. Brakke, F. Berge and C.T. Hess "Atmospheric Deposition in Norway During the Last 300 Years as Recorded in SNSF Lake Sediments. IV. Synthesis and Comparison with New England" in Ecological Impact of Acid Precipitation, D. Drablos and A. Tollan, Eds. (Conference Proceedings, Sandefjord, Norway, Mar. 11-14, 1980), pp. 274-5.

[11] View expressed in discussion with GAO by Dr. R.B. Davis, lead author of previous reference and other papers on lake sediment deposition studies.

--Reconstructed estimates of emission trends over a longer period suggest that acid deposition became substantial in the northeastern United States early in this century. This may have implications for the prospects of damage from acid deposition developing elsewhere in the future.

Total precipitation acidity in the Northeast has been relatively unchanged since the 1960's

Comparisons of eastern U.S. precipitation acidity in the 1950's, 1960's, and 1970's, done in the mid-1970's, had reported increasing acidity and expansion of the region of high acidity. These conclusions were based on efforts to reconcile different and discontinuous data sets and, in part, on inferred rather than measured acidity. These approaches were the grounds cited in arguments against the conclusion that acidity had been increasing.

Two sets of continuous data bearing on the Northeast are now available for the period from the mid-1960's, and both support opposite conclusions from the earlier work. Measurements at Hubbard Brook Experimental Forest in New Hampshire show variability, but no consistent trend in mean annual precipitation pH from 1964 to 1977.[12] Measurements of the acidity of bulk precipitation, which includes a part of dry deposition as well as all of wet deposition, carried out since 1965 in a nine-station network in New York State ". . . indicate little change in pH from 1965-1978 within New York State as a whole . . . ."[13] This report records about a 0.2-unit pH decrease in the western part of the state and a similar increase in the east, but it indicates the possibility of variable neutralization of samples by particles from dry deposition which may mask regional changes, and it reinforces this view by suggesting the desirability of separating dry and wet contributions in future studies.

Precipitation acidity in the Southeast U.S. has increased over the past 20 to 30 years

Recent views of historic precipitation acidity in the eastern United States, surveyed by the Atmospheric Science Sub-Group of

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[12] G.E. Likens, F.H. Bormann and J.S. Eaton, "Variations in Precipitation and Streamwater Chemistry at the Hubbard Brook Experimental Forest During 1964 to 1977," in Effects of Acid Precipitation on Terrestrial Ecosystems, T. Hutchinson and M. Havas, Eds. (1980), pp. 443-464.

[13] N.E. Peters, R.A. Schroeder and D.E. Troutman, Temporal Trends in the Acidity of Precipitation and Surface Waters of New York, U.S. Geological Survey, Water Supply Paper No. 2188, 1982.

the U.S.-Canada Atmospheric Sciences and Analysis Work Group[14] conclude that there is evidence of a southward and westward spread of high precipitation acidity in the United States since the 1950's.

An example of research which we have reviewed supporting this finding about the Southeast is a study of 1978-79 precipitation in Florida.[15] This paper reports a 1.3- to 1.6-fold increase in sulfate concentration, and a 3.7- to 4.5-fold increase in nitrate concentration, compared with precipitation records from 1956, which implied pH decreases averaging close to a full pH unit over the interval. Furthermore, the report found total sulfate deposition increased even more, up to fourfold, in the northern part of the state, which had the most acidic precipitation in the recent measurements. Similarly, another report indicating lowered precipitation pH in the southeastern United States, in Great Smokey Mountains National Park on the Tennessee-North Carolina border,[16] notes a decline from an average precipitation pH of 5.3 in 1955 to 4.4 in 1973.

Nitric acid has been increasing in proportion to sulfuric acid in Eastern U.S. precipitation

In three of the locations discussed above, where comparisons were possible between earlier and recent deposition of nitrate and sulfate, research has found that the proportion of nitric to sulfuric acid was higher in recent precipitation than in precipitation 1 to 2 decades earlier.

At Hubbard Brook, New Hampshire, data extending from 1964 to 1979 showed that the importance of nitric acid had increased about 50 percent relative to sulfuric acid.[17] In the U.S. Geological Survey New York State network, data from 1965 to 1978 showed declines in sulfate concentrations at most stations, while nitrate

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[14] Atmospheric Sciences Review, Report No. 2-14, Prepared by Work Group 2 Under U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, July 10, 1981.

[15] P.L. Brezonik, E.S. Edgerton and Charles D. Hendry, "Acid Precipitation and Sulfate Deposition in Florida," SCIENCE, Vol. 208, pp. 1027-29, May 30, 1980.

[16] R. Herrmann and J. Baron, "Aluminum Mobilization in Acid Stream Environments, Great Smokey Mountains National Park, U.S.A.," pp. 218-219, in Ecological Impacts of Acid Percipitation, D. Drablos and A. Tollan, Eds., Proceedings of Symposium, Sandefjord, Norway, Mar. 1980.

[17] J.N. Galloway and G.E. Likens, "Acid Precipitation: The Importance of Nitric Acid," Atmospheric Environment, Vol. 15, pp. 1081-85, 1981.

concentrations did not change sufficiently to establish a trend.[18] Thus, the relative proportion of sulfate declined compared with nitrate, as was also found at central New York stations between 1955 and 1973 in another study mentioned in the U.S. Geological Survey paper.

Finally, in the Florida study,[19] the increases in precipitation nitrate concentrations between 1956 and 1979 were much greater on a percentage basis than those for sulfate, as shown by the proportions of the two, which were about 1 nitrate:7 sulfate in 1956, but only about 1 nitrate:2.4 sulfate in 1979.

While all these cases still show greater amounts of sulfuric than nitric acid in precipitation, if these trends were to continue over several more decades, then nitric acid could become the dominant form of acid in precipitation. If this were to become the case, then it might be necessary to reconsider the finding (noted in the previous chapter) that sulfuric acid is the most damaging deposited acid for ecosystems.

Recent trends in deposited acidity show some correlations with emission trends but also show that other factors influence deposition

The trends of precipitation acidity in the United States noted in the previous sections correlate to some extent, but not completely, with regional emission trends.[20] This points to complexities in the relations between emissions and deposition.

The increase in the importance of nitric as compared with sulfuric acid in precipitation in the Northeast between the mid-1960's and the late 1970's parallels an increase in the relative proportions of NO_x as compared with SO_2 in emissions in that region. However, the relative change of emissions is greater than the relative change in deposition. Also, counted in terms of potential acidity, the total emissions in the Northeast fell somewhat over that interval, yet total deposited acid was relatively unchanged.

For the entire Southeast, total emissions of SO_2 and NO_x each increased about 2-1/2 times from 1955 to the middle and late 1970's, consistent with the direction of change of deposition, but

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[18] Peters, et al., op. cit.

[19] Brezonik, et al., op. cit.

[20] For historic emissions estimates, see Work Group 3-B, Emissions Costs and Engineering Assessment, Final Report, June 15, 1982. Prepared Under U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, pp. 33-39. Older emission estimates are recognized to be less accurate than recent ones.

not with its overall change in magnitude. Similarly, the greater increase of nitric acid as opposed to sulfuric acid deposition in Florida correlates with a larger increase in NO_x than in SO_2 emissions in the state, but again does not match it quantitatively.

Various authors have used the correlations noted here to support arguments either about the relative importance of local versus transported pollutants or about the nature of the reactions controlling the transformations of emitted oxides to acids. However, different explanations often appear possible for the same observed effects, so that definitive conclusions about the relationship between emissions and deposition have not been agreed to on the basis of these correlations alone. This demonstrates that deposition in an area of only a few states cannot be explained solely on the basis of emissions within the area, which points to a need for study of transformations and transport, as well as trend data, to better understand how acid deposition occurs.

Estimates of early 20th Century acid deposition and its relation to the timing of damage

On the basis of emission estimates reconstructed from fuel use data, acid deposition appears to have been high in the northeastern United States since the early 20th century. This may explain the early appearance of damage there, and raises the possibility of damage appearing in other parts of the United States, where deposition increased more recently.

These recently published estimates[21] show total SO_2 emissions in the United States growing extremely rapidly from about 1895 to 1915 and then fluctuating around that level for most of this century. Through the first surge, emissions were highest in the Northeast and Ohio River Valley. By the late 1950's, northeastern emissions had declined somewhat, while midwestern emissions had increased. Then, by the mid-1970's, the highest emissions were centered in the Ohio River Valley and Midwest, while southeastern emissions had risen enough to match or exceed those of the Northeast, which had decreased again.

Thus, the Northeast is the section of the Nation which would have sustained the largest cumulative deposition, just from its own emissions, starting in the first decades of this century. In contrast, high deposition in the Southeast would not have begun until the last 2 or 3 decades, as was, in fact, noted in a previous section. (See p. 44.) This raises the possibility that damage from acid deposition, which began to appear in the Northeast

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[21] R.B. Husar and J.M. Hollaway, Sulfur and Nitrogen Over North America, Ecological Effects of Acid Deposition, background paper from 1982 Stockholm Conference on the Acidification of the Environment, National Swedish Environment Protection Board, Report SNV PM 1636, 1983, pp. 95-115.

only after one-third to one-half a century of heavy deposition, may begin to develop soon in sensitive areas in other regions, such as the Southeast, where deposition increased only more recently. Whether, and to what extent this may be true, however, is part of the uncertainty about acidification rates discussed in the previous chapter. (See pp. 15-16.)

TRANSFORMATIONS: HOW DO EMITTED SO_2 AND NO_x CONTRIBUTE TO ACID DEPOSITION?

Scientific studies of the ways in which emitted SO_2 and NO_x are transformed into deposited sulfuric and nitric acids have explained many parts of the process, but understanding is not yet complete. Therefore, while some of the information can be described quantitatively, other parts can only be described approximately or, as scientists term it, "semi-quantitatively." The behavior of the sulfur compounds is simpler and somewhat better understood, so it is discussed first, followed by a briefer treatment of the nitrogen compounds. The two key issues to consider are

--the relative amounts and effects of wet as opposed to dry deposition and

--whether the amounts of acidic and acidifying sulfur and nitrogen deposition are directly proportional to the amounts of SO_2 and NO_x emissions, or whether the amounts deposited are affected by the availability of other pollutants, such as oxidants.

Scientists working on acid deposition agree that sulfur and nitrogen oxides are not accumulating in the atmosphere. Rather, all emitted oxides are deposited after a short stay or "residence time" in the atmosphere, usually estimated as averaging several days. This section will focus mainly on the amounts deposited in the original forms or as transformed products, and the pathways by which the transformations occur. The complementary question of source-receptor connections--the geographic relationship between locations where particular emissions are released and where they or their products are deposited--will be covered in the following section on transport.

The physical pathways to deposition of SO_2 and its products

Emitted SO_2 can be found free in the air, dissolved in cloud water, or deposited back on earth, in each case either as SO_2 or transformed to sulfate. It can take any one of four possible pathways to final deposition. Of these four, two are dominant in eastern North America--sulfate in precipitation and dry deposition of SO_2 gas. The other two--SO_2 in precipitation and dry deposition of small sulfate particles--make much smaller contributions to total deposition.

Deposition in precipitation is only one of the possible fates of emitted SO_2, and indeed only a minority of all emitted SO_2 is deposited wet in eastern North America. The ratio of wet deposition to total man-made emissions has been computed from recent precipitation-monitoring data in two different studies. One gives an estimate of 25 to 30 percent,[22] and the second 26 percent.[23] This means that only a quarter or slightly more of all eastern North American sulfur emissions are deposited in precipitation on North America as sulfate.

The ratio of total wet deposition to total man-made emissions differs significantly between the eastern United States (the 31 states east of or touching the Mississippi River) and eastern Canada (Ontario and east). For the United States the ratio is lower, and for Canada it is higher, than the 25- to 30-percent estimate averaged over the eastern parts of both countries. This is because U.S. emissions are a number of times greater than those from Canada, which, combined with the mean wind directions, results in substantially more deposition in Canada due to emissions from United States sources than there is deposition in the United States from Canadian sources.[24] For the eastern United States alone, the ratio of total wet deposition to total man-made emissions was estimated in 1980 at about 15 percent.[25]

Very little emitted SO_2 is deposited wet as unoxidized SO_2--usually only a small fraction of the quantity of sulfate in precipitation. In fact, unless special precautions are taken, the SO_2 in precipitation is oxidized to sulfate before samples are analyzed, and the SO_2 is not even detected.[26] The total amount, however, averages only a small percentage of total emissions.

The balance of the SO_2 emitted in eastern North America, some 70 percent or more, was either deposited dry on land as SO_2 gas or sulfate particles or "vented," that is, carried away from the region to be deposited, in either its original or transformed state, mostly in the Atlantic Ocean.

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[22] Ibid.

[23] M. Oppenheimer, "The Relationship of Sulfur Emissions to Sulfate in Precipitation," Atmospheric Environment, Vol. 17, pp. 451-460 (1983).

[24] J.N. Galloway and D.M. Whelpdale "An Atmospheric Sulfur Budget for Eastern North America," Atmospheric Environment, Vol. 14, pp. 409-417 (1980).

[25] L. Newman, Important Considerations on the Incorporation of Sulfur and Nitrogen into Rain, in Conference on Acid Rain and the Atlantic Salmon, Portland Maine, Nov. 1980, pp. 13-15.

[26] J.M. Hales and M.T. Dana, "Regional-Scale Deposition of Sulfur Dioxide by Precipitation Scavenging," Atmospheric Environment, Vol. 13, pp. 1121-1132 (1979).

Direct measurement of the amounts of dry deposition and venting for eastern North America has not been possible because of insufficient data from monitoring dry deposition or from measurements of actual transport at regional boundaries. However, two approximate estimates have been made using simple modeling approaches. The first approach, based on measurements from the late 1970's, when only a fraction of current monitoring stations were in operation, indicated that for eastern North America as a whole, dry deposition was slightly larger, accounting for 32 percent of total SO_2, compared with 30 percent vented.[27] Again, this paper showed significant differences between the two countries, with total dry deposition substantially greater than wet for the United States, but with the relative amounts reversed for Canada. The second study, done more recently, estimated that on the average, only about 25 percent of all emissions are vented, leaving almost half--45 to 50 percent--deposited dry.[28] While the uncertainty in these two estimates is large (the first study indicates a factor of two uncertainty for its dry deposition value), they agree in finding dry deposition larger than venting, which lends more support to this view than either study would separately.

Finally, comparing the two forms of deposition, total dry deposition on eastern North America appears to be of about the same magnitude as, but probably somewhat larger than, wet deposition as a source of sulfur deposition. A main reason is that precipitation only occurs part of the time, while dry deposition occurs continuously. Studies which compare dry and wet deposition generally find that the ratio of dry to wet is greater in the United States than in Canada because there are more sources closer together in the United States, and dry deposition is always stronger near sources than farther away. Also, a recent study[29] has found that forested mountain regions in the eastern United States (which would include the Adirondacks and other parts of the Appalachian chain to the south, as well as New England where the study was done) receive annually several times more sulfate and nitrate deposition directly when immersed in clouds than they receive from

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[27] J.N. Galloway and D.M. Whelpdale, op. cit. This earlier study assigned a greater share, 38 percent, to wet deposition than has been found in more complete recent monitoring. However, one of the authors told us that they would now give a lower estimate for wet deposition, which they indicated in their paper that they suspected was true.

[28] R.B. Husar. Paper given at American Geophysical Union Meeting, Philadelphia, June 4, 1982.

[29] G.M. Lovett, W.A. Reiners and R.K. Olson, "Cloud Droplet Deposition in Subalpine Balsam Fir Forests: Hydrological and Chemical Inputs," Science, Vol. 218, pp. 1303-4 (Dec, 24, 1982).

falling precipitation.[30] Because precipitation collectors do not gather this cloud deposition, its contribution must also be counted as dry deposition.

Of the other two forms which are more usually counted as dry deposition, unaltered SO_2 gas predominates over sulfate particles by a large margin, probably greater than 10:1. There are two reasons for this fact. First, only a minority of all emitted SO_2 is transformed into sulfate particles while it is still in the atmosphere,[31] so more sulfur is available for dry deposition as SO_2 gas than as particulate sulfate. The Electric Power Research Institute's (EPRI's) Sulfate Regional Experiment (SURE)[32] found, using a regional model for the eastern United States, that on an annual average, only about 6 percent of emitted SO_2 is found as particulate sulfate over the SURE region--less than one-tenth of the SO_2 which is not deposited wet. The second reason is that particulate sulfate is found predominantly in very small particles, between 0.1 and 1 micron in size, which are deposited more slowly than SO_2 gas,[33] and therefore remain suspended longer and are more likely to be vented. The conclusion that SO_2 gas provides the major portion of dry deposition is also confirmed by measurements at Hubbard Brook,[34] which estimated that only about 5 percent of dry deposition came in the form of sulfate particles, compared with the great majority, about 95 percent, being deposited as SO_2 gas.

In addition to the particulate sulfate formed by atmospheric oxidation of SO_2, which is called "secondary" sulfate, there is also "primary" sulfate, which is sulfate already formed in combustion systems before it is released to the atmosphere. There are

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[30] This heavy deposition in cloud-covered zones on mountains is what is referred to in the statement ". . . by comparison to most of the forests in the East, the remote mountain peaks of the Appalachians are highly polluted environments," by Prof. A.H. Johnson, Univ. of Pennsylvania, in testimony before Senate Committee on Environment and Public Works, Feb. 7, 1984.

[31] H. Rodhe, "Budgets and Turnover Times of Atmospheric Sulfur Compounds," Atmospheric Environment, Vol. 12, pp. 671-680 (1978), and L. Newman, op. cit.

[32] Electric Power Research Institute, EPRI Sulfate Regional Experiment: Results and Implications, Summary Presentation EPRI EA-2165-SY-LD, Dec. 1981.

[33] International Symposium on Sulfur in the Atmosphere, Workshop 2, in Atmospheric Environment, Vol. 12, p. 14 (1978).

[34] J.S. Eaton, G.E. Likens and F.H. Bormann, The Input of Gaseous and Particulate Sulfur to a Forest Ecosystem, Tellus, Vol. 30, pp. 546-551 (1978).

reports that residential oil burning releases a greater proportion of fuel sulfur as primary sulfate than does coal burning, and it had been argued that this could make a major contribution to acid deposition. However, preliminary modeling of sulfate formation, reported by a U.S./Canada work group,[35] found that amounts of primary sulfate in the air over eastern North America did not even match those of secondary sulfate, except within a few tens of miles of the New York City area, and even there only in winter--the season when total sulfate concentrations are generally lower. This indicates that the primary sulfate contribution to acid deposition could not be an important source of regional acid deposition.

Transformations of SO_2 to sulfate

Having identified the two main paths followed by sulfur compounds being deposited, we now must examine how the transformation (chemically known as an oxidation) of SO_2 to sulfate occurs. Although a number of points about this chemistry are not fully understood, it involves more than one process. This is shown by the fact that the rate of SO_2 oxidation can vary 1,000-fold, from about 0.1 percent per hour to 100 percent per hour, depending on humidity, solar radiation intensity, the presence of clouds, and other pollutants. However, enough is known to allow a reasonable determination of what controls the extent of oxidation, that is, whether sulfate formation is in 1:1 proportion to the amount of SO_2 available or whether any other components have a role in controlling sulfate formation in a way which could make it less than proportional to SO_2 concentrations.[36]

Dry deposited SO_2 can only be oxidized to sulfate after deposition. However,

> "Sulfur dioxide is converted to sulfate soon after it arrives at all moist plant, soil or water surfaces. It also is converted to sulfate on biologically inert surfaces soon after they are wetted. Thus, atmospheric sulfur creates acidity whenever it is oxidized (either in the atmosphere or after deposition)."[37]

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[35] U.S.-Canada Work Group 2, Atmospheric Sciences and Analysis, Working Report No. 2-15, prepared under Memorandum of Intent on Transboundary Air Pollution, July 10, 1981, ch. 10.

[36] The proportionality of deposition to emissions is sometimes referred to as "linearity." However, this term has another meaning in chemical usage which can lead to some confusion, so we will avoid it and refer only to proportionality.

[37] E.B. Cowling, "Biological Perspectives Regarding Acid Deposition and Its Effects," Memo, June 18, 1982.

This view was confirmed by unanimous agreement of a diverse panel of acid rain scientists:

> ". . . in the biologist's view the geographical distribution of total sulfur and nitrogen oxide deposition is really most important. The chemical form produced in the atmosphere is of relatively little consequence compared with the biological assimilation taking place in the ground and in the surface waters."[38]

Thus, for dry deposited SO_2, the transformation step does not limit sulfate production or the effects of sulfate on ecosystems, so we can turn to examine the controlling factor of the deposition step.

The extent of dry deposition is directly proportional to the concentration of the depositing substance in the air.[39] While some variation occurs due to mixing turbulence in the air, and also due to the nature of the surface onto which deposition is occurring (faster on wet or rough surfaces than on dry or smooth surfaces, and faster on tall vegetation than on short vegetation), in all cases the proportionality to the concentration of the depositing substance remains. In the first section of this chapter (p. 33), we noted that essentially all SO_2 in the eastern North American atmosphere comes from man-made emissions, which means that SO_2 concentrations at any location in the region will be controlled by the amount of man-made emissions which are transported to that location. Therefore, at least with regard to biological effects, a major share of acidity, the part delivered by SO_2 dry deposition, will be proportional to SO_2 emissions as they are distributed by transport.

Sulfate in precipitation is the second of two major forms of deposited sulfur and is most familiar because it is the form measured by monitoring stations. Extensive debate has taken place regarding whether its deposition is proportional to SO_2 emissions. In particular, the possibility has been raised that sulfate levels in precipitation are controlled by the supply of strong oxidizing substances in the atmosphere, rather than by SO_2 concentrations, so that wet sulfate deposition would not be directly responsive to changes in SO_2 emissions. Strong oxidizing substances in polluted atmospheres are mostly products of reactions that start from

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[38] G.M. Hidy, speaking for himself, R.W. Brocksen, R. Husar, J.S. Jacobson, G. Likens, O. Loucks and V. Mohnen, in testimony before the Subcommittee on Natural Resources, Agriculture Research and Environment, House Committee on Science and Technology, Nov. 19, 1981.

[39] D. Fowler, "Removal of Sulfur and Nitrogen Compounds from the Atmosphere in Rain and by Dry Deposition," in Ecological Impact of Acid Precipitation, D. Drablos and A. Tollan, Eds., Conference Proceedings, Sandefjord, Norway (1980), pp. 22-32.

hydrocarbon and nitrogen oxide (NO_x) pollutants. Thus, if the supply of oxidants does control the rate of precipitation sulfate production, then the proper emission control approach to lower sulfate deposition in precipitation would be to reduce hydrocarbon and/or NO_x emissions. In the event of such "oxidant control," lowering SO_2 emissions would not be effective in reducing wet sulfate deposition, or at least not until the relative amounts of other pollutants were changed enough to leave SO_2 supply as the controlling factor in precipitation sulfate formation.

Recent scientific work has led to some agreement that precipitation sulfate is formed mostly by the oxidation of SO_2 in cloud water, rather than from the uptake of particulate sulfate formed in the air. Furthermore, scientific work published in 1983 has shown growing agreement that a close proportionality exists between wet sulfate deposition and SO_2 emissions. Two different theoretical approaches have been used to examine this question.

One approach is a statistical analysis which estimated how much SO_2 was exposed to clouds and, from that, how efficiently the clouds oxidized it.[40] This study found that the oxidation process is relatively efficient and, on a regional basis, not substantially limited by oxidant availability but rather by the amount of SO_2 which was exposed to clouds. It estimated that, for eastern North America, a minimum of 87 percent of all SO_2 exposed to clouds was oxidized and precipitated. Thus, wet sulfate deposition, according to this study, was very close to proportional to SO_2 concentrations and therefore would respond quite readily to changes of SO_2 emissions.

The other approach was taken in two studies of a chemical model of the atmospheric reactions involving the three pollutants: SO_2, NO_x, and reactive hydrocarbons (RHC) and their products which interact in atmospheric transformation reactions. The first study used a model involving 19 chemical reactions, but recognized that, while this model was quite complex, it still did not treat all possible reactions and conditions involved in the atmospheric processes of interest.[41]

Use of this simplified chemical model in the first study led to a finding that precipitation sulfate formation was substantially influenced by the availability of oxidants. However, reexamination of the question, by a committee chosen by the National

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[40] M. Oppenheimer, op. cit. (footnote 23).

[41] P.J. Samson, On the Linearity of Sulfur Dioxide to Sulfate Conversion in Regional-Scale Models, Office of Technology Assessment, Contractor Report, June 1982.

Academy of Sciences (NAS),[42] indicated that this finding occurred because a main chemical reaction used in the first study had assumed that sulfate formation resulted in the consumption of oxidant, without the oxidant being replaced. Actual experiments in controlled conditions showed that oxidant was not consumed by oxidation of SO_2 to sulfate, showing that the assumption in the earlier study was not valid.

The results of the NAS committee's examination of the corrected model showed that sulfate formation should not be significantly influenced by oxidant availability. This was one of the main reasons for the committee's conclusion that precipitation sulfate concentrations are closely proportional to SO_2 emissions, and that changes in regional SO_2 emissions should therefore result in equal proportional changes in regional precipitation sulfate deposition.

The chemical model approach and the statistical analysis approach both indicate that wet sulfate deposition would be altered by changes in SO_2 emissions, and they essentially agree that, on a regional basis, sulfate deposition in precipitation should change in proportion to changes in SO_2 emissions. Both of the approaches reached this conclusion, however, on the basis of the usual "all other things being equal" assumption. Therefore, while the conclusion ought to be valid for a reduction of SO_2 emissions made without major changes in other atmospheric pollutants, the question might require reexamination if there were also significant reductions of emissions of the other pollutants--particularly reactive hydrocarbons--which are involved in oxidant production.

Transformations of NO_x to nitric acid

Nitric acid is a lesser contributor to acid deposition than sulfuric acid in both eastern North America and western Europe, but its contribution to atmospheric acidity in both regions is growing, and reports have been made of high concentrations of nitrate--sometimes representing a majority of the acid--in acid deposition in some locations in the western United States.

Emitted NO_x is extensively involved in complex atmospheric chemical reactions which also include reactive hydrocarbon pollutants, and these give rise to ozone and a number of other oxidants in addition to nitric acid itself. These processes make an accounting of the pathways and fates of emitted NO_x a more complex matter than is the case for SO_2.

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[42] Committee on Atmospheric Transport and Chemical Transformation in Acid Precipitation, Acid Deposition: Atmospheric Processes in Eastern North America, National Research Council, June 1983, pp. 75-84.

According to a recent estimate based on data from precipitation monitoring networks, about 20 percent of the nitrogen oxides emitted in eastern North America can be accounted for as nitrate in precipitation.[43] Precipitation nitrate is formed by transformation of NO_x to nitric acid, largely in the air rather than in cloud water,[44] and it forms more quickly after emission than sulfate does, but its proportionality to emissions is not as well investigated experimentally or theoretically as far as we could determine. However, the fact (noted on p. 44) that wet nitrate deposition has increased in recent years, a period when increases of NO_x emissions were occurring, is a demonstration of at least partial proportionality between NO_x emissions and nitrate deposition, though not necessarily exactly 1:1.

Scientific studies give evidence that (as is the case with the sulfur oxides) wet deposition of unoxidized NO_x and dry deposition of nitrate in particles are negligible in comparison to precipitation nitrate and other dry deposited forms. However, dry deposition, in the nitrogen case, will include nitric acid vapor in addition to unoxidized NO_x gas, and possibly other nitrogen-containing products from the complicated set of NO_x-RHC reactions.

The relative amounts of these dry deposited forms and the overall extent of dry deposition as opposed to venting off the continent have not been directly measured to our knowledge. However, comparisons between geographical patterns of wet nitrate and sulfate deposition have led to some agreement that NO_x and its oxidation products do not travel as far as the sulfur compounds. This would suggest that, in comparison to SO_2, a smaller percentage of NO_x is vented and even more is dry deposited, in whatever combination of forms it takes.

Scientists indicate less certainty that dry deposited NO_x is all oxidized to nitrate than with the sulfur compounds. In addition, we have not seen evidence as to whether the formation of other dry deposited forms is controlled by other factors or is proportional to NO_x emission levels. As a result, the overall relation between NO_x emissions and total nitrates delivered by deposition appears somewhat less clear, at least on the theoretical level at this time, than is the case for SO_2 and sulfate. On the other hand, there is the actual observation (which is even more convincing than theory) that time trends in the last few decades clearly show some proportionality between wet nitrate deposition and NO_x emissions.

Summary of transformation findings

--Emitted SO_2 and NO_x may be transformed to the more acidic forms, sulfate and nitrate, either before or after they are

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[43] R.B. Husar and J.M. Holloway, op. cit. (footnote 21).

[44] L. Newman, op. cit. (footnote 25).

deposited, and can be deposited either "wet" (in precipitation) or "dry" (as gases, particles, or fog or cloud droplets).

--Overall, dry sulfur deposition in the eastern United States is greater than wet deposition, while the opposite is true for eastern Canada. At particular sites in the United States, far from major sources, wet deposition can exceed the dry contribution.

--Dry deposition of sulfur occurs almost entirely (more than 90 percent) as untransformed SO_2 gas, and will show 1:1 proportionality to SO_2 emissions. (There is one exception to this--in the zone on mountains which is often immersed in clouds. Sulfate and nitrate deposition by cloud droplets onto vegetation may be several times greater than from precipitation, and would be expected to respond to emission changes in the same way as does wet deposition.)

--Transformation of dry deposited SO_2 to sulfate, after deposition on biological materials, is essentially complete and fast enough so that the majority of acidic sulfur deposition on living ecosystems will be in 1:1 proportion to emissions, and will change in direct proportion to changes of SO_2 emissions.

--Wet deposition of sulfur in precipitation occurs almost all as sulfate. While this is the most commonly recognized form, it represents only a large minority of total acidic sulfur deposition in the eastern United States, compared with the larger dry deposition.

--Of the transformations that produce wet deposited sulfate, the majority occur in cloud water, with a smaller share also taking place in the air. The amount of wet sulfur deposition will be essentially directly proportional to SO_2 emissions--a conclusion reached on the basis of work just published in 1983. This means that total sulfate deposition regionally should decline in proportion to SO_2 emission reductions, but does not necessarily apply to smaller areas.

--Nitrogen emitted as NO_x is deposited wet as nitrate, and dry as nitric acid vapor, untransformed NO_2, and possibly as other nitrogen-containing compounds.

--Wet nitrate deposition has increased in recent years as NO_x emissions have increased, so it should change in response to further changes in NO_x emissions, though not necessarily in exact 1:1 proportions.

--Oxidation of NO_x to nitric acid in the air is faster than the oxidation of SO_2 to sulfate, but relatively little particulate nitrate is found in the atmosphere. This fits with the view that more NO_x is deposited and less vented off the continent than SO_2, but the proportionality of dry nitrogen deposition to NO_x emissions appears less clearly established than for the sulfur compounds.

TRANSPORT: WHICH WAY, HOW FAR, AND HOW MUCH?

The third major question to examine in the acid deposition process is transport, the carrying in the atmosphere of acidic and acidifying materials from their emission points to the locations where they or their products are deposited. This issue is usually phrased in terms of source-receptor relations, and models being developed to describe the process often have their results presented in a "transfer matrix" showing the share of deposition in each receptor area contributed by each source area. In the following pages, we will describe what experimental information and simple models indicate about directions, distances, and amounts of transport, and then examine what the more complex present models can tell and are telling about the overall process from emission to deposition.

Direction of transport

Deposition monitoring alone cannot determine where deposited materials originate, but only the amounts of acid deposited, and its time distribution. However, a technique called "back-trajectory analysis," which uses recorded wind data to determine the previous path of air parcels from which individual precipitation events came, has been used for over a decade to establish the direction from which deposited material arrived.

Back-trajectory analysis has two kinds of limitations as a technique for establishing transport directions: one based on the sparseness of available meteorological data and the second arising from the complexities of atmospheric behavior in some kinds of precipitation events.

In order to trace back the path of an air parcel, it is necessary to have detailed information on the winds that transported it, including possible vertical movements, and the variations in wind speeds and directions at different altitudes and times. However, except for winds at ground level, the routinely available wind data for North America is limited to measurements made at 12-hour intervals, at a limited set of altitudes, from a network of stations spaced hundreds of miles apart. This means that the winds that acted on an air parcel must be approximated by interpolating between the available measurements. As a result, a calculated back-trajectory is only approximate, and becomes increasingly uncertain as the time and distance of travel increases. While there is reason to believe that the uncertainties will tend to balance out when averages are taken over back-trajectories for a number of events, this is still a limitation on the accuracy of the technique for any single event.

A second, and probably more serious limitation arises from the problem of accounting for vertical movements of air in general, and particularly in certain types of precipitating systems. Sophisticated methods have been developed which can treat part of the vertical movement question, but even these are not adequate to deal with the complex air movements that occur at weather fronts. These are cases where different air masses collide and move over and under each other, and where some air parcels can move long distances, up to the range of hundreds of miles, along the front--an occurrence called "ducting." Because air movements near fronts change sharply over even very small space and time differences, the approach of interpolating between the wind data from widely spaced observing stations clearly is not able to give accurate back-trajectories--instances can be shown in which the end points of two different plausible trajectories from the same point in the vicinity of a frontal storm are separated by several hundred miles in 24 hours. While these frontal storm systems do not produce the most acidic precipitation, which is usually seen in local thundershowers, they are responsible for a substantial share of the precipitation in eastern North America. Therefore, back-trajectory analysis, at least at this time, will still have somewhat limited accuracy. However, it still can give useful indications of the direction of pollutant movements.

An example, from joint international studies in Europe, examined wet deposition at two sites in southwestern Norway. It showed that, of total acid deposition, 55 percent came from the direction of the United Kingdom; 24 percent from France, Holland, and the Ruhr in West Germany; and 8 percent from East Germany and Poland. The remaining 13 percent arrived from other directions, and showed acidity of only one-half to one-quarter of that from the three above sectors.[45]

A similar study at Whiteface Mountain, in the Adirondacks, showed the highest sulfate, nitrate, and acid concentrations, and the highest total contributions of these substances, in samples arriving from the directions of the Appalachians, Midwest, and Ohio River Valley.[46] Somewhat lower levels came from the upper Great Lakes and the Canadian area north of them, while the lowest amounts and concentrations came from the entire eastern half of the compass, which included the New York City area, all of New

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[45] OECD Report on Long Range Transport of Air Pollutants, Organization for Economic Cooperation and Development, quoted in Effects of SO_2 and its Derivatives on Health and Ecology, report of a working group sponsored by the International Electric Research Exchange, Nov. 1981, Vol. 2, p. 2.20.

[46] J. Wilson, V. Mohnen and J. Kadlecek, Wet Deposition in the Northeastern United States, Atmospheric Science Research Center, State University of New York at Albany, Dec. 1980. pp. 45-51.

England, and Canadian areas east of Montreal. In contrast, back-trajectory analysis on precipitation at Champaign, Illinois, in the same study,[47] did not show differences in concentrations of these pollutants in precipitation from different compass sectors. However, the majority of precipitation arrived with air masses from the southwest direction leading to greater overall contributions from that direction, which includes major sources of emissions in the St. Louis area.

These results and others show that the direction of air movements plays a major role in determining acid deposition at a site. Winds in the eastern United States are predominantly in the eastward and northeastward directions, which is a major reason for assertions that Ohio River Valley and midwestern emissions are the sources of the heavy deposition in the northeastern United States and southeastern Canada.

Back-trajectory analyses done on simultaneously collected samples at a set of nearby sites have also allowed comparison of local and more distant contributions to acid deposition. In a study in Nova Scotia,[48] wet deposition of sulfate and acid were compared in individual precipitation events at two sites about 15 and 90 miles southwest of, and so supposedly upwind from, the city of Halifax. It was found that wind variability during storms transported local emissions in all directions to account for about half of total acid deposition at sites near the city, which is a moderate emitter of SO_2. In contrast, at the site about 90 miles away upwind, the effect of the city appeared negligible. With the contribution of Halifax separated out, back-trajectory analysis traced the balance of the wet deposition to areas on the mainland: about three-fifths from the United State below northern New England and the Great Lakes, one-third from the St. Lawrence Valley and lower Great Lakes, and less then a tenth from the Canadian region farther to the north and east.

Evidence on transport distances

While back-trajectory studies at receptors are capable of explaining the directions from sources to receptors, they usually lack information on the distances over which transport occurs. An air parcel arriving at the Adirondacks from the southwest direction, for example, might have passed sequentially over regions with substantial emissions in Missouri, Illinois, Kentucky, Indiana, Ohio, Pennsylvania, and northwestern New York. All of these regions would have contributed SO_2 to the air, and the relative proportions could not be determined from the trajectory alone.

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[47] Ibid. pp. 128-131.

