IMPACT MODELS TO ASSESS REGIONAL ACIDIFICATION

edited by

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CHAPTER 14

Atmospheric Deposition Assessment Model: Applications to Regional Aquatic Acidification in Eastern North America

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14.1. Introduction

US research efforts on the problem of "acid rain" are directed at improving current scientific understanding in critical areas, including sources of precursor emissions; the transport and transformation of pollutants in the atmosphere; the deposition of acidic species; and the chemical and biological effects of acid deposition on aquatic systems, materials, forests, and crops (NAPAP, 1986). The general goals of this research are to characterize the current situation and to develop analytical "models" for predicting the future response of systems to changes in key parameters.

Methods to link the various components of the problem so that the results of scientific research may be better related to the needs of policymakers also are being developed (Balson and North, 1982; Center for Energy and Environmental Studies, 1984; Alcamo *et al.*, 1987). The goal of such "assessment" methods is to provide information useful for decisions about the need for – or consequences of – policy measures to abate acid deposition. A systematic characterization of uncertainties is an important part of assessment methods development.

The analysis presented here focuses on the effect of future emissions on regional aquatic acidification in eastern North America. Specifically, we wish to estimate the degree to which further acidification of lakes, and the consequent loss of fish life, might be expected in the absence of an acid rain control program,

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and the extent to which such effects might be reduced or reversed as a result of policy measures to reduce emissions of sulfur dioxide (SO_2). Estimating the magnitudes and sources of uncertainties in predicted impacts is a key element of the integrated analysis.

14.2. Methodological Overview

An integrated modeling framework called the Atmospheric Deposition Assessment Model (ADAM) has been developed at Carnegie-Mellon University to analyze acid rain issues in North America (Rubin *et al.*, 1986a). A schematic of the framework is shown in *Figure 14.1*.

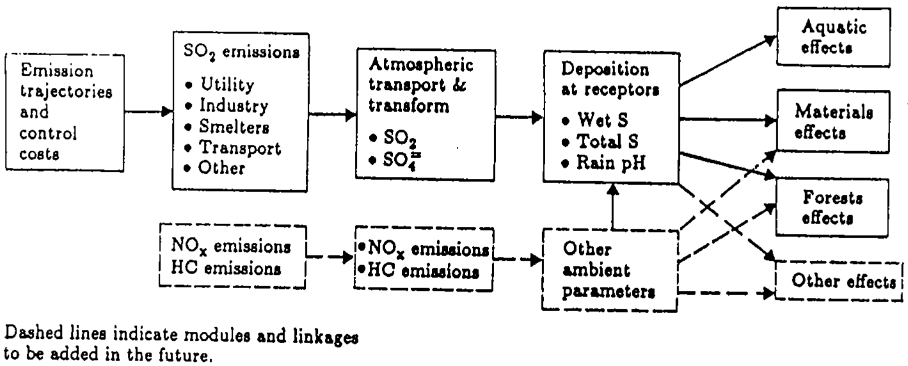

Figure 14.1. The integrated modeling framework.

The major components of the problem are linked in a computer environment designed to facilitate the testing of alternative hypotheses and the analysis of uncertainty. More detailed descriptions of the model and the rationale for its development are presented elsewhere (Center for Energy and Environmental Studies, 1984; Rubin *et al.*, 1986a; Marnicio *et al.*, 1986). The general approach is to seek simplified representations of state-of-the-art models of individual components of the acid deposition problem, and to link them systematically in an integrated framework. Simpler representations often are derived by exercising the more detailed models over a range of assumptions, then fitting the results to simpler algorithms using multivariate statistical analysis. Alternatively, results of detailed models may be employed directly in the form of a table or matrix. This process of simplification is necessary since the size, complexity, data requirements, and design of many analytical models being developed for individual components of the problem preclude their being directly linked for assessment purposes. The simplified models, however, retain the scientific credibility of the parent models on which they are based.

Figure 14.2 identifies the emission source and receptor regions currently included in the ADAM framework. The model contains up to 66 source regions, including the 48 continental states, the District of Columbia, and the 10 Canadian provinces (divided into 17 subregions). Five source sectors representing SO_2 emissions from utility combustion, industrial combustion, nonferrous smelters, transportation, and other miscellaneous sources are modeled for each region. Other pollutant species (e.g., nitrogen oxides and hydrocarbons) may be included in the emissions inventory at a future date.