[48] R.W. Shaw, "Deposition of Atmospheric Acid from Local and Distant Sources at a Rural Site in Nova Scotia," Atmospheric Environment, Vol. 16, pp. 337-348 (1982).

Examples discussed in the following pages help to indicate the range of transport of substances that contribute to acid deposition. Some are based directly on observations, while others are based on simple models.

Observed transport distances

One kind of direct evidence comes from studies in which emission plumes from individual sources, or entire urban plumes, have been traced for distances of 200 to several hundred kilometers.[49] Another indication of the range of transport, specifically in North America, comes from comparing maps of deposition and emission patterns.

The deposition map in figure 1a (page 11) shows the annual average pH's of wet deposition at a number of monitoring stations in North America for the year 1980. The curves on the map connect locations estimated to have the same pH values, as marked on the curves, while closed curves surround areas in which precipitation was at or below the marked pH. The map shows a large area at pH 4.5 and below, covering much of eastern North America, and two areas within it that are at and below pH 4.2.

Maps of deposition patterns shown in figure 1 (p. 11) for other components of precipitation beside the acid (H^+) ion show that sulfate and nitrate (figs. 1c and 1d) are the predominant ions that accompany H^+, in generally similar geographic patterns. Since sulfate is the major acidifying form deposited, an approximate comparison can be made of figure 1c to the map in figure 2 (p. 41) roughly showing the geographic distribution of eastern North American SO_2 emissions. This comparison is shown in the map in figure 3, in which the curves of sulfate deposition from figure 1c have been overlaid on the SO_2 emission source map of figure 2.

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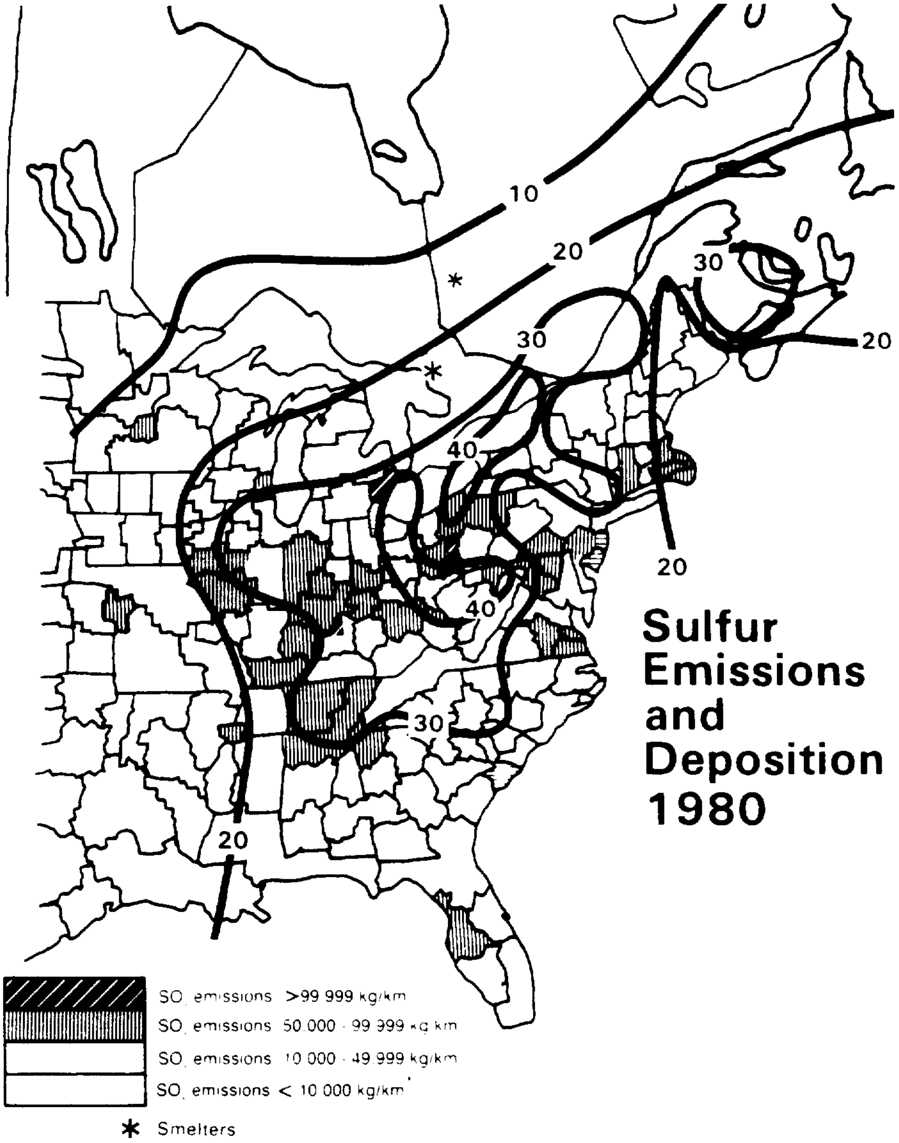
[49] International Symposium on Sulfur in the Atmosphere, Atmospheric Environment, Vol. 12, pp. 22-23, (1978).

Figure 3



Comparison of Annual SO_2 Emissions and Annual Wet Sulfate Deposition in Eastern North America for 1980

Shading shows emission densities for U.S. Air Quality Control Regions as explained in key, and asterisks show two large eastern Canadian smelters. Curves join points of approximately equal total wet sulfate deposition.

Source: GAO overlay of maps from figure 1, panel (b) on page 11 and figure 2 on page 38 of this report.

The comparison shows a general correlation of areas of high emissions and high wet deposition, but with the boundaries somewhat displaced from exact overlap. In particular, the high-emission area does not reach northeastern Pennsylvania or upper New York State, yet wet deposition of 30 units and above does occur in those areas and even extends to parts of the St. Lawrence River Valley and the northern boundaries of New York, Vermont, and New Hampshire. This amounts to a displacement of high wet deposition up to about 500 miles east-northeast from maximum emission densities, and even shows the highest wet deposition, 40 units and above, extending over parts of New York State and Lake Ontario, which contribute much lower or even zero emissions. A similar displacement is also noticeable for the southern edge of the 30-unit deposition zone, which does not extend significantly west of high emission areas, but extends on the east over a sizeable part of North Carolina, up to 300 miles or more east and northeast of the high emissions areas.

These displacements show that the main effects of SO_2 emissions on wet sulfate deposition are, on the average, displaced from the locations where the emissions are released to areas up to 300 to 500 miles in the direction of prevailing air movements.

Another kind of evidence has been widely cited recently as indicating that acid deposition is not transported beyond distances of about 200 miles, based on studies of tracer elements in aerosol. However, the author of this work introduced it with a set of caveats including:

> "(5) It deals with aerosol, not precipitation . . . and
>
> (7) It is now recognized that acidic aerosol is not the major source of acid precipitation."[50]

Also, in a response to a critical letter about a news report on his research, the author denied assuming that the tracer elements had the same sources as acid precipitation:

> ". . . having no results on acid precipitation per se, I made no such assumption. The link between tracer elements in aerosol and precipitation has not yet been determined for the Northeast."[51]

Thus, while this technique may be useful in future studies of acid deposition, its results do not bear on the question of the transport range for deposition at this time.

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[50] K.A. Rahn, New Evidence on Source Areas of Pollution Aerosol in the Northeast, testimony before Senate Committee on Environment and Public Works, May 25, 1982.

[51] K.A. Rahn, Letter, Science, Vol. 217, p. 688, Aug. 20, 1982.

In a few instances, back-trajectories over water allow estimation of transport distances without the confusing factor of sequential source areas. In the European studies, back-trajectories from southern Norway showed that the majority of wet deposition came from Great Britain, over distances of at least about 500 miles. Also, the study in Nova Scotia discussed above found the majority of nonlocal deposition came from an area of the United States which is least 400 miles from the deposition site. Furthermore, a recent report from Bermuda[52] traced most of the observed rain acidity to air masses coming from the North American continent, a distance of at least 600 miles, but noted that acidity was reduced by a factor of three during the transit.

Simple modeling of transport distances

In another approach, a transport model following a single air parcel was used to estimate contributions of sulfur pollutants from a series of regions from Florida and the Gulf Coast up to Adirondack Park in New York.[53] This model used actual emission and air movement data but had to assume average values for other parameters describing mixing, transformation, and deposition.

The results showed that the contribution to the total sulfur in the air parcel from each region diminished during transport, because material deposited out as the parcel moved after passing the region. In the case studied, it took on the order of 700 to 1,000 miles for any contribution to drop to half of its original maximum.[54] Therefore, when the air parcel arrived at Adirondack Park, 24 percent of its total sulfur was composed of contributions from western New York, 44 percent was from midwestern (mostly Ohio River Valley) contributions, 10 percent from TVA, and 22 percent from a combination of other southeastern sources and minor natural contributions. While cases with other trajectories and wind speeds would differ, and result in smaller average distances, this example suggests the maximum distances and proportions of transport that are possible for a single trajectory, given average values for the parameters of pollutant behavior in the atmosphere.

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[52] T. Jickells, et al., "Acid Rain on Bermuda," Nature, Vol. 297, pp. 55-57 (May 6, 1982).

[53] L.M. Reisinger and T.L. Crawford, "Interregional Transport: Case Studies of Measurement versus Model Predictions," Journal of the Air Pollution Control Association, Vol. 32, pp. 629-633 (1982).

[54] If wet deposition had occurred to remove some sulfur during transport, some of the distances would have been shortened.

Another simple model approach was used in an earlier study of the direction and range of movement of air parcels in Europe.[55] Actual trajectories for 24-hour and 60-hour periods were followed over the course of a year from a site in West Germany. The results showed that the trajectories scattered widely, with the average end point displaced east and somewhat north of the starting point, by about 200 miles in 24 hours and over 400 miles in 60 hours, with many parcels moving much farther. Allowing for the deposition that would occur during these travel times, these results imply that emissions from any source are distributed widely, dropping off with distance but still noticeable as far as 200 to 300 miles in all directions, and out to distances of 500 to 700 miles along the direction of prevailing air movements.

Summary of transport distances

These observations and simple model results show an approximate bracketing of the effective transport range of wet deposited sulfur emissions. Following along prevailing wind directions, major contributions can be deposited at distances out to the range of 200 to 500 miles, as shown by the displacement distances in North America and the cases of transport to Norway and Nova Scotia. By the time distances of 500 to 700 miles are reached, contributions to wet deposition begin to drop off substantially, as shown by the Bermuda measurements and from the European trajectory modeling.

With regard to dry deposition, scientists recognize that the average range of transport for dry deposited sulfur compounds is less than that for wet deposition, meaning that dry deposition is relatively more important near sources, while wet deposition becomes more significant at longer distances. A review of deposition mechanisms[56] summarized the relationship as showing that the ratio of dry to wet deposition in areas with high pollutant concentrations close to sources could go as high as 8:1 to 12:1, while it should drop down to, or even below, 1:1 beyond about 200 miles from source areas. However, this review pointed out that dry deposition still makes large contributions to total acid deposition even at long distances from sources.

Actual observations are very limited, but available results are consistent with this view. In one watershed studied in an area near major sources in England,[57] dry deposition accounted

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[55] H. Rodhe, "A Study of the Sulfur Budget for the Atmosphere Over Northern Europe," *Tellus*, Vol. 24, pp. 128-138 (1972).

[56] D. Fowler, op. cit. (footnote 39).

[57] J.E. Rippon, *Studies of Acid Rain on Soils and Catchments* in *Effects of Acid Precipitation on Terrestrial Ecosystems*, T.C. Hutchinson and M. Havas, Eds. (Toronto Conference, May 1978), Plenum Press, pp. 499-524.

for two-thirds of the sulfate entering the watershed. In comparison, estimation of dry deposition at Hubbard Brook, New Hampshire,[58] which was somewhat further from major sources, found the opposite relation--specifically, a little less than one-third of the sulfate came from dry deposition.

These findings indicate that changes in emissions of SO_2 would have their greatest effects on total sulfur deposition in the first few hundred miles from the sources, but significant effects out to 500 to 700 miles downwind, with the changes beyond the first few hundred miles coming as much or more in wet as in dry deposition. Deposition of nitrogen compounds apparently occurs at shorter range than for sulfur compounds, so effects on their deposition from changes in NO_x emissions would likely also be of somewhat shorter range. However, the relative ranges for wet versus dry deposition of nitrogen compounds are not as well understood because of the greater complexity of the NO_x transformations.

Long-range transport modeling

The separate examination of transport and transformation processes allows a general understanding of the factors that control the relations between emissions and deposition. However, a full quantitative description requires combining all the relevant factors and their variation over time and space: emission locations, heights and timing; the rates and controlling factors for chemical transformations; atmospheric mixing in both horizontal and vertical directions; precipitation timing, intensity, and location; and the deposition velocities of individual pollutant forms--that is, the rates at which they appear to settle to the ground. The approach that is taken to do this is the use of large computer models of long-range transport.

Joint European studies in the mid-1970's modeled sulfur pollutant transport over western and central Europe. In the 1980's, at least eight different models have been developed and tested for eastern North America, as reported in coordination with the joint U.S.-Canada Work Group studies on long-range transport of air pollutants.

The results of the European modeling were presented as annual total sulfur budgets for each country, showing the shares contributed by each other country,[59] with an accuracy estimated at ± 50 percent. The results showed that, in a number of low-emitting countries, more sulfur deposition was imported than came from

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[58] J.S. Eaton, G.E. Likens and F.H. Bormann, op. cit. (footnote 34).

[59] B. Ottar, "An Assessment of the OECD Study on Long Range Transport of Air Pollutants," Atmospheric Environment, Vol. 12, pp. 445-454 (1978).

domestic sources. In all cases except Norway (for which Great Britain was the largest single source), however, each country was the largest contributor to its own sulfur deposition. The implication drawn is that, generally, acid deposition was shown to be both a local and a long-distance transport problem.

In the current North American modeling, a variety of estimates and approximations is being used in different models to treat the individual factors that determine pollutant behavior. Some of these approximations are necessary because exact data are not available--for example, on deposition velocities for individual pollutant forms--and others are used because available data are incomplete--upper air winds, for example, as noted on p. 57, are only known for a few heights and only at widely spaced geographic and time intervals. In addition, all the models treat the sulfate formation process as proportional to SO_2 concentrations, so the possibility of control by oxidant supplies is not considered (except indirectly, in some cases where rates are varied with time of day or season). Finally, the models have not yet been used to treat nitrogen compounds but rather are estimating concentrations and amounts of deposition of sulfur compounds. This is being done on a monthly or annually averaged basis, over large areas, because many of the necessary input data are only available or predictable as averages over long times (e.g., annual emissions or average precipitation) or at large distance intervals (e.g., winds from stations spaced 200 to 250 miles apart).

In the most recent report which we analyzed in detail,[60] results from seven models showed wide differences in many cases, occasionally as large as a factor of 10, in their absolute estimates of annual sulfur transport between particular regions. On this basis, it would be premature to use these results to indicate source-receptor connections. However, the model results are more consistent on a relative basis on a number of points. Considering the wide range of values and approaches used on a number of significant variables in the models, the results on which they are more consistent are quite likely to hold as valid in a semiquantitative fashion in many respects. (This does not necessarily apply, however, to a question such as the proportionality of oxidation to SO_2 concentration, since the models essentially all used the same approach to the proportionality question.) Two types of issues on

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[60] Modeling Subgroup Report No. 2-13, U.S.-Canada Atmospheric Sciences and Analysis Work Group 2, Phase II, Under Memorandum of Intent on Transboundary Air Pollution, July 10, 1981. Modeling work in the next (final) phase of the Work Group's activity was done at a finer level of geographic detail. However, it did not give much greater accuracy in results, which would be expected since the models were not made much more realistic in their chemistry or meteorology in the final, as compared with these Phase II, runs. The Phase II results are therefore described here, since they allow a simpler and more easily understood description.

which these models showed consistency in their results bear particularly on transport distances: one is the range of transport, and the other the relative ranges of dry and wet deposition.

The Phase II analysis for the U.S.-Canada Work Group divided the modeled area into eight regions covering the eastern United States and three for eastern Canada, then calculated the contributions in wet and dry deposition from each region to nine receptors spread around the same area.

In all instances, transport was calculated to be much greater toward the east and northeast than in all other directions, showing the influence of prevailing winds. As an example, the Pennsylvania region, on average, was calculated to provide about 13 percent of deposition in the Adirondacks, to the northeast of Pennsylvania, whereas all of New York and New England, despite larger total emissions, could only provide an average of about 3 percent of deposition at a receptor in western Pennsylvania, which lay to the southwest.

The effect of local sources and those at intermediate distances ("mesoscale" is the term used in technical studies, meaning out to about 180 miles) as opposed to distant sources was calculated to be significant, but only rarely dominant. When averaged over all models, only one receptor (on the Minnesota-Ontario border) showed more than half of its deposition coming from one region, the one south and west of it which included Minnesota. This receptor had by far the lowest calculated deposition of any studied, because of a lack of any major sources in range to its west, which can also explain its domination by the more local sources.

The amount of transport in the downwind direction did show a clear decline with distance. As an example, averaging all models, a receptor in New Hampshire was estimated to receive 60 to 70 percent of its deposited sulfur from regions no farther away than Pennsylvania and Ontario, while the major source areas south of Pennsylvania and extending from Ohio and Michigan through the Midwest, with substantially greater total emissions, contributed only 30 to 40 percent. Another example, for the Adirondacks receptor, showed sources in Ohio and eastern Indiana, at about 400 to 800 mile distances, calculated to contribute about 14 percent, or twice as much sulfur as the 7 percent transported from sources of the same total strength in western Indiana and Illinois, at about 800 to 1,000 mile distances.

Finally, on the relative range of wet, compared with dry deposition, the models generally showed the dry contribution to be larger than the wet contribution from the region within which a receptor was located. The reverse was true, with wet contributions being greater than dry ones from distant regions, while some nearby regions also made larger dry than wet contributions. However, this changeover was not estimated to be very steep. Averages for all models showed only a few cases where a distant region's calculated contribution to dry deposition was as low as 60 percent of its wet contribution, and most remained no lower than about 75 percent. Thus, the models indicate that, on average, dry deposition stemming from any source region remains close to equal to its contributions to wet deposition, as both diminish to low levels at long travel distances.

THE CAUSES OF ACID DEPOSITION--A SUMMARY

Our review of scientific literature and judgments on the specific subquestions on how acid deposition occurs indicates that there is substantial, though not complete, scientific understanding of this process; much of it is the result of scientific work in the last several years.

--Geographic patterns of sulfuric and nitric acid deposition show that man-made emissions of sulfur and nitrogen oxides are the predominant sources of acid deposition in eastern North America. Evidence of time trends is consistent with this, showing that the intensity of current acid sulfur deposition did not occur before the Industrial Revolution. High deposition has occurred in the northeastern United States for much of this century, but has spread in recent decades.

--Dry deposition of sulfur oxides, predominantly as SO_2, is a substantial source of sulfate delivered to ecosystems in eastern North America and is probably, by a small margin, the majority source of sulfur deposition for the eastern United States. This part of acid deposition is proportional to the level of SO_2 emissions and would be reduced in direct proportion to reductions in emissions.

--Research first published in 1983 indicates that sulfur deposited in precipitation ("wet deposition"), mostly as sulfuric acid, will also be essentially proportional to SO_2 emissions on an overall regional basis for eastern North America. Therefore, a uniform percentage reduction of emissions would result in about the same percentage of deposition reduction throughout the region. Furthermore, a reduction of emissions anywhere in the region would result in comparable reductions of deposition, although the precise locations where changes in deposition would occur are not well known.

--Emissions from any source will be spread somewhat in all directions, but predominantly to the east and northeast, and will contribute to deposition over a wide area. Deposition at any location will be made up of contributions from many sources, so no single source can be held predominately responsible, except possibly within short distances (tens of miles) from major sources.

--Deposition from a source or area will diminish with distance, but can remain important at distances up to 500 miles in the direction of prevailing air movements. At distances of the order of 500 to 800 miles, contributions from any source will be several-fold less than those from a similar source within the first few hundred miles of a receptor.

--Dry deposition caused by any source will, on the average, be greater than its contribution to wet deposition near the source, and then become equal or less beyond about 200 miles. Actual data on this are sparse, as are data on the relative contributions of wet and dry deposition at greater distances, and on the overall extent of dry deposition in most areas. Recent work indicates extremely high deposition from clouds onto forested mountain areas they rest upon, which must be counted as dry deposition since it is not gathered in precipitation collectors.

--The processes leading to nitrate deposition from NO_x have received less attention and appear less well understood than those involving the sulfur compounds. This includes dry deposition, which may involve a significant share of nitric acid as well as NO_2, and possibly other nitrogen-containing compounds.

--Accurate quantitative estimates of the shares of deposition at any receptor area, stemming from emissions from specific source regions, are not yet available. Further work will be required to improve and validate long-range transport models for making these estimates. However, the findings on transport distances already cited indicate that these shares will be significant over distances of up to hundreds of miles along prevailing wind directions from the sources.

CHAPTER 4

IMPLICATIONS OF ALTERNATIVE ACID DEPOSITION CONTROL STRATEGIES

The need for additional controls to alleviate acid deposition, beyond those already in place under the Clean Air Act, has been a controversial subject, involving disputes about whether or when added controls are needed, what their costs and effects would be, and who should bear their costs. Existing SO_2, NO_x, and particulate emission controls have been expensive, promoting understandable concern over the need for further costly regulations.[1] This concern, however, should be weighed against the act's inability, as presently structured, to lessen the acid deposition problem in this century.

This chapter examines the issues involved in controlling acid deposition. It starts with an examination of existing controls and their ability to affect acid deposition, then considers which emissions would be the best to focus on in attempting to reduce the amount or risk of damage from acid deposition. Finding that SO_2 emissions deserve the greatest efforts in order to reduce the damage stemming from acid deposition, the chapter describes the possible methods to control these emissions, and the main strategies that might be taken in laws and/or regulations to accomplish this aim.

The costs and other impacts of SO_2 emission controls have been examined in a wealth of studies, leading to a wide range of cost estimates. There have also been differing findings on the regional distribution of side-effects, particularly on coal production and employment, that might arise from alternative control strategies. An examination of the results of many of the cost and impact studies, and of the information and assumptions upon which they are based, is carried out in the fourth section of this chapter. This examination leads to identification of a general range for estimated control costs and focuses the areas of uncertainty regarding coal production and employment impacts and their relationship to control costs.

The final two sections treat, more briefly, the control of NO_x emissions and the alternate approach to avoiding damage from acid deposition by liming, that is, by treating the areas where acid deposition occurs with lime or other alkaline materials to neutralize the acid.

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[1] The major Clean Air Act programs affecting SO_2 and NO_x emissions are discussed in our previous report, EMD-81-131, Sept. 11, 1981, app. III.

EXISTING EMISSION CONTROLS UNDER THE CLEAN AIR ACT

Existing emission controls have been expensive

Clean air in an industrialized society is not cheap. While the technology exists to remove over 90 percent of the SO_2 emissions from a coal-fired power plant, the capital cost for the necessary flue gas desulfurization (FGD), or "scrubber" system would be high. Utility and scrubber industry analysts estimate that retrofitting conventional limestone scrubbers on a single 1,000-megawatt (MW) coal-fired power plant would require an investment of from $150 million to $240 million, and in some cases this could exceed $300 million.

Table 3, drawn from material published by the Council on Environmental Quality several years ago,[2] gives an approximate sense of these capital costs on a national scale. The table includes estimates of all costs that were expected to be incurred over the period 1979 to 1988, in response to current federal air pollution control legislation. Total annual costs were estimated to be $22.3 billion in 1979, and to rise to $37.6 billion in 1988 (all in 1979 dollars). Cumulative costs for federal air pollution control requirements for the 10-year period were projected to be nearly $300 billion dollars. Electric utility and mobile source[3] controls account for the largest share of costs in all categories.

While actual air pollution control costs may differ somewhat from these estimates, costs of this magnitude appear quite substantial. Compared with the scale of the whole U.S. economy, they are noticeable but relatively small--just under 1 percent of the gross national product in 1979, and rising to about 1.5 percent of the gross national product by 1988. Some regional effects, or effects on specific industries, however, have been more significant. Disproportionately larger impacts, for example, have occurred in the steel industry and on utility rates in some areas of the country.[4]

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[2] Council on Environmental Quality, 11th Annual Report, Dec. 1980, p. 394. CEQ has not published estimates of these expenditures in its subsequent reports. While these are several years old, we believe that they can still serve to indicate the approximate scale of air pollution control expenditures, because there have been no major changes in clean air legislation since the time these estimates were made.

[3] Mobile sources include highway vehicles, aircraft, railroads, vessels, and other off-highway vehicles.

[4] National Commission on Air Quality, To Breath Clean Air (Washington, D.C.), Mar. 1981, pp. 4.1-18 and 19.

Table 3

Estimated Incremental Air Pollution Abatement Expenditures,[a] 1979-88

Program	1979			1988			Cumulative (1979-88)		
	Operation and maintenance	Annual capital costs	Total annual costs	Operation and maintenance	Annual capital costs	Total annual costs	Operation and maintenance	Capital costs	Total costs
	(billion of 1979 dollars)			(billions of 1979 dollars)			(billions of 1979 dollars		
Public	$1.2	$0.3	$1.5	$2.0	$0.5	$2.5	$15.8	$3.7	$19.5
Private									
Mobile	3.2	4.9	8.1	3.7	11.0	14.7	32.1	83.7	115.8
Industrial	2.0	2.3	4.3	3.0	4.1	7.1	25.8	33.0	58.8
Electric utilities	5.5	2.9	8.4	7.6	5.7	13.3	62.3	42.7	105.0
Total	11.9	10.4	22.3	16.3	21.3	37.6	136.0	163.1	299.1

[a] Incremental costs are those made in response to federal legislation beyond those that would have been made in the absence of that legislation.

SOURCE: Council on Environmental Quality, 11th Annual Report (December 1980) p. 394.

Why consider additional controls?

The substantial cost of existing emission controls under the Clean Air Act has led to a heightened awareness of the impacts of possible new control measures. This awareness underscores the need to properly justify controls before implementing them and, if they are justified, to implement them cost-effectively. It does not mean, however, that new measures are not necessary or that the current act adequately deals with all air pollution-derived problems. Indeed, evidence discussed in the following pages suggests that, despite the Clean Air Act's cost, it does not deal effectively with acid deposition, visibility reduction in many areas of the country, nor other more general aspects of the long-range transport of air pollutants.

Current levels of SO_2 and NO_x emissions are high enough to contribute to continuing acid deposition at levels which, as explained in the previous chapter, may be damaging valuable resources. Furthermore, despite the emission limits imposed under the Clean Air Act, which make current emissions lower than they otherwise would have been, particularly the stringent SO_2 emission limits for new major stationary sources, United States SO_2 and NO_x emissions are still expected to increase over the balance of this century.

Utility and coal industry analysts[5] often point out that gradual replacement over this and the next 1 or 2 decades of older, heavily polluting plants by new ones, which will be more strictly controlled under the act's New Source Performance Standards (NSPS) will lead to significant SO_2 emission reductions. However, while the shift of coal-burning to plants meeting these tighter emission standards will eventually help to reduce total U.S. SO_2 emissions, the paramount question is one of time. Most existing power plants are new enough to be operable at or beyond the turn of the century. Furthermore, the cost of constructing replacement plants is large enough to make early retirement economically unappealing and, as discussed in appendix I (pp. 142-144) this is leading to efforts to extend the lives of these older plants beyond the traditional 40 years. They could, therefore, continue to emit large amounts of SO_2 well into the next century under current law and regulations.

THE CLEAN AIR ACT HAS NOT BEEN EFFECTIVE IN DEALING WITH INTERSTATE AND TRANSBOUNDARY POLLUTION

Interstate and transboundary air pollution are at the heart of the debate over the Clean Air Act's ability to control acid deposition. The Clean Air Act Amendments of 1970 were designed to alleviate local air quality problems. This was accomplished partly by reducing overall emissions, but also by using tall stacks to disperse emissions away from the sources--thereby raising the possibility of polluting downwind, and especially rural areas. While the 1977 Clean Air Act Amendments addressed the possible problems caused by sending emissions across state and international boundaries, its provisions to reduce such pollution have not been effective.

The Clean Air Act Focuses on Local Ambient Air Quality Problems

The Clean Air Act addresses local air quality problems for several "criteria" pollutants, two of which are SO_2 and NO_x.[6] The methods for controlling these pollutants are the National Ambient Air Quality Standards (NAAQS). A NAAQS sets the maximum

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[5] For example, testimony by K.E. Yeager, EPRI, before the Subcommittee on Energy Development and Applications and Subcommittee on Natural Resources, Agriculture Research and Environment, House Committee on Science and Technology, Sept. 20, 1983, p.20.

[6] Ozone, a pollutant produced by reactions involving hydrocarbon and NO_x emissions, is also a criteria pollutant.

allowable ambient[7] concentration for each criteria pollutant. States are given flexibility to decide how each NAAQS will be met within their borders. Two problems exist, however, which can lead to larger total emissions and to transport of pollution across both state and national boundaries.

First, while NAAQS's limit concentrations of SO_2 and NO_x, they do not limit concentrations of sulfates and nitrates, products into which SO_2 and NO_x can be transformed. Despite the well-established origin of these transformed products from their precursor oxide emissions, the transformation products themselves are not regulated under the act. Thus, while the transformations of SO_2 and NO_x can contribute to acid deposition, they can, ironically, facilitate compliance with NAAQS's.

Second, many stationary sources have satisfied local ambient air quality requirements by using dispersion technology--notably tall smokestacks--to send some of the emissions away from their areas of origin. It has been argued that this practice (1) allows greater total emissions than would occur without the tall stacks and (2) increases the range of transport of SO_2 and NO_x, and therefore allows greater transformation into sulfates and nitrates. Scientists disagree somewhat on just how much injection of sulfur and nitrogen oxides higher in the atmosphere affects their average transport, with the estimates in the range of 10 to 50 percent. There is little disagreement, however, that dispersion technology, as opposed to emission reductions, allows greater total emissions under a scheme which regulates only local ambient air quality. Although one may argue over how far dispersed emissions go, and in what forms they are deposited, it is clear that they add to total regional atmospheric pollutant loadings.

The Clean Air Act Amendments of 1977 have had limited effectiveness in addressing transported pollution

Recognizing that the Clean Air Act of 1970 did not address transported pollution, the Congress attempted to deal with the problem through several provisions in the Clean Air Act amendments of 1977.[8] However, as we noted in a previous report[9] these provisions have had limited effectiveness in dealing with this problem.

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[7] Ambient, in the context of the Clean Air Act, refers to the concentration of a pollutant in the air at or near ground level within a specified geographical area.

[8] Sections 110(a)2E, 115, 123 and 126 were all added in an attempt to limit transboundary air pollution.

[9] U.S. General Accounting Office, op. cit. (footnote 1).

Modeling limitations have most directly inhibited the effectiveness of these provisions to alleviate interstate and transboundary air pollution. Even close to the source, available EPA-approved models can only estimate pollutant concentrations with moderate accuracy. The models are even less accurate concerning long-range transport of air pollutants, such as between the Midwest and Northeast. Furthermore, presently approved models are not capable of dealing with acid deposition, because they cannot adequately model transformations such as the conversion of SO_2 to sulfate.

Thus, existing mechanisms under the Clean Air Act have not been able to deal with the complexities of the transported pollutants issue. Furthermore, they have not been able to resolve many of the disputes that have arisen between states, industry, environmental groups, and EPA over the problem.

PROPOSED CONTROLS AND WHERE THEY SHOULD BE FOCUSED

The Clean Air Act's inability to deal effectively with interstate and transboundary pollution, and specifically with acid deposition, has led many to advocate new environmental protection strategies to reduce the damage or risk of damage from acid deposition. Most of the proposals focus on reducing total regional SO_2 and/or NO_x emissions--a different approach from the current NAAQS focus on ambient pollutant concentrations--while another strategy proposed has been to treat vulnerable receptors to mitigate damage.

As discussed in chapter 3, there is not complete understanding of the extent to which acid deposition would be affected by any particular emission change. However, the available knowledge does indicate that, on the average, regional sulfate deposition would be reduced essentially proportionately to decreases in regional SO_2 emissions. Thus, a percentage decrease of SO_2 emissions applied uniformly over eastern North America is likely to result in the same percentage reduction of sulfate deposition throughout the region. For acidic nitrogen compounds, actual trends in recent years have shown that increases of nitrate deposition have roughly paralleled NO_x emission increases, so that nitrate deposition decreases can be expected to follow about proportionately to NO_x emissions reductions.

These findings provide a rationale for concentrating on SO_2 and NO_x emissions, if an emission control approach is to be taken to lessen the extent and effects of acid deposition.[10] Also, information on the amounts and effects of acid deposition, (see ch. 2, p. 12 and ch. 3, pp. 32 to 33) particularly effects on aquatic

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[10] The other possible approach, attempting to neutralize acid at the place(s) where it is deposited, is usually referred to as liming. It is discussed in the final section of this chapter.

and terrestrial ecosystems, justify placing the primary control focus on SO_2 emissions at this time.[11]

Examining recent and anticipated SO_2 emissions in eastern North America (see table I-10, app. I), the greatest single contributor by far is, and is expected to remain, U.S. utilities, mostly from coal burning, while industrial combustion emissions in the United States and smelter emissions in Canada are also quite substantial and anticipated to increase somewhat. Other source categories are, or are expected under current policy to become, much smaller than these and have therefore not been a major focus of control studies. We will review control strategies being considered for coal-burning utilities after briefly indicating the status of SO_2 emission controls on other fuels.

Natural gas-fueled facilities produce essentially no SO_2 emissions, since the sulfur-containing compounds found with gas are removed before the gas is consumed. Oil-based fuels can give rise to differing amounts of SO_2 emissions depending on the sulfur content of the crude oil from which they are produced and the extent of refinery processing to which the oil is subjected. In many older refineries, a large proportion of the sulfur in crude oil was channelled into the heavy product called residual oil--usually referred to as "resid"--which is the predominant oil-based fuel used by utilities and which also meets a substantial share of industrial oil demand.

Desulfurizing of resid plays a major role in SO_2 emission control in Japan[12] and was expected to be the leading SO_2 emission reduction technique for western Europe, according to a study by the Organization for Economic Cooperation and Development.[13] Some desulfurizing of resid occurs in the United States now and, until 1979, there seemed to be room to achieve substantial further reductions in U.S. SO_2 emissions by expanding refinery capacity for resid desulfurization.

However, two trends in the last few years have reduced this opportunity substantially. First, U.S. refineries have been progressively "heavying-up," that is, increasing their capacity to refine heavy high-sulfur oil, and produce less high-sulfur resid.

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[11] Continuation of NO_x emission increases (if they bring acidic nitrogen deposition in wide areas up, to equal or exceed the contributions from acidic sulfur deposition) might justify a shift in control emphasis. However, the NO_x increases anticipated for the balance of this century, while not inconsequential, would not yet require this in eastern North America.

[12] Environment Agency, Government of Japan, Quality of the Environment in Japan 1981, p. 79 and p. 208.

[13] Organization for Economic Cooperation and Development, The Costs and Benefits of Sulphur Oxide Control--A Methodological Study, Paris 1981, p. 52.

Second, a steep decline in resid use has occurred in the United States, in the wake of the 1979-80 round of world oil price increases. As of late 1983 and the first half of 1984, this decline appears to have ended, with total national resid consumption down to less than half, and utility consumption down to around 40 percent, of 1977 amounts. On this basis, by comparison to consumption and emission data from the 1970's, we estimate that total SO_2 emissions from resid use in all types of facilities in the United States are down to a rate of about 1-1/3 million metric tonnes a year, with less than two-thirds of it being attributable to the eastern 31 states. This eastern share amounts to only about 4 percent of total national SO_2 emissions--well under the contribution from nonutility coal combustion, and about 7 percent or less of the emissions from utility coal combustion. Thus, on the basis of the relative size of SO_2 emission shares, it is now appropriate to focus most SO_2 emission control approaches on utility coal combustion, with an eye to the future to possible ways to restrain growth of industrial emissions anticipated in the coming decades.

CONTROLLING SO_2 EMISSIONS

There are three well-known methods of reducing the amount of SO_2 released from coal combustion: coal washing, switching to lower-sulfur coal, and desulfurizing flue gases after combustion (scrubbing). In addition, another method now under development, furnace injection, is being considered as a possible candidate for SO_2 emission control. The form in which this method would be used--combining SO_2 and NO_x control--is called limestone injection with multistage burners (LIMB).

In the balance of this section, we explain the four methods and describe some regulatory strategies proposed to bring them into use. More detailed descriptions of these control methods and comparisons of their capabilities, limitations, costs, and side effects are presented in appendix III. The appendix also describes another method of SO_2 control--fluidized bed combustion--which is not likely to be used as a retrofit technique because it would involve replacement of much of a facility. Other techniques in early research stages, and therefore not likely to be widely available for commercial use in this century, have not been included in our study.