The 30 receptors identified in *Figure 14.2* reflect 25 regions in the USA and five in Canada believed to be potentially sensitive to acid deposition effects on aquatic systems, materials, forests, crops, or human health. For each receptor, six deposition quantities are estimated, including ambient SO_2; ambient sulfate; wet, dry, and total sulfur deposition; and precipitation acidity (pH). The model is capable of evaluating up to three types of effects at each receptor. Effects on aquatic systems, materials, and forests currently are of prime concern. A time period of up to 50 years (1980 to 2030) may be simulated in steps of one year or more.

The model is implemented in a new software environment called DEMOS (for Decision Modeling System), developed by Henrion and co-workers (see Henrion and Wishbow, 1987), to facilitate the testing of alternative models and the analysis of uncertainty. Some ways in which the uncertainty in individual model parameters may be expressed are shown in *Figure 14.3*. These include three standard distributions (normal, uniform, and log-normal) as well as any user-specified values of discrete distribution or fractiles. These input distributions are sampled randomly using a Monte Carlo procedure to generate multiple sets of results from which probability distributions of model outputs can be obtained. Replication is performed automatically for a given sample size. This analytical capability forms the basis of the uncertainty analysis described below.

14.3. Scope of Analysis

The integrated model is used in this chapter to estimate the effects and uncertainties of acid rain control policies on lake acidity and fish viability in two potentially sensitive regions of eastern North America: the Adirondack Park area of upstate New York and the Boundary Waters region of northern Minnesota and southern Ontario.

Two scenarios are considered. The first is a "base case" situation in which no new policy initiatives are taken to abate acid deposition other than the announced Canadian plan to reduce current SO_2 emissions by approximately 40% by 1994 (*Canadian Control Program*, 1987). The second scenario examines the consequences of an acid rain control strategy for the United States typical of plans recently proposed in the US Congress (Long, 1986–1987). The methods used to model each component of the problem and to characterize key uncertainties are described below, followed by a summary of key results.

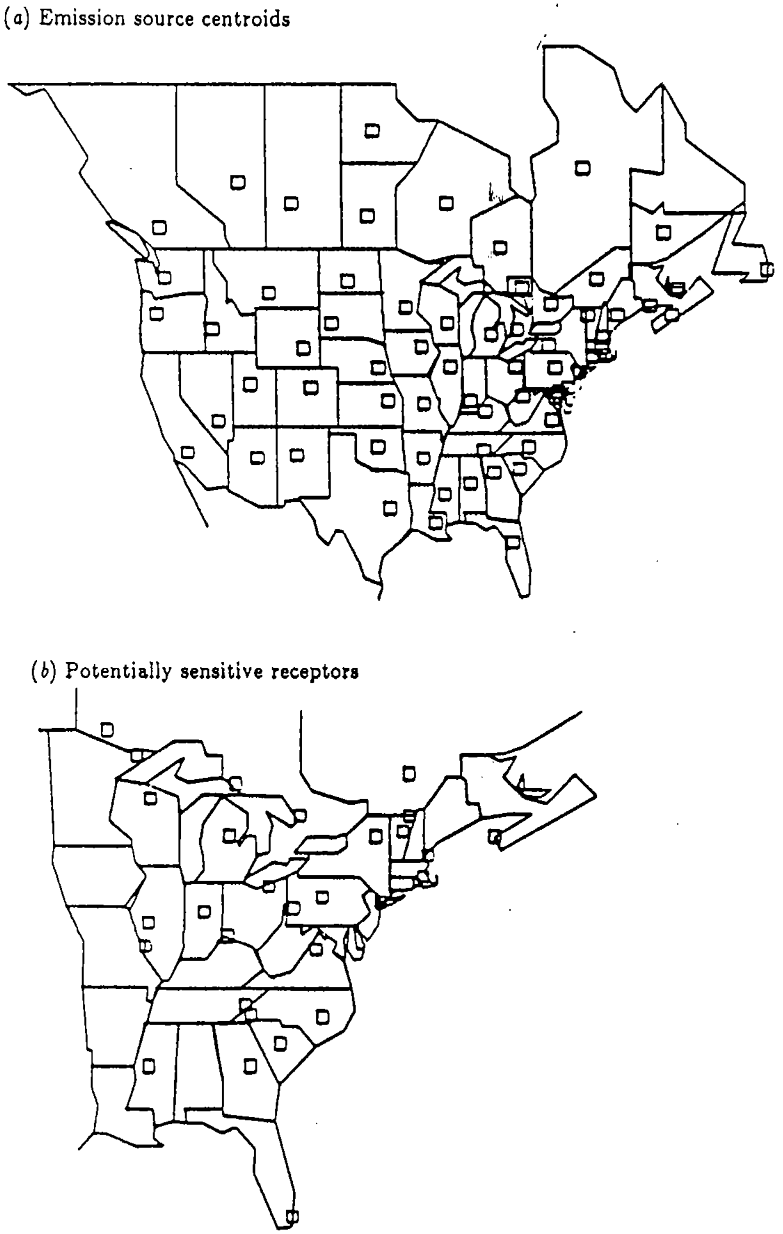

Figure 14.2. Model source (a) and receptor regions (b).

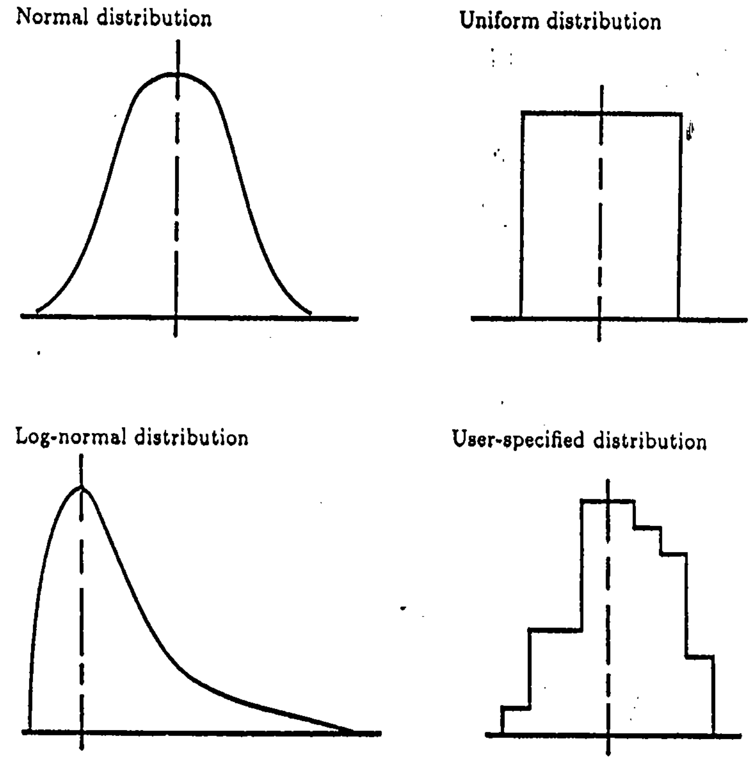

***Figure 14.3.* Some ways of expressing parameter uncertainty. Parameter values on x-axis versus probability of that value on y-axis. Dashed lines represent the mean (expected) value for each distribution.**

14.3.1. Emissions and control costs

Acid deposition effects on aquatic systems have been linked primarily to emissions of sulfur dioxide and its reaction products.[1] To analyze aquatic impacts over the next several decades, emissions of SO_2 for the USA and Canada were estimated for the 30-year period from 1980 to 2010.[2] All emission projections were based on studies recently published by the National Acid Precipitation Assessment Program (NAPAP), the US Environmental Protection Agency (EPA), the US Department of Energy (DOE), and the Electric Power Research Institute (EPRI).

Table 14.1 shows the total base year (1980) emissions by sector (Industrial Environmental Research Laboratory, 1984). US emissions account for 84% of the North American total, with electric utility emissions the dominant source (56% of the total for North America). Industrial combustion emissions account for 17% of the total; nonferrous smelters, 11%; transportation, 3%; and other sources (industrial processes, residential, and commercial fuel combustion, etc.), 13% of the total. The geographical distribution of SO_2 emissions is weighted heavily toward the eastern USA, particularly the midwestern states containing large numbers of coal-fired power plants.

Table 14.1. The 1980 SO_2 emissions for North America (millions of short tons per year).