Control methods for SO_2

Coal washing

This is the common term for physical coal cleaning. It can remove the pyritic part of the sulfur from coal before it is burned.[14] The process has been demonstrated to be effective when used with high-sulfur coals such as those from northern Appalachia and the Illinois Basin. While not a panacea for the acid deposition problem, this technique can produce moderate sulfur removal, in the range around 20 to 35 percent, for some high-sulfur coals.

Coal washing is already being successfully used by some coal companies in the Midwest. For example, in 1981 a Peabody Coal Company representative reported that the company washed about 80 percent of its midwestern coal, and achieved an average sulfur dioxide emission reduction of about 27 percent.[15] This representative also noted some side benefits of coal washing; it reduces ash as well as sulfur content, which increases the heat content per pound of delivered coal, thus reducing transportation, storage, and handling costs and improving boiler operation and maintenance.[16]

Coal washing's ability to reduce emissions depends largely on which coals are to be treated. Some high-sulfur coals, in areas such as western Kentucky, Indiana, and Illinois, can be washed quite effectively, at costs estimated by EPRI as low as $200 per ton of SO_2 removed. By contrast, with low-sulfur coals such as those found in southern West Virginia and eastern Kentucky, washing could cost thousands of dollars per ton of SO_2 removed. Thus, washing's main value is confined to high-sulfur coal, much of which is already being washed: according to a report prepared for EPA, 72 percent of Illinois utility coal and 52 percent of Indiana utility coal was washed before delivery in 1979.[17] In an estimate given in recent EPA testimony, current coal washing was cited

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[14] Some sulfur found in coal is chemically part of the coal. This is called "organic" sulfur, and cannot be removed by ordinary washing. The remaining sulfur is present in the form of a different mineral, iron pyrites, which can be separated by physical processes, usually flotation based on the density difference between pyrites and coal.

[15] C. Farrand, Vice-President, Peabody Coal Company, testifying before the Senate Committee on Environment and Public Works, Oct. 29, 1981.

[16] Ibid. EPRI confirms improvements of plant operations and efficiency. See EPRI Journal, Nov. 1981, pp. 40-41, June 1982, pp. 24-29.

[17] Teknicron Research, Inc., Coal Resources and Sulfur Emission Regulations: A Summary of Eight Eastern and Midwestern States, 1981 (Berkeley, CA), p. 2-2.

as already reducing the potential annual SO_2 emissions from coal by 1.8 million tons.[18]

A further limitation on the amount of SO_2 emission reduction achieveable by coal washing might arise if a major national program of SO_2 control included shifting from high-sulfur to medium-or low-sulfur coals. This would lessen the amount of sulfur control that washing could accomplish at relatively low cost.

On the other hand, washed coal is a better fuel for plants with scrubbers, since it decreases the frequency of malfunctions which can be a major cause of cost increases in scrubber operations.

Expanded coal washing could probably reduce annual utility SO_2 emissions between 0.5 and 1.5 million tons.[19] While this could be a useful contribution, at best it would amount to less than 8 percent of SO_2 emissions in the 31 eastern states.

Coal switching

Coals currently used in U.S. power plants range in sulfur content from below 0.3 percent to above 6 percent.[20] From this it might appear, given enough low-sulfur coal, that very substantial reductions in SO_2 emissions could be accomplished by switching from high- to lower-sulfur coals. In fact, given some lead time--on the order of 5 to 10 years--U.S. low-sulfur coal supplies could probably be increased by amounts in the range from 100 to 200 million tons. The key questions about coal switching, therefore, are not on the production side, at least nationally. Rather, they focus on whether potentially available low-sulfur coals could be delivered to, and used in, eastern high-sulfur coal-burning plants at costs that would compete with scrubbing or furnace injection.

Three constraints and one socioeconomic concern--one relatively short-term and the others more fundamental--might make the costs of coal switching higher than could be acceptable for large-scale use. The short-term effect would occur if many users attempted to switch to low-sulfur coals before increased mining and transportation capacity was ready, and the prices were bid up in the short-term by excess demand. As we noted, if extra demand

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[18] F.T. Princiotta, Director, Industrial Environmental Research Lab, EPA, testimony at joint hearing of Subcommittee on Energy Development and Applications, and Subcommittee on Natural Resources, Agriculture Research and Environment, House Committee on Science and Technology, Sept. 20, 1983.

[19] K.E. Yeager, EPRI, op. cit. (footnote 5).

[20] Energy Information Administration, Electric Power Quarterly, April-June 1983, DOE/EIA-0397 (83/2Q), Sept. 1983, table 14.

were to phase in over a decade, we would expect that new supplies of low-sulfur coals in the range of hundreds of millions of tons could be ready, so that severe price fluctuations could be avoided.

The two more fundamental contraints that could make large-scale coal switching very expensive are transportation costs and the costs of modifying existing plants to use different coals. I both cases, we have seen conflicting views presented as to how severely and how widely the costs of switching would be affected, so we regard these as questions still requiring resolution at thi time.

The issue of transportation costs to eastern coal-fired plants appears to hinge largely on whether eastern low-sulfur coa production could be expanded if a major increase in eastern deman took place, or whether western coals would have to be used. For example, in 1982 there were 224 million tons of higher sulfur coals used in the East, from eastern districts with average coal sulfur contents ranging from 1.9 to 3.9 percent. In comparison, eastern utilities used 132 million tons of lower sulfur coals fro eastern districts with average coal sulfur contents between 1.1 and 1.3 percent, but only 30 million tons of western coals, even though they offered substantially lower sulfur content, with district averages ranging from 0.4 to 0.7 percent sulfur.

Much of the reason for this may lie with transportation costs. For example, coal from District 17 in Colorado, with an average delivered price under $23 per ton in-state, was priced between $58 and $66 delivered in the only three midwestern states where it was used. Similarly, Montana coal, priced under $10 per ton in-state, drew an average delivered price in Illinois and Indiana of about $47 per ton. These prices paid for western coal in the Midwest, consisting predominantly of transportation costs, made the western coals among the most expensive used in the few eastern states that did use them. This would seem to show why western coals did not significantly penetrate markets further east, beyond the Mississippi and Ohio River valleys. The questions that remain open on this issue are whether major expansion of eastern low-sulfur coal output is feasible and, if not, whether the extra cost of transporting western coal to the East would still allow it to compete with retrofitting SO_2 control technology.

The second uncertain issue about coal switching is the cost impact of accommodating a plant to coals different from the one for which the plant was designed. In the case of mine-mouth plants (those adjacent to mines) this appears quite unlikely. But, more generally, coals differ in many ways, including not onl sulfur content, but also heat value, grindability, and ash content and composition--the latter two being particularly important in determining tendencies to form slag and cause fouling in a furnace. Switching to lower-sulfur coals is likely to involve changes of some of these other properties which can result in a wide range of effects on the operation of a generating plant.

Some changes can require capital expenditures, for example, replacing pulverizers for a harder coal or expanding precipitator capacity to handle greater amounts of ash. Modifications in the furnace itself may be required if slagging properties are changed. Changes in heat value can be particularly important since, unless all of a plant's material-handling equipment is modified, using coal with a lower heat value could result in derating the plant and thus raising generating costs in inverse proportion to the reduction in heat value. The significance of this can be seen in the fact that most eastern coals average over 12,000 Btu's per pound, and even those of the Illinois basin have heat values that fall between 10,900 and 11,500. In contrast, three-quarters of western coals have heat values below 9,000 Btu/lb.; only about 18 percent even average above 10,700 (all in the Southwest); and less than 2 percent are above 12,000 Btu/lb. Thus, unless increased eastern low-sulfur coal output could meet much of the demand, the prospects for coal switching seem further clouded by the cost impacts from modification or derating of plants to accommodate different coals.

Finally, significant localized socioeconomic impacts could occur in eastern high-sulfur coal-producing regions if a large amount of coal switching were to take place. The impacts could be losses (or foregone growth) of employment for miners and others, and generally depressed or stagnant economies in these regions. The regions affected would be northern Appalachia (areas in Pennsylvania, Maryland, northern West Virginia, and eastern Ohio) and the Illinois Basin (areas in Illinois, Indiana, and western Kentucky). While a large portion of these employment and economic effects would be balanced out on a national basis by increases in low-sulfur coal-mining activity in other areas of the country, the impacts in the high-sulfur regions could be quite substantial. The Department of Energy estimated that for each million tons of SO_2 emissions controlled by switching, between 5,100 and 6,400 high-sulfur coal mining jobs would be lost if the replacement coal had a 1-percent sulfur content.[21] Allowing for a wider range of sulfur content in the replacement coals, between 0.5 percent and 1.2 percent sulfur compared with the 1 percent assumed in the Department of Energy calculation, we estimate a wider range: between about 4,300 and 7,000 mining jobs in high-sulfur coal areas would be lost for each million tons of SO_2 emissions controlled by switching.

These calculations are projections for the future, probably between 1990 and 1995, when all the needed low-sulfur coal would have been developed for the switching plants, so it would not be likely to actually result in extensive losses of jobs by current miners in the next several years. In fact, given the expected growth in overall coal use, some of the affected areas might

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[21] Dept. of Energy, Office of Policy Planning, and Analysis, Costs to Reduce Sulfur Dioxide Emissions, DOE/PE-0042, Mar. 1982, app. p. 13.

experience only reduced growth or constant employment from the net effect of losing some markets because of switching and gaining some markets from growth in other uses. However, most studies still show that large-scale switching would result in there being fewer mining and mining-related jobs in the early 1990's than now exist in some of the high-sulfur coal areas affected.

Scrubbing

Retrofitting existing coal-fired power plants with flue gas desulfurization (FGD) systems, commonly called scrubbers, is the most effective--but also the most capital intensive--means of reducing SO_2 emissions commonly available or in use today. Scrubbers are required equipment on all new coal-fired power plants. Most current systems are "wet scrubbers," designed to combine the SO_2 in the flue gas with a circulating slurry containing lime or limestone, that is, calcium oxide or calcium carbonate. The SO_2 reacts with this material to form nonsoluble calcium compounds, which are subsequently disposed of as sludge. A few units now in place are spray driers, in which a slurry with lower moisture content dries as it is sprayed into the gas stream. The dry product is then collected at the same time as other particulate material on the precipitator or filter.

Scrubbers could achieve a 90-percent reduction of SO_2 emissions for many of today's highest emitting sources. This is significant because a relatively small number of large coal-fired power plants are responsible for much of the SO_2 emissions in the eastern half of the United States. A study prepared for DOE[22] estimated that the 50 largest utility emitters in the eastern United States were responsible for 8.6 million tons of SO_2 per year and that scrubbing could reduce these emissions to 0.9 million tons per year, a reduction of about 7.7 million tons.

Scrubber maintenance can be troublesome and its cost is quite substantial, running higher than that for maintenance of the entire balance of a plant. Many utilities prefer to avoid scrubbers if possible, so as not to get involved in ". . . running a chemical plant as well as a power plant." A related issue with scrubbers has been low reliability, which could result in very high costs for buying replacement power or maintaining extra generating capacity. To avoid such costs on new plants, it is common to build multiple scrubber units, including a spare to serve when one unit malfunctions. This practice contributes additionally to capital outlays for scrubbers.

While scrubber capital costs are always high, there are a number of reasons why the capital costs for retrofit scrubbers are a heavier burden than they are for new ones. First, the cost of building scrubbers as retrofits is generally higher than the cost

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[22] PEDCO Environmental, Inc., Control Strategies for Coal Fired Utility Boilers, DOE/METC-82-42, May 1982.

for building them for a new plant, because modifications will have to be made to an existing plant in addition to building the entire scrubber. Examples include extra duct work to carry flue gases to and from the scrubber, which can seldom be ideally located to maximize efficiency, and extra support structures for scrubber components placed in unusual locations because of space constraints. This "retrofit cost penalty" is roughly estimated as an extra 30 percent added to the scrubber cost, but is actually very site-specific, and can run as high as 150 percent.

A second reason why retrofitting increases the impact of scrubber capital costs is that while a scrubber built with a plant can be amortized over the full life of the plant, a retrofitted scrubber will have a shorter life so that a greater proportion of its cost will have to be paid for each year. Also, scrubbers use energy from the plant to operate blowers, pumps, and gas reheaters. Consequently, they lower the plant's electric output and cause costs per unit of electricity sold to increase further, since costs must be recovered from smaller sales. This scrubber energy penalty tends to be in the range of 3 to 6 percent of total plant output--not extremely large, but significant because all these extra cost factors multiply together to potentially result in extremely high increases in the cost of power from a plant retrofitted with a scrubber.

The extra cost burden from scrubber retrofitting may be lessened in two ways. One would be to omit building a spare scrubbing unit and allow the plant to operate with less than complete scrubbing on occasions when a scrubber unit malfunctions. This could be considered in cases where scrubbers were installed to limit total emissions, as long as periods of high emissions were balanced by averaging with other low-emission periods, and the high-emission periods did not violate ambient air quality standards.

The second way to avoid extremely high cost impacts from scrubber retrofitting is to focus it on plants which will continue to operate for longer lifetimes. New SO_2 emission restrictions effective as of July 1983 in West Germany do this by only requiring control retrofitting on plants which will remain in use beyond a set number of hours of operation. (See app. IV.) This approach would produce less emission reduction by scrubbers than would come from wider use of scrubbing, and could add some costs to the overall emission reduction achieved, to cover accelerated construction of replacement generating capacity. But there is the possibility that the added costs could be balanced in part by improved efficiency and reliability of new as compared with several-decade-old plants.

LIMB

Limestone injection with multistaged burners uses an SO_2 emission control technique which had been tested and rejected in the 1960's, but has been revived and become the subject of new development efforts in a number of countries in the past several years. The technique entails injecting a finely divided SO_2 sorbant (an absorbing material) directly into the furnace, so it can act in the zone where the pulverized coal/air mixture burns.

Poor results in previous efforts at furnace injection occurred in large part because flame temperatures were so high that they inactivated the sorbant. Therefore, when furnace operating conditions came under study, in efforts to decrease NO_x emissions, tests were made which showed that the conditions which favor less NO_x formation also make in-furnace SO_2 capture more effective. These conditions emphasize lower flame temperatures and gradual mixing of air and coal, starting out with a fuel-rich mixture and then slowly adding more air in stages.

LIMB still requires an excess of sorbant calcium over fuel sulfur, usually in the range of between 2:1 and 3:1, compared with a calcium/sulfur ratio nearer to 1:1 in a scrubber. At the 2:1 or 3:1 ratio, LIMB is expected to achieve SO_2 capture rates of 50 to 70 percent, compared with rates up to and even above 90 percent in scrubbers.

Using two or three times as much limestone as sulfur would result in a major increase in production of particulates, which would have to be removed from the combustion products. For this reason, retrofitting LIMB would demand expansion and/or replacement of the electrostatic precipitator or fabric baghouse particulate collector on the plant. However, this is likely to be the main capital investment required for LIMB; the limestone storage, grinding, and injection equipment will be off-the-shelf items, and the burner modifications, even if they have to be custom-made, will be small enough to have relatively minor costs. In sum, LIMB will require much lower capital costs than scrubbers. In recent testimony, preliminary estimates were made that capital costs for retrofitting LIMB will be less than one-fifth (EPA)[23] or between about one-fifth and two-fifths (EPRI)[24] those for retrofitting scrubbers on the same plant.

While its lower capital and maintenance costs stemming from its greater simplicity could make LIMB less unattractive than scrubbers for retrofitting, LIMB would not have as great an advantage in cost-effectiveness when looked at as an SO_2 control technology alone. First, because it would use two or three times as much sorbant as a scrubber would (per unit of SO_2 emitted), LIMB would have greater operating costs, partly offsetting its capital and maintenance cost advantages. Also, while its overall cost per kilowatt-hour would be much lower than scrubbing, its lower level of SO_2 removal--probably about 50 percent on retrofits, compared with 90 percent for scrubbers--would mean that the costs would be

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[23] F.T. Princiotta, EPA, op. cit. (footnote 18).

[24] K.E. Yeager, EPRI, op. cit. (footnote 5).

distributed over a smaller amount of SO_2 removed, which would make the cost per unit of SO_2 removed closer to that for scrubbers.

Because LIMB is still in the developmental stage, only approximate cost comparisons are justified at this time. However, EPRI estimates suggest that costs per ton of SO_2 removed would be between about 13 and 35 percent lower for LIMB than for scrubbing.[25] Such savings in SO_2 control costs, if achieved, would certainly be desirable, but some time and effort are still required before they can be regarded as assured or before it can be known how widely LIMB could be applied, and exactly when it could be ready. Research and development (R&D) efforts presently under way could make it possible to have LIMB retrofits in place on a share of existing coal-fired boilers by the mid-1990's, with some other boilers possibly following a few years later. This might be accelerated by a few years, but it is unlikely that LIMB could be in use to any substantial extent before 1990.

The overall scope of applicability of LIMB will be determined by the R&D effort. Work to date has emphasized two major types of existing boilers--tangential (or corner-fired) and wall-fired--which together make up 66 percent of pre-NSPS coal-fired capacity. A recent EPRI estimate is that about half of these, some 35 percent of all pre-NSPS capacity, already appear likely to be compatible with LIMB, and preliminary studies on a third type of boiler, cell-fired (another 14 percent of pre-NSPS capacity), have begun. Thus, efforts underway could result in LIMB application on up to 80 percent of pre-NSPS boilers and, therefore, might be able to lower total SO_2 emissions from older coal-fired plants by as much as 40 percent.

Two LIMB retrofit projects in West Germany are further advanced than U.S. efforts. As of late 1983, each was about to install LIMB on an operating large-sized boiler, one of 700 MW and another of 300 MW. While the coals are lower in sulfur than U.S. coals, and the boiler designs are not very similar to those here, success in these German projects could help increase utility confidence in the technology. This is important because, until utilities are ready to trust LIMB's reliability, it is not likely to be applied widely because of the high costs that can come from loss of generating capacity that could occur due to reliability problems. Indeed, some utility representatives have suggested that, as things now appear, they would avoid LIMB and prefer to accomplish the same amount of emission reductions by scrubbing a smaller portion of their generating capacity, unless LIMB proves to be substantially more reliable than scrubbers. They explained that, because LIMB equipment would be located in the furnace, a LIMB breakdown would force the affected plant to shut down, whereas an inoperable scrubber unit could be replaced by another and still allow the plant to operate.

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[25] Ibid, pp. 14 and 17.

In summary, LIMB is a developing technique which may offer SO_2 control in retrofits at much lower capital cost, and somewhat lower cost per ton of SO_2 controlled, than scrubbers. It could not be widely used until some time in the 1990's, and its scope of applicability is not yet clear, though it already appears to be compatible with at least one-third of pre-NSPS coal capacity. Its future hinges on the progress of ongoing development work, plus the possibility that its ability to lower NO_x as well as SO_2 emissions could become particularly desirable in future air pollution control considerations.

Regulatory strategies for SO_2 control

A wide range of possible regulatory approaches could be taken to bring about a desired reduction in SO_2 emissions. At one extreme, under existing air pollution control policy, U.S. SO_2 emissions will decline as older coal-fired sources retire and are replaced by lower emitting units which meet the strict NSPS. Those concerned about acid deposition, however, find waiting for retirements to take their course to be unacceptably slow, as illustrated by projections that under that policy, eastern U.S. SO_2 emissions will actually rise at least until 2000.

Some economists suggest that a market approach, such as charging an emission fee per unit of SO_2 released, would be the most efficient approach to control. They argue that, by suitable adjustment of the fee, the desired level of emissions would be reached at minimum cost by market forces, since emitters would adjust their facilities to minimize the total cost of controls plus fees. While this approach is used in Japan, it has made little headway here.

Other market-type approaches have been used in the United States to a limited extent, by encouraging "emissions trading" between sources, and by "bubbling." Trading can lead to emission reductions being made by facilities which can do it at lower costs than would have to be paid by the sources which were originally required to reduce their emissions under existing regulations.[26] Bubbling allows a firm to choose which and how many among a number of its sources it will control, to give an overall level of emissions that meets the standards in effect. Firms can do this rather than having to place controls on all sources, despite possibly wide differences in costs of control at the different sources.

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[26] We examined trading and found it offered prospects for lowering control costs. See U.S. General Accounting Office, A Market Approach to Air Pollution Control Could Reduce Compliance Costs Without Jeopardizing Clean Air Goals, PAD-82-15, Mar. 23, 1982.

A number of more traditional control strategies have been examined in studies released over the past several years.[27] These include requirements for

- --uniform percentage reductions of emissions throughout the United States, or in the eastern 31 states;
- --caps limiting emission rates from individual facilities, which would result in proportionately greater reductions by regions and sources that currently emit at higher rates;
- --uniform percentage rollbacks of emission limits as set, for pre-NSPS sources, in state implementation plans (SIPs);
- --combinations of SIP rollbacks plus requirements for removing a minimum percentage of the potential SO_2 that would be emitted on the basis of fuel sulfur content;
- --various coal-washing requirements;
- --compulsory retirement of fossil-fueled plants at a fixed age;
- --NSPS compliance by older units; and
- --least emissions dispatching.[28]

This range of approaches would result in very different levels of SO_2 emissions over the next 2 decades and would have highly variable effects on emissions in different states.

The strategies that include minimum percentage removal and/or stringent caps on individual emission rates would generally lower total SO_2 emissions most and, in the process, assign the greatest emission reductions to states with many large sources of high emission rates, that is, states in the Midwest and Ohio River Valley. While this geographic imbalance in emission reduction requirements has given rise to strong opposition from the states potentially most heavily affected, it was carried through in the first legislative proposal on acid deposition to make significant progress through the Congress: Senate bill 3041 in the 97th Congress, reported out by the Senate Environment and Public Works Committee on November 15, 1982. This approach was based, at least partly, on a view that the sources with highest emission rates would generally offer the most cost-effective opportunities for

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[27] Many of these strategies are described, and some of their cost and effects are compared in Emissions, Costs and Engineering Assessment, Interim report of Work Group 3B, Under U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Feb. 1981, Section E1.

[28] See app. III, p. 153.

emission reductions. However, it also expressed the views of Northeast states that they are already paying high costs to limit emissions from their electric-generating plants (by using oil rather than coal) and that therefore, states with high emission rates should also accept increased prices to lower emissions, particularly since those emissions contribute substantially to acid deposition in downwind, that is, Northeast, states.

This approach, asigning emission reductions largely to states whose utilities omit SO_2 at high rates, has remained at the center of most proposals for acid rain control legislation. However, a major difference in the financing of controls has been suggested in a number of legislative proposals. These would have the costs of emission controls shared, to various extents, by a wider group of persons than those who would bear the costs if controls were paid for entirely by those emitters taking control actions. The cost sharing would be accomplished by payment, for some part of the control cost, being made with money from fees to be collected by a governmental body. The fee would be based in most cases on all, or at least some part of, the electricity generated or used in the United States, with different proposals leaving various geographic areas or types of power sources (e.g., hydro and/or nuclear) exempted from the fees; fees based on emissions rather than on electricity have also been proposed. We have not studied the ways in which these costs would be distributed under different proposed financing approaches, since these questions have been treated extensively in reports from the Office of Technology Assessment (OTA), the Congressional Research Service (CRS) and the Congressional Budget Office (CBO),[29] among others.

COSTS AND COAL MARKET IMPACTS OF PROPOSED SO_2 CONTROLS

The cost of reducing SO_2 emissions will, of course, depend mainly on the size of the reduction desired, and also on the control methods used, as well as on other variables such as the time allowed to accomplish the reduction. A number of studies have attempted to estimate the costs of emission controls in response to two types of questions:

--What would be the lowest-cost way to reduce SO_2 emissions down to some particular level? and

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[29] Office of Technology Assessment, Acid Rain and Transported Air Pollutants, Implications for Public Policy Report No. OTA-O-204, June 1984; L.B. Parker and D.A. Thompson, Acid Rain Legislation and the Future of the Eastern Low-Sulfur Coal Industry, CRS Report No. 84-89-ENR, Apr. 6, 1984; and Congressional Budget Office, Curbing Acid Rain: Allocating Sulfur Dioxide Control Costs Under an Emissions Control Program, Staff Working Paper, Jan. 1984 and previous reports cited in these studies.

--What would be the cost of a particular emission control strategy, and how would its cost compare with the lowest-cost way?

In addition, a good deal of work has been devoted to estimating the extent to which coal-switching would occur under different strategies, because of the concern about employment impacts in high-sulfur coal areas.

The results of the studies can provide a range of estimates for the costs and coal-market impacts of a particular control strategy. Probably more useful, however, is an examination of how and why the results of various studies differ, because this will focus on the differences in assumptions and/or methods among the studies, and allow us to understand how well the costs of control and the coal-market impacts of different strategies are known.

Many legislative proposals to deal with acid deposition have been made during the 97th and 98th Congresses, ranging from futher study to very strong control measures. These have been analyzed to various extents by many parties to the debate. In this section we have focused on legislative proposals that address two major types of approaches, one prescribing controls to be paid for entirely by the sources which would control their emissions and another which would spread the costs of control more widely.

This section starts by describing the acid deposition control strategy in the bill (S. 3041), reported to the Senate in November 1982, and then examines the results and differences among a number of modeling studies of the costs of this bill and similar control strategies. The section next turns to coal market shifts that might arise from this bill, and then examines an alternative strategy offered in a House bill (H.R. 3400), proposed in 1983 in order to avoid these shifts. Modeling studies of costs and coal market impacts, the major kind of study examined up to this point, agree that H.R. 3400's strategy would protect high-sulfur coal markets, but would be significantly more expensive than the Senate bill strategy. However, different studies disagree sharply over the extent to which coal market shifts would differ between the two approaches. This leads to consideration of the same questions as viewed by the other kind of analyses available on these subjects, site-specific studies, and how their results differ from those of modeling studies, in particular on the relative costs of the H.R. 3400 strategy and that of S. 3041. The section ends with an overview of knowledge about the costs to be expected for acid deposition control, leaving unresolved the question of whether avoiding coal market shifts would lead to higher control costs.

The Senate Environment Committee bill--modeling and overview studies

The first legislative approach to emission reduction for acid deposition control which went beyond hearings was contained in

S. 3041 in the 97th Congress, as reported out by the Senate Committee on Environment and Public Works as part of its proposed Clean Air Act Amendments.[30] The key acid deposition provision in this bill was the requirement for a reduction of annual SO_2 emissions in the eastern 31 states by 1995 to a level 8 million tons below 1980 emissions, with the individual state reductions determined by an "excess utility emissions" formula, as explained in the next paragraph.

This type of formula for allocating emission reductions had originally been contained in another bill, S. 1706, proposed in 1981,[31] which had aimed to achieve a larger, 10-million ton, reduction of SO_2 emission in the 31 eastern states. The Environment Committee bill first provided for the states to try to agree on an allocation of the emission reductions among themselves. If this was not achieved, the bill then called for distributing the reduction requirements in proportion to each state's share of "excess" utility SO_2 emissions in the region in 1980. Excess emissions were defined as those above a limiting rate from fossil-fuel-fired utility plants. This rate was set at 1.2 pounds of SO_2 per million Btu's of fuel heat content in the earlier bill, and at 1.5 pounds per million Btu's in the Committee bill. While the bills both called for state SO_2 reduction requirements to be determined by the excess utility emissions formula, each state was to be left to assign its required reduction to any combination of sources, with the expectation that this would allow market-like decisions that could approach a least-cost response to the reduction requirement. To further the objective of reaching a least-cost response, the Committee bill also authorized trading of emission reductions within and between states in the eastern region.

The bill included other provisions which could change its total emission reduction effects, such as a requirement for further reductions to offset increases in emissions or emission rates after 1980, and oportunities to decrease the required SO_2 emission reductions by substituting reductions of NO_x emissions. These additional provisions, as well as questions of interpretation of the main parts, led to some disagreement on the proper basis for analyzing the costs of this bill.[32] However, a reasonably consistent comparison is possible by focusing on the total amount of SO_2 emissions which is to be controlled in each analysis.

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[30] S. 3041 (97th Cong.), Nov. 15, 1982. This bill was not brought up on the Senate floor, but was reintroduced verbatim in the 98th Congress as S. 768, Mar. 10, 1983, and was reported out of the committee with amendments on May 3, 1984.

[31] Senator George Mitchell and others, S. 1706 (97th Cong.) Oct. 6, 1981.

[32] We examined the effects of many of these other provisions in <u>Analysis of the Acid Rain Proposal Approved by the United States Senate Committee on Environment and Public Works on July 22, 1982</u>, U.S. General Accounting Office, B-209020, Sept. 24, 1982.

A number of studies of the costs of S. 3041 and S. 1706 were carried out by the consulting firm ICF, Inc., with its Coal and Electric Utility Model, using input assumptions specified by the various clients for whom the studies were done. The results of five of these ICF analyses, and one of S. 1706 done by OTA, are presented in table 4.

The major effect of the different input assumptions appears in the range of amounts of total SO_2 emission reductions that are shown in the results, while the range of cost estimates largely follows from this range of reduction requirements.

The results show a roughly continuous pattern of costs increasing with amount of emission reduction, except that the calculation for an 8.6-million-ton SO_2 emission reduction, done for

Table 4

SO_2 Emission Reductions and Their Costs for Various Interpretations of "Excess Utility Emissions" Formulae

Bill (97th Congress)		S.1706	S.1706	S.1706	S.3041	S.1706	S.1706
Interpretation		EEI[a]	EEI[a]	NWF[b]	EPA[c]	OTA[d]	"OTA" EEI[e]
Analysis by		ICF	ICF	ICF	ICF	OTA	ICF
SO_2 emission reduction (million tons)		13.1	12.0	10.0	9.2	8.6	8.1
Annualized control cost (billions of 1982 $)	Without interstate trading	$6.9	$5.4	-	$3.3	$4.2	$2.6
	With interstate trading	-	$4.3	$2.8	-	$3.9	$2.2

[a]EEI = Edison Electric Institute.

[b]NWF = National Wildlife Federation.

[c]EPA = Environmental Protection Agency.

[d]OTA = Office of Technology Assessment.

[e]"OTA" EEI = EEI understanding of OTA's interpretation.

Source: ICF analysis of S.1706: Summary of Acid Rain Analyses Undertaken by ICF for the Edison Electric Institute, National Wildlife Federation and Environmental Protection Agency, Prepared by ICF Inc., for Edison Electric Institute, May 1982.

ICF analysis of S.3041: ICF memo from K.S. Schweers & B. Braine to M. Wagner, J. Austin and R. Brenner, EPA, Revised Sept. 16, 1982.

OTA analysis of S.1706: OTA Staff Briefing Memo for Senate Committee on Environment and Public Works, Revised June 4, 1982. Converted for inflation from 1979 dollars.

OTA, has a cost that is about one-third higher than would be expected from the series of ICF calculations. We regard this as an indication of differences between the models used, and do not believe that one or the other is necessarily more correct.

Cases that assume interstate trading have lower cost estimates than cases that exclude it, in the three examples where calculations were done for both. Cost increases of 18 and 26 percent are estimated for excluding trading in two examples treated by ICF, while the calculation by OTA shows about an 8-percent cost increase for excluding trading between states. In our analysis of S. 3041, we noted that while intrastate trading appeared feasible, the prospects for interstate trading did not seem good, because of political and institutional factors,[33] so the size of this cost difference may well be academic.

Table 4 shows that control costs increase with increasing amounts of emission reductions, and that the increase is greater at higher levels of control. This is to be expected in any cost-conscious approach to control, since emitters will use, or trade for, cheaper control methods first, and only go to the more expensive methods as cheaper ones are exhausted.

The results also show that the cost increase becomes steeper quite rapidly at high levels of control. Looking at the ICF cases which exclude interstate trading, the 8.1-million-ton reduction is estimated to have an annualized cost of $2.6 billion, or $321 per ton of SO_2. The cost for the step from 8.1 to 9.2 million tons is an increase of $700 million per year for 1.1 million tons, or $636 per ton--about twice the average cost of the first 8.1 million tons. The next step, from 9.2 to 12 million tons is estimated to cost still more--$2.1 billion dollars per year for another 2.8 million tons, or $750 per ton--and the final step--from 12 to 13.1 million tons--is estimated to result in a cost increase of $1.5 billion dollars for 1.1 million tons, for an average of $1,364 per ton, which is more than four times as expensive as the average for the first 8.1 million tons. This kind of steep increase in control costs at large emission reductions was also found by OTA in a series of calculations with its model.[34]

Another quite different approach to estimating control costs was also released in 1982, by DOE.[35] Using an engineering analysis approach rather than the econometric modeling method, to estimate minimum costs, the DOE study found that a 5-million-ton reduction of annual SO_2 emissions would have an annualized cost

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[33] GAO, ibid.

[34] Office of Technology Assessment, Acid Rain and Transported Air Pollutants, Implications for Public Policy, Report No. OTA-O-204 June 1984, p. 168.

[35] Department of Energy, op. cit. (footnote 21).

(adjusted to 1982 dollars) of a minimum of \$1.5 billion, while a 10-million-ton reduction would have a minimum annualized cost of \$4.1 billion. Looking at incremental costs again, this says that the first 5-million-ton reduction would average at least \$300 per ton of SO_2, while the second 5 million tons would have an average cost of at least \$530 per ton of SO_2.

Comparing these three studies, the ICF work would indicate that reductions of annual SO_2 emissions by 10 million tons would show an annualized cost of nearly \$4 billion, very close to the DOE estimate's minimum of \$4.1 billion, while OTA's results would appear to fall near or slightly above \$5 billion a year. Our experience with prospective cost estimates, especially when made at this stage of a project before the particulars of the sites are taken account of and engineering design is done, is that they are unlikely to be accurate enough to distinguish a difference as small as the 25-percent range between these three estimates. Therefore, these results, taken together, appear to indicate a range of about \$4 billion to \$5 billion per year in 1982 dollars as the annualized cost of a 10-million-ton reduction of annual SO_2 emissions in the eastern United States. One caveat should be placed on this estimate however. DOE's estimate is described as a minimum so that, if some factors not counted in that estimate turn out to be substantial, then the DOE estimate of annualized cost could go above the \$5-billion-per-year level.

As an indication of the overall national effect of the estimated cost of these control proposals, the \$4-billion to \$5-billion range can be compared to the total cost of electricity in the United States, on the premise that the control costs would be paid by the users of power from plants whose costs were increased by the controls. In 1982, 2.09 trillion kilowatt hours (kwh) were sold to consumers in the entire United States, of which 1.92 trillion kwh, or 92 percent, were sold to consumers by privately owned utilities, yielding total revenues of \$110 billion.[36] Adding somewhat less than \$10 billion for the remaining 8 percent of sales by publicly owned utilities would bring total electricity costs in the United States to a bit under \$120 billion a year, so that the estimated increase of \$4 billion to \$5 billion would amount to a total increase of between 3.3 and 4.2 percent of 1982 costs, or quite likely a somewhat smaller percentage of costs as of the 1990 to 1995 time when the control programs were to be fully in effect.

While this effect does not seem severe, compared with an average increase of total national privately owned utility revenues which averaged 13.4 percent per year from 1977 to 1982, it has been a focus of understandably strong concern to the states in which the cost effects would be concentrated, where electricity

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[36] Energy Information Administration, Electric Power Annual, 1982, DOE/EIA-0348 (82) Aug. 1983, tables 112 & 117.

cost increases could be several times the national average.[37] This distributional issue, and another regional concern about possible adverse impacts on mining employment and local economies in high-sulfur coal-producing areas, have combined to maintain strong opposition to the type of emission reduction approach in the bills just discussed.

Coal market impacts and a proposal for avoiding them

The impact the previous bills would have on the regional distribution of coal production has been a major focus of concern about acid deposition control legislation. ICF's analyses had regularly examined this issue, and the effect of the 1982 Senate Environment Committee bill was estimated to result in major changes in regional coal production.

In particular, while ICF's analysis showed negligible effects on total national coal output, it indicated that, if the bill went into effect, then 1995 coal production would be about 56 million tons lower in northern Appalachia and about 96 million tons lower in the Midwest (mainly the Illinois Basin) than without the control legislation. ICF's studies indicated that these large losses would be balanced by increases of 1995 coal production--about 62 million tons in central Appalachia and 90 million tons in the West--again compared with 1995 projections without new controls. The regional losses projected in these calculations were large enough to overbalance expected overall growth in the use of coal from the high-sulfur regions. The 1995 coal production levels were projected to be down 22 million tons (12 percent), from 1980 output in northern Appalachia; down 23 million tons (17 percent), from 1980 levels in the Midwest; and down about 5 million tons (19 percent), from 1980 levels in southern Appalachia.

Indeed these declines, for a total reduction in annual SO_2 emissions of 9.2 million tons, were larger than ICF had projected in earlier studies of the effects of even larger emission reductions, ranging up to 12 million tons of annual SO_2.[38] This increase in coal production shifts was explained by ICF[39] as arising because they had modified their model to take account of reliability penalties from retrofit scrubbers for this analysis, compared with the model as used in the earlier studies. The result of this change in scrubber costs was to change the relative

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[37] We have not analyzed cost distributions among states ourselves since they have been looked into closely in a number of studies by OTA and CRS. See footnote 29.