Sector	*USA*	*Canada*	*Total*
Utility	17.4	0.8	18.2 (56%)
Industrial	4.6	0.9	5.5 (17%)
Smelters	1.2	2.4	3.6 (11%)
Transport	0.9	0.2	1.1 (3%)
Other	3.1	1.0	4.1 (13%)
Total	27.2 (84%)	5.2 (16%)	32.4 (100%)

(Source: Industrial Environmental Research Laboratory, 1984.)

The uncertainty in historical and current SO_2 emissions is not well characterized, although several efforts have been initiated in the area (Chun, 1987). One measure of uncertainty comes from comparing different published inventories of 1980 state-level emissions for each sector (Industrial Environmental Research Laboratory, 1984). In general, such comparisons show good agreement for the electric utility sector (sample standard deviations on the order of 5% of the mean), with larger deviations (14 to 25% of the mean) for the non-utility sectors, which are characterized by large numbers of small sources spread over large geographical areas. While recent studies suggest that the uncertainty in emissions from individual sources can be quite high, especially for short averaging times (Chun, 1987), existing data suggest a relatively high degree of confidence in emissions from the major source category (utilities) when averaged over a period of one year and aggregated to the state level. More recent estimates of 1985 non-utility emissions (Braine and Steubi, 1987; Placet *et al.* 1986) show better agreement than earlier estimates for 1980, suggesting some improvement in inventory techniques, data quality, or both.

A much larger uncertainty is associated with projections of *future* SO_2 emissions. *Figure 14.4* shows the range of results to the year 2010 from three recent studies of the US electric utility sector (Braine and Steubi, 1987; Placet *et al.*, 1986; McGowin *et al.*, 1986). Aggregate results for the non-utility sectors also are shown, along with total Canadian emissions.

For the scenarios involving no acid rain emission controls, the projected US utility emissions in 2010 differ by approximately eight million short tons per year (Mtpy) [3], or nearly 50% of the total 1980 utility SO_2 emissions. This wide range arises principally from different assumptions made by each organization regarding the future demand for electricity, the retirement age of the coal-fired power plants (which affects the replacement of current plants with lower-emitting new plants), and the utilization of coal for power generation in the future.

The highest SO_2 emissions are projected by EPA, who anticipate extended plant lifetimes of 60 years, leading to a continually rising trend in emissions through 2010 (Braine and Steubi, 1987). DOE and EPRI scenarios, however, suggest a gradual decline of emissions in the future. The DOE National Energy Plan (Placet *et al.*, 1986) projects somewhat lower electricity growth rates and shorter plant lives than those assumed by EPA, leading to lower estimated SO_2 emissions. Still lower SO_2 emission estimates come from the EPRI "base case" scenario reflecting historical trends. This assumes an average electricity demand growth rate of 2.3%; a 20% planning reserve margin; no nuclear additions after 1994; and plant lives of 40 years for coal, oil, and gas plants and 30 years for nuclear facilities (McGowin *et al.*, 1986). Alternative EPRI scenarios (not shown in *Figure 14.4*) give different results. A scenario extending the life of fossil fuel plants to 60 years raises emissions in 2010 by 6 Mtpy, while increasing the electricity demand growth rate to 3.3% raises emissions by an additional 1 Mtyp, producing results similar to the EPA projections. Thus, the uncertainty introduced by alternative views of the future clearly dominates the "measurement" uncertainty associated with quantifying current emissions. Projections beyond the year 2010 (not considered in the present analysis) show even greater diversity.[4]

Future SO_2 emissions from non-utility sectors are related principally to the assumed use of coal in the industrial sector and the extent of emissions control from smelters. Current EPA and DOE projections for the USA (*Figure 14.4*) show relatively constant emission rates over the next two decades, although detailed studies of non-utility emission trends have not been widely undertaken.

Canadian SO_2 emissions in *Figure 14.4* are based on current projections, assuming full compliance with planned SO_2 emission reductions from smelters and other sources. The allocation of emission reductions to specific provinces was based on a detailed inventory of sources to be controlled by 1994 (*Canadian Control Program*, 1987). Since Canadian emissions are dominated primarily by smelters, rather than coal-fired power plants, future SO_2 emission trends are far less sensitive to assumed electricity demand growth than in the USA.