[38] ICF Inc., May 1982. See first source reference in table 4 above.

[39] Schweers & Braine, Sept. 1982. See pp. 8 to 10 in second source reference in table 4 above.

economics of switching versus scrubbing, so that about 15 gigawatts (GW) of electric generating capacity would probably switch instead of scrubbing. This relative balance between the costs of switching and scrubbing is an important dimension of acid deposition control, to which we will return in the next section.

As a response to the regional concerns about the distribution of both total control costs and coal production impacts, a different legislative approach to acid deposition control was proposed in H.R. 3400 in June 1983.[40] This included a combination of two major changes from the previous bills--one to lessen coal production shifts and the other to spread the burden of control costs more widely over the nation.

The coal production effects were to be lessened mainly by a provision requiring that scrubbers be installed on the 50 power plants in the country which emitted the largest total amounts of SO_2 in 1980, chosen from those plants emitting at rates equal to or greater than 3 pounds of SO_2 per million Btu's. As a result of this provision, many plants believed likely to switch coals would, instead, be required to install scrubbers, thus avoiding a large part of the anticipated coal production shifts. In addition, the bill offered the possibility of contributing to the costs of scrubbers on other plants which, if it were to actually occur, could tip some additional utility decisions from switching to scrubbing.

The other new feature of H.R. 3400 was to establish an Acid Deposition Control Fund, which would pay 90 percent of the capital costs of installing scrubbers at the "top 50" plants and also, if funds were available, the same percentage of capital costs for emission control technology at other plants required to reduce emissions under this bill. Those other plants were to be chosen by the states, to meet state allocations of emission reductions based, as in previous proposals, on an "excess utility emissions" formula. These allocations were to be calculated for all of the lower 48 states, rather than only the 31 in the East, from a baseline of 1.2 lbs of SO_2 per million Btu's, but giving credits to any state for the reductions accomplished by those plants in the state which were among the national "top 50." The money for this fund was to be collected from a fee of one mill (one tenth of a cent) paid by utilities for each non-nuclear kilowatt hour generated in or imported into the lower 48 states starting in 1985.

Costs and effects of this proposal were analyzed separately by ICF, Inc., and by OTA, using econometric models as in their previous work, with each study examining the effects of both the proposed H.R. 3400 and a similar emission reduction program without the scrubber requirement on the "top 50" plants. In each case, the results showed that the requirement of scrubbers raised

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[40] H.R. 3400, Representatives G. Sikorski, H. Waxman, and J. Gregg, June 23, 1983.

the cost of the emission reduction substantially, as shown in table 5.

Table 5

Annualized Costs of Emission Reduction Under H.R. 3400

Study by	Top 50" scrubbers required	No scrubbers required
	(billions of 1982 dollars)	
OTA[a]	$4.5 to $5.3	$2.9 to $4.0
ICF[b]	$4.2	$3.2

[a] OTA Oceans & Environment Program An Analysis of the Sikorski/Waxman Acid Rain Control Proposal: H.R. 3400, The National Acid Deposition Control Act of 1983. Staff memorandum, Revised July 12, 1983.

[b] D.E. Klein, ICF Inc., Analysis of Coal Production and Employment Impacts of the Waxman-Sikorski Bill and a Cost-Effective Equivalent, memo to Alliance For Clean Energy, Sept. 22, 1983.

Again, as in the previous studies, we see that OTA's cost estimates were somewhat higher than ICF's. However, both agreed on the central points--eliminating the scrubber requirement would lead to much less scrubbing and much more switching, and would save $1 billion a year or more, about one-quarter to one-third of the total cost of the programs.

ICF's study is explicit about the amount of scrubbing involved in the two cases. Under its "cost-effective equivalent" program, only 2.5 GW of generating capacity would retrofit scrubbers for acid deposition control, while in the case with scrubbers mandated, 67.7 GW of capacity--over one-third of all pre-NSPS coal-fired capacity--would have scrubbers retrofitted.

In contrast, OTA does not directly estimate amounts of each control method which would be used in the two cases it analyzes. However, the relative extent to which plants freed from the scrubber requirement would turn to switching can be roughly compared between the two models by looking at the amounts of coal production shifts that would occur in that event.

Coal market impacts--Coal production by regions for three cases, one under H.R. 3400, a second for a "cost-effective equivalent" program without the scrubber requirement, and a third for a base case with no acid deposition control, was projected for 1995 by ICF as shown in table 6.

Table 6

Regional Coal Production Impacts as Projected by ICF for 1995

	1980 (Actual)	1995 Base case	H.R. 3400	"Cost-effective equivalent"
	(millions of tons)			
East				
Northern Appalachia	185	213	202	157
Central Appalachia	233	295	299	350
Southern Appalachia	26	26	26	27
Midwest	134	182	187	88
West				
Western Northern Great Plains	125	250	250	270
Rockies	32	73	78	133
Southwest	29	61	59	78
Balance of West	65	200	200	196
U.S. Total	830*	1,300*	1,303*	1,299

*Totals do not add because of independent rounding in the source.

Source: D.E. Klein, ICF Inc., op. cit. (note b to table 5).

The base case shows that, according to the model, all major regions would share to some extent in the anticipated growth of total coal production. The West would generally show greater growth, with most regions at least doubling their output, while the three main eastern regions would see some increases, by 27 percent in the lower sulfur central Appalachia region, and by 15 and 36 percent, respectively, in the higher sulfur northern Appalachia and Midwest regions. In keeping with the very large amount of scrubbing that ICF projected under H.R. 3400, their model shows that this bill would cause only minor changes in coal production in 1995 compared to the base case. Indeed, all regions except for southern Appalachia are projected to remain well above their 1980 production levels, and the high-sulfur Midwest even shows a small increase.

The effect of an equivalent emission reduction without the scrubber requirement, however, is very substantial, according to the ICF model. The two major high-sulfur regions, northern Appalachia and the Midwest, lose, between them, 150 million tons of production compared with the 1995 base case, and would even be down significantly from their 1980 levels, by 28 million tons, or 15 percent, in the former and by 46 million tons, or 34 percent, in the latter. According to ICF's model, a little over one-third of this loss would be made up by central Appalachian increases, while the majority would be replaced by western coals, predominantly those from the Rockies.

The model used by OTA did not break down coal production changes in the detail done by ICF. Instead it reported changes in utility industry coal use by sulfur content classes, as if control programs on existing plants had been put into place in 1980 to produce the amount of emission reduction required in H.R. 3400. The effect of H.R. 3400, with its scrubber requirement, was estimated to result in a decline of 17.5 million tons in production of coals with greater than 3 pounds of SO_2/million Btu's, all in the East; an increase of 31.3 million tons in production of coals between 1.2 and 3 pounds, over 90 percent of it occurring in the East; and, for coals below 1.2 pounds, an increase of 16.3 million tons in the West partly balanced by a drop of 4.2 million tons in the East. OTA anticipates these shifts to be more than balanced by growth in coal demand from new plants, so that they should not push coal production in the 1990's below 1980 levels in any region.

Removing the scrubber requirement, the OTA model showed a further decline of 33.2 million tons in coals greater than 3 pounds of SO_2/million Btu's, again all in the East. This was balanced by an increase of an additional 26.3 million tons of 1.2- to 3-pound coal in the East (and a small decrease--3.3 million tons--in production of these coals in the West), and a further increase of coals under 1.2 pounds, by some 17.8 million tons in the West.

Comparing the ICF and OTA results, the main effect of an SO_2 emission reduction program of the 10-million-ton-size required in H.R. 3400 without the scrubber requirement would be a loss of 150 million tons of production in the eastern high-sulfur regions according to ICF's model, but a loss of only 50.7 million tons of eastern high-sulfur coal output, only one-third the amount, according to OTA's model. ICF's results in this case show definite drops in production in the two high-sulfur regions, even when the general increase of coal production is taken into account, while OTA says it anticipates that total high-sulfur coal production would probably drop from 1980 levels, but might be offset in part by demand from new utility plants.

In contrast, for H.R. 3400 including the scrubber requirement, OTA's model projection is a 17.5-million-ton drop in production of coals over 3 pounds, compared with no control program, while ICF's projection is only a net 6-million-ton drop of coal production in the two main high-sulfur coal regions. Taken together, then, ICF projects protection of 144 million tons of high-sulfur coal production by H.R. 3400's scrubber requirement, whereas OTA only sees 33.2 million tons, less than one-quarter as much, protected. In short, the model used by OTA projects a good deal less scrubbing under H.R. 3400 than does ICF's model. Furthermore, this comparison demonstrates that the two models must differ significantly in their assignment of cost differences between switching and scrubbing, since their least-cost approaches show so much of a disagreement in the amount of switching that would occur under an approximately 10-million-ton SO_2 emission reduction requirement.

Site-specific control studies--Is there a major switching-scrubbing trade-off?

Almost all of the cost and coal production analysis discussed up to this point has been modeling work, with the costs of each increment of each control technique specified by assumptions included in the model.

There is an alternative, site-specific, approach that could be used to examine control programs. Ideally, it would employ a description of each generating unit at every plant potentially affected by a control program, including the extent, feasibility, and cost of each control technique that could be used to control SO_2 emissions from each unit. To date, we do not know of any complete analysis of a SO_2 control program having been carried out by this method. However, partial approaches have been made, and their results are quite different from those of the modeling studies in two major ways:

--control cost estimates from site-specific studies are substantially higher than those from modeling studies and

--the site-specific studies we have examined show that much less switching would occur under a least-cost approach than do either of the modeling systems.

The site-specific studies we have reviewed are based on the results of individual utilities examining the control options they would employ to meet emission reduction requirements. Some caution is needed in considering the results of these studies, because many of these utilities could be greatly affected by the outcome of an acid deposition control program. On the other hand, these studies contain a wealth of information based on practical experience with and detailed analysis of the particular characteristics of individual generating units and, for that reason, merit consideration.

The first site-specific studies presented were analyses of the effects of S. 1706, the emission control bill introduced in the Senate in October 1981, done independently by several utilities and presented at a Senate Environment Committee briefing.[41] These showed a variety of control methods being used, but their main thrust was some substantially higher control costs than came from the modeling studies. One leading example, from the American Electric Power Company, reported $2 billion in annual costs, for a reduction of only about 1.3 million annual tons of SO_2 emissions.[42] Scaling this to an overall 10-million-ton reduction would indicate costs of about $15 billion, compared with the $4 billion to $5 billion in the modeling studies examined above.

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[41] Senate Committee on Environment and Public Works, Acid Rain: A Technical Inquiry, Serial No. 97-H53, Appendix, "A Briefing on the Costs of Controlling Acid Rain," June 2 and 3, 1982.

[42] J. Dowd, Sr. Vice-President, American Electric Power, in Senate report (previous note) pp. 672-687.

Discussion of this study in the briefing showed higher cost estimate assumptions for scrubber retrofitting than were used by most other utilities, an assumption that coal switching would be barred by states protecting local high-sulfur coal sources, and 31 percent of annual costs arising from building new capacity both to replace prematurely retired units and to replace capacity lost due to scrubber energy consumption and reliability penalties. While some of these items may be challengeable, the scrubber energy and reliability penalties are examples that had not been considered sufficiently in previous modeling studies. Other considerations brought to light by site-specific studies, such as the barriers to coal switching presented by mine-mouth plants and long-term coal contracts, have also contributed to recognition that major SO_2 emission reductions could be more complex and narrowly constrained, and therefore more expensive, than simpler studies had found.

Another result of the utility studies presented in the 1982 Senate Committee briefing which differed from modeling studies was a very low estimate of the amount of coal switching that would be used in a least-cost 10-million-ton SO_2 control program. Some utilities reported that they would scrub most of their larger units and switch only on smaller, less frequently used ones. A major reason expressed for this preference was that scrubber capital costs go up very sharply for smaller sized units. A 1981 study[43] done for EPA, cited in answer to briefing questions, estimated scrubber capital costs at $162/kw(e) for a 1,000-MW plant, rising somewhat--to $216/kw(e)--for a 250-MW plant, and rising much more sharply--to $324/kw(e)--for a 100-MW plant. Also, the addition of a scrubber to an infrequently used plant would result in very high costs per unit of electric output because the scrubber's high capital cost would not be spread over as large an amount of power as it would be for a base-load plant.

The individual utility studies presented at the briefing were only from a sampling of utilities and were not added up to a region-wide result. However, a preliminary study[44] based on individual utility analyses did do this. In contrast to ICF's "cost-effective equivalent" to H.R. 3400, which would switch away from some 150 million tons of high-sulfur coal, thus contributing the majority of a 10-million-ton SO_2 reduction, the preliminary study combining the utility analyses found that only 20 percent, or 2 million tons of SO_2 reductions, would come by switching in a least-cost, 10-million-ton control program.

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[43] J.G. Ball and W.R. Menzies, Flue Gas Desulfurization Cost Estimates for Acid Rain Studies: Technical Report, Radian Corp., Report No. DCN 81-203-001-12-23, Sept. 1981.

[44] J. Gruhl, Issues Relating to the Costing of Acid Rain Control Technologies, Gruhl Associates, P.O. Box 36524, Tucson, AZ. 85740, Apr. 1983.

This study also combined the various site-specific utility estimates of costs of each type of control method, to give preliminary "supply curves" for SO_2 control.[45] These curves would suggest that, while somewhat over a million tons of SO_2 control could come from switching at lower costs than scrubbing, at a point somewhere near 1.5 million tons of SO_2, the cost of switching would rise rapidly to levels well above the costs for equivalent increments of control by scrubbing. The preliminary nature of this finding, and the limited set of data it comes from, are not sufficient grounds on which to base a firm view on the switching-scrubbing mix. However, this does give notice that coal switching may have much less cost-effective SO_2 control to offer than ICF's modeling in particular would indicate.

A more comprehensive utility analysis was released in September 1983, based on results from an earlier survey of companies which would be responsible for about 44 percent of the emission reductions under the Senate Environment and Public Works 8 million ton SO_2 reduction bill, S. 768. The results of the survey[46] do not bear directly on a 10-million-ton SO_2 reduction, or on the specific matter of H.R. 3400 and the switching versus scrubbing choice, since they are based on S. 768. However, a follow-up analysis carried out by Temple Barker & Sloane, Inc. (TBS) for the Edison Electric Institute (EEI) has been focused directly on H.R. 3400 and this question.[47]

The TBS analysis examined the 50 highest emitting plants which would be targeted for scrubbers under Subpart I of H.R. 3400. It assumed that scrubbing would be applied on 80 units, amounting to 47.7 GW of capacity and responsible for 81 percent of total "top 50" emissions. It assumed that 111 other units (18.6 GW of "top 50" capacity) would be retired, mostly because of a combination of advancing age and small size, while the remaining 1.6 GW of capacity in the group is already being scrubbed. The scrubbers installed under this provision would provide 4.9 million tons of SO_2 emission reductions, while replacement of the output of the retired units was assumed to lower emissions by another 1.2 million tons.

This would leave 3.9 million tons of reductions to come from controlling other plants, under the state-administered excess

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[45] See fig. III-1 and pp. 174 to 177 in app. III.

[46] National Economic Research Associates, A Report on the Results From the Edison Electric Institute Study of the Impacts of the Senate Committee on Environment and Public Works Bill on Acid Rain Legislation (S.768), Prepared for EEI, June 30, 1983.

[47] Temple, Barker & Sloane, Inc., Evaluation of H.R. 3400, the "Sikorski/Waxman" Bill for Acid Rain Abatement, Briefing paper presented to Steering Committee of the Clean Air Act Issue Group, Sept. 20, 1983.

emissions formula in Subpart II of H.R. 3400. To provide this remaining reduction, the TBS study assumed the use of a combination of all three of the usual control methods. This included scrubbing an additional 30 GW of generating capacity, which gave 2.1 million tons of SO_2 reductions; switching for 35 GW of generating capacity, which reduced SO_2 emissions by 1.6 million tons; and washing of coal for an added 20 GW of capacity, which gave the remaining 0.2 million tons of SO_2 emission reduction.

Combining the two SO_2 control Subparts of the bill, TBS estimated that annual costs would be $7.3 billion, a good deal larger than the $4.2 billion to $5.3 billion estimated in OTA and ICF analyses of H.R. 3400 (all estimates in 1982 dollars). Much of this difference, as with the separate utility studies mentioned earlier, arises from the individual site-specific cost estimates being higher than the values used in modeling studies.

Even more striking than the high cost estimates in the TBS study, however, are the results of changing the mix of switching and scrubbing assumed in the study for the state-administered SO_2 controls under Subpart II of H.R. 3400. This was examined in sensitivity analyses in the study, which showed that overall costs would be decreased for a mix including more scrubbing, while costs would be increased for a mix that included more switching.

These results are not exactly comparable to the case of relaxing the scrubber requirement in Subpart I of H.R. 3400, as was done in ICF and OTA studies, because different sets of plants would be involved--plants from the "top 50" set in the ICF and OTA cases, as opposed to other plants in the TBS study. However, the general point does bear comparison--OTA and, even more particularly, ICF studies include many cases where switching is found a good deal more cost-effective than scrubbing, while the TBS study finds scrubbing, to a large extent, more cost-effective than switching.

We discussed the nature of and basis for the inputs to the TBS analysis with EEI staff, and found that a central reason for these divergent results lies in different assumptions about the future price differentials between low- and high-sulfur coals. In the TBS study, these differentials are assumed to show continuing growth through the coming decades. It was explained that this was anticipated because low-sulfur coal producers would avoid commitment to long-term, fixed-price contracts in the expectation that demand for their coal would grow progressively and allow continuing price escalation. While this reasoning may seem plausible, we do not believe that it alone could explain switching not being competitive with scrubbing, for the following reason. If, at current low-sulfur coal prices, switching could be done more cheaply at some plants than scrubbing, then some low-sulfur coal suppliers would see long-term contracts with those plants as being in their interest, rather than losing them as customers to scrubbers, since the plants would no longer be potential future customers once scrubbers were installed. Therefore, low-sulfur coals may not undergo unlimited price escalation.[48]

In summary, we have found a wide range of views on the extent to which switching to low-sulfur coal could be competitive with scrubbing for controlling SO_2 emissions from the older coal-fired power plants which predate the New Source Performance Standards of the Clean Air Act. If switching is cost-competitive for a significant fraction of these plants, then a major SO_2 emission reduction program could cause substantial economic harm in eastern high-sulfur, coal-mining areas. Available information, however, does not appear to be sufficient to determine whether there is, in fact, a major switching-scrubbing tradeoff.

Overview of SO_2 control costs

The studies we have reviewed agree that major amounts of SO_2 emissions can be controlled in the United States at incremental costs that would increase progressively more steeply at higher levels of control. Estimates of these costs vary, depending on the factors considered and assumptions made in the various control cost studies in the literature reviewed.

For a program involving a 10-million ton reduction of annual SO_2 emissions, estimates from econometric modeling studies fall as low as $2.8 billion (1982 dollars) in annualized costs allowing for emissions trading. We believe, however, that actual annualized costs are likely to be at least a few hundred million dollars greater than these estimates. This is because the use of both intrastate and interstate emissions trading is assumed to lower these annual costs by between $0.5 billion and $1 billion, whereas we have pointed out (see p. 93) that the interstate portion of this trading will be hampered by political and institutional difficulties. On the other hand, a compilation of site-specific studies indicates that annualized costs for a 10-million-ton SO_2 reduction will be near $7.3 billion, though somewhat lower if utilities used even more scrubbing and less switching than assumed in this study. The annualized costs of a 10-million-ton reduction, therefore, fall somewhere between over $3 billion and about $7 billion, a range of about a factor of 2.

This range could probably be narrowed somewhat by approaches that combine the methods used in both kinds of studies. However,

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[48] A recent CRS report (cited in footnote 29) points out that even if the cost of supplying low-sulfur coal could allow it to be provided at a saving compared with scrubbing, it is possible that the delivered price for utilities could result in the same control cost for switching as for scrubbing. This would occur if the mining and/or transportation sectors were to take the difference as an additional profit, rather than allowing it to be passed along to the utility and consumer as a saving.

experience with advance cost estimates leads us to doubt that they could be predicted to an accuracy as close as 25 percent; therefore, we are probably nearing the point of diminishing returns in cost studies.

Our review of these cost studies indicates that the most significant area of disagreement is the extent to which coal switching can compete cost effectively with scrubbing. The answer to this question hinges on the availability and delivered price of low-sulfur coals that are compatible with high-emitting power plant boilers. Because different studies show a wide range of projections of this supply curve, it is not clear where the correct version lies and, therefore, whether (or how much) coal production employment would be affected by a major SO_2 control program.

Future technology developments are possible that could lower control costs, but not dramatically and not very soon. The earliest that one of these developments could be proved and start to make a significant contribution to lowering SO_2 emissions now appears to be the mid-1990's, for LIMB. LIMB might lower costs per ton of SO_2 removed by 13 to 35 percent for the amount of control that it could achieve. However this would be, at maximum, about half as much as scrubbing, so its usefulness would diminish if control requirements grew much above 50 percent of total SO_2 emissions. Furthermore, knowledge of how large the SO_2 control potential of LIMB could actually be is still several years away. Its availability might be brought a few years earlier by support of early demonstration projects, but even in that event it is unlikely that LIMB would be in operation to any substantial extent before the early 1990's.

CONTROLLING NO_x EMISSIONS

As we noted earlier, NO_x emissions are expected to rise more sharply than SO_2 emissions over the balance of this century. Therefore, while advocates of acid deposition controls have focused most of their efforts on controlling SO_2 emissions, control of NO_x emissions has been receiving increased attention.

Table 2 (p. 36) showed that the two largest sources of manmade NO_x emissions in the eastern United States in 1980 were the transportation sector (44 percent) and the utility sector (34 percent). As shown in appendix I, utility NO_x emissions are expected to grow slowly through the end of the century, while transportation emissions are expected to fall in this decade and then rise in the 1990's. As a result, the projections for 2000 (table I-11, app. I) still show these two sectors dominating NO_x emissions, but the relative shares should be closer to equal, with transportation at 39.6 percent and utilities at 37.4 percent. In 1980, coal burning accounted for 78 percent of national utility NO_x emissions, and this share should increase as coal continues to take a larger role in electricity generation. Separate regional breakdowns of NO_x emissions by fuels are not available, but applying the national percentage to the East would indicate that coal-burning utilities emitted 27 percent of all eastern United States NO_x in 1980 and will emit at least 29 percent of eastern NO_x in 2000.

Utility NO_x emissions

The costs of controlling utility NO_x emissions are comparable, and in many cases lower, than the costs of controlling utility SO_2 emissions. Commercially available retrofittable methods, however, can only cost effectively control an average in the range of 20 to 30 percent of a utility's emissions.

In the United States and Canada, "combustion modification" is the most common method of NO_x control. This involves changing the rates and proportions at which combustion air and fuel are mixed, in order to reduce the amount of NO_x generated during the combustion process. This is usually accomplished by injecting the combustion air in stages or by using "low-NO_x burners." Not all utility boilers, however, can be retrofitted with combustion modification equipment.

The PEDCO study,[49] found that of the 50 largest utility NO_x emitters, 36 would be able to retrofit NO_x controls. PEDCO estimated total annualized costs at about $46 million for reducing NO_x emissions in these plants by 483,000 tons per year. These results show moderate levels of NO_x control to be relatively cost-effective--$95 per ton removed--compared with SO_2 controls at costs of hundreds of dollars per ton. An EPRI-sponsored review reported that combustion modifications can control NO_x emissions for as little as $60 per ton.[50]

Despite this relative cost advantage, two problems presently inhibit effective control of NO_x emissions from existing utilities. First, combustion modifications can only achieve average reductions of about 25 percent, and are not technologically or economically feasible in all utility boilers. Second, baseline data on current utility NO_x emission are very poor, making verification of reductions from current levels extremely difficult.

More effective NO_x control technologies are expected to become commercially available within the next decade. Second generation, low-NO_x coal burners, capable of achieving NO_x control of 50 percent or more when retrofitted to existing furnaces, are projected to become available within the next few years. Higher levels of control are possible with changes in furnace geometry, but this would involve major equipment replacements not likely in retrofits. Other techniques offering greater NO_x control, notably

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[49] PEDCO Environmental Inc., op. cit. (footnote 22).

[50] Decision Focus Inc., Acid Deposition: Decision Framework, EPRI Report No. EA-2540, Aug. 1982, p. 2-35.

flue gas treatment which could be retrofittable but would be substantially more capital-intensive, are likely to face much opposition and take longer to develop. LIMB, the technology discussed under SO_2 controls, could substantially reduce both SO_2 and NO_x emissions, but is not expected to be available until the 1990's.

Mobile source NO_x emissions

Table 7 gives 1982 U.S. NO_x emissions from each mobile source category. It shows that automobile emissions account for the largest single share of mobile source NO_x emissions, 35 percent, although the total emissions of other types of vehicles still contribute the majority of U.S. mobile source NO_x.

Table 7

U.S. Mobile Source NO_x Emissions, 1982
(million metric tonnes, as NO_2)

Highway vehicles		
Automobiles	3.4	
Heavy-duty trucks	3.3	
All other highway vehicles	1.2	
Total highway vehicles		7.8[a]
Off-highway vehicles		
Aircraft	0.1	
Railroads	0.7	
Vessels	0.2	
Other off-highway vehicles	0.9	
Total off-highway vehicles		1.9
Total		9.6[a]

[a] Totals differ because of independent rounding

Source: U.S. Environmental Protection Agency, National Air Pollutant Emission Estimates, 1940-1982, EPA-450/4-83-024, Feb. 1984.

Examination of trends in U.S. NO_x emission estimates from 1970 to 1982[51] shows that the leading four categories of emission sources in 1970, with their approximate percentages, were: utilities--25 percent, industrial combustion--22 percent, passenger cars--20 percent, and heavy duty highway vehicles--9 percent. In the 1970-82 interval, estimated utility NO_x emissions apparently peaked in 1980-81, after increasing by over 40 percent; industrial combustion NO_x emissions decreased about 30 percent, to rank fourth at 2.7 million metric tonnes after peaking in the early 1970's; and passenger car NO_x emission declined about one-fourth after peaking in 1973. But heavy duty highway vehicles almost doubled their NO_x emissions, advancing to third place, only

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[51] See same source reference as table 7.

about 0.1 million metric tonnes behind passenger cars. The increase in heavy duty highway vehicle emissions continued by about 3 percent, even in the recession year 1982, when no other category of NO_x emissions increased. This has virtually all been due to diesel powered trucks, whose NO_x emissions are estimated to have increased more than 150 percent since 1970 while NO_x emissions from gasoline-powered heavy trucks in 1982 were negligibly different from those at the start of the 1970's. This growth in truck NO_x emissions, however, is likely to be balanced, at least in this decade, by a downward trend in passenger car NO_x emissions which can be expected to continue through the 1980's under present policy. This is because the auto fleet, which takes over a decade to turn over, is in the process of being replaced by cars conforming to the lowered NO_x emission limit--one gram per mile of travel--that only went into effect in 1981.

LIMING

Liming is the use of limestone (calcium carbonate) or other alkaline materials to neutralize excess acids in lakes, streams, and soils. It has been successfully used in agriculture for years to maintain desired pH levels for soil. One need only modify the quantity of lime to offset the effects of acid deposition on these soils.

Liming differs from other control strategies in that it focuses on alleviating damage at the receptor (i.e., downwind areas), rather than controlling emissions at the source. The technique has benefits, but also has limitations. Its positive effects for acidified lakes and soils would be experienced immediately, while the benefits of reducing emissions would be more long-term and are still very hard to quantify. However, liming's capacity to significantly reduce possible future large-scale lake and forest impacts is limited, and it cannot always restore affected waters to their previous condition.

Research in Europe and North America suggests that liming can also mitigate acidification of forests and aquatic ecosystems--up to a point. Long-term Swedish research has documented positive effects on animal and plant life by increasing lake pH and alkalinity through liming.[52] Affected fish populations were able to successfully reproduce again, and the condition of other organisms in the food chain also improved. In addition, certain undesirable features associated with acidified waters, such as extensive sphagnum growth and surficial sediments, were eliminated. However, while acidity was corrected, other chemical conditions of lakes were not necessarily restored to their original quality.

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[52] Swedish Water and Air Pollution Institute, Liming of Acidified Lakes and Streams--Perspectives on Induced Physical/Chemical and Biological Changes, Apr. 1981, Stockholm, p. 1.

Ontario, Canada's Ministry of the Environment reported having successfully restored the pH of four acidified lakes near the province's Sudbury smelters to normal, at a cost of about $50 per acre. In the United States, liming projects in the Adirondacks have demonstrated improved pH and greater trout survival.

While these findings are encouraging, liming has limitations. First, its beneficial effects are temporary. Applications must be repeated at intervals of one to several years to maintain proper pH, as long as acid inputs continue. Furthermore, when liming is used to restore waters already damaged (as opposed to its use to provide enough alkalinity to prevent damage from occurring), it has been shown in some cases to have two biologically significant disadvantages.[53] One is that it can produce waters more vulnerable to acidification than they were originally, because they lack some of the acid-neutralizing capacity that had originally been present in the form of dissolved organic carbon compounds. The second disadvantage is that, in acidified waters where concentrations of dissolved aluminum or toxic heavy metals had been increased to harmful levels during acidification, these metals will not be removed from the solution immediately. And when they do precipitate out, they remain readily available to redissolve in case of reacidification. As a result, such waters can become harmful to organisms much more quickly upon reacidification than when they were first acidified.

The major problem, however, is the difficulty of liming wide geographic areas where acidification is a problem. While it is not a major task to lime a limited number of commercially important lakes, it would be much more difficult to deal with the number of lakes that may be affected by acid deposition.

Liming is also limited in its ability to reduce possible forest soil acidification, because of the vast number of unmanaged acres that would have to be treated. While Sweden has practiced forest liming on a limited scale, financial considerations limit wider application. According to an EPRI study of the Swedish experience,

> "From an economic standpoint, dollar returns per unit area for most forest products do not justify investing in large applications of agricultural chemicals, even

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[53] C.T. Driscoll, J.R. White, G.C. Schafran and J.R. Rendall, "$CaCO_3$ Neutralization of Acidified Surface Waters," _Journal of the Environmental Engineering Division_, Amer. Soc. of Civil Engineers, Vol. 108, No. EE6, pp. 1128-1145, Dec. 1982.

> when these applications cause appreciable increases in rates of growth. Forest liming is further complicated because the inaccessibility of forests makes application difficult."[54]

These findings support the claims of many scientists, industry representatives, and agency officials who assert that, while liming is useful for alleviating damage to crops and some lakes, it is not by itself a remedy for the large-scale and long-term problem that acid deposition may pose.

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[54] General Research Corporation, <u>Feasibility Study to Utilize Liming as a Technique to Mitigate Surface Water Acidification</u>, EPRI Report No. EA-2362, April 1982 p. 2-3.

CHAPTER 5

ACTIONS TO CONTROL ACID DEPOSITION:

GAO OBSERVATIONS ON ISSUES IN THE DEBATE

The previous three chapters have reviewed the available information on acid deposition in detail. This concluding chapter will briefly summarize the current state of knowledge about the subject, and then examine to what extent, and how, this knowledge can be used to guide policy decisions on the question of controlling acid deposition.

SUMMARY OF THE EXTENT AND LIMITS OF KNOWLEDGE ABOUT ACID DEPOSITION

The causes

Acid deposition in eastern North America is predominantly of man-made origin, arising from SO_2 and NO_x emissions. The United States produces the majority of these emissions, with eastern Canada contributing an estimated 17 percent of the SO_2 and 8 percent of the NO_x. SO_2 emissions are the source of the largest share of this deposition and pose the greatest risk of acidification today. Transport studies indicate that substantial contributions to deposition at almost all sites come from a range of distances up to 500 miles or more; however, precise assignments of the shares at any site that come from each source area are not yet available.

Scientific work released in 1983 has now confirmed that, for eastern North America as a whole, acid sulfur deposition will change in essentially a 1:1 proportion to changes of SO_2 emissions. Therefore, a uniform reduction in SO_2 emissions will result in a proportional uniform reduction of acid sulfur deposition. The proportionality, however, may differ from 1:1 for emission reductions targeted to control deposition in particular locations, depending on source-receptor relations which are known only imprecisely at this time.

Damage

There is substantial uncertainty in attempting to establish a quantitative relationship between acid deposition and damage. Some of the effects of acid deposition are documented adequately enough to indicate, at least qualitatively, that damage has already occurred--acidified lakes and streams in certain areas in northern Europe and eastern North America, materials damage in many urban areas, and acidification of some drinking water supplies. However, even the extent of this recognized damage is not clear since, for example, the share of materials damage attributable to acidic pollutants is not known.

Even wider uncertainty exists regarding the prospects for damage in the future. The threat of greatest concern is that future damage of recognized kinds, or other kinds of damage that are suspected but not conclusively proven, such as that to forests, could be found to be occurring in other regions and/or to a greater extent than has been recognized to date. As we noted in chapter 2, lakes and streams in a number of sections of the eastern United States, as well as in large areas of eastern Canada, either show evidence of some stress from acidity, or appear vulnerable to acidification. This indicates that greater and more widespread aquatic damage is possible, but leaves substantial uncertainty about whether it will occur and, if it does, its amount, location and timing. Similar uncertainties exist in other damage categories, in some cases to an even larger extent. One particular uncertainty is whether forest damage occurring in Europe is a foreshadowing of similar effects in the United States.

Control methods and costs

In contrast to the great uncertainty about damages, knowledge of control methods and their costs is relatively good. The leading source of SO_2 in the United States is electric utility fuel combustion. Utilities were responsible for 74 percent of eastern United States SO_2 emissions of about 22 million metric tonnes in 1980; almost all came from coal burning, and most of that came from older pre-NSPS plants emitting at high rates. These are logically the main targets for SO_2 emission reductions, with attention also merited to avoiding large increases in industrial SO_2 emissions anticipated in the 1990's. Sulfur dioxide emissions are expected to increase at least until 2000, and may not decrease until a decade or two into the next century, if proposals to extend the lives of older coal-fired utility plants are carried out without reducing their emissions.

There is not exact agreement among different studies of the costs for SO_2 emission reductions from coal-fired utilities. However, estimates of the cost for a 10-million-ton reduction lie within a range of about a factor of two, from somewhat over $3 billion to nearly $7 billion per year. It is agreed that the marginal costs of least-cost emission control strategies will increase progressively more steeply as the extent of emission reductions increases.

Three major types of SO_2 emission control methods are currently available--coal washing, switching to lower sulfur coals, and flue gas scrubbing. It is agreed that, using currently available technology, washing can only offer modest reductions, possibly up to about 1.5 million tons of SO_2. Scrubbing can remove 90 percent or more of the SO_2 from a plant and therefore could offer reductions of up to 8 million tons, and possibly more. It would, however, have the highest capital costs. No clear agreement has been reached on the availability of low-sulfur coals that are compatible with existing high-sulfur coal-burning plants at costs competitive with those for scrubbing. Hence, estimates of the extent of cost-effective SO_2 emission reductions available by switching range widely--from 1.5 million tons of SO_2 to over 6 million tons.

Utility NO_x emission control, at much lower cost per ton than for SO_2 control, is feasible now at levels up to about 20 or 30 percent reductions. However, deposition derived from NO_x is currently of much less concern as regards damage stemming from acid, although the expected continued growth of NO_x emissions may be a cause for greater concern in the next century.

LIMB, a new technology which could offer reductions of 50 percent or more of both SO_2 and NO_x emissions, may be available for retrofit application to coal-fired utility plants in the 1990's. LIMB is estimated to offer total costs, per ton of SO_2 controlled, in a range from about 13 to 35 percent lower than scrubbing, with much lower capital costs. Although its total scope of applicability will not be known for several years, research to date already indicates that it could work on at least 35 percent of older coal-fired utility plants. LIMB is not likely to be in widespread use before the mid-to-late 1990's, and even if development work on it is accelerated in the next few years, it is unlikely to be accepted widely by utilities before the early 1990's.

CAN COST-BENEFIT ANALYSIS BE USED TO HELP REACH A DECISION ON ACID DEPOSITION CONTROL?

Cost-benefit analysis, as discussed in our recent report,[1] is a method that can be applied to guide decisions on pollution control. It involves examining the costs and benefits of alternative control policies, to help identify the policy or range of policies which would result in the greatest net benefits. To use this method to help decide about acid deposition control, it would be necessary to have estimates of the costs and benefits of alternative control proposals.

The costs of acid deposition control would be just the costs of the actions required to prevent damage, either by treating areas and objects that are subject to acid deposition to protect them from damage, or by reducing the emissions which are responsible for acid deposition. To estimate these latter costs, it would be necessary to know which emissions are responsible for acid deposition, and it would be necessary to know that the responsible emissions could be controlled. If this is known, then the costs of any amount of acid deposition control can be estimated as the costs to reduce the particular emissions involved to the desired extent. The critical scientific information required to carry out

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[1] U.S. General Accounting Office, Cost-Benefit Analysis Can Be Useful in Assessing Environmental Regulations, Despite Limitations, GAO/RCED-84-62, Apr. 6, 1984.

this estimation would be the quantitative proportions between emissions and deposition, combined with information relating the sources of pollutants to the shares of deposition they are responsible for at each location--the source-receptor relationship.

The benefits of acid deposition control would be evaluated in this approach by estimating the extent of damage which would occur as a function of the amount of acid deposited, and counting as potential benefits the value of the damage avoided by any given amount of control action. The critical scientific information needed to evaluate the benefits of control would then be the quantitative relationship between amounts of acid deposition and amounts of damage--the dose-response relationship.

In short, an indication of the economically efficient range of control actions for acid deposition and an answer on whether to start control actions promptly or to wait for better information, could be given if scientific data were available which could be used to identify the responsible pollutants, estimate the proportionality between their emissions and the resulting acid deposition, and determine the source-receptor and dose-response relationships. As was noted in the previous sections, identification of the responsible pollutants, and determination of the proportionality between total emissions of these pollutants and the resulting acid deposition have largely been accomplished. Furthermore, while source-receptor relationships have not been determined precisely, they can be approximated to an extent that can offer some guidance on policy decisions. However, the uncertainty about the prospects of damage from acid deposition--the dose-response relationship--is so great that it prevents identifying any narrow range of estimates of the benefits of acid deposition control at this time. Therefore, cost-benefit analysis can give only a very imprecise indication of the efficient level of control actions.

HOW FAR DOES CURRENT KNOWLEDGE LEAD? GAO OBSERVATIONS ON ISSUES IN THE ACID DEPOSITION DEBATE

Current scientific and technical knowledge does not give either a strong "yes" or "no" to the question of whether any further emission control action would reduce damage enough to justify its control costs. Ongoing and planned or proposed research is likely to provide better knowledge on damages, but only after some years. At this time, scientists can predict neither what those results will be, nor what the probabilities are that they will indicate either that control actions will be worthwhile, or that the costs of control actions will not be justified by their results. Therefore, since there is not a scientific answer to the question, it is necessary to turn to a weighing of the relative risks of alternative decisions.

Frequent statements from both sides of the debate have pointed out significant risks in either course. As one example, available information does not specify the value of prevention or reduction of future damage that would come from any particular level of emission controls. These emission controls, however, would increase utility capital requirements and rates, and might adversely effect employment and the overall economies of particular regions. Thus, acting in the face of incomplete information would increase costs with the risk that there would be few or no benefits in the form of damage prevented.

On the other hand, since damage, or the probability of damage, tends to increase with the amount of harmful material delivered, greater damage could develop over time if emission reductions are not required. This risk is intensified by the fact that, under current policy, eastern U.S. SO_2 and NO_x emissions are projected to increase for the balance of the century, further increasing the rate of acid deposition and, therefore, the risk and extent of potential damage.

Some opponents of immediate regulation have argued that, by waiting until more information is developed about acid deposition's causes and effects, a more efficient control strategy could be designed. Proponents of regulation respond that this is doubtful--in light of emerging knowledge about transport--and that delay would still result in higher, potentially more damaging, deposition levels in the intervening years. Some scientists have argued for early partial reductions which, they have pointed out, would reduce damage rates or risks without risking the possible inefficiency of an excessive control program that aimed to prevent all potential or possible damage immediately.

Any decision on whether to enact controls now, or wait for better scientific information, must weigh the risks of economic and other dislocations that may result from taking additional control actions now (without precise estimates of environmental and possible health benefits), versus the risks of further avoidable environmental and possible health impacts that could occur if additional control actions are not taken. This choice involves an allocation of the burden of risk among industry, consumer, labor, environmental, and other interests and between different regions of North America. It is, therefore, more than a scientific and economic decision. It involves political judgments on how our society's resources--and burdens--should be spread among these constituencies.

To aid the Congress in examining the issues involved in the acid deposition debate, we are providing the following observations, on the basis of our analysis of the scientific and economic information on the acid deposition problem.

Observations on damage stemming from acid deposition:

1. Limited damage caused by transported acid deposition has been confirmed in only a few areas of North America to date. However, the major reason for concern is the possibility of much greater damage which may occur in the same or other regions, and be of the same or other types, as that which has already been confirmed.

Recognized ecosystem damage clearly attributable to acid deposition in North America has been limited to lost fish populations in about 180 of the lakes at high altitudes in the Adirondacks in New York, a few dozen lakes in a small area of Ontario, and 10 salmon rivers in Nova Scotia. Other recognized damage includes a share--not at all well quantified--of materials damage, largely in urban areas, and acidification of some drinking water supplies--again with the scale of impact not well quantified. A larger number of lakes appears to have been somewhat acidified, but not yet seriously damaged biologically. However, the greatest concern is that ecosystem damage is largely cumulative, hence, there is a possibility of future damage of greater intensity in more areas and of more types.

Potentially vulnerable lakes and streams in the United States, besides those in New York, are found in New England, some parts of the Appalachian Mountains, the Southeast, upper Midwest, and possibly also in mountain areas in the West. In Canada, areas considered vulnerable are found in the Maritime Provinces and in much of Ontario and Quebec. Hundreds of thousands of lakes on poor soils in each of the latter two provinces, of which many thousands are classed as vulnerable, make lake damage a major concern to Canada.

Evidence has been found this year that growth rates have been slowed in the past 1 to 3 decades for several species of trees in forests in many parts of the eastern United States. In addition, deaths of trees have been occurring for several years among a smaller number of species in a few high-altitude areas. Since scientists have found that usual natural causes alone do not explain these tree declines and deaths, air pollutants are likely involved. However, acid deposition is only one of several possible contributing pollutants. While it is not yet clear how, or even whether, acid deposition contributes to this problem, there is a possibility of substantial damage to forests, which are an economically important resource in much of eastern North America. Thus, this has become a leading concern regarding acid deposition, particularly in light of a possible parallel to widespread and rapidly increasing damage to forests in West Germany.

2. Uncertainties about anticipated damage rates, more than any other of the scientific uncertainties about acid deposition, make it difficult to predict the need for and effectiveness of control actions.

Fundamental uncertainties exist about whether damage caused by acid deposition is occurring or will occur to some categories of resources (e.g., forests, crops, human health) and about how much damage will occur at present and anticipated deposition rates in all categories. Quantifying these effects will be difficult until better data exist on how much damage is caused by a given amount of acid deposition (i.e., the "dose-response" relationship). For some types of ecosystems, for example, available scientific information does not show if decreased deposition would cause damage to decrease proportionately, or if a threshold level exists instead--below which no effects will occur and above which effects increase with increasing deposition. For aquatic ecosystems, where a threshold does apparently exist, scientists are uncertain about its exact level, and about how much damage would occur at deposition rates between the threshold and the current rate.

Scientists generally indicate that, where damage is occurring or is likely to occur, any decrease in deposition will decrease the damage or the risk of damage, and an increase in deposition will increase the damage or risk. However, there is not yet evidence about which scientists have reached agreement that indicates what the timing and level of damage will be under present deposition rates. Therefore, it is not possible at this time to say quantitatively how much damage a given reduction in deposition would avoid, or how much more damage any particular increase in deposition would cause, until better dose-response information is available.

Observations on the causes of acid deposition and damage:

3. The substances from which acid deposition is produced, in and near major industrial and populated areas such as eastern North America and western and central Europe, are predominantly the man-made pollutants SO_2 and NO_x.

Research reported several years ago already had established that man-made contributions of sulfur compounds to the atmosphere over eastern North America were at least 10 times greater than natural emissions. While earlier reports had suggested that natural contributions of NO_x might be comparable to the man-made ones, recent examination has shown that older analyses had substantially overestimated NO_x emissions from natural sources. When this overestimation is corrected, the concentrated man-made NO_x contributions in areas with major industry and vehicle use again dominate natural contributions.

Eastern North American SO_2 and NO_x emissions come mostly from the United States, with eastern Canada producing only an estimated 17 percent of the SO_2 and 8 percent of the NO_x. About three-quarters of eastern U.S. SO_2 is emitted by electric utility fuel combustion, predominantly from coal and mostly from plants that pre-date the stringent new source performance standards. Industrial fuel consumption is a distant second as a U.S. SO_2 source, but is expected to grow substantially in the 1990's, largely because of expanded use of coal. Over half of eastern Canada's SO_2

comes from nonferrous metal smelters, while utility and industrial combustion together contribute almost two-thirds of the balance.

The leading eastern U.S. NO_x source is transportation, with electric utilities coming in second--but expected to nearly equal transportation by the end of the century. Industrial combustion ranks third. In eastern Canada, transportation produces about 62 percent of all NO_x--a share expected to rise to 80 percent by 2000. Most of the remainder of eastern Canadian NO_x comes from utility and industrial fuel combustion.

4. In eastern North America, sulfur compounds, predominantly emitted as SO_2, are the main damaging acidic pollutant.

About two-thirds of the acidic material deposited in the East is in the form of sulfate, or SO_2 which is converted to sulfate shortly after deposition; most of the remainder is in the form of nitrate or materials that convert to nitrate. These are the acidic forms of sulfur and nitrogen, which derive, respectively, from SO_2 and NO_x emissions. The relative proportions of sulfate and nitrate deposition have been changing in the past 2 to 3 decades, with nitrate's share increasing but still not anticipated to become predominant in this century. Western deposition shows higher proportions of nitrate--in some cases a majority--roughly corresponding to the greater relative levels of NO_x compared with SO_2 in emissions there.

Input-output balances on watersheds usually show that sulfate passes through unaltered to contribute acidity to the waters. In contrast, the majority of nitrate does not pass through, but instead reacts with plants in the watershed, resulting in the release of base that neutralizes the acid accompanying the nitrate.

Thus, with nitrate only about one-third of deposited acid, and a majority of that neutralized by living plants, it is sulfur compounds that provide over four-fifths of all acidifying deposition on ecosystems. In winter, when many plants are dormant, less neutralization of nitrate takes place so it can accumulate in the snowpack. Because of this, nitrate contributes to the "acid shock" which can occur at spring snowmelts, even though it does not contribute to progressive long-term acidification. Over the course of a year, however, sulfuric acid contributes much more to total ecosystem acidification than nitric acid in the East.

5. Precise estimates of the shares of sulfur compounds deposited at any particular location which are contributed by individual sources or source areas are not yet available. However, approximate estimates indicate that most areas, except possibly those within a few tens of miles of large sources, receive partial contributions to their deposition from many sources spread over large areas up to hundreds of miles away, with the majority coming from within about 500 miles in the average upwind direction, and from shorter distances in other directions.

Because emitted gases remain aloft for several days on the average, they will be distributed widely by air movements that vary over time. Consequently, they will be deposited over a broad area so that deposition at any location, averaged over time, will be made up of overlapping contributions from a wide range of sources extending up to hundreds of miles away, particularly along prevailing wind directions.

Current transport models still differ so much that they cannot provide accurate information on the sources of acid deposition in each receptor area. However, they are generally consistent in their qualitative features which show that, while each of 11 eastern North American receptor areas gets significant deposition contributed by sources in its own region, most receive the majority of their deposition from the combined emissions of other regions. Contributions decrease at longer distances but remain at levels in the range of one-tenth to one-third at distances up to 500 miles or more along prevailing wind directions. Only at locations that are very close to large sources (within a few tens of miles) can a single source or small group of sources dominate deposition.

<u>Observations on techniques for controlling the effects of acid deposition</u>:

6. <u>Mitigation actions taken where deposition occurs, such as liming of lakes, can prevent damage in some cases. However, they have limited capabilities both because they cannot control all kinds of damage, and also because they could not be applied economically to large unmanaged areas such as forests</u>.

Liming can be used to prevent pH decline and biological damage in lakes and streams, when applied either directly to the waters or to the surrounding watershed, and can also protect agricultural and forest lands. If the prospect of further damage in North America remains limited to aquatic effects on a scale similar to, or not much larger than, the few hundred lakes already damaged, then liming could be an adequate response, recognizing that repeated applications would be necessary at intervals of several years.

On the other hand, if acid deposition is found to have a significant share of responsibility for damage which may be developing in forests on a wide scale, and/or if acid damage is recognized to be approaching in the tens of thousands of lakes classified as extremely or moderately sensitive, then the scale of liming called for could be too large to be feasible. Also, it must be recognized that liming of forest and agricultural land would be capable of dealing only with types of damage that are caused by deposited acid acting in or through the soil, but not with damages that may be caused directly by acid deposited on above-ground parts of plants. Liming is therefore limited to dealing with only some types of potential damage, and on scales not greatly in excess of the levels of damage already observed here.

7. If deposition reduction is desired, to control the risk of damage stemming from acid deposition, the greatest reduction in risk would come from lessening the deposition of acidic sulfur compounds, which could be accomplished best by reducing SO_2 emissions.

Research findings indicate that, if deposition reductions are undertaken in eastern North America to reduce acid deposition, a greater emphasis should be put on controlling deposition of sulfur, as opposed to nitrogen acids. This is both because the sulfur acids predominate in deposition, and because the nitrogen acids tend to be less harmful to biological systems. This might require reexamination, however, if the proportion of nitrogen to sulfur deposition were to grow to make the nitrogen compounds dominant, either because of increases in nitrogen deposition greater than those anticipated in this century, or because of substantial reductions in sulfur deposition.

The evidence also indicates that the most effective type of control measure to accomplish this aim would be to reduce SO_2 emissions, which would result in essentially proportional reductions of acidifying sulfur deposition. Reductions of reactive hydrocarbon or NO_x emissions might also lower sulfur deposition, but not as much as would direct controls on SO_2 emissions, largely because levels of these other emissions would not substantially affect the extent of dry deposition of SO_2.

This responsiveness of deposition to emissions indicates that a sizeable SO_2 emission reduction would be an effective way to reduce the risks associated with sulfur deposition. However, as noted in item 2 above, existing data cannot accurately quantify the expected benefits from a particular deposition reduction.

8. Because deposition at almost any location includes significant contributions from sources spread over a wide area, emission controls intended to produce substantial reductions of acid deposition, even at one location, would be needed over a wide area rather than at one source or a narrowly localized set of sources.

While the sources that contribute to acid deposition at any location cannot yet be determined with high accuracy, existing knowledge shows that it will receive significant contributions from sources up to 500 miles or more in the direction from which prevailing winds come, except for cases where a major source may make the dominant contribution at receptors located within a few tens of miles of the source. When account is taken of the broad geographic range of areas believed to be vulnerable to acidification--from the southern Appalachians through New England as well as the upper Midwest and across much of southeastern Canada--emission reductions over essentially all of the eastern half of the continent would be required to lower deposition, and thus the probability of damage, in all areas now suspected to be risk.

Further research, if it were to give more accurate dose-response information, might narrow the list of areas where deposition reductions would be required to lessen the risk of damage, and thereby somewhat reduce the scope of the region where emission reductions were needed. However, the wide distribution of emissions from each source will preclude narrowing of the areas where emission reductions are needed to sizes much smaller than the 500 miles or more observed in known transport situations, unless it is shown that only a very few areas are at risk of damage from anticipated levels of deposition.

Observations on attempts to determine the proper level of control actions:

9. While cost/benefit analysis can often aid in identifying a range of economically efficient pollution control policies, current scientific uncertainty about the value of the benefits expected from proposed levels of acid deposition control is so great that cost/benefit analysis is of limited value in deciding whether additional controls on SO_2 emissions would or would not have benefits that justified their costs.

Advocates on both sides of the debate have suggested that a comparison of the benefits and costs of control actions should be used in deciding whether and what type of action to take. However, while cost ranges can be estimated for control expenditures to within about a factor of 2, major uncertainties exist (1) over whether damage caused or contributed to by acid deposition is occurring or will occur in forests and (2) over the extent of damages to be expected in forests and other categories such as freshwater aquatic ecosystems, water supplies, and man-made materials. As a result, there is a great deal of uncertainty about what the benefits would be of controlling acid deposition. This also extends to other benefits that would occur if SO_2 emissions were decreased. These include improvements of visibility, since sulfate particulates derived from SO_2 emissions are a major cause of visibility impairment, and also a lessening of materials damage in and near areas where SO_2 is emitted, since local deposition is responsible for most materials damage and a reduction of SO_2 emissions would decrease this damage.

In addition, a comparison of benefits and costs should account for the distribution of these impacts across geographical areas. Such an analysis cannot be done at this time, however, for two reasons. One is the extreme uncertainty about the value of benefits and about their distribution. The second is because of uncertainty about the extent to which coal switching would be used as a control technique, which means that there is also uncertainty about the distribution of control costs.

10. Marginal costs of emission reductions increase at greater levels of reduction, so that if reductions were chosen seeking to eliminate damage completely, the last increments of reduction would be very costly compared with the last increments of damage prevented.

The studies we have reviewed all indicate that, as higher SO_2 emission reductions are sought from sources in a region, the marginal cost per ton of SO_2 removed (i.e., the incremental cost of removing each succeeding ton of SO_2) increases. The reason is that, as the less expensive emission reduction techniques become exhausted, sources must resort to increasingly expensive methods. At the same time, the varying susceptibility of different receptors to damage will mean that, at increasing levels of emission reductions, the rate and extent of further damage will be progressively decreased. This decrease will come, however, at an increasingly greater cost of control actions. With the knowledge that successive levels of controls will have greater marginal costs, a point would come where prevention of the next increment of damage would cost more than the value of the resource protected. While the quality of damage information to date cannot specify where this crossover point would occur, it would be at an emission level where some damage is still occurring. Preliminary applications of a number of approaches to estimating a zero-damage deposition level for aquatic ecosystems in eastern North America, for example, have led to suggestions of acid sulfur deposition reductions in the range of 50 to 75 percent to protect the most sensitive waters. Emission reductions aimed to achieve this level of deposition reduction would result in marginal costs for controls which would be very high compared with the marginal costs of the damage they prevented.

11. <u>The control method of switching to low-sulfur coal may or may not offer lower overall costs. However, it would have indirect costs, in the form of employment shifts between regions, which would disproportionately effect limited areas where high-sulfur coal is mined.</u>

As was pointed out in chapter 4, widespread switching to low-sulfur coals would have adverse effects on coal-mining employment and regional economies in the high-sulfur coal-mining areas in northern Appalachia and the Illinois Basin. If a great deal of switching were possible at lower cost than scrubbing, then a side effect of a national least-cost control policy would be to disproportionately focus a part of the cost heavily on these two regions. This possibility has led to suggestions that compensation of some kind be paid, to avoid an extremely uneven distribution of the burden of emission reductions costs. Under this approach, the utilities achieving a cost reduction would redistribute a portion of their savings to the affected parties in the high-sulfur coal-producing regions. However, studies of the availability and cost of switching, as compared with scrubbing, leave wide uncertainty about how much switching would be included in a least-cost strategy. It is not clear, therefore, whether this concern is real or only hypothetical.

12. <u>Agreement on an approach to the acid deposition problem is likely to be aided by separating the question of when and in which areas of the country control actions should occur, from the question of how the control actions are to be financed.</u>

The equity arguments in the acid deposition debate are probably the most difficult to deal with, since they focus on the uneven distribution of damages and the costs and impacts of proposed control actions. In many control proposals, it has been assumed that the costs of any control or mitigation actions would be borne by the parties required to reduce emissions, or by the states or localities where mitigation was undertaken.

A number of proposals have been made which offer alternatives that might help to resolve these issues. One example was a bill introduced in the Senate in 1982 (S. 2594, 97th Cong.) under which emission control costs at utilities would have been paid for from a trust fund gathered from equal payments on all the electricity generated in an established acid deposition impact region. Other possible methods of paying for acid deposition control, proposed in the 98th Congress, start with a study of a trust fund called for in S. 768 and include several alternative revenue sources for such a trust fund:

--fees on fossil-fueled electricity (S. 2215),[2]

--fees on non-nuclear electricity (H.R. 3400), and

--SO_2 and NO_x emission fees (S. 2001).

In addition, proposals for federal expenditures for damage mitigation activities are included in S. 454, S. 766, H.R. 1405 and H.R. 3904.

General observations on the acid deposition issue:

13. Because the Clean Air Act currently focuses on concentrations of pollutants near their sources, any air pollution control approach to deal with acid deposition in this century would necessitate additions to, or a basic reorientation of, the ambient air quality standard approach in the present act.

Current clean air law focuses on local air quality problems for several pollutants, two of which are SO_2 and NO_x. The levels of air quality required to protect public health (and, in some cases, welfare) are set forth as National Ambient Air Quality Standards (NAAQS). The NAAQS focus on the concentration of these pollutants in the air at or near ground level, within specified geographical areas. They do not set standards for the total amount of these pollutants emitted in a region if enough of the material is dispersed away from the local areas, for example, by means of tall smokestacks. It is the total amount of the pollutants emitted, however, that is of primary concern in dealing with

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[2] Analyzed in L.B. Parker, Distributing Acid Rain Mitigation Costs: Analysis of a Three-Mil User Fee on Fossil Fuel Electricity Generation, Congressional Research Service, Library of Congress, Apr. 11, 1983.

acid deposition. Any acid deposition control measure which seeks to alleviate the problem through emission controls would have to focus on reducing total regional emissions, rather than improving local ambient air quality.

New source performance standards do limit emissions directly, so they can be expected to begin to lower total regional emissions as older facilities, particularly coal-fired power plants, are retired. However, these retirements are not expected to lower total emissions until after the turn of the century, or possibly 10 or 20 years later if proposals to extend these plants' lifetimes are carried out.

14. The dispute persists over whether it would be advisable to establish emission controls promptly to reduce acid deposition or to wait further. However, at a minimum, having control plans ready could save time, and therefore spare resources, if/when a need for rapid action becomes evident.

On one hand, many of the potential ecological effects of acid deposition are cumulative, and do not develop until after some years of deposition, so that delaying control action increases the risk of the damage point being reached in vulnerable ecosystems. This potential risk would be enhanced by increases in U.S. SO_2 and NO_x emissions that are projected during the balance of this century under current regulations, and which can be expected to lead to increases in acid deposition. Also, it would take some years after passage of any control legislation to develop specific emission reduction plans and implement them. For example, the emission controls required in three bills proposed in the 98th Congress--H.R. 3400, S. 2001, and S. 2215--would not be fully in effect until about 10 years after passage. These points argue for early action, to speed emissions reductions and thus lessen the risk of further damage, and further emphasize that waiting allows a greater risk of damage than prompt action would.

On the other hand, electric utility SO_2 emissions are projected to decline starting at the end of the century, since older high-emitting power plants are anticipated to be retired in increasing numbers.[3] Time used to reach more accurate assessments of the risk and rate of future damage might be well spent, if the assessments yielded results showing that major damage was not expected over the next 2 decades, and the nation was therefore spared from making unnecessary or misapplied control expenditures. Even a few years' time might be useful to allow further development of LIMB, an emission control technique which appears likely to offer somewhat lower control costs.

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[3] This time may be extended by a decade, or even longer, by proposals to stretch out the lives of existing power plants. (See pp. 143-144.)

A number of acid deposition control programs have been proposed which take account of both of these views, seeking to "buy" time in different ways. The National Governor's Association and the American Public Power Association have each recommended a two-stage emission reduction program. The first reductions, in the range of 5 million tons of SO_2, would be made promptly, with a second stage to be considered several years later, when more detailed information would be available. Among proposals widely reported to be under consideration in EPA's 1983 option analysis were ones which would have an emission reduction program covering areas of various sizes between 4 and 12 states, with total reductions as small as 2 million tons. In addition to reducing the risk of damage, an explicitly stated purpose of such a plan was to serve as an experimental demonstration of the actual effect on deposition of emission reductions. At a minimum, it has often been proposed that early analysis and planning of control strategies and actions be carried out, in order to have plans ready in the event it is learned that prompt control action is required.

15. Further scientific work on acid deposition will be needed for a number of years, no matter what decisions are made on control actions in the short run.

Those opposing action now ask for further research before making control decisions, while those favoring early action point to the need to monitor the results of any control program, to determine its effects and to learn whether further action may be needed.

Our discussions with scientists about the uncertainties, particularly on damage questions, tend to indicate that relatively long-term research is likely to be needed to answer the most significant of these questions--at least several years on aquatic systems; 5 or more years on agricultural effects, and likely a decade or longer on forest effects because of the long lives of the plants involved, the complexity of forest ecosystems, and the need for long-term field experiments to supplement field surveys and simpler experimental studies. Similarly, a few years will be required to develop techniques for measuring or estimating dry deposition, followed by several years of monitoring to obtain good data on this sizeable share of deposited acid. Only after experimental studies of the relationship between deposition and damage give reasonably quantitative dose-response data, and long-term measurements are made of deposition and its associated effects to confirm these experimental results, will it be possible to make final, as opposed to interim, decisions on acid deposition controls.

MAN-MADE SO_2 AND NO_x EMISSIONS IN NORTH AMERICA: PRESENT, PAST, AND PROJECTED

CURRENT EMISSIONS

Man-made SO_2 and NO_x emissions in the United States and Canada were estimated for 1980 by personnel from the two countries' governments, for the final report by one of the joint work groups operating under the 1980 U.S.-Canada Memorandum of Intent on Transboundary Air Pollution.[1] These estimates are presented in tables 1 and 2 of chapter 3. They have also been retained as the basis for much of our analysis, because they represent the most recent estimates done in agreed fashion for both countries for the same period.

Emissions since 1980 in the United States

Emission estimates more recent than those for 1980 are available for the United States. They are prepared by EPA and issued annually in a continuing series of publications.[2] The estimates for 1980, 1981, and 1982 from the most recent EPA publication--the one covering 1940-82--are presented in table I-1. Looking first at the figures for the United States in 1980, the two sets of estimates--those from EPA in this table and those from the MOI work group in table 1 of chapter 3--show the same rankings of individual emitting sectors for each pollutant, and roughly similar magnitudes for total emissions and for contributions from individual sectors. However, there are differences between the two sets of estimates both in total emissions and in the absolute amounts of each pollutant estimated to come from each of the major sectors. The differences between sector estimates range from 10 to 15 percent, with the exception of the dominant SO_2-emitting sector, electric utilities, for which the two sets of estimates agree within 2 percent. The differences between the EPA and the MOI work group sector estimates are not all in the same direction, however, with the result that the estimated totals do not differ as much as the sector estimates: the two values for 1980 total U.S. NO_x emissions differ by 7 percent, while the values for total 1980 U.S. SO_2 emissions differ by about half as much--3.4 percent.

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[1] Work Group 3B, Emissions, Costs and Engineering Assessment, Report No. 3B-Final, June 15, 1982.

[2] Office of Air Quality Planning and Standards, EPA, National Air Pollutant Emission Estimates, 1940-1980, EPA-450/4-82-001, Jan. 1982; 1970-1981, EPA-450/4-82-012, Sept. 1982; and, most recently National Air Pollutants Emission Estimates, 1940-1982, EPA-450/4-83-024, Feb. 1984.

Table I-1

Estimates of Man-Made Emissions of SO_2 and NO_x in the United States, 1980-1982

Emitting sector	SO_2			NO_x (as NO_2)		
	1980	1981	1982	1980	1981	1982
	(millions of metric tonnes)			(millions of metric tonnes)		
Electric utilities	15.5	14.7	14.3	6.4	6.5	6.2
Industry	4.8	4.6	4.3	3.7	3.7	3.3
Transportation	0.9	0.9	0.9	9.6	9.7	9.7
Nonferrous smelters	1.2	1.5	1.1	-	-	-
Residential and commercial	0.9	0.8	0.8	0.7	0.7	0.7
Other	-	-	-	0.3	0.3	0.3
Total	23.3	22.5	21.4	20.7	20.9	20.2
Combustion emissions by fuel						
Coal	15.7	15.3	15.0	5.6	5.7	5.7
Oil[a]	2.9	2.4	2.2	0.9	0.8	0.6
Transportation[b]	0.9	0.9	0.9	9.5	9.8	9.6
Natural gas	-	-	-	3.4	3.3	3.0
Other	0.1	0.1	0.1	0.2	0.2	0.2
Total	19.6	18.7	18.2	19.6	19.7	19.1

[a] Excluding transportation sources

[b] Essentially all from oil.

Source: National Air Pollutant Emission Estimates, 1940-82 (see footnote 2.), tables 8, 9, 13, 14, 18, 19, and 23.

These differences of estimates can be taken as an indication of the level of accuracy currently attainable in emissions data. As a confirmation of this view, the summary of each of the annual compilations of EPA emissions estimates points out that

> "Since these data are estimates only and do not represent the results of any program for the measurement of actual emissions, their accuracy is somewhat limited."[3]

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[3] Ibid (summary, p. 1).

The summaries also explain that the objective of compiling the data is as an indication of recent emission trends, and cautions against comparing data in the present publication with data from any of the preceding editions. We have observed this caution throughout our emissions comparisons, by only comparing trends within individual sets of data that were prepared in a consistent fashion. Thus, the emission estimates in table I-1, which we will examine to consider current trends in man-made U.S. emissions, come entirely from the most recent EPA report, that covering 1940-82.[4]

Estimates made over the 2-year interval--1980-82--show noticeable decreases of SO_2 emissions each year, together totalling about an 8-percent decrease, and a small increase of NO_x the first year, followed by a decrease the second year which more than offsets the first year increase, netting out to a decrease of somewhat over 2 percent for the 2 years. Looking first at the breakdown of combustion emissions by fuels, in the lower section of the table, the largest percentage changes occurred in emissions from stationary source combustion of oil--a decrease of 24 percent in SO_2 emissions and 33 percent in NO_x emissions. These are associated with a steep decrease in the use of residual oil (the oil-based fuel which provides the major share of all sulfur in petroleum fuels), which fell 17 percent from 1980 to 1981 and 18 percent from 1981 to 1982, and then continued with a further 18-percent decrease from 1982 to 1983, in a trend that has started with the sharp oil price increases of 1978-79[5]. The major share of these oil-based emission declines, for both pollutants, were from utilities, although there were also decreases in NO_x and SO_2 emissions from other stationary sources. With regard to NO_x emissions, the balance of the 1980-82 decrease occurred in natural gas combustion, and was of roughly the same magnitude as the 9-percent decrease in total gas consumption in the United States over the interval.[6]

While NO_x emission decreases from 1980 to 1982 all came in oil and gas combustion, the decrease in SO_2 emissions came about as much from coal combustion as from oil combustion, with a decline of about 0.9 million tons, or 6 percent, in utility coal-burning emissions partly balanced by an approximately 0.2-million-ton increase in SO_2 emissions from coal burning in other sectors. The decline in coal-fired utility SO_2 emissions from 1980-82 is surprising at first glance, since utility coal consumption was

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[4] See footnote in table I-1.

[5] Energy Information Administration, Monthly Energy Review, DOE/EIA-0035 (82/12[3]), Dec. 1983 [3], p. 45.

[6] Ibid, p. 54.

slightly higher in 1982 than in 1980[7]. On these grounds, if no other changes had occurred, coal-fired utility SO_2 emissions would have been expected to rise rather than fall, and, in fact, a slight rise occurred in NO_x emissions from coal burning over the 1980-82 interval, which is more consistent with the increase of coal use than is the decline in coal-based SO_2.

Closer examination of coal use data, however, shows that the 1980-82 interval saw a decrease of about 13 million tons in utility use of eastern coals, which are generally higher in sulfur content, and a slightly larger increase, about 21 million tons, in utility consumption of the lower sulfur western coals.[8] This offers a very plausible explanation of coal-fired utility SO_2 emissions in 1982 being lower than in 1980, since displacement of eastern coal by western coal would be expected to lower the total sulfur content of the coal used and therefore to lower SO_2 emissions. This is, indeed, confirmed by the fact that utility coal burned in the United States in 1982 had an average sulfur content of 1.5 percent, compared with the 1.6 percent values for 1980.[9] With this understanding of current SO_2 emission trends, then, we can anticipate that the decline of coal-fired utility SO_2 emissions will have stopped, and an increase will have occurred for 1983, since coal consumption data for that year indicate that utility use of eastern as well as western coal increased about 5 percent compared with 1982.[10]

In summary, current trends in U.S. SO_2 and NO_x emissions have seen about an 8-percent decrease in SO_2 emissions and a net 2-percent decrease of NO_x emissions from 1980 to 1982. For 1983 there can be expected an approximate stabilization of SO_2 emissions, with some increases from increased coal combustion about matching a decrease from further reduction of residual oil combustion, and very little change in NO_x emissions.

PAST TRENDS IN EMISSIONS

Past trends in man-made emissions in North America have been estimated directly only since the early 1970's. Estimates for

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[7] Energy Information Administration, Electric Power Annual, 1982 DOE/EIA-0348 (82), p. 79.

[8] Energy Information Administration, Cost and Quality of Fuels for Eletric Utility Plants: 1980 Annual and 1982 Annual, DOE/EIA 0191 (80) and DOE/EIA-0191 (82) table 53 in each volume.

[9] Ibid.

[10] Energy Information Administration Electric Power Annual, 1983, DOE/EIA-0348 (83) July 1984, p. 49.

earlier years have been reconstructed from industrial activity and fuel use data and therefore are regarded as less accurate, because many factors influencing emissions are less well characterized for those earlier times.

Table I-2

Estimates of Historical Emissions of SO_2 and NO_x for North America

Year	United States[a] SO_2 (million metric tonnes)	United States[a] NO_x[b] (million metric tonnes)	Canada SO_2 (million metric tonnes)	Canada NO_x[b] (million metric tonnes)
1940	17.4	6.5	-	-
1950	19.6	9.3	-	-
1955	-	-	4.54	0.63
1960	19.2	12.7	-	-
1965	-	-	6.59	0.85
1970	27.9	18.5	-	-
1973[c]	28.4	20.4	-	-
1974	27.0	20.1	-	-
1976	26.4	20.9	5.31	1.86
1978	24.8	21.5	-	-
1980	23.7	20.7	4.77	1.83

[a] These estimates for the United States were made using internally consistent methods but did not use the same approaches as, and so are not meant to be compared with, the estimates in the other tables of emissions estimates in this report.

[b] Counted as NO_2.

[c] 1973 was the peak year for U.S. SO_2 emissions in the data we obtained.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B Final, June 15, 1982, tables A.2.3, A.2.4, B.1.1 and B.1.5.

National trends

National estimates made by these methods for the United States and Canada are given in table I-2. In both countries, SO_2 emissions rose from the earliest date shown, reached a high (1965 in Canada and 1973 in the U.S.), and have declined somewhat since the high; NO_x emissions increased steadily until 1976 in Canada and 1979 in the United States and then showed a slight decrease to 1980. For the United States, the net change from 1940 to 1980 showed SO_2 emissions increased by 36 percent, while NO_x emissions more than tripled. Over the shorter interval shown for Canada, 1955 to 1980, SO_2 emissions were up 5 percent while NO_x emissions almost tripled. For the most recent decade, 1970-80, U.S. SO_2

emissions decreased some 15 percent, and a similar decrease of 19 percent took place for the nearest parallel period for Canada, 1965-76. However, changes for NO_x were not so similar in the two countries. NO_x emissions in the United States between 1970 and 1980 grew only 12 percent, whereas in Canada the 1965-76 interval still saw rapid growth, with NO_x emissions more than doubling.

One other trend over recent decades which has received extensive attention is that much of the SO_2 and a significant share of the NO_x emitted in the United States is now released at substantially greater heights above the ground than was the case before the 1950's. The main part of this change is associated with utilities, and the height increase can be recognized as a result of very widespread low-altitude SO_2 releases from residential, commercial, railroad, and small-scale industrial coal usage being replaced by utility emissions from tall stacks, which were put in place quite widely from the 1950's to 1970's. The effect of this change in release height is discussed on p. 74.

Breakdowns of types of emission sources over the recent decades for each country are shown in detail in tables I-3 through I-5.

Table I-3

Estimates of SO_2 Emissions in the United States, 1940-80

	1940	1950	1960	1970	1980
	-----(millions of metric tonnes)-----				
Combustion:					
Industry	4.5	4.4	3.3	3.9	2.3
Residential/commercial	3.1	3.3	2.0	1.3	0.8
Transportation	2.9	2.3	0.4	0.6	0.9
Electric utilities	2.2	4.2	7.4	15.6	15.9
Total	12.7	14.2	13.1	21.4	19.9
Smelters	3.3	3.5	3.8	4.1	1.8
Others[a]	1.4	1.9	2.3	2.4	2.0
Total	17.4	19.6	19.2	27.9	23.7

[a] Includes other industrial processes and miscellaneous burning.

Source: National Air Pollutant Emission Estimates, 1940-1980, U.S. EPA Report 450/4-82-001 (Jan. 1982), table 3.