To characterize the magnitude and uncertainty in future SO_2 emissions, the scenario ranges for the utility and non-utility sectors in *Figure 14.4* were taken as the bounds established by current analyses. US utility emissions in any future year were assumed to be characterized by a uniform distribution ranging from a high value given by the EPA scenario to a low value given by the EPRI "base case" scenario. Thus, any value in this range was assumed to be equally probable. For the non-utility sectors, we assumed that future emissions remained constant constant at the average between the DOE and EPA estimates, and that the uncertainty was characterized by a normal distribution. A standard deviation of 14% of the mean (the sample value for these estimates) was assumed. Since future emissions were not reported on a sector-by-sector basis, all non-utility emissions were aggregated for purposes of this study.[5]

To evaluate the effects of an acid rain control policy for the USA, we modeled a scenario requiring a reduction in SO_2 emissions of 10 Mtyp below 1980 levels, to be achieved by 1995. This emission reduction plan is similar to a

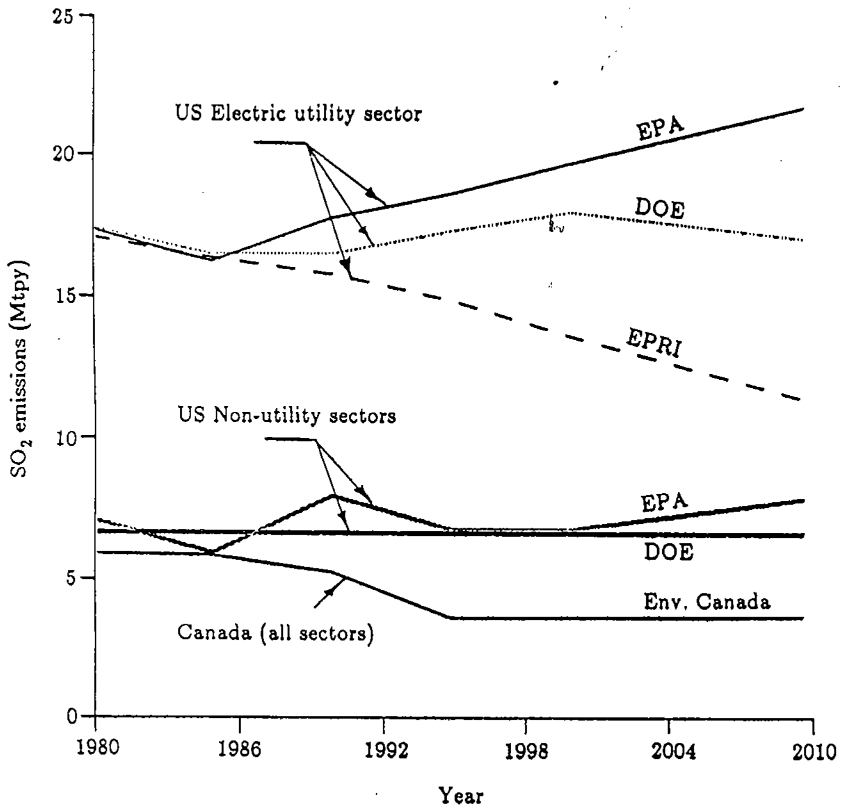

Figure 14.4. SO_2 emission projections for the USA and Canada (base case scenario).

number of proposals recently introduced in the US Congress (Long, 1986–1987). Emission reductions were allocated to individual states using a formula based on emissions exceeding 1.2 lbs SO_2/MBtu (520 ng/J) in 1980, as also proposed in several Congressional plans (Morrison and Rubin, 1985).

For purposes of cost estimation, it was assumed that all emission reductions came from the electric utility sector. To the extent that other sectors also are controlled, emission reduction costs would be roughly compared with utility costs (Rubin, 1981), although such costs still need to be better defined. Results from a detailed utility simulation model (Rubin *et al.*, 1986b) were used to obtain a range of cost estimates reflecting different technological options for SO_2 emissions reduction as well as varying plant lifetimes for coal-fired units (Salmento *et al.*, 1987). Uncertainties in other technical and economic factors also affect SO_2 control costs and could further broaden the range of costs reported later in this chapter (Rubin *et al.*, 1986b; Cushey and Rubin, 1987).