Table I-4

Estimates of SO_2 and NO_x Emissions in Canada, 1955-76

Emmitting sector	1955		1965		1976	
	SO_2	NO_x[a]	SO_2	NO_x[a]	SO_2	NO_x[a]
	---------(millions of metric tonnes)---------					
Smelters	2.89	-	3.90	-	2.60	-
Power plants	0.06	0.01	0.26	0.06	0.61	0.21
Other combustion	1.21	0.23	1.13	0.25	0.88	0.45
Transportation	0.08	0.32	0.05	0.51	0.08	1.02
Others	0.30	0.07	1.26	0.03	1.13	0.19
Total	4.54	0.63	6.59[b]	0.85	5.31[b]	1.86[b]

[a] Counted as NO_2.

[b] Totals do not agree because of independent rounding.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982, table B.1.5.

Table I-5

Estimates of NO_x Emissions in the United States, 1940-80

Emitting sector	1940	1950	1960	1970	1980
	--(millions of metric tonnes, NO_2)--				
Transportation					
Highway vehicles	1.3	2.0	3.3	5.6	6.6
Other	0.9	1.4	1.3	1.9	2.5
Transportation total	2.2	3.4	4.6	7.5	9.1
Stationary combustion					
Industry	2.1	2.8	3.6	3.8	3.3
Electric utilities	0.6	1.3	2.5	5.0	6.7
Residential/commercial	0.4	0.6	0.7	0.7	0.6
Other[a]	1.2	1.2	1.3	1.5	1.0
Total	6.5	9.3	12.7	18.5	20.7

[a] Includes industrial processes, forest fires and other burning.

Source: National Air Pollutant Emission Estimates, 1940-80, USEPA Report EPA-450/4-82-001, Jan. 1982, table 4.

Sulfur dioxide

In Canada, smelters were the leading SO_2 source throughout the 1955-80 period, but their absolute emissions by 1980 were down 45 percent from a 1965 peak, as their share declined from nearly two-thirds to less than one-half of the national total.[11] Power plants made up for much of this decline, increasing their emissions more than 10-fold and rising to about one-sixth of the total in 1980. Combustion outside utilities was another significant SO_2 source, but it declined about one-third both in actual quantity and in percentage share from 1955 to 1980, with the industrial sector leading the decline. Finally, other noncombustion sources, relatively minor before the 1960's, grew to contribute about one-fifth of all SO_2 from 1965 on, largely arising from increased amounts of petroleum and natural gas being processed.

In the United States, fuel combustion remained the leading SO_2 contributor from 1940 to 1980, but the sources responsible changed dramatically. In 1940, fuel combustion provided 73 percent of all SO_2 emissions, with the order of contributions showing industry leading, followed by the combination of commercial and residential sources, then transportation (almost entirely from railroads), and utilities, contributing about 13 percent, last. At that time smelters, the leading noncombustion source, emitted another 19 percent of total U.S. SO_2.

By 1980, emissions from the three leading combustion sectors had dropped steeply. Industry emissions dropped by about one-half, transportation emissions by over two-thirds, and residential and commercial emissions by nearly three-quarters, so that together they provided only 17 percent of all U.S. SO_2. This was the result of coal's being displaced as a fuel to a great extent in industry and almost completely in the other sectors. The electric utility sector, on the other hand, grew sharply. In 1977, when the sector's SO_2 emissions peaked, they were about eight times the 1940 level, and in 1980 they represented 67 percent of all U.S. SO_2 emissions.[12] Finally, smelter SO_2 emissions grew moderately until 1970 and then dropped by 1980 to less than 8 percent of total emissions, at a level just over half their 1970 amount.

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[11] Canadian data for 1980, to complete the series in table I-4, were taken from table I in chapter 3 above, and from other sources in Work Group 3B Report No. 3B-Final (footnote 1).

[12] As noted earlier, on p. 125, these time series data are not completely consistent with the best estimates for 1980 in table 1. This shows in the difference of estimates for the utility share of 1980 SO_2 emissions: 67.1 percent in these data compared with 65.6 percent in table 1.

Oxides of nitrogen

Unlike the situation with SO_2, shifts in the relative shares of NO_x contributions were not very great, as most sources contributed to the substantial overall increase.

In Canada, transportation emissions remained the majority source, ranging from 51 percent of all NO_x in 1955 to 61 percent in 1980. Total stationary-source combustion grew enough to remain responsible for between 38 and 35 percent of NO_x, while the largest change saw utilities keep increasing their emissions, going from 2 to 13 percent of the total, as other combustion sources actually saw their NO_x emissions drop for the first time in the late 1970's.

In the United States, transportation's share of NO_x emissions remained less than a majority but increased from 34 to 44 percent. This was driven by the growth of highway vehicle contributions which, by 1980, had gone from 20 to 32 percent of all NO_x emissions. Stationary-source combustion held its share close to one-half the national total, but industry, which in 1940 was responsible for most of this and 32 percent of the national total, declined to 16 percent of the total in 1980. In this interval utilities increased their NO_x emissions over 10-fold, going from 9 percent to over 32 percent of the total and thus catching up with highway vehicles as the leading class of sources.

Past trends in eastern North American emissions

The regional emissions estimates available before the 1970's are not as complete, in either the United States or Canada, as are those at the national level. Specifically, Canadian data we obtained showed emissions for 1955, 1965, and 1976 broken down by source types but only for the whole of eastern Canada, with no further disaggregation by areas. In contrast, U.S. estimates we obtained covered each state consistantly at 5-year intervals from 1950 to 1978 (although their accuracy is particularly uncertain before 1965) but showed no breakdown by source types. This allowed us to identify trends in eastern North American emissions, but did not provide explanations of the trends in as complete a way as for national emissions.

Sulfur dioxide

Eastern U.S.--Trends in SO_2 emissions in the eastern United States show major differences between different parts of the East, with the Northeast showing the earliest and greatest decreases and the Southeast still not showing clear decreases as of 1980. Since some of the estimates for 1950 and 1955 were reported to be questionable, our discussion will concentrate on changes starting with 1960, although patterns for the 1950's will also be noted if they are not based on particularly questionable data. These data are shown in detail in table I-6.

Table I-6

Estimates of SO_2 Emissions in the Eastern United States, 1950-78

Region	1950	1955	1960	1965	1970	1975	1978
	(millions of tons)						
Northeast[a]	3.37[f]	3.92[f]	2.71	3.35	3.18	2.14	2.03
Middle Atlantic[b]	1.66[f]	3.10	3.29	3.59	3.49	3.05	2.82
Southeast[c]	1.09	2.29	2.90	3.84	5.32	5.89	5.76
Midwest[d]	3.27[f]	7.32	9.45	11.16	10.40	10.04	9.54
Trans-Mississippi[e]	1.67	3.25[f]	1.58	1.83	2.28	2.23	2.55
Total (31 Eastern states and D.C.)	11.06[f]	19.88[f]	19.93	23.77	24.67	23.35	22.70

[a] Northeast: New England, New York, and New Jersey.

[b] Middle Atlantic: Delaware, District of Columbia, Maryland, Pennsylvania, and Virginia.

[c] Southeast: Alabama, Carolinas, Florida, Georgia, Kentucky, Mississippi, and Tennessee.

[d] Midwest: Illinois, Indiana, Michigan, Ohio, West Virginia, and Wisconsin.

[e] Trans-Mississippi: Arkansas, Iowa, Louisiana, Minnesota and Missouri.

[f] Questionable data included.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report 3B-Final, June 15, 1982, table B.1.3.

For the Northeast, SO_2 emissions rose by about 23 percent from 1960 to a peak in 1965 and then dropped by about 39 percent through 1978, giving a net decrease of 25 percent between 1960 and 1978. As a result, from 1960 to 1978, the Northeast's share of total U.S. SO_2 emissions fell nearly half, from 13 to 7 percent.

The Southeast, defined as EPA Region IV, showed a very different pattern, with SO_2 emissions rising steadily to an apparent peak in 1975[13] and then dropping about 2 percent to 1978. The increase from 1960 to 1975 was 103 percent, and even with the small decrease through 1978, emissions in this area were essentially twice as great as in 1960. Furthermore, this was superimposed on an approximately 2.7-fold increase from 1950 to 1960, so that, overall, the Southeast's share of national SO_2 emissions went from about 5 percent in 1950 to 23 percent in 1980, as they increased about 5.6-fold in absolute amount. Trends in other eastern U.S. regions fell between these extremes.

The mid-Atlantic area showed a 16-percent SO_2 emission increase from 1955 to a peak in 1965, a slight decrease until 1970, and then a steeper drop to a level in 1978 that was 21 percent below the 1965 peak and about 9 percent below the 1955 level. In its share of national emissions, this area fell from about 16 to 11 percent between 1960 and 1980.

The Midwest showed a sharp increase of SO_2 emissions, some 52 percent, from 1955 to a peak in 1965, and then a relatively steady drop to a level in 1978 about 14 percent below the peak, but still some 30 percent higher than in 1955. This region's share of national SO_2 emissions was about 45 percent in 1960, but dropped to near 34 percent by 1970 and was still at that level in 1980.

The last group, the five trans-Mississippi states, showed continuous SO_2 emission increases up to 1978, to a level 53 percent above that of 1960, with their share of national SO_2 emissions remaining roughly the same, between about 7 and 9 percent. Within this group, emissions in the two more northerly states peaked and then declined--Minnesota peaking about 1955 and Iowa about 1965--but emissions in the southern three continued upward through 1978, enough to keep the total rising for the group as a whole.

Eastern Canada--In Canada, eastern SO_2 emission trends, detailed in table I-7, were generally closely similar to national ones, with two exceptions: smelters produced a larger share of eastern than of national SO_2 emissions because almost all smelters are located in the East; and emissions from oil and gas processing, which grew rapidly at the national level from 1955 to 1965, had little effect on eastern Canadian emissions because most of these came from gas processing in western Canada.

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[13] This peak is called "apparent" because 1980 data showed an emission level higher than 1975 data, and also a higher percentage share of national SO_2 emissions than in any earlier year.

Table I-7

Estimates of SO_2 Emissions in Eastern Canada,[a] 1955-80

Emitting sector	1955	1965	1976	1980
	(million of metric tonnes)			
Smelters	2.89	3.90	2.60	2.09
Electric utilities	0.05	0.25	0.55	0.67
Other combustion	1.07	0.99	0.79	0.74
Transportation	0.05	0.03	0.05	0.12
Other	0.23	0.41	0.38	0.35
Total	4.29	5.59[b]	4.39[b]	3.98[b]

[a] Eastern Canada includes Manitoba, Ontario, Quebec, and the Maritime Provinces.

[b] Totals do not add exactly because of rounding.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memoradum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982, tables B.1.6 and B.2.4.

Finally, in comparing overall SO_2 emission trends, the eastern parts of both the United States and Canada showed the same type of trends as each country taken as a whole, with emissions increasing to a peak and then dropping in more recent years. There were two significant differences in the nature of the decreases, however. First, Canada achieved emission reductions earlier, and in greater proportions than the United States. Second, eastern Canadian SO_2 reductions have been proportionately greater than those at the national level; in the United States, SO_2 emissions have not dropped as much in the East as they have on a national basis because of continued increases in the Southeast and Trans-Mississippi states. The reductions in Canada were largely accomplished in smelters and nonutility combustion sources, while utility SO_2 emissions in Canada actually continued to increase through 1980, in contrast to their having begun to decrease somewhat by the late 1970's in the United States.

Oxides of nitrogen

Most of the time-trends of NO_x emissions in the eastern parts of the United States and Canada followed closely with the national patterns, showing only moderate differences between regions within the United States and similarly moderate differences between eastern and total Canadian emissions. These data are detailed in tables I-8 and I-9.

Table I-8

Estimates of NO_x Emissions in the Eastern United States, 1950-78

Region	1950	1955	1960	1965	1970	1975	1978
	----------(millions		of tons	as NO_2)----------			
Northeast[a]	1.12	1.32	1.66	1.97	2.29	2.04	2.14
Middle Atlantic[b]	0.82	1.12	1.59	1.90	1.95	1.91	1.97
Southeast[c]	1.24	1.69	2.06	2.67	3.42	4.08	4.12
Midwest[d]	2.03	2.92	3.55	4.14	4.51	4.74	4.79
Trans-Mississippi[e]	0.93	1.10	1.40	1.77	2.28	2.52	3.10
Total (31 Eastern States and D.C.)	6.14	8.15	10.26	12.45	14.45	15.29	16.12

[a] Northeast: New England, New York, and New Jersey.
[b] Middle Atlantic: Delaware, District of Columbia, Maryland, Pennsylvania, and Virginia.
[c] Southeast: Alabama, Carolinas, Florida, Georgia, Kentucky, Mississippi, and Tennessee.
[d] Midwest: Illinois, Indiana, Michigan, Ohio, West Virginia, and Wisconsin.
[e] Trans-Mississippi: Arkansas, Iowa, Louisiana, Minnesota, and Missouri.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982, Table B.1.4.

All regions of the eastern United States showed major increases of NO_x emissions between 1950 and 1978, but the extent of increase was lowest, at about 1.9-fold, for the Northeast and highest, at about 3.3-fold, for the Southeast and Trans-Mississippi states. The Mid-Atlantic, with about a 2.6-fold NO_x increase, and the Midwest, with about a 2.4-fold increase, again fell between these extremes. The differences between the regions were mostly seen in the 1970's. Between 1970 and 1978, northeastern NO_x emissions actually fell 7 percent, Trans-Mississippi emissions rose 36 percent, and emissions in other regions increased by lesser percentages. Over the entire period, eastern U.S. NO_x emissions rose from 60 percent of the national total in 1950 to 74 percent in 1960 and then fell slowly to 66 percent in 1980. The largest change in any eastern region's share was the Southeast's, rising from 12 to 18 percent.

The only substantial difference between national and eastern NO_x emission trends in Canada was in nonutility stationary source combustion, for which eastern Canada showed only a 31-percent increase between 1955 and 1980; the increase nationally was 73 percent. This difference, and also the decrease in the share that eastern NO_x emissions represented in the national total, from 71 to 62 percent between 1955 and 1980, are attributable to a westward shift of industry and population.

Table I-9

Estimates of NO_x Emissions in Eastern Canada,[a] 1955-80

Emitting sector	1955	1965	1976	1980
	--(millions of metric tonnes as NO_2)--			
Transportation	0.21	0.34	0.65	0.70
Electric utilities	0.01	0.05	0.14	0.17
Other combustion	0.18	0.16	0.25	0.24
Other	0.05	0.01	0.09	0.03
Total	0.45	0.56	1.14[b]	1.13[b]

[a] Eastern Canada includes Manitoba, Ontario, Quebec, and the Maritime Provinces.

[b] Totals do not add exactly because of rounding.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982, Tables B.1.6 and B.2.5.

PROJECTED TRENDS IN EMISSIONS

Projecting future emission patterns is subject to a number of uncertainties, arising from possible changes in factors including environmental standards, economic growth rates, and the overall and comparative costs of fuels, all of which can effect both the amounts of those activities that give rise to emissions and the amounts of emissions released from each unit of activity.

For this section, we have taken the most recent projections of U.S. and Canadian emissions which were made under the U.S.-Canada Memorandum of Intent on Transboundary Air Pollution.[14] These projections were based on maintenance of present environmental standards and recently updated economic assumptions, but actual emissions may differ from the projections as time progresses.

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[14] Work Group 3B, Emissions, Costs and Engineering Assessment U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982. Section B.3. Detailed projected emissions are in tables I-10 and I-11.

As one example, the Canadian emission projections do not take into consideration an indicated 50-percent reduction of SO_2 emissions from a large high-emitting smelter in Quebec because, at the time of our last review of this report, the Quebec provincial government had not yet served a final order for this reduction. As another example, some U.S. coal-fired power plants were assumed to continue to operate at their recent capacity factors as they aged, while the capacity factors for plants fueled by oil or gas were assumed to decrease with time because of their higher fuel costs. Yet, if the cost or availability of oil or gas improves, then earlier patterns could recur: that is, coal-plant capacity factors would decrease with age and thus lead to lower SO_2 emissions from the same amount of electricity generation. Further sources of uncertainty lie in predicting future demand for the products of emitting industries and processes, and therefore their levels of emissions. Thus, the following projections must be recognized as subject to substantial uncertainty.

Projected SO_2 emissions

In both the United States and Canada, SO_2 emissions are projected to decline modestly from 1980 to 1990, but the decreases are attributed to different types of sources and show different geographic distributions. These projections are shown in detail in table I-10.

Table I-10

Projections of SO_2 Emissions, United States and Canada, 1990 and 2000

	United States						Canada					
	1980 (actual)		1990		2000		1980 (actual)		1990		2000	
	(million metric tonnes)						(million metric tonnes)					
	National	East	National	East	National	East	National	East	National	East	National	East
Electric utilities	15.8	14.58	16.1	14.69	16.4	14.82	0.74	0.67	0.65	0.58	0.66	0.47
Nonferrous smelters	1.4	0.16	0.6	0.02	0.5	0.03	2.13	2.09	2.32	2.29	2.32	2.29
Transportation	0.8	0.49	0.8	0.53	1.0	0.68	0.16	0.12	0.16	0.12	0.16	0.12
Residential & commercial	0.8	4.60	1.0	0.78	0.9	0.67	0.21	0.20	0.08	0.07	0.03	0.02
Industrial combustion	5.3		3.5	2.97	6.6	5.25	0.62	0.55	0.33	0.28	0.23	0.18
Industrial processes			1.2	0.71	1.4	0.78	0.92	0.35	1.16	0.35	1.11	0.34
Total	24.1	19.83	23.2	19.70	26.8	22.23	4.78	3.98	4.70	3.69	4.51	3.42

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution Report No. 3B Final, June 15, 1982, combined from data in tables A.2.3, B.2.1, B.2.4, B.3.1-A, B.3.6, B.3.7, B.3.11, B.3.13, B.3.16 to 18, 6.1, 6.3, 6.5, 6.6, 6.8, and 6.10.

In the United States, about a 4-percent decrease is projected nationally, mostly in the West and mainly because of an almost two-thirds drop in smelter emissions, while eastern emissions are projected to drop less than 1 percent. In contrast Canada, which projects less than a 2-percent decrease of SO_2 emissions nationally, anticipates about a 7-percent decline in the East, mainly occurring in nonutility combustion as gas displaces sulfur-containing oil and/or coal; emissions are projected to increase significantly in the West because of growth in the processing of tar sands, gas, and oil.

The limited extent of the decrease in SO_2 emissions in the United States projected for the 1980's may appear surprising, since trends over the 1973-80 and 1980-82 period showed a progressive decline amounting to over 23 percent, including an 8-percent decrease just from 1980 to 1982. However, the projection is more understandable in light of three facts. First, the United States has already come quite close to achieving the goals of the current SO_2 standards: in 1980, the National Commission on Air Quality projected that very few areas were likely to continue to exceed the SO_2 standard after 1982.[15] Thus, little further decrease would be expected based on current SO_2 standards. Second, 1981 and 1982 saw the United States decline into a recession, and slowed economic activity resulted in reduced industrial energy consumption, an event which generally results in lessened emission of pollutants. Finally, as discussed on p. 77, the 1978-83 period saw a major decline in the use of residual oil, a decline which stopped in 1983 and so should not continue as a reason for further SO_2 emission decreases.

Going on to the next decade, changes for 1990 to 2000 are expected to diverge sharply, as U.S. emissions are anticipated to increase substantially, while those in Canada are projected to continue a moderate decline.

In Canada, the projection is for a 4-percent decrease nationally, driven by another 7-percent drop in eastern SO_2 emissions, essentially all coming in stationary combustion sources, which should more than overcome increases in the West that are expected from utilities.

In contrast, an increase of about 16 percent is projected for the United States in the 1990's, with the eastern contribution up some 13 percent and the West up 31 percent. Throughout the country these increases are projected to occur overwhelmingly from industrial combustion, whose SO_2 emissions are estimated to almost

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[15] To Breathe Clean Air, Report of the National Commission on Air Quality, Washington, 1981, p. 3.4-32.

double in the 1990's. This striking increase is the result of a combination of three factors controlling the industrial SO_2 emission projections of this study:

--The models used showed coal capturing over 90 percent of the market for new industrial boilers.

--Continued differentials assumed between prices for coal as opposed to those for oil and gas are projected to lead to many early retirements of oil- and gas-fired boilers, and their replacement by coal.

--Existing new source performance standards covering industrial boilers are not as stringent as those for utilities, and do not extend down to cover boilers smaller than 250 million Btu/hr (73 MW) sizes.

In addition to the projected increase of U.S. industrial SO_2 emissions, two major features of projected North American SO_2 emissions show significant differences between the same sectors in the two countries: one bears on the comparative trends in leading sectors, while the other applies to the intensity of emissions from fossil-fueled utilities.

SO_2 emissions from leading sectors

As noted previously, the leading SO_2-emitting sectors are power plants in the United States and smelters in Canada. In each country, SO_2 emissions from the leading class of sources have shown sizeable decreases over recent years, but they are currently projected to resume growth, though at slower rates, in the next decade or two. In contrast, the sector which leads in emissions in one country is projected to show major declines of emissions in the other country over the coming years. Thus, electric utilities in eastern Canada are anticipated to show about a 30-percent decrease of SO_2 emissions by 2000, which should drive a net 11-percent decrease of utility emissions nationally, despite substantial additions of fossil-fueled generating capacity in the West. In contrast, United States utility emissions, which declined from 1973 to 1982, are projected to increase slightly both in the East and West. On the other hand, smelter SO_2 emissions in the U.S. are anticipated to drop about 53 percent from 1980 to 1990, and another 17 percent in the following decade--a total of 64 percent by the end of the century--while Canadian smelters are projected to increase their SO_2 emissions about 9 percent by 1990 and hold that level until 2000.

These projections show that in each country, major SO_2 emission reductions, which would have to occur largely in the leading emitting sector, can only be expected in this century if emission control regulations on these leading emitters are made significantly more stringent than they are at present.

Utility SO_2 emission rates

The second comparison which shows differing trends between the United States and Canada is the intensity, or rate, of SO_2 emissions from power plants. Emission rate is a measure which is commonly used with regard to emissions from combustion sources in the United States, and is counted in pounds of pollutant per million Btu's of fuel energy (lb/mmbtu). For the United States in 1980, on a national basis, combining all coal- and oil-fueled power plants, the average emission rate was 2.38 lb SO_2/mmbtu; projections indicate that this rate will decline to 1.96 lb/mmbtu in 1990 and 1.43 lb/mmbtu in 2000. For Canada, the national average was 2.45 lb/mmbtu in 1980, and the projected values decline more sharply, to 1.57 lb/mmbtu in 1990 and 1.01 lb/mmbtu in 2000. This general decline can, in large part, be attributed to the fact that new plants coming on-line in both countries will have to meet SO_2 emission limits that are much more stringent than those affecting existing plants--e.g., 0.6 to 1.2 lb/mmbtu for coal-fired plants in the United States. However, in each country the amount of new fossil-fueled generating capacity scheduled to start operation is not sufficient to explain the projected decline of emission rates by itself: rather, there is also some decrease in emission rates anticipated from existing plants. As might be expected, this decline is projected to be greatest in the one jurisdiction where emission reductions have been ordered for the specific purpose of reducing acid deposition by decreasing total emissions, rather than improving ambient air quality. This jurisdiction is the Canadian province of Ontario, where fossil-fueled generation is projected to grow from 31 billion to 56 billion kwh hours per year between 1980 and 2000, while SO_2 emissions are projected to decline from 398,500 metric tonnes to 260,000 metric tonnes, resulting in a drop of average emission rate from 2.80 to 1.03 lb SO_2/mmbtu.

Utility SO_2 emissions in the United States beyond 2000

The projections reviewed so far in this appendix are all from the joint U.S.-Canada work group report, which did not present emission estimates beyond the year 2000. Any attempt to project beyond that date will be subject to even greater uncertainty, because of the longer interval in which unanticipated events could intervene. However, because of the dominance of SO_2 emissions by utility contributions and the controlling role that currently existing coal-fired plants have in those emissions, we shall look briefly at the longer-term expectations for this sector.

For this purpose, we show a widely used set of baseline projections, made by ICF Inc., in its Coal and Electric Utilities Model.[16]

Utility SO_2 Emissions in the United States, 1980-2010

	1980	1985	1990	1995	2000	2010
	(millions of tons)					
31 eastern states:						
Existing plants	16.19	16.75	17.25	16.84	14.36	10.30
New plants	-	0.30	0.69	1.14	2.71	5.45
31-state total	16.19	17.05	17.94	17.98	17.07	15.75
Western states	1.19	1.76	2.09	2.23	2.48	2.85
Total U.S.	17.38	18.81	20.03	20.21	19.55	18.60

These projections show total and eastern U.S. SO_2 emissions peaking in 1995, with plants that already were operating in 1980 still dominating eastern U.S. emissions. Even in 2000, these older plants still will be emitting at 89 percent of their 1980 level and producing 84 percent of all eastern utility SO_2. Plant retirements are projected to accelerate after 2000 in this analysis, on the basis of 45-year plant lifetimes.[17] Thus, by 2010 the pre-1980 eastern plants should be down to 64 percent of their 1980 emissions (60 percent of the emissions at their 1990 peak) and, for the first time, total eastern utility SO_2 emissions should drop below the 1980 level.

The exact level of utility SO_2 emissions over the next several decades differs somewhat from one set of projections to another, but the general picture remains the same if the basic assumptions are not changed. Thus, a set of projections examined in an EPRI Research Overview and Strategy study[18] also used 45-year coal-plant lifetimes, and showed about an 8-percent increase in total U.S. utility SO_2 emissions from 1980 to 2000, compared with the approximately 12-percent increase in ICF's projections.

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[16] ICF base-case projections remain similar in a number of studies using the model. We cite values from a Sept. 16, 1982, ICF memo to EPA policy office staff.

[17] ICF, Inc., Alternative Strategies for Reducing Utility SO_2 and NO_x Emissions, draft report prepared for Argonne National Laboratory, Department of Energy, and Environmental Protection Agency, Sept. 1981, pp. 4-7.

[18] EPRI Planning and Evaluation Division, Overview and Strategy 1982-1986 R&D Program Plan, P-2156-SR, EPRI, Palo Alto, California, Nov. 1981, pp. 158-159.

However, if these older coal-fired plants were to stay in operation beyond the assumed 45 years, then SO_2 emissions would not begin to drop until further beyond the turn of the century. In fact, a number of different parts of the EPRI strategy study focus on how valuable it will be to the utility industry and its clients if the useful lives of existing plants, particularly coal-fired ones, are extended beyond 45 years to 50 years or even more. This is described as an "essential" research area, and one of the four highest priority program areas for EPRI's research.[19] If this result is achieved, then the time when eastern U.S. utility SO_2 emissions would drop below the 1980 level could be extended by a decade or more, to 2020 or beyond.

Projected NO_x emissions

Projections of NO_x emission levels for 1990 and 2000, and actual 1980 levels, for the United States and Canada and their eastern regions are shown in detail in table I-11. While contributions are shown from a number of different sectors, the main features of the projections are dominated by transportation contributions in Canada and by transportation and utility emissions in the United States. In both countries, the expectation is that NO_x emissions will grow more than those of SO_2.

At the national level for the 1980's, total NO_x emissions are projected to hold steady in the United States and increase by about 8 percent in Canada. Growth of utility emissions is the driving force in the United States, and is estimated to balance out decreases in transportation and industrial contributions. In Canada the opposite situation is expected, with transportation increases overpowering decreases in utility and other sectors.

In the 1980's, Canada anticipates that eastern utility and other combustion NO_x emissions will decrease enough to just counterbalance a 20-percent growth of transportation NO_x emissions, while in the United States increases of eastern utility and other combustion NO_x emissions will overpower the decline of eastern transportation NO_x, leading to an increase of about 7 percent.

Much greater growth of NO_x emissions is expected in the 1990-2000 period, with no important sector showing decreases in either country. As a result, total NO_x is projected to increase by 21 percent in Canada and 25 percent in the United States in the decade, with slightly slower growth in the East in each country.

Transportation NO_x is projected to increase by about one-fourth in the 1990's both in the eastern parts and nationally in each country. As a result, in the year 2000, transportation is expected to account for 40 percent of all NO_x both nationally and in the East for the United States, for 69 percent of the national total and, strikingly, fully 80 percent of the eastern share in Canada.

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[19] Ibid, p. 107, p. 185.

Table I-11

Projections of NO_x Emissions, United States and Canada, 1990 and 2000

Source	United States						Canada					
	1980 (actual)		1990		2000		1980 (actual)		1990		2000	
	National	East	National	East	National	East	National	East	National	East	National	East
	(million metric tonnes, as NO_2)											
Transportation	8.5	5.63	7.8	5.29	9.7	6.63	1.11	0.70	1.34	0.84	1.67	1.04
Electric utilities	5.6	4.34	6.8[d]	5.14[d]	8.7	6.27	0.25	0.17	0.19	0.10	0.26	0.09
Industrial processes[a,b]	4.2	0.57	0.8	0.41	1.1	0.45	0.08	0.03	0.08	0.03	0.08	0.03
Industrial combustion		2.19	3.0	2.26	4.0	2.93	0.30	0.17	0.30	0.11	0.33	0.11
Residential & commercial	0.7		0.8	0.57	0.7	0.48	0.09	0.07	0.07	0.04	0.07	0.03
Other[c]	0.3	-	-	-	-	-	-	-	-	-	-	-
Total	19.3	12.73	19.3	13.67	24.2	16.76	1.83	1.13	1.98	1.12	2.41	1.30

[a] Very small contributions from non-ferrous smelters (less than 0.01 million metric tonnes for Canada, smaller for the U.S.) are included with other industrial process NO_x emissions.

[b] Canadian values small and assumed constant rather than projected explicitly.

[c] Solid-waste disposal, forest fires, and other miscellaneous sources. No projections done for these sources.

[d] State-by-state projections add to only 6.83 million metric tonnes despite national projection giving 7.2, about a 5-percent discrepancy.

Source: Work Group 3B, Emissions, Costs and Engineering Assessment, U.S.-Canada Memorandum of Intent on Transboundary Air Pollution, Report No. 3B-Final, June 15, 1982, combined from data in tables B.2.3, B.2.5, B.3.1-B, B.3.2-B, B.3.7, B.3.12, B.3.14, B.3.21, 6.2, 6.4, 6.7, and 6.11. Columns may not add to totals because of independent rounding.

The modest share of NO_x projected to be contributed by utilities in Canada drops somewhat from 1980 to 2000, and falls to less than half its level in the East. In contrast, utility NO_x emissions in the United States are estimated to increase by about one-half, as their percentage share rises to almost match that from transportation.

Two key differences seem to largely explain the variance in NO_x projections between the two countries: different vehicle emission standards, and different levels of stringency in requirements for reducing utility NO_x emissions.

Motor vehicle NO_x emission rates and projections

Automobile emission standards in the United States have been at 1 gram of NO_x per mile (1 gpm) since 1981. In contrast, in 1981 the Canadian Parliament Sub-Committee on Acid Rain said:

> "The Sub-Committee is appalled that motor vehicles in Canada emit three times as much NO_x per vehicle mile as vehicles in the United States."[20]

The Subcommittee recommended that NO_x emission standards for new cars sold in Canada, currently at 3.1 gpm, be made at least as stringent as those in the United States. The Sub-Committee pointed out that, if this were done by 1985, then vehicle NO_x emissions in Canada could drop 20 percent by 1990, instead of rising by more than half by 2000.

To date, neither this change, nor a relaxation of vehicle NO_x emission standards in the United States, which was discussed during the depressed period of United States auto production in the early 1980's, has occurred, so the present projections stand. However, this analysis shows that vehicle standards represent a significant source of possible divergence from the NO_x projections described here.

Utility NO_x emission rates and projections

Emissions of NO_x by fossil-fuel-burning utilities in 1980, expressed as rates per unit of fuel energy, averaged 0.67 lb/mmbtu in the United States and 0.75 lb/mmbtu in Canada. As noted in chapter 4, current technology offers means for moderate reductions in these levels, by possibly 25 percent, at relatively modest costs compared with the costs estimated for similar or larger SO_2 reductions. Techniques for larger reductions are not yet available, but are believed likely to reach the market during 1985-90. The utility NO_x emission projections for the United States are based on current standards, which do not require major changes of NO_x controls, so that the average emission rate for the United States is not expected to decrease--in fact it is projected to increase slightly, to 0.72 lb/mmbtu by 2000. In contrast, the average Canadian NO_x emission rate is projected at only 0.39 lb/mmbtu by 2000 as the result of the application of more stringent standards. This is most evident in Ontario again, as with SO_2. By 2000, Ontario is projected to increase its fossil-fueled electricity output by more than three-quarters, while decreasing its utility NO_x output by more than 60 percent. Consequently, the province's NO_x emission rate is projected to drop from 0.71 to 0.16 lb/mmbtu. Without arguing whether the Ontario projection is too optimistic, or the U.S. projection too lenient, it is clear that there is room here for substantial divergence between actual utility NO_x emissions and the projected values.

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[20] Still Waters: The Chilling Reality of Acid Rain, Sub-Committee on Acid Rain, House of Commons, Ottawa, Canada (1981), p. 47.

INQUIRY LETTERS

We sent an inquiry letter containing the text shown below to a wide spectrum of persons in the United States involved in the acid rain issue, as scientists and/or advocates.

Dear ____________________

The United States General Accounting Office is carrying out a study of the "acid rain" issue. We recently issued an interim report on the subject, a copy of which is enclosed. This report was aimed at describing the range of views which have been expressed on a number of separate facets of the issue, and attempted to indicate the level of scientific understanding or uncertainty on these points.

Our follow-up work will focus on the costs of damage caused by acid precipitation and the costs and potential reductions of emissions resulting from proposed regulatory strategies. In addition, we intend to clarify our understanding of the causes of acid precipitation, the role of fossil fuel emissions and, to the extent that these emissions are implicated, the effect on deposition of reducing emissions. Our findings in these areas will be the basis for our conclusions and recommendations on whether or what kind of regulatory action is now appropriate.

You could be of great assistance to us if you would take a few moments to indicate your views on acid precipitation, in light of the current level of understanding of the subject, particularly concerning directions for public policy and possible governmental action. An explanation of the basis for your views would also be very useful. Topics on which we hope you could touch include:

--the extent and type of governmental action you think would be appropriate,

--the specific questions you believe may need to be resolved before decisions should be made on a regulatory program, and

--the type of research needed and the time you expect it may take to answer these questions.

Reponses to this letter were received from the following individuals:

PUBLIC OFFICIALS

Ms. Jennie E. Bridge
Environmental Scientist
New England Interestate Water
Pollution Control Commission

Dr. Wolfgang Fuhs
Director, Environmental Health
Laboratory Institute
New York State Department of Health
Office of Public Health

Ms. Wayne Nichols
Director, Ohio Environmental
Protection Agency

Mr. F. Peter Fairchild
Executive Director,
Northeast States for Coordinated
Air Use Management
Boston College Weston Observatory

Dr. Anne LaBastille,
Commissioner
Adirondack Park Agency, New York

INTERESTED PARTIES AND ADVOCACY GROUPS

Mr. David C. Branand
Senior Counsel
American Mining Congress

Ms. Susan L. Falvo
Environmental Scientist
Edison Electric Institute

Mr. S. David Freeman
Commissioner
Tennessee Valley Authority

Mr. John J. Jansen
Research Specialist
Southern Company Services, Inc.

Mr. Phillip X. Masciantonio
Vice-President, Environment
and Energy
U.S. Steel Corporation

Dr. Michael Oppenheimer
Senior Scientist
Environmental Defense Fund

Mr. Bradley H. Spooner
Manager, Air and Environmental
Resources Programs
New England Power Service Company

Mr. Al Courtney
Utility Air Regulatory Group
and Commonwealth Edison, Chicago

Howard Fox, Attorney
Sierra Club Legal Defense Fund

Messrs. Steve Howards and
Kenneth Kamlet
National Wildlife Federation

Mr. Francis M. Lee
Manager, Environmental Affairs
Division
Boston Edison Company

Ms. Pam McClelland
Trout Unlimited

Dr. George T. Patton
American Petroleum Institute

RESEARCH SCIENTISTS

Dr. John Carroll
Institute of Natural and
Environmental Resources
University of New Hampshire

Dr. Leonard D. Hamilton, Director
Biomedical Environmental Assessment
National Center for Analysis of
Energy Systems
Brookhaven National Laboratory

Dr. Jay S. Jacobsen
Boyce Thompson Institute
Cornell University

Dr. Orie L. Loucks
Science Director
The Institute of Ecology
Indianapolis

Dr. Leonard Newman, Head
Environmental Chemistry Division
Brookhaven National Laboratory

Dr. G.K. Voight
School of Forestry and
Environmental Studies
Yale University

Dr. Eville Gorham
Department of Ecology and
Behaviorial Biology
University of Minnesota

Professor Rudolf B. Husar
Center for Air Pollution Impact
and Trend Analysis
Washington University, St. Louis

Dr. Allen S. Lefohn
ASL and Associates
Helena, Montana

Dr. Volker Mohnen, Director
Atmospheric Science Research
Center
New York State University at
Albany

Dr. Ralph Perhac, Director
Environmental Assessment
Department
Electric Power Research
Institute

In addition, a letter with the following somewhat different text was sent to a number of foreign persons.

Dear ________________

The Congress of the United States is currently considering renewal of the Clean Air Act. In that process, the issue of acid precipitation is being widely examined. As a support agency of the Congress, the General Accounting Office has been asked to study this issue. We recently issued an interim report on the subject, a copy of which is enclosed. This was aimed at describing the range of views which have been expressed on a number of separate facets of the issue, and attempted to indicate the level of scientific understanding or uncertainty on these points.

Our follow-up work will focus on the costs of damage caused by acid precipitation and the costs and potential reductions of emissions resulting from proposed regulatory strategies. In addition, we intend to clarify our understanding of the causes of acid precipitation, the role of fossil fuel emissions and, to the extent that these emissions are implicated, the effect on deposition of reducing emissions. Our findings in these areas will be the basis for our conclusions and recommendations on whether or what kind of regulatory action is now appropriate.

You could be of great assistance to us if you would take a few moments to indicate your views on acid precipitation, in light of the current level of understanding of the subject. An explanation of the basis for your views would also be very useful. Topics on which we hope you could touch include:

--the extent and type of action you think would be appropriate at this time;

--the specific questions you believe may need to be resolved before further decisions should be made; and

--the type of research needed, and the time you expect it may take to answer these questions.

Responses to this letter were received from:

Dr. Gunnar Abrahamsen
Norwegian Forest Research
Institute

Dr. Donald A. Chant, Director
Canadian Centre for Toxicology
University of Toronto

Dr. W.R. Effer, Manager
Environmental Studies and Assessments
Ontario Hydro

Dr. Ib Johnsen
Institute of Plant Ecology
University of Copenhagen

Dr. J.E. Rippon
Central Electricity Generating Board
Research Division
Great Britain

Mr. J. Stuart Warner
Vice-President Occupational
Safety and Health
International Nickel Company
of Canada

Dr. B.W. Bache
Macauley Research Institute
Agriculture Research
Council of Great Britain

Dr. William Dickson
National Swedish Environmental Protection Board
Research Department

Drs. Arne Henriksen and
Lars Overrein
Norwegian Institute for
Water Research

Mr. George Rejhon
Environmental Counselor
Embassy of Canada
Washington, D.C.

Professor Bernhard Ulrich
Institut fur Bodenkunde
and Waldernahrung
Universitat Gottingen
Federal Republic of
Germany

OVERVIEW OF SULFUR OXIDE CONTROL STRATEGIES

A wide range of proposed strategies exist for reducing sulfur oxide emissions. These have been reported at length in a wide range of publications,[1] and their anticipated costs have been analyzed in many studies. Therefore, we did not think it necessary to do an independent analysis of control methods and their costs and impacts. Rather, we will give brief explanations of the major methods, noting some of their potential effects, extent of applicability, and advantages and disadvantages. We will also examine some of the cost estimates that have been published, but mainly with the aim of comparing costs between methods, and noting important factors that contribute to costs, without attempting to decide on a single "correct" estimate.

In this treatment, we are only considering methods to control the emissions from older (pre-NSPS) plants so, among technological approaches, we will concentrate on those useful for retrofit, as opposed to those limited to new units. The usual division of technological approaches to emission control into three groups can serve here, separating the possibilities into those applied before, during, or after combustion, with institutional approaches included with the precombustion group.

PRECOMBUSTION SO_2 EMISSION CONTROL METHODS

Methods of reducing SO_2 emissions before combustion can focus on reducing the amount of fuel used, on preferentially using plants which release less SO_2, or on lessening the amount of sulfur contained per unit of fuel burned.

Energy conservation actions

Any steps which reduce consumer use of electricity, either by improvements in efficiency or by reductions in the amounts or kinds of services electricity is used for, would decrease the amount of electricity that must be generated. This would allow utilities to lower their fuel consumption and therefore reduce their SO_2 emissions in proportion to the amount of sulfur in the

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[1] Two very useful recent sources are a Congressional Research Service report entitled Mitigating Acid Rain With Technology: Avoiding the Scrubbing-Switching Dilemma, prepared for the Subcommittee on Natural Resources, Agriculture Research and Environment, House Committee on Science and Technology, Serial L. June 1983, and an earlier report entitled Institutional Aspects of Transported Pollutants by G. Wetstone and P.D. Reed, for the National Commission on Air Quality, Environmental Law Institute, Wash., D.C., Feb. 18, 1981. The papers in the CRS report also give extensive technical literature references.

fuel saved. Similarly, efficiency improvements by utilities, in the generation and/or transmission areas, would allow them to serve their loads with less fuel and therefore lower their emissions. While all of these steps could be effective in lowering emissions (if the generating fuels saved are ones which give rise to significant emissions), we have not explicitly examined the conservation alternative in this study, since an extensive literature already exists on the subject.

It should be noted, in this context, that many of the technological approaches to emission reduction have an effect in the opposite direction from energy conservation, because they require significant energy to operate, which will usually come from the plant to which the technology is applied. This will reduce the plant's net power output, making it consume more fuel to meet demand and thus somewhat increasing the amount of potential SO_2 emissions it will have to control.

Preferential use of low-emitting generating plants

For any utility with more than a single generating plant, the question of what plant or mix of plants will be used to meet its load at any time is subject to constant scrutiny. Making these decisions from minute to minute and day to day is called "dispatching," and, within constraints that include (1) plant availability, (2) the responsiveness of each plant to changes of demand, and (3) maintenance schedules, dispatching is usually aimed at minimizing the cost of generation, in keeping with regulatory requirements.

If regulatory policies were to be changed, to put a higher priority on emission reduction than on cost minimization, then the resulting least emission dispatching (LED) could possibly lead to substantially lower SO_2 emissions. LED would be accomplished by maximizing the use of nuclear and hydro capacity and lower emitting, fossil-fueled plants (gas, low-sulfur oil, and post-NSPS coal) in preference to high-sulfur oil or older coal plants. A central problem to establishing LED would be the need for policy changes, which would likely result in higher electricity rates, to be made by the many independent regulatory commissions that oversee utility operations. This would have to be done in the face of widespread concern about, and opposition to, recent and anticipated electricity rate increases. Also, in addition to having higher prices, gas and low-sulfur oil are premium fuels which national energy policy has tried to conserve and replace with coal or nuclear fuel.

The idea of choosing low-emitting generating methods has been proposed in other forms besides LED. One example is the suggestion that high-emitting plants be cut back to lower output levels or turned off entirely during a particular season, or at times when major pollution episodes occur or threaten. While this might lessen acid deposition in one season or in individual weather events, there is little evidence that it would result in lower total deposition over a year, unless the emissions in question were actually foregone, as opposed to only being displaced to a different week or season.

Another way of reducing acid deposition deriving from the emissions of one utility (or group of utilities) could be to generate power elsewhere and transmit it to the area served by the utility in question. If the distances between the sending and receiving areas were long enough, and/or in the right direction, then it might be possible that the same amount of emissions could be tolerable in a region with less vulnerable receptors, beyond the atmospheric transport range of the receptor areas affected by the first utility. This would amount to exporting emissions, however, and could be expected to raise pollution concerns in the areas where emissions and deposition would increase. In addition, it would likely require construction of additional transmission lines, making for a rather rigid long-term commitment to the power exchange.

Of course, replacement of power generated by high-emitting plants with that from lower emitting units in other areas could avoid the concern with exported pollution. However, in most of the United States there is opposition to construction of new power plants, particularly nuclear ones--the main form of nonsulfur-emitting units now being built here. The alternative prospect, having the United States buy excess output from planned Canadian nuclear or hydro plants, has led to charges that Canadian calls for reduction of U.S. emissions are actually an attempt by Canada to develop an electricity export market here, to the disadvantage of midwestern coal and utility firms and workers.

A final method of preferentially using lower emitting power plants--the early retirement of high-emitting pre-NSPS plants and their replacement by new low-emitting units--should be mentioned here. While replacing older plants is clearly not a retrofitting approach, it offers the possibility of improved generation efficiency as well as lower emissions compared to older units, and is one approach likely to be used to meet stringent new SO_2 emission reduction regulations in effect since July 1983 in West Germany. (See app. IV.) The prospects for such action by U.S. utilities seem very unlikely at this time, however, without changes in financial or regulatory conditions. Indeed, as mentioned in app. I (p. 144), utilities are very interested in extending the lifetimes of existing plants.

Lowering the sulfur content of fuels used

The sulfur content of fuels burned in utility plants can be lowered either by choosing fuels which have lower sulfur content or by treating the fuels to remove some of their sulfur. Clearly, these methods will overlap to some extent, since fuel producers could, and in many cases do, prepare lower sulfur forms of fuels before delivery. This is often done because it is logistically more feasible and/or of lower overall cost to process fuel before delivery, rather than at the power plant.

The range of choices of sulfur content available in fuels can be seen in statistics on the quality of utility fuels compiled by the Energy Information Administration.[2] For the first 3 months of 1983, as an example, heavy oils used as fuel in different power plants showed monthly average sulfur contents from below 0.2 percent to as high as 3.1 percent, with a nationwide average about 1.2 percent; while coal sulfur contents ranged from 0.3 percent up to 5.6 percent with an average about 1.5 percent--about a 16-fold range for oil and about an 18-fold range for coal. While this cannot be taken to mean that all utilities could obtain fuels with the lowest values of sulfur content shown, it does suggest that a great deal of the sulfur emissions from utilities could be avoided by some combination of switching to lower sulfur fuels and/or desulfurizing fuels before combustion.

Oil desulfurization

We discussed desulfurization of residual oil (resid) briefly in chapter 4, (pp. 76-77) where we noted that it is a major approach to reducing sulfur emissions in Japan and Europe. In comparison, we found its potential in the United States is relatively limited by the sharp decline in resid usage in the wake of the 1979-80 world oil price increases. Indeed, even in the three East Coast Census Divisions--the main areas where oil is used for electricity generation--emissions that would be released if all the sulfur in resid burned by utilities were turned to SO_2 would amount to only about three-quarters of a million metric tonnes, only about one-tenth of total SO_2 emissions in these states or less than 4 percent of total SO_2 emissions in the eastern United States. This upper limit, combined with the progressive expansion of desulfurization capacity that has been underway in U.S. oil refineries, indicates that there is relatively little more in the way of SO_2 emission reductions to be gained from desulfurization of oil burned in U.S. utilities.

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[2] Energy Information Administration, Electric Power Quarterly, January to March 1983, DOE/EIA-039 (83/1Q) July 1983, Table 14.

We have not found analyses that offered estimates of the cost of resid desulfurization in the United States over a range of sulfur content, or of the cost per amount of sulfur removed. An OECD study[3] of sulfur control in Europe reported that desulfurizing high-sulfur resid had a cost, per ton of SO_2 removed, that was "fairly similar" to the cost for post-combustion flue gas desulfurization at the same facility. For a sense of costs here, we can look to EIA data,[4] which show significant premiums paid for lower sulfur resid compared with higher sulfur resid, with price differences of about 9 to 18 percent between the lowest sulfur and highest sulfur categories of resid sold in the three main utility resid-consuming Census Divisions.

Coal desulfurization

Removal of sulfur from coal before combustion is one of the three techniques currently used substantially in the United States to lower SO_2 emissions.[5] The technique, known as physical coal cleaning or coal washing, is capable of removing only the pyritic part of the sulfur found in coal.[6] Methods are being explored which would accomplish separation of organic sulfur as well, but these chemical coal cleaning techniques are in early stages of development, and can approach coal liquefaction or gasification in complexity and scale; therefore, we will not cover them further here.

Coal washing is best applicable to those coals with substantial proportions of their sulfur in the pyritic form. As noted in the text, it is already being used successfully by some coal companies in the Midwest, but would be less effective, and much more expensive per ton of sulfur removed, on lower sulfur coals.

The studies we have reviewed suggest that, depending on how an expanded coal-washing program is implemented, SO_2 emissions

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[3] Organization for Economic Cooperation and Development, The Costs and Benefits of Sulphur Oxide Controls--A Methodological Study, OECD, Paris, 1981, p. 52.

[4] See table 8 in footnote 2.

[5] The other two, to be discussed below, are: switching to lower sulfur coal and flue gas desulfurization.

[6] Some of the sulfur found in coal is chemically part of the coal. This is called "organic" sulfur, and cannot be removed by ordinary washing. The remaining sulfur is present in the form of a different mineral, iron pyrites, which is much denser than coal and can be separated by crushing the mixture and separating the lighter coal from the denser pyrites.

could be reduced by somewhere in the range of 0.5 million to 1.5 million tons per year. It should be cautioned, however, that in addition to the limitations of washing low-sulfur coal, several factors noted in the text limit the usefulness of establishing an expanded national or regional coal washing program:

--A lot of high-sulfur coal is already being washed in some states. According to a report prepared for EPA, 72 percent of Illinois utility coal and 52 percent of Indiana utility coal was washed before delivery in 1979.[7] This limits the extent to which further SO_2 emission reductions could be achieved in these states from an across-the-board national program to wash high sulfur coal.

--The total potential emission reductions that a relatively cost-effective coal-washing program could produce are not very large. Even the highest limit of the range presented above, 1.5 million tons of SO_2, would be only about 6 percent of total national SO_2 emissions, or less than 8 percent of emissions in the eastern 31 states.

--A major SO_2 emission reduction program would probably involve some shifts of coals used, with medium- or low-sulfur coals replacing some high-sulfur coals. If such shifts took place, they would reduce the potential for emission reductions by coal washing.

Despite these limitations, coal washing can be effective in making modest, first-step emission reductions or as one part of a more comprehensive emission reduction program.

Switching to lower-sulfur fuels

The final pre-combustion method of lowering SO_2 emissions is replacing the fuel currently being burned by other fuel with lower sulfur content. While a good deal of early utility emission control was done by switching from coal to oil or gas, particularly in the Northeast, this is no longer considered an available method, because oil and gas prices are now much higher than coal prices, per unit of energy. Indeed, the opposite step, conversion back to coal, is being taken by a number of utilities and was a

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[7] Teknicron Research, Inc., Coal Resources and Sulfur Emission Regulations: A Summary of Eight Eastern and Midwestern States, 1981 (Berkeley, Calif.), p. 2-2.

major aim of 1978 energy legislation.[8] Thus, the choice of fuel to reduce SO_2 emissions from coal-fired plants now appears largely limited to lower-sulfur coal.

Large coal-burning facilities are often designed to operate using a particular coal to which they are committed by long-term contracts, and much of a plant will be chosen to match particular physical and/or chemical properties of that coal. Coals come in a wide variety with substantial differences, for example, in Btu content, ash content, and ash properties. Therefore, changing coals could result in a mismatch that might require modification of one or more of several parts of a plant. Possible areas could include pulverizers, ash handling equipment, suspended particulate collectors, and any other material-handling system. There might even be reason to change the combustion chamber itself if, for example, a new coal produced an ash that melted to form a slag which could solidify and block movement of materials in the furnace. Therefore, coal switching may well require some capital investment in addition to the likely changes of both purchase and transportation costs for fuel.

U.S. coal consumption is recognized to be demand-limited, meaning that substantial additional identified deposits of coal exist and could come into production fairly readily if demand for it materialized. Thus, we regard it as feasible that the increased demand for low-sulfur coal, which would come with widespread coal switching, could be met given the resources and the time (years, but not decades) needed to develop new mines and expand coal transportation facilities.

The extent of SO_2 emission reduction possible by coal switching appears quite large upon first examination. As an indication, a report prepared for the Department of Energy[9] found that over two-thirds of the 8.6 million tons of SO_2 emitted annually by the 50 largest utility sources in the eastern United States could be avoided by coal switching. However, there may be substantial limitations on the actual amount by which SO_2 emissions could be reduced by coal switching. These will be discussed further on pp. 167 to 168.

SO_2 CONTROL DURING COMBUSTION

A potentially important approach to SO_2 control at intermediate levels (a range of 50 to 70 percent) in combustion systems has

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[8] Powerplant and Industrial Fuel Use Act of 1978, P.L. 95-620, 95th Cong., Nov. 9, 1978.

[9] PEDCO Environmental Inc., Control Strategies for Coal-Fired Utility Boilers, Cincinnati, May 1982.

surfaced in the past 2 to 3 years. This is a revival of the technique of injecting an alkaline sorbant material such as limestone directly into a furnace, where it will absorb SO_2 as the gas is formed, giving a solid product which can readily be collected before it is released. We call this a revival because furnace injection of sorbants was tested in the 1960's and early 1970's, but was abandoned because it did not capture a sufficiently high percentage of SO_2, compared with post-emission flue gas treatment.

The key to this revival is the recognition that conditions in the furnace are central to determining the extent of sulfur capture by the sorbant. In particular, lower combustion temperatures and more gradual fuel-air mixing result in higher percentages of sulfur capture by sorbants injected into a furnace. This means that the method could potentially be very effective in combination with either of two combustion techniques which are currently under development or entering use at utilities-- fluidized bed combustion (FBC) and combustion modifications used to lower NO_x emissions.

In a fluidized bed, combustion occurs in a bed of pulverized coal and limestone, made fluid by blowing air through the bed from below. FBC is already in use in the United States on small units mostly used only for heat in industrial and commercial systems. It is also being used in a 20-MW(e) pilot scale electric plant located at a Tennessee Valley Authority (TVA) site where it began operation in May 1982. In addition, plans are being developed for a joint EPRI-TVA sponsored 160-MW(e) utility demonstration FBC unit to begin operation, also at a TVA site, late in this decade. If these projects are successful, then FBC may become available for utility use in the 1990's.

However, use of FBC in an existing plant would require replacement of the entire combustion system as well as either major modification or replacement of much of the auxiliary equipment. This would involve a capital investment so large that it would not likely be used as a retrofit technique, except possibly as an onsite replacement for a unit whose combustion system was being retired early. Therefore, we will not consider FBC as a separate retrofit option in this analysis.

In contrast to FBC, combustion modifications being developed and used to lower coal-fired power plant NO_x emissions involve only relatively minor changes in plant facilities. A recent survey of EPRI-sponsored work in this field indicated that:

> ". . . promising NO_x retrofit options with costs as low as $5/kW will be available for many coal-fired units in the next 3-5 years."[10]

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[10] M.W. McElroy, Retrofit NO_x and SO_2 Controls for Coal-Fired Utility Boilers, Coal Combustion Systems Division, EPRI, Palo Alto, Calif., May 1983, p. 2-1.

The additional changes in a power plant that would be required, to retrofit furnace sorbant injection for SO_2 as well as NO_x control, would add more than the $5-per-kilowatt estimated for NO_x control alone. These changes would consist of two things: first, equipment to store and prepare the sorbant and inject it into the furnace and, second, additions or modifications to the post-combustion particulate collection system (whether a fabric baghouse or electrostatic precipitator) to handle a substantial increase in the amount of particulate material produced. However, in many cases they could be accomplished using commercially available equipment, and would result in an overall system substantially simpler, and likely to have much lower capital costs, than the other major retrofittable technology--flue gas desulfurization (FGD) or "scrubbing."

The sorbant most likely to be used in furnace injection is limestone, while a major feature of the NO_x emission control technique with which it would be combined would be staged combustion, using a series of burners with different fuel/air ratios--an arrangement similar to the stratified charge technique used in some automobile engines. Combining these features, the proposed system is usually called Limestone Injection with Multi-staged Burners, or LIMB.

LIMB description and costs

Work on the development of LIMB has been stimulated recently by concern about the high costs of currently available retrofittable SO_2 emission control technologies which would be likely choices for reducing acid deposition. Both EPA and EPRI testified on the subject in joint hearings by two subcommittees of the House Committee on Science and Technology in September 1983.[11] These two statements suggest that retrofitting of LIMB would have capital costs, including necessary particulate collector modifications, in the range of about one-fifth to two-fifths that of retrofitting FGD on the same plant. Also, since LIMB would be a good deal simpler mechanically than FGD, it is anticipated that it should be substantially more reliable. This is an important consideration to utilities, since equipment of low reliability can reduce a plant's availability and require either construction of additional generating capacity or purchase of replacement power at high cost during times of high demand.

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[11] Acid Rain: Implications for Fossil Energy R&D, Joint hearing of Subcommittee on Energy Development and Applications and Subcommittee on Natural Resources, Agriculture Research and Environment, House Committee on Science and Technology, Sept. 20, 1983.

While the capital cost of LIMB is anticipated to be a small fraction of the cost of FGD on the same plant, LIMB is expected to have relatively high operating costs because of a major difference in the chemical process of the reaction between SO_2 and sorbant. In FGD systems the reaction is close to stoichiometric, that is, the amount of sorbant (limestone or other material) used is only a small percentage greater than the minimum amount needed to absorb all the SO_2 produced if every atom of sorbant calcium combined with one SO_2 molecule. In contrast, LIMB systems have shown much less than stoichiometric reaction, with 50 percent or greater SO_2 capture generally requiring between a 2:1 and 3:1 ratio of calcium to SO_2. This means that sorbant costs, which are an important part of total sulfur-removal operating cost, will be two or three times as great for LIMB as for FGD. In addition, the extra sorbant results in substantially more solid waste for disposal, which is another contribution that makes estimated LIMB operating costs relatively high. Still, variable costs for LIMB would not likely be prohibitive, because its maintenance is not expected to be as complex and costly as that for FGD.

According to an EPRI estimate[12] the combination of much lower capital cost and higher operating cost would lead to increases in electricity costs per kilowatt hour for retrofitting LIMB which would be only 30 to 60 percent of those for retrofitting FGD to the same plant burning high-sulfur coal. While this comparison appears to be highly favorable to LIMB, it is not as one-sided when put on a different basis--the cost per unit of SO_2 removed. Because FGD can attain 90 percent sulfur removal, compared to about a 50-percent target for LIMB, the ranges of costs per ton of SO_2 removed for the two methods are relatively close according to EPRI, with the higher estimates for LIMB about overlapping the lower estimates for FGD.[13]

As the comparison stands, the technology to choose would depend on the level of SO_2 control desired, and on whether the added NO_x control from LIMB would also be regarded as a desirable or necessary feature. If moderate SO_2 control, on the order of 50 to 60 percent at a particular plant, were satisfactory, then LIMB would appear to be the preferred technique, provided its reliability was sufficiently well proven that it would not risk causing expensive losses of plant availability. The key question remaining is how long it will be before the reliability of LIMB retrofitting to existing power plants is proven at commercial scale.

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[12] K.E. Yeager, Director, Coal Combustion Systems Division, EPRI, Testimony at hearing, footnote 11.

[13] The EPRI estimates are $650 to $1,000 per ton of SO_2 for LIMB and $1,000 to $1,150 for FGD. (See footnote 12.) We do not necessarily regard these estimates as correct, but we believe they are consistent for comparison purposes.

Current status of LIMB

The current status of LIMB in the United States is that it is at a relatively early developmental stage, mainly at laboratory scale. Both EPA- and EPRI-supported efforts are now underway, looking toward construction of prototype-size (20-80 MW) retrofit units in 1985-86. A usual development schedule would have some testing on a prototype unit followed by construction and then testing of a unit for a boiler of commercial scale (at least several hundred MW). If this program went well, then other utilities might be willing to order similar units starting in the early 1990's, and therefore they could come into use during the middle and later part of that decade.

An important limitation of LIMB retrofitting is the likelihood that it will have to be designed and proven separately for each type of boiler, because the technology is so critically dependent on boiler geometry and conditions. Of the four major types of boilers which make up 93 percent of older pre-NSPS coal-fired generating capacity, we noted in chapter 4 that development work aimed toward prototype design and construction is underway with the two largest groups--tangential or corner-fired (43 percent) and wall-fired units (23 percent). In addition, exploratory work is underway on a third type of unit, cell-fired (another 14 percent), so that programs already in process are aimed at LIMB retrofitting on up to 80 percent of old coal-fired capacity. The fourth large group, cyclone burners (another 13 percent of capacity) is regarded as very difficult to adapt for LIMB, while the remaining 7 percent of the capacity is divided among too many different designs to have received any generic study. EPRI has recently reported that half of the tangential-fired capacity and about three-quarters of the wall-fired capacity appears suitable to LIMB retrofit,[14] about 35 percent of pre-NSPS coal-fired capacity, but this is a minimum estimate based on currently available commercial equipment. In subsequent discussions, EPRI staff has indicated to us that further R&D work is aimed at making LIMB compatible with a substantially greater percentage of pre-NSPS coal-fired utility boilers.

Work on furnace sorbant injection has also been underway in other countries, including Japan, Austria, and Canada, but has been most extensive in West Germany. Steinmuller, a German boiler manufacturer, has demonstrated 80 percent SO_2 removal from a low-sulfur coal by sorbant injection at a 3:1 calcium-to-sulfur ratio in a pilot plant, using its own burner design for wall-fired boilers, and has tested the process in a section of a 700-MW unit. This large unit is now being fully refitted for the new method,

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[14] M.W. McElroy, op. cit., p. 2-13.

with testing planned to start this year (1984). Also, a German electric utility, RWE, has demonstrated 60 percent SO_2 removal by sorbant injection in a 60-MW boiler and is scaling the process up to a 300-MW unit. This application is being done with "brown coal," a lignite which has unusually high (40-60 percent) moisture content (which makes for furnace conditions that are very favorable to effective sorbant injection) and with very low (0.3 to 0.5 percent) sulfur content.

The applicability of these German systems to U.S. coal-burning facilities may be questionable, depending on the compatibility of the Steinmuller burner with U.S. boilers or on the RWE method's being usable with higher-sulfur, lower-moisture coals. However, they do seem to show that furnace sorbant injection can be an effective SO_2 removal method and, therefore, give reason for some optimism about its prospects here.

Time to readiness

A key question about the possible use of LIMB in retrofits to reduce SO_2 emissions for acid deposition control is how long it will take to be ready. The development schedule described above would appear likely to bring LIMB into use in one or two types of boilers (tangential and/or wall-fired) over a period spanning the mid-to-late 1990's, with other boiler types following a few years later. In contrast, most proposed large-scale control legislation, even recognizing the time required to set individual plant as well as state reduction requirements, has aimed to have emission reductions achieved within 10 years, or at most 12 years. Further, programs proposing phased reductions call for first steps in significantly less than a decade.

On the basis of information we have reviewed about LIMB, it seems clear that it could not provide any substantial part of the first step of a phased reduction program. Indeed, even for LIMB to be widely used within 10 to 12 years, its development and demonstration schedule would have to be accelerated from that described earlier. As is often the case, an accelerated schedule would likely require greater expenditures to operate multiple prototype and demonstration units, and/or would face higher risks of difficulties and disruptions at the test facilities if they were started without waiting for full results from previous developmental stages. In the latter case, it would also be expected that substantial added costs would occur to pay for replacement power from unanticipated outages of test units.

POST-COMBUSTION SO_2 CONTROL METHODS

The final group of retrofit emission control methods to be considered are those used to remove SO_2 from combustion products after leaving the furnace--methods generically known as flue gas desulfurization (FGD) or scrubbing.

Scrubbers have become the required approach to sulfur emission control on new coal-fired utility sources in the United States, with 73 already in operation on over 31.9 GW of new capacity as of December 1983. They have been installed in Japan on over a thousand sources--oil-fired as well as coal-fired--many being retrofits, and have also been retrofitted on 31 U.S. utility generating units, totalling over 7 GW of capacity, as of December 1983.

Most current U.S. systems are "wet" scrubbers in which flue gas SO_2 is absorbed by a slurry containing an alkaline material, usually calcium based, such as limestone or lime. These units have the scrubber located after the particulate collector, and the insoluble calcium sulfite/sulfate product is collected separately and disposed of as a sludge. A few spray driers ("dry" scrubbers) in operation in the United States treat the flue gases before the particulate collector. This is done using an absorbing slurry with a low moisture content that evaporates to result in a dry solid waste product collected along with fly ash in the particulate collector. Wet scrubbers can capture 90 percent or more of the SO_2 produced from burning high-sulfur coals, using little more than a minimum 1:1 (stoichiometric) ratio of sorbant to sulfur. Spray driers, using similar sorbant to sulfur proportions, are known to be capable of high sulfur removal on medium-sulfur coals--EPRI estimates SO_2 removal of up to 85 percent from a 2.5-percent sulfur coal[15]--but have not accumulated as much experience in the United States as wet scrubbers.

One other post-combustion method which is not yet in full-scale operation in the United States, but has been successfully demonstrated on a 22-MW unit, involves injecting a dry pulverized alkaline sorbant into the flue gas stream after it leaves the furnace but before it reaches a fabric filter. In this all-dry method, the sorbant (sodium bicarbonate or sodium carbonate have been used successfully to date) collects on the filter and reacts there with the SO_2 in the flue gas stream passing through. The method forms a solid product which can be disposed of with the fly ash that also collects on the filter. It is expected to have much lower capital costs than the other scrubbing approaches because of its lesser complexity, but costs for its sodium-based sorbants are expected to be higher, to the point of controlling total costs. As a result, this dry post-furnace injection is likely to be limited to use with low-sulfur western coals, where its capability of 70 to 80 percent SO_2 removal could be sufficient to meet new source standards or, if required, to accomplish similar emission reductions when retrofitted to pre-NSPS plants. However, unless other lower-cost sorbants are found, this method will be too expensive for use on higher-sulfur coals. Therefore, for use in

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[15] K.E. Yeager, op. cit. (footnote 12) p. 14.

retrofit applications on medium- or high-sulfur coal-fired boilers in the eastern or mid-western United States, to reduce acid deposition, the post-combustion SO_2 removal techniques that can be used will be restricted to wet scrubbers and spray driers.

COMPARISON OF SO_2 CONTROL METHODS

Of the range of SO_2 emission control methods just described, no single one would be the best to apply at all existing SO_2 emitting sources.[16] Rather, the best approach to use would be that combination of different methods at different sources which would yield the desired total emission reduction at the lowest overall cost. Finding this optimal combination, however, would require a great deal more specific knowledge about the individual sources, including their financial condition and the applicability of different methods to them, than is currently available in any single set of information. This is particularly true for retrofitting, because plants already in existence will present many technical constraints that will tend to require individual study, and thus will mitigate against standardized or generic solutions.

As we saw in chapter 4, one approach used to estimate the overall mix of control actions to reach a particular level of SO_2 emission reductions from coal-burning utilities, and the cost and effects of that mix, has been the use of large computer simulation models, such as the one used by ICF Inc., in studies for EPA and other organizations or another used by OTA. These models have several advantages: they use consistent methods and assumptions to treat all facilities; they may be able to take account of overall effects on fuel availability and cost in the event of widespread fuel switching; and they can avoid the possibility of cost estimates being exaggerated to discourage control legislation, which could come from asking individual sources to assess their own control methods and costs. On the other hand, the models lack the detailed knowledge about individual facilities, in particular their compatability with different kinds of fuels, operating methods, and retrofit equipment, which could be included by obtaining site-specific estimates of the methods and costs of controls from operators of the individual sources.

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[16] The one exception to this generalization would be energy conservation actions since, to the extent that they were cost-effective in their own right, they could all aid in reducing SO_2 emissions with net positive effects. Analyzing the prospects for emission reductions by this approach, however, is beyond the scope of this study.

Reconciliation of the results of the numerous studies carried out using these different approaches has not been accomplished--we saw in chapter 4 particularly that the relative cost-effectiveness of scrubbing and coal switching still seems to be uncertain. In the following pages, we will compare the methods available for retrofitting to reduce SO_2 emissions, in order to understand their capabilities and differences. The comparison will be focused on the following 4 categories:

--Maximum SO_2 reduction capability.

--Major constraints on applicability and possible ways to overcome them.

--Comparative costs and factors that determine costs.

--Side effects and infrastructure considerations.

Maximum SO_2 emission reduction capability

Of the four methods being compared, it is clear that coal washing is most limited in its capacity for SO_2 emission reductions. This is primarily because it can only remove pyritic sulfur (the form not chemically bound to the coal), and also because a good deal of eastern utility coal is already being washed. Application to the remaining high-sulfur coal being burned might increase its emission reduction contribution by as much as 1.5 million tons of SO_2 annually, or possibly a bit more in the relatively unlikely event that some plants choose to change to higher sulfur fuels and retrofit scrubbers. Beyond this level, washing would rapidly become prohibitively expensive--up to thousand of dollars per ton of SO_2 removed--to apply to low-sulfur coals, because its costs are largely proportional to the amount of coal processed, rather than to the amount of sulfur removed.

Application of LIMB, if it is successfully developed, could remove 50 percent or more of the SO_2 produced in the plants to which it is applied. This amount would depend on what portion of the pre-NSPS coal-fired capacity was found to be compatable with LIMB. This portion has already been estimated as at least 35 to 38 percent, on the basis of presently available equipment, and there appear to be prospects that it can rise to two-thirds or more. Thus, LIMB should be able to control at least somewhat over 2 million tons of SO_2 annually, with a possible increase to 5 or even more than 6 million tons, if it were to perform at the high end of its anticipated removal rate on most of the pre-NSPS plants. However, even with prompt acceleration of LIMB development efforts, there is little chance that much of this could occur until the 1990's, and without such acceleration, it would not be until the middle-to-late part of that decade.

Switching to lower sulfur coals has, on its face, the capacity to reduce utility SO_2 emissions by almost 75 percent because, while a good deal of western U.S. coal contains less than 0.5 percent sulfur, coals burned by utilities in 1982 averaged 1.51 percent sulfur nationally, and 1.88 percent in the East.[17] Thus, for the eastern coal-burning utilities, widespread switching to low-sulfur coals could apparently reduce SO_2 emissions by the order of 10 million tons. However, this estimate is much higher than could actually be reached for two major reasons.

The first is that most eastern coals average over 12,000 Btu's of heat value per pound, and even the somewhat lower heat value Illinois basin coals still average over 11,000 Btu's per pound. In contrast, the most geographically accessible western low-sulfur coals average under 9,000 Btu's per pound, and some, those from North Dakota, average under 6,600 Btu's per pound. Therefore, somewhere between about 25 and 90 percent more of these coals would have to be burned per unit of heat output than the amount of eastern coal substituted for, so the actual reduction of sulfur content would be proportionately less than the comparison of sulfur percentages implies.

There is an area of eastern coal sources--Districts 7, 8 and 13 in parts of Virginia, West Virginia, Kentucky, Tennessee, Alabama, and Georgia--which yield relatively low-sulfur coals with heat values of over 12,000 Btu's per pound, but their sulfur content averages in the range of 1.1 to 1.3 percent. Thus, even if these eastern districts could expand their outputs to meet the potential demand from the rest of the East, the resulting reduction of annual eastern SO_2 emissions would only be about half as much as the 10 million tons estimated earlier.

The second reason why coal switching would result in a significantly smaller reduction of SO_2 emissions than the simple estimate made earlier, is that changing a plant to a lower-sulfur coal with a lower heat value than the coal it was designed for would result in "derating" (lowering) the output of the plant. Even in the absence of any other conversion costs, such a derating would raise the cost of the power produced by the plant to an extent directly determined by the reduction in heat value. For example, if an 8,700 Btu/lb Powder River Basin coal was substituted for an 11,600 Btu/lb Ohio coal, then the plant's output from the same weight of coal would drop by one-quarter, making the cost per unit of electricity produced rise by one-third. Such an increase would be quite severe, compared with the small percentage increases in electricity rates that have been estimated in a number of region-wide studies of acid deposition control costs. Furthermore, the

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[17] Energy Information Administration, Cost and Quality of Fuels for Electric Utility Plants, 1982 Annual, DOE/EIA-0191 (82), Aug. 1983, tables 53, 54, and 56.

cost increase would almost surely be higher, since it is quite probable that modifications to the coal preparation and particulate control systems in the plant would also be required to accommodate the change of coals. Thus, switching to very low heat value Northern Great Plains coals seems unlikely to be used as an SO_2 emission control measure on any major scale in the East.

Some southwestern coals have somewhat higher average heat values, ranging from about 10,000 to over 11,600 Btu/lb. However, the total tonnage produced from all of these areas--Districts 16, 17 and 18 in Colorado, New Mexico, and Arizona, and District 20 in Utah--was just over one-third of that produced in the northern Great Plains in 1982. Furthermore, they are produced in areas more distant from most eastern coal-burning plants, so they do not appear likely to offer as much coal for substitution in the east as a major switching pattern would demand. Taken together, these three considerations--relatively high sulfur contents in the lower-sulfur eastern coals, low heat values in most of the lowest sulfur western coals, and long distances for the high-heat value low-sulfur western coals--suggest a limit to the SO_2 emission reduction capacity of coal switching far below the first estimate of 10 million tons per year. While the correct value is an important unresolved question, one estimate given for this limit, in a recent EPRI statement, is between 1.5 million and 3 million tons of SO_2 per year.[18]

Finally, the SO_2 emission reductions potentially obtainable by retrofitting FGD systems are the greatest for any of the methods under examination. Scrubbers can be built to remove 90 percent of the SO_2 emitted from a plant. Therefore, the upper limit to the possible reduction by scrubbing will be 90 percent of the emissions from all pre-NSPS units, diminished only to the extent that

--some of these units already have scrubbers in place or under construction and

--others may have space constraints at their sites which will prevent the installation of scrubbers.