14.3.2. Atmospheric transport and transformation

The transport and transformation of air pollutants in the atmosphere remain subjects of intense study. The acid deposition phenomenon is believed to be strongly influenced by the long-range transport of air pollutants. A number of mathematical models have been developed to predict long-range source–receptor relationships for time scales ranging from one year to episodic events lasting only days or hours. Key issues in atmospheric transport modeling include the "linearity assumption" of a constant proportionality between SO_2 emissions and sulfur deposition; the influence of year-to-year meteorological variations over large geographic areas; and the role of nitrogen oxides (NO_x), volatile organic compounds (VOC), and other species in atmospheric deposition chemistry. While NO_x and VOC are known to affect the formation of photochemical oxidants, which, in turn, may affect sulfur chemistry, the precise nature and importance of such interactions remain a subject of ongoing research (NAPAP, 1986). At present, modeling of annual average sulfur deposition is based on a linear source–receptor relationship, which appears to be reasonable for long-term regional analysis (National Research Council Committee on Atmospheric Transport and Chemical Transformation in Acid Precipitation, 1983).

Two options currently are available for representing source–receptor relationships in ADAM. One is to specify a "transfer matrix" giving the magnitude of an ambient concentration or deposition quantity at a given receptor resulting from a unit of emissions at a given source region. Total impacts then are calculated, assuming a linear relationship between SO_2 emissions and ambient concentrations or deposition on an annual average basis. The ADAM database contains default transfer matrices based on the ASTRAP model developed at Argonne National Laboratory (Shannon, 1985). This model has been widely used by EPA and DOE researchers for acid deposition analysis. Recent work with ASTRAP has focused on the uncertainties in predicted deposition due to annual meteorological variations. Niemann (1986a) used ASTRAP transfer matrices for six consecutive years (1976–1981) to calculate wet sulfur deposition rates at nine ASTRAP receptor sites in eastern North America. For constant SO_2 emission rates, variations of up to 2 to 3 kg SO_4 ha^{-1} yr^{-1} were found. Analysis of longer periods of meterological data using simplified models was recommended, and this work is in progress. Uncertainty analyses involving atmospheric chemistry parameters in ASTRAP, however, do not appear to be available.

The most straightforward way to characterize source–receptor uncertainties in ADAM using a long-range transport model like ASTRAP is to run different sets of transfer matrices reflecting different meteorological or atmospheric chemistry assumptions, or both. This requires multiple runs of the detailed model to obtain the transfer matrices for specified sources and receptors. However, if the variability and covariance structure of a transfer matrix can be specified directly, this can be used in ADAM to characterize uncertainty. This remains a difficult and cumbersome task, however, for a matrix of the required size.

An alternative method of representing source–receptor relationships in ADAM is to specify an analytical model that calculates atmospheric concentrations at a given receptor based on chemical and meteorological input parameters. This is the approach used in the present study since it allows uncertainties in both meteorological and chemical parameters to be characterized more simply and explicitly than in mechanistic long-range transport models (which are relatively difficult and expensive to use, and usually are not directly available to groups other than the model developer).

The analytical model chosen was developed by Fay *et al.* (1985), and is based on multivariate regression analysis of monitoring data for wet sulfate deposition at 109 sites in North America for the years 1980–1982. Although based on relatively simple formulations of atmospheric chemistry and meteorology, predictions from this model have been found to be comparable with those of more complex mechanistic models (Matthias and Lo, 1986). A version of this model also is used by EPA for acid deposition policy analysis (Niemann, 1986b). In this study, uncertainties were assigned to three key meteorological parameters and four chemical rate constants, affecting ambient sulfate concentration, which is used to estimate precipitation acidity. Nominal parameter values and ranges, reflecting available data and reported estimates, are shown in *Table 14.2*.