We have not obtained a detailed inventory showing which plants would be candidates for scrubbing after excluding those in

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[18] K.E. Yeager, op. cit. (footnote 12) p. 7. This estimate is as low as it is due both to the technical and economic factors cited above and also to what the author calls "institutional constraints" arising from situations such as mine-mouth plants which are not equipped to accept coal shipments from elsewhere, or potential state interventions to protect the economies of local high-sulfur coal-producing regions.

the two classes just noted, but it seems clear that substantially more than half of the pre-NSPS coal capacity would be included, and the potential for SO_2 emission reductions from these plants would be over 8 million tons per year.

We base this estimate on comparison to the study,[19] done for EEI, of H.R. 3400, a House bill under which scrubber retrofitting is mandated for a significantly more limited set of older plants.

The TBS study found that H.R. 3400's first requirement, of scrubber retrofitting at the 50 highest emitting plants, would cover some 195 units with a total of 67.9 GW of generating capacity which emitted 7.9 million tons of SO_2 in 1980. If all those plants could be retrofitted with scrubbers removing 90 percent of their SO_2, it would reduce annual SO_2 emissions by about 7 million tons. The study, however, excluded from the scrubber requirement not only the two groups of plants previously noted--those already being scrubbed, and those where scrubber installation would be prevented by space constraints--but also older and smaller units where scrubbers were not anticipated to be economic. This last exclusion covered all units smaller than 200 MW built before 1965, and all units of larger sizes built before 1960. Taken together, the exclusions covered 20.2 GW of capacity in 115 units which together were responsible for 2.3 million tons of SO_2 emissions. The study then found that 4.93 of the remaining 5.5 million tons of SO_2 emissions could be controlled from the 80 units to be scrubbed.

The study also considered the possibility that further emission reductions, to be carried out on other plants to reach the bill's total goal of a 10-million-ton reduction of annual SO_2 emissions, could include scrubbing another 40 GW of capacity. This would control an additional 2.8 million tons of SO_2 emissions, giving a total 7.7-million-ton reduction of annual SO_2 emissions by scrubbing. We need only add to these a small amount of additional capacity being scrubbed--for example, some of the plants excluded from the EEI study because of their age if their owners were to decide to extend the plants' lifetimes--and scrubbing could be capable of reducing annual SO_2 emissions by over 8 million tons.

Constraints limiting SO_2 emission reduction techniques

The constraints which limit the capabilities of available SO_2 emission reduction techniques are, in general, peculiar to the

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[19] Temple, Barker & Sloan Inc. (TBS) Evaluation of H.R. 3400, The "Sikorski/Waxman" Bill for Acid Rain Abatement, Presentation to Steering Committee of the Clean Air Act Issue Group, Sept. 20, 1983.

individual techniques. In some cases there may be ways of overcoming the constraints, while in other cases, the constraints are intrinsic to the technique and not likely to be overcome without turning to another method.

Coal washing

Coal washing is only capable of removing sulfur which is mixed with the coal in the form of pieces of a separate compound, iron pyrites. The separation is carried out by crushing the raw coal and then using some physical property which differs between the two substances--usually density, since pyrites is several times as dense as coal. Therefore, the constraints on coal washing are the proportion of sulfur in a coal which is chemically separate from the coal, plus the limitations which are intrinsic to physical separations.

Coals with substantial proportions of non-chemically bonded sulfur are mostly restricted to those which have high sulfur contents, so the scope of applicability of coal washing for sulfur removal is not likely to spread to other coals beyond the types presently being washed. As to improving the separation process, the two factors involved are the size to which the raw coal is crushed before the separation step, and the amount of useful coal allowed to be discarded as waste. Since both of these factors have a direct impact on the cost of the energy supplied in the finished coal, decisions on the extent of crushing and on the proportion of coal lost as waste are made on economic grounds. The utility industry is in the process of optimizing its coal-washing activities with the help of an EPRI-sponsored testing plant, and we would expect little more progress beyond this, except to the extent that techniques were developed with greater sensitivity in separating coal from non-coal substances. The only prospect we know of is the possibility of chemical coal cleaning techniques, which would be able to remove sulfur chemically bound to the coal, in addition to the separate compounds physically mixed with the coal. This, however, is a method which is far from ready for application, so it would not be likely to be available for wide scale use in this century.

Coal switching

The key constraint on coal switching, as we understand it, is the deliverability of coal which has both low sulfur content (near or under 1 percent) and other physical properties compatible with units now burning high-sulfur eastern coals, that is, high heating values (near or above 12,000 Btu/lb) and similar grindability and ash properties.

Lower-sulfur eastern coals would likely be satisfactory substitutes, although they would not produce as great a reduction in emissions as some western coals might because the lower-sulfur eastern coals' sulfur content tends to run at or somewhat over 1 percent, compared with western coals whose sulfur content runs as low as 0.5 or even 0.4 percent. However, while coal in the United States overall is demand-limited, there is not an unlimited supply of low-sulfur eastern coal. If it were being sought for switching by a large proportion of eastern utilities, its price could be bid up to the point of raising costs above those for other methods of reducing SO_2 emissions.

Higher heat value low-sulfur coals produced in some areas of the Southwest would seem to be the second choice for replacement in eastern plants, but in this case the cost of transportation could once again raise prices above those for other methods of emission reductions. Some prices for high-heat-value coals, to illustrate this point, are shown in table III-1.

Table III-1

Average Delivered Price per Ton for High-Heat-Value Coal,
Local High-Sulfur vs Southwestern Low-Sulfur, as
Used in Midwestern and Southwestern States, 1982

Consuming State	State of Origin, Heat Value, and Price					
	Intrastate or (local)		Colorado, District 17		Utah, District 20	
	Heat Value	$ per ton	Heat Value	$ per ton	Heat Value	$ per ton
Illinois	10,727	$32.87	12,257	$57.97	-	-
Indiana	10,928	28.94	11,268	64.74	12,287	$83.23
Mississippi	(Kentucky) 12,259	42.73	11,573	66.32	12,076	73.80
	(Illinois) 12,203	41.26				
Colorado	-	-	10,497	22.84	-	-
Utah	-	-	-	-	11,541	32.58

Source: Energy Information Administration, Cost and Quality of Fuels for Electric Utility Plants, 1982 Annual Report, DOE/EIA-0191(82), table 53.

In two midwestern states which produce much of their own coal, Illinois and Indiana, local coals cost about $33 and $29 per ton in 1982, while Colorado coal cost about $58 and $65, and Utah coal, used only in Indiana, cost about $83. Even in a nonproducing state, Mississippi, local high-sulfur coals cost about $41 and $43 per ton, while the western coals cost about $66 and $74, respectively. Thus, there were substantial price premiums, from $25 to $54 per ton, paid where these two high-heat-value low-sulfur western coals were used in the Midwest. In addition, neither of the two coals was used elsewhere east of the Mississippi--other than in the three states mentioned in the table--suggesting that, despite their quality, their transportation costs become prohibitive at longer distances.

In fact, of all utility coal produced in the Mountain States--Districts 16 through 22--including coals of lower heating value, only 30 million of 201 million tons (about 15 percent) were used east of the Mississippi in 1982, and almost all of that in the East North Central region. Considering the entire East, this coal only provided 8 percent of the total tonnage consumed by utilities. Essentially no Mountain State coal at all was burned by utilities in any of the New England, Middle Atlantic, or South Atlantic States, and the 1.1 million tons of Colorado and Utah coal burned in Mississippi was the only western coal used by utilities in any of the East South Central States.

It would seem, therefore, that despite their low sulfur content, Mountain State coals have been and are likely to remain constrained by transportation costs from making substantial contributions to SO_2 emission reductions anywhere in the East, outside the East North Central States. As for the prospect of overcoming this constraint, it would appear more likely that the difficulty of displacing eastern with western coals will increase, rather than decrease, because of the concerns over regional economic effects, to be treated in the final section of this appendix.

LIMB

The amount of SO_2 emissions that could be controlled by LIMB retrofitting is limited by the proportion of sulfur which can be captured in any particular furnace, and the extent to which the range of existing furnace designs can be adapted to the technology. At this time, sulfur capture rates at or possibly above 50 percent are believed to be achievable. However, exact values, and also the extent to which capture rates can be increased or the combination of capture rate and control cost per unit of sulfur can be optimized will only be determined from experience. Similarly, currently available equipment is believed to be retrofittable on over one-third of existing coal-fired capacity, with studies underway to extend this to as much as 80 percent. Again, limits will be determined as the result of ongoing and proposed work. In both areas, the main step to determine, and possibly overcome, these limits would be to intensify and accelerate the pace of those development efforts, which would require additional financing in the next few years.

The other constraint which might limit utility use of LIMB is possible reliability problems in some of the equipment which would be installed on a LIMB-equipped boiler. If such problems occurred, they would force the boiler, and therefore the entire plant, to shut down for maintenance, thus requiring the utility to either have extra generating capacity available or purchase replacement power. Some utility representatives have told us that, because LIMB equipment would be part of the boiler rather than being in a separate part of the plant as a scrubber would, they would prefer to meet an SO_2 control requirement by scrubbing half their plants rather than putting LIMB on all. In this way, they explained, scrubber maintenance could be done without causing overall plant shutdowns, and thus avoid the threat of major cost penalities arising from reliability problems in sulfur control equipment.

Scrubbing

Aside from cases where scrubbers were already installed or where space was not available to fit them, the major factors which could constrain scrubber use are the prospect of reliability problems and the fact that scrubber costs are more capital-intensive than those of other sulfur control methods.

Most scrubbers in the United States are installed on plants which are required to have continuous scrubbing, and which are not allowed to operate if the flue gases are not being scrubbed. If a scrubber unit at one of these plants were not available when required, the boiler would have to shut down. In this event, the scrubber's reliability could limit the reliability of the entire plant.As a way of avoiding very high costs stemming from lowered plant reliability, utilities have installed spare scrubbing units, thus paying a capital cost premium to avoid even higher costs for replacement capacity or power. In this way, the reliability and high capital cost problems are compounded for scrubbers.

However, in the case of scrubbers installed to reduce total emissions, as would be done to lessen acid deposition, a spare scrubbing unit need not necessarily be required. Instead, periods of higher emissions, when no scrubber was available, could be balanced against other periods of lower emissions, as long as total emissions over a longer interval remained below the required limit, and the high-emission episodes did not violate ambient air quality standards. Thus, if laws or regulations to reduce acid deposition were adopted that did not require continuous scrubbing, the affected sources could avoid this compounding of the reliability and capital cost problems.

In addition, reliability problems can be alleviated by combining coal washing and scrubbing. EPRI has noted that coal washing has a particular value in improving scrubber reliability.[20]

In the past decade, rising and fluctuating interest rates, as well as dramatically increased costs and times for new plant construction, have made utilities less willing, and often less able, to undertake other major capital expenditures. While this situation appears to have eased somewhat since 1980, it is still quite likely to result in utilities preferring a lower rather than a higher capital-cost approach to possible added emission control requirements. This could possibly even lead to utilities choosing a control method having greater overall costs but lower capital costs than an alternative, just as some utilities have continued to use higher-cost oil- or gas-fired plants, rather than face the difficulties of raising capital to pay for coal conversion, even though the total cost of generation could be lowered by the conversion.

Scrubber installation would be expected to be limited because of this preference, particularly because acid deposition control would occur in retrofit situations, where scrubber capital costs usually run higher than in new installations. The major way that has been proposed to overcome this capital cost constraint seems to demonstrate just how seriously the limit is regarded. H.R. 3400 would not only mandate full-time scrubbing on the 50 high-emission-rate power plants in the United States with the greatest annual SO_2 emissions--to ensure that scrubbing was used--but would also reimburse the utilities for 90 percent of their capital costs for these scrubbers, and possibly also further capital costs of emission control technology such as scrubbers used on other plants.

Comparative costs and factors that determine costs

A main finding about emission control costs that we have reached on the basis of our work is that there is no single lowest-cost method for controlling SO_2 emissions. Rather, for each facility, the local equipment and conditions will combine in a unique way which can result in one method being more cost-effective than others at a certain level of emission reduction, and then make another method, or several others, more cost-effective at higher or lower levels of control.

To compare the costs of the different possible methods on a consistent basis, it is necessary to have a common measure which

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[20] K. Yeager, op. cit. (footnote 12) p. 14.

can be used for all of them. The best measure of cost-effectiveness we have found is the cost per unit of control, expressed as dollars per ton of SO_2 removed. This cost, for any given control method, will vary between sites and vary with the percentage of control at any site, so no single value will describe it adequately. Possibly the best overall picture can be obtained by drawing a "supply curve" for each control method, that is, a graph of how cost-effectiveness varies with the amount of control, and then comparing the supply curves for the different methods. This has been done in figure III-1, which compares a set of supply curves for coal washing, coal switching, and wet scrubbers. These curves were drawn[21] on the basis of data compiled in preliminary studies. Therefore, we present them without necessarily confirming their detailed shapes or specific dollar values, but rather as an example to illustrate the kinds of differences that can exist between the costs of different methods. In addition, this figure emphasizes the point that for each method, the cost per unit of SO_2 controlled increases at higher levels of control, with the cost per ton eventually increasing very sharply, as the method is applied to less and less compatible sources.

Figure III-1 also shows that coal washing and switching offer the first modest levels of SO_2 control at lower costs per ton than scrubbers. However, at intermediate levels of control, they reach the limits of their favorable applications, and their costs increase rapidly to values per ton substantially greater than scrubbing. As discussed above, there is more disagreement about the shape of the switching curve than the others in the figure, and our examination of switching's capacity suggests that it might actually remain competitive with scrubbing up to a substantially greater SO_2 tonnage than the 1 million to 1.5 million tons shown in the figure. EPRI[22] has suggested that, taking account of both technical and political constraints, switching's capacity may be limited to between 1.5 million and 3 million tons of SO_2 controlled.

Supply curves for control can be developed separately for each technique. In contrast, an overall approach to any widespread emission reduction program, if it were to aim at minimizing costs, would want to use the lowest-cost increments of each method, rather than follow any one method to its limit. Deciding on which combination of locations, control methods, and amounts of control to choose in order to minimize overall costs could not be done on a centralized basis, however, because of the lack of complete information on the details of possible levels and methods of control at all emission sources.

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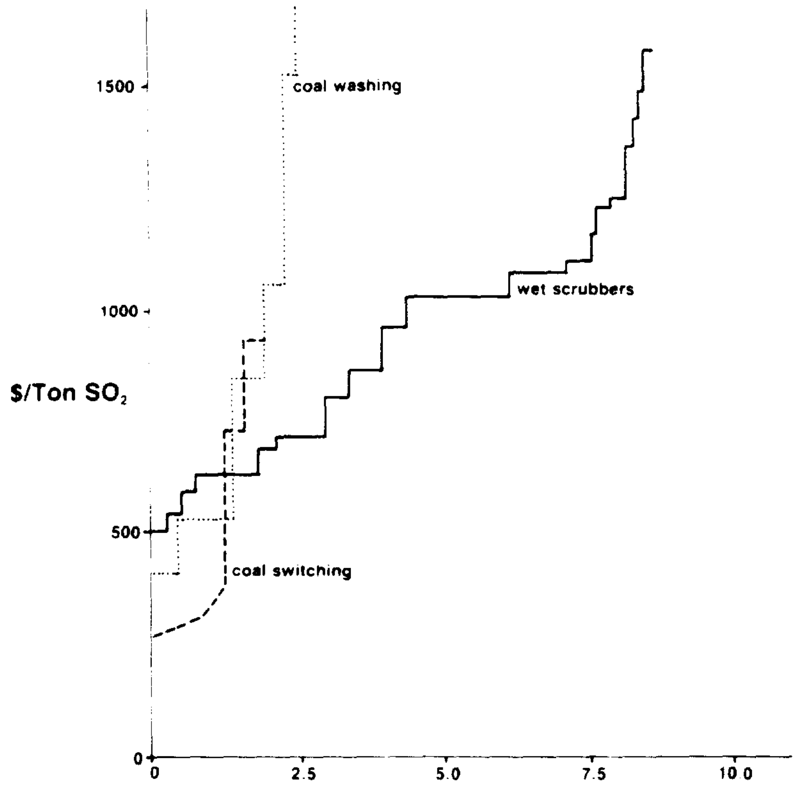
[21] J. Gruhl, Issues Relating to the Costing of Acid Rain Control Technologies, Gruhl Associates, P.O. Box 36524, Tucson, Ariz. 85740, Apr. 1983.

[22] K.E. Yeager, op. cit. (see footnote 12).

Figure III-I

Preliminary "Supply Curves" for SO_2 Controls



Million Tons of SO_2 Controlled

Preliminary "Supply Curves" for SO_2 Emission Control in the Eastern United States

Each zigzag line shows the relationship between amount of SO_2 controlled and the cost per ton controlled for one control method, as it could be applied in the eastern U.S. Scaled up from estimates covering less than half of utilities in the region. An overall control strategy could not combine all the lowest parts of all three curves, because some facilities will have low costs for more than one control method.

Source: J. Gruhl, Issues Related to the Costing of Acid Rain Control Techniques, Gruhl Associates, P.O.B. 36524, Tucson, AZ., 85740, April 1983, pp. 43-51 and figure 3.1-2. (Based on utility analyses published in Senate Environment and Public Works Committee report cited in chapter 4, Footnote 41.)

To deal with this situation, some control proposals have suggested that emission reductions be assigned on a rough basis to major sources, and that those sources could then seek to lower their costs by trading emissions with other sources which were able to reduce their emissions more cheaply. While this method, using a "market" approach to reduce costs, seems in principle a useful way to deal with the lack of detailed information in the hands of regulatory or policy-making authorities, in practice there would be barriers limiting the extent to which emission trading could operate on a wide scale. As noted in chapter 4, we examined the prospects for such trading in a bill analysis, and said that there was some precedent for and successful experience with it on a local basis. However, we found that extending it over wider areas, and particularly across state lines, would be constrained by the difficulty of obtaining cooperation and approval from regulatory bodies in different jurisdictions, suggesting that it would fall significantly short of attaining an ideal least-cost approach to a desired level of emission reductions.

Still, it would seem that once required levels of emission reductions were established, allowing the affected sources to choose their own methods to reduce emissions would tend to result in lower costs than would come from prescribing methods. This should hold except to the extent that particular constraints (such as utility difficulties with raising capital, which could make scrubbers hard to finance) tipped some decisions away from methods with potentially better overall cost-effectiveness.

An important cost-effectiveness question is the extent to which waiting for the developing technology, LIMB, could offer significant savings in emission reduction costs, both at individual sources and in a region-wide program. Much attention has recently turned to LIMB because of indications that it may offer substantial savings in the cost of controlling SO_2 emissions.

As we noted in chapter 4, capital costs for LIMB, including required modifications in particulate collectors, have been estimated in the range of one-fifth to two-fifths those of retrofitting FGD on the same plants. Also, total increases in electricity cost from installing LIMB have been estimated at between 30 and 60 percent of the cost increases stemming from installing scrubbers on the same plants burning high-sulfur coal. However, when converted to our overall cost-effectiveness criterion, dollars per ton of SO_2 controlled, the advantage of LIMB seemed much less impressive. Because scrubbing can attain 90 percent sulfur removal, compared with a target of about 50 percent for LIMB, the ranges of cost-effectiveness for the two methods are relatively close according to EPRI, with the high end of the cost range for LIMB overlapping the low end of the range for FGD.[23] Thus, the advantage of LIMB for SO_2 control appears to be a difference of only 15 to 35 percent in cost, rather than a factor of 2 or 3. Seen in this light, delaying scrubber retrofitting to await LIMB development would appear harder to justify, unless the added advantage of LIMB, in offering substantial NO_x control as well as SO_2 control, was found to be of significant value.

Costs unique to retrofits

Finally, some cost factors are often not taken into account in examining relative and overall costs of retrofit SO_2 emission controls. Two of these, discussed in chapter 4, are the "retrofit cost penalty," which arises in the difficulty of finding space and arranging for the layout of control equipment on a plant which was not planned to accommodate added equipment, and the higher capital charges that would have to be paid each year for a plant which has only a relatively short remaining lifetime over which to amortize its capital cost. This latter reason is why there have been suggestions that plants over a certain age, say 15 to 20 or 25 years, be exempted from a requirement to retrofit scrubbers.

However, as discussed in appendix I, there is substantial interest in extending the lives of older plants to delay the need for new plant construction. One possible approach to this issue, recently adopted in emission control requirements in West Germany, is to exempt plants from retrofit requirements if they are decommissioned after a limited further amount of operation, but to require retrofits on plants which are to continue operating longer.[24] This could allow utilities to decide on which plants would be kept in operation over extended lifetimes, with sufficient time to amortize retrofit controls at lower annual costs, and which ones, to be closed by the chosen cut-off limit, could be exempted from adopting controls.

Three other kinds of costs can arise out of retrofitting:

--Replacement energy or capacity, to substitute for the affected unit during construction.

--Replacement energy or capacity for that used in regular operation of the control system.

--Replacement capacity needed by the utility to maintain reserve margins, in the event that the control system's reliability lowers the overall reliability of the affected unit.

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[23] See footnote 13.

[24] See App. IV.

Of these three costs, the first (substituting for the affected unit if it must be shut down during some part of the controls construction) would be a one-time cost, likely to be treated as part of the control capital cost and amortized over the period of subsequent operation. This could be minimized if the construction shutdown can be scheduled to overlap with regular shut-downs for maintenance.

The cost to replace energy consumed in regular operation of the control system can be a significant one. Scrubbers are estimated to take 3 to 5 percent of total plant energy for their operation, mainly for pumps and blowers, and to reheat flue gases to avoid possible downwash of the plume and/or damage to the smokestack from condensation. This should be recognized as a legitimate part of control costs, and included as an ongoing cost of operation. On the other hand, this cost would be exaggerated if it were accounted for by amortizing, as part of the control system, the full cost of building new replacement capacity, since such new capacity would be quite likely to long outlive the older unit and its retrofit controls. Thus, the proper method of accounting for this cost would be to charge to the control cost only the routine cost of generating or buying replacement energy for that used by the retrofit control system during its lifetime.

The third extra retrofit cost will occur if the retrofitted control system has low reliability of performance, which results in reducing the overall reliability of the unit on which it is installed. In this case, again, there will be recurring costs either to replace the energy the unit would produce or to maintain additional capacity to keep up a necessary reserve margin. If the control system's reliability is low enough,[25] and the unit could not be operated without the control system, this could become the largest of the three extra retrofit costs treated here. On the other hand, if the unit could keep generating while the control system was repaired and restarted, then this cost could be minimized. Spare scrubbers units are used to accomplish this on NSPS plants but, as noted above (p. 173) this problem need not occur, and the cost of the extra scrubbing unit could be avoided, if continuous scrubbing is not demanded on older plants where scrubbers are retrofitted only to control total emissions.

Side-effects and infrastructure considerations

Each of the potential methods of emission control would have some interactions with the rest of society, either positive or negative, which could be important to the prospects for use of the

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[25] EPRI has stated that scrubber availability on medium- and high-sulfur coal was 53 percent in 1978, and had reached only 85 percent by 1981. See footnote 12, p. 12.

method. These include transportation and waste disposal issues, availability of industrial capacity to build or install necessary equipment, the amount of time needed to apply the method on a wide scale, and, of most sensitivity, employment impacts on affected industries and regions. We review these here by methods.

Coal washing

The main point to note here about coal washing is the fact that, by separating out a sizeable portion of non-coal matter from the raw coal, washing produces a product of improved quality. The washed coal will have a higher heat value, lower ash content, and reduced content of hard-to-grind foreign matter, in addition to lowered sulfur content. Indeed, washing is not necessarily done soley to remove sulfur--both Peabody Coal Co., and EPRI have noted that it can reduce transportation, storage, and handling costs and also improve boiler performance, thus giving savings beside sulfur removal that can help pay for its costs.[26]

On the other hand, coal washing does pose one difficulty; it produces a combination of solid and liquid wastes which may present disposal problems. We have not explored this issue, but it would not necessarily seem to pose much more or less of a problem than the wastes gathered from the emission-control and ash-handling portions of a power plant after combustion.

Scrubbing

Two points about the timing of potential scrubber installations merit attention here, both relate to capacity considerations. If scrubber retrofitting were to be adopted as a major part of an acid deposition control strategy, the amount of generating capacity involved could be substantially larger than the entire amount of scrubbers operating today. These would have to take a number of years to design and construct, simply because the existing control equipment industry would need time to expand to meet such a surge of business. By the same token, the utility industry would also need to stage scrubber installation over a number of years, so that only a limited share of generating capacity would be shut down for scrubber installation and start-up at any one time, allowing enough capacity to remain on-line to meet demand. Thus, for reasons arising in both the control and utility industries, a major scrubber program would have to be phased in over at least a major part of a decade, if not more.

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[26] C. Farrand, Vice-President of Peabody Coal Co., testimony before Senate Committee on Environment and Public Works, Oct. 29, 1981, and EPRI Journal, Nov. 1981, p. 41.

The construction, installation, and subsequent operation of a greatly increased amount of scrubbing capacity would have some impact on the labor market, providing a noticeable increase in employment for certain technical specialties including plant construction workers, chemical process operators, and engineers.

Finally, a major increase in scrubbing would correspondingly increase the rate of production of waste products--either sludge from wet scrubbers or solids from spray driers--which will greatly exacerbate a disposal problem that is only now starting to receive significant attention in the United States. We would note that in Japan, a somewhat more densely crowded nation where scrubbing has been practiced proportionately more and earlier, this issue has been important enough to lead to a great deal of effort to see that scrubber wastes can be dealt with conveniently. This has led to some use of regenerable scrubbing, which yields much less waste, or at least to seeing that the calcium/sulfur waste is fully oxidized to gypsum (calcium sulfate), which can be more easily collected as a solid and potentially made into useful by-products.

LIMB

Wide-scale retrofitting of LIMB could involve some of the same capacity issues as scrubbing: not all the units to be retrofitted could be done at the same time, both because of limited capacity in the boiler industry to do the work and because of the need to stagger installation so that most plants could remain online to meet power demands. As a result, extensive LIMB installation would have to take some years after adequate demonstration of its reliability had been carried out. On the other hand, LIMB's ability to give significant control of NO_x as well as SO_2 may be seen as progressively more desirable over time, as the effects and impacts of NO_x emissions become better understood. Such an enhancement of LIMB's value could provide it further support and incentive, and result in its wider application.

Coal switching

Three infrastructure/side-effect issues will affect the prospects for coal switching, and all three are likely to slow its growth and/or limit its extent.

The lesser two issues, mining and transportation capacity, could give rise to a transitional period of some years during which switching costs could be extremely high, and wide use of switching would be delayed, until the economy had time to accommodate in both the mining and transportation sectors. A small amount of switching could likely start very soon. However, it would take about 5 or more years to finance and develop new low-sulfur coal mines and the transportation equipment and capacity to move the coal. Attempts to accelerate switching faster than these industry sectors could respond would likely result in sharp price increases, if there were more users bidding for supplies than the amounts that could be mined and delivered in the short run. It appears likely, however, that supply and transportation would not constrain major shifts on time scales running a decade or more, considering, for example, that U.S. coal production grew by over 260 million tons in the 1971-81 decade,[27] which was more than 60 percent of total utility coal consumption in the eastern 31 states in 1982.

Much more important though, would be the impacts that widespread switching would have both on mining and related employment, and on regional economies, in the high-sulfur coal producing areas in the East. The main regions that could experience mining and employment losses are northern Appalachia (Pennsylvania, Maryland, eastern Ohio, and northern West Virginia) and the Illinois Basin (Illinois, Indiana, and Western Kentucky).

We estimated, in chapter 4, that between about 4,300 and 7,000 high-sulfur coal miners would be affected for each million tons of SO_2 emissions reduced by switching. These figures suggest considerable employment impact for any SO_2 emission reduction program which relies heavily on this technique. Two factors would partially offset these effects:

--Long-term increases in eastern coal production are expected over the next 2 decades.

--In some states, such as West Virginia, the availability of new or expanded low-sulfur coal mines might help to offset some of the negative effects in other high-sulfur coal mines.

Nevertheless, we would expect that coal production and mining-related employment in high-sulfur coal regions would be affected by any large-scale utility shifts to low-sulfur coal. Even in West Virginia, it would be an over-simplification to say that the negative effects in one part of the state could be offset by the positive effects in the other, because miners could not easily relocate--both because of strong local traditions and the very different types of mining which are practiced in the two parts of the state.

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[27] Energy Information Administration, <u>1982 Annual Energy Review</u>, DOE/EIA-0384(82), Apr. 1983 p. 5.

STRENGTHENED EMISSION CONTROLS

IN WEST GERMANY

Responding largely to widespread and rapidly increasing damage to and death of trees, especially evergreens, occurring in forests throughout much of the country, the government of the Federal Republic of Germany, on June 22, 1983, published the final version of a new Ordinance on Large Firing Installations (Verordnung uber Grossfeuerungsanlagen), which took effect on July 1, 1983.

This, the Thirteenth Ordinance Implementing the Federal Immission Control Law (13th BImSchV), established new emission limits for a number of pollutants from what would, in the United States, be called large stationary fuel-burning sources. The pollutants covered included particulates, carbon monoxide, halogen compounds and, particularly relevant to the acid deposition question, oxides of nitrogen and oxides of sulfur. Also relevant to the acid deposition question is the fact that the ordinance establishes emission limits for existing as well as new sources. Most significant is the limit for SO_2 emissions from existing sources larger than 300 MW(t) (about 100 to 120 MW(e), which is a relatively small electric generating unit): if the plant is to be used for more than 30,000 hours--which amounts to less than 6 years at a 60-percent capacity factor--then the plant must be controlled to meet the new source limit.

These limits are stated as maximum concentrations of the pollutant allowed in exhaust gas from the source, whereas U.S. emission limits are stated in terms of maximum weights of pollutant that may be released per unit of heat supplied in the fuel burned, expressed as pounds per million Btus (lb/mmBtu). To compare these with United States limits, it is necessary to know how much excess air is mixed with the fuel during combustion, which is specified in the German limits by indicating the proportion of unconsumed oxygen left in the exhaust gas. Variation in excess air levels leads to the occurrence of the small range of values, instead of a single value, shown for the equivalences in table IV-1.

The table shows that SO_2 emission limits for most large new coal-fired sources in the United States are almost 50 percent higher (less stringent) than those in West Germany, while those for sources using very high sulfur coals are nearly twice as high (half as stringent) in the United States as in West Germany.

Table IV-1

SO_2 Emission Limits for Solid Fueled Sources Larger Than 300 MW(t)

New sources	German limit	Equivalent in U.S. units[a]	U.S. limit
Most cases	400mg/m^3 and 15 percent	0.40-0.43 lb/mmBtu	0.6 lb/mmBtu and 30 percent
Very high-sulfur fuels	650 mg/m^3	0.65-0.68 lb/mmBtu	1.2 lb/mmBtu
Existing sources			
To be used: up to 10,000 hrs	Existing limit	--	Varies by plant
10,000-30,000 hrs[b]	2500 mg/m^3	2.5-2.7 lb/mmBtu	Varies by plant
over 30,000 hrs[b] and after April 1993	400 mg/m^3	0.40-0.43 lb/mmBtu	Varies by plant
Very high-sulfur fuels	650 mg/m^3	0.65-0.68 lb/mmBtu	Varies by plant

[a] As calculated by EPA, Industrial Environmental Research Laboratory, per letter Apr. 2, 1984, confirming approximate calculations by GAO.

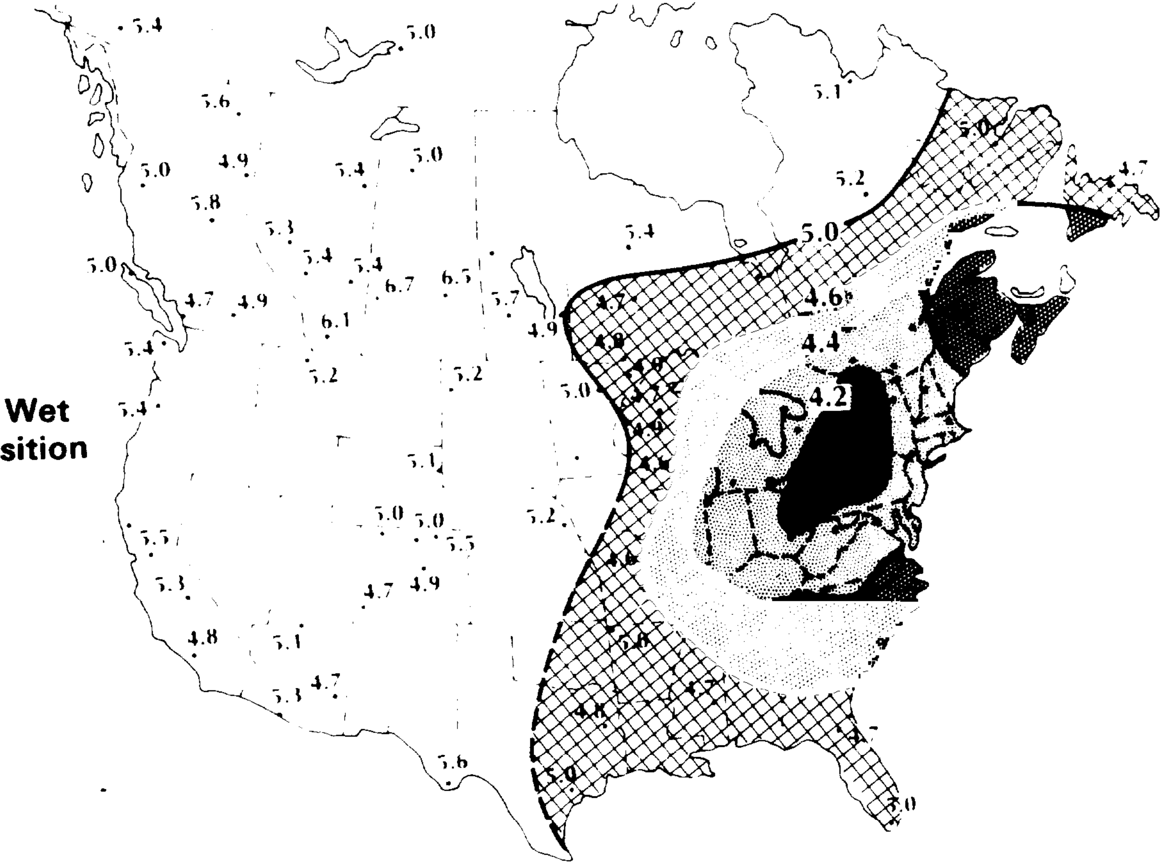
[b] If the SO_2 limit on an existing source is to be met by switching fuel only, then compliance must be reached within 2 years, i.e., by July 1, 1985.

Sources: Federal Republic of Germany, 13th BImSchV, June 22, 1983, and EPA letter Apr. 2, 1984.

Federal limits for NO_x emissions from new solid-fueled sources in the West German ordinance are not substantially different from those in place in the United States: the German limit is 800 milligrams/cubic meter (mg/m^3), equivalent to about 0.64 lb/mmBtu, compared with the current U.S. limit of 0.7 lb/mmBtu. For existing solid-fueled sources, limits between 1,000 and 2,000 mg/m^3 (about 0.8 to 1.6 lb/mmBtu) must be met within 5 years. However, all the German NO_x limits, for all fuels, direct also that all opportunities must be used to further reduce stationary source NO_x emissions by applying new combustion technology or other state-of-the art control technology. In addition, one of the German states, Baden-Wuerttemberg, requires that as of January 1984, new boilers must meet a lower 650 mg/m^3 (0.52 lb/mmBtu) limit on NO_x emissions, and by 1988 and 1990, the two primary types of pulverized coal boilers in that state will have to meet a much more stringent 200 mg/m^3 (0.16 lb/mmBtu) limit.

ACIDITY OF PRECIPITATION ON NORTH AMERICA, 1982

Figure V-I



pH of Wet Deposition 1982

Map of Annual Average pH of Wet Deposition on North America, 1982

Dots show the locations of deposition monitoring stations, while accompanying numbers show annual (precipitation-weighted) average pH's of wet deposition at each station. The four curves connect points showing approximately the same pH, as given by the number on each line. Shaded areas between lines have average pH's between the values on the surrounding lines.

Source: Interagency Task Force on Acid Precipitation, Annual Report, 1983, to the President and Congress, Wash. D.C., June 1984, p. 36.

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