Table 14.2. Source–receptor model parameter uncertainties

Parameter[a]	*Base value*[a]	*SD*[b] *or Range*[c]
Meterological factors		
Wind speed (horizontal)	4.0 m/s	0.74 m/s
Wind speed (vertical)	5.9 m/s	0.74 m/s
Horizontal diffusivity	$4.3 \cdot 10^6$ m^2/s	$0.43 \cdot 10^6$
Chemical factors		
Time constants for:		
Conversion ($SO_2 \rightarrow SO_4$)	$1.9 \cdot 10^5$ s	$(1.0 - 7.5) \cdot 10^5$
Wet SO_2 deposition	$11.3 \cdot 10^5$ s	$(1.5 - 14.0) \cdot 10^5$
Dry SO_4 deposition	$12.5 \cdot 10^5$ s	$(0.25 - 25.0) \cdot 10^5$
Wet SO_4 deposition	$0.6 \cdot 10^5$ s	$(0.14 - 2.5) \cdot 10^5$

[a] Based on Fay *et al.*, (1985).
[b] Standard deviations based on historical data.
[c] Ranges based on literature values.

14.3.3. Precipitation acidity

Acid deposition effects on aquatic systems are related in this study to precipitation acidity, which is determined by the annual average hydrogen concentration. An extensive study of the correlation between precipitation acidity and ambient sulfate concentrations for various regions of the USA has been undertaken by Hales (1982). The regression relationship is shown in *Table 14.3*, along with parameter values for Adirondack Park and Boundary Waters receptor regions. Mean annual rainfall values and variances for each region are based on historical data (Atmospheric Sciences and Analysis Working Group 2, 1982), while the uncertainty in regression coefficients has been estimated from the work of Hales and others.

Table 14.3. Parameter values and uncertainty estimates for precipitation acidity model.[a]

		Adirondack Park		*Boundary Waters*	
Parameter	*Definition*	*Mean*	*SD*	*Mean*	*SD*
a_i	Regression coefficient	$16.5 \cdot 10^{-6}$	$1.65 \cdot 10^{-6}$	$2.3 \cdot 10^{-6}$	$0.23 \cdot 10^{-6}$
b_i	Regression coefficient	1.618	0.162	1.573	0.157
R	Annual rainfall (mm)	1045	91	680	68

[a] Based on Hales *et al.* (1982). Precipitation acidity is given by:
$(pH)_i = - \log (a_i + 3.125 \cdot 10^{-3} \frac{\chi_{ws}}{R} b_i)$ where χ_{ws} = wet sulfate deposition rate (kg S ha^{-1} yr^{-1}).

Rainfall acidity is related directly to the rate of wet sulfur deposition predicted by the atmospheric transport model. Note that because the relationships are empirically fit to current wet deposition chemistry data, they implicitly incorporate the effects of other acid and base species, such as nitrates and soil dust. In most areas of eastern North America, sulfate contributes about two-thirds of the acidity of precipitation and nitrates about one-third (Verry and Harris, 1988). Extrapolation of the pH–sulfate relationship to conditions of significant change in the level of sulfate therefore represents a source of uncertainty in the model predictions since the relationships are unlikely to be closely maintained unless there are concomitant reductions in the level of atmospheric nitrate.

Another reason for using sulfate as the driving deposition input in the integrated assessment model is that sulfate is generally quite mobile in the terrestrial environment, except in certain locations, such as the southeastern United States, where sulfate adsorption occurs in the soil. As noted by Seip (1980), nitrate is, in contrast, "taken up in the catchment, and only a small fraction . . . appear(s) in runoff." As such, nitrate in lake water is generally less than 10% of sulfate on an equivalent basis (Wright, 1988), although nitrate can play an important role during snowmelt periods (Verry and Harris, 1988; Driscoll and Schafran, 1984). Long-term lake acidification models thus commonly use sulfate as the measure of acid input (Cosby *et al.*, 1985b; Henriksen and Brakke, 1988). A more complete analysis of acid deposition and its impacts should, in the future, include delineation of the contribution of nitrogen species.

14.3.4. Deposition uncertainty

To characterize the uncertainties in sulfur deposition, the sensitivity method of Hornberger and co-workers (1985a, 1986a) was applied to the source–receptor relationships from which deposition values are calculated. The method is implemented by assuming prior distributions for uncertain model inputs, then determining which combinations of these inputs give results consistent with observed atmospheric chemistry data. This allows estimation of the posterior distributions of output results.

Ranges of values were first ascribed to the individual parameters of the Fay *et al.* (1985) model, then Monte Carlo simulation was used to calculate the precipitation acidity at six geographically dispersed locations in eastern North America (*Figure 14.5*). Combinations of the uncertain chemical rate constants that produced "acceptable" values of precipitation pH at all six receptors were chosen for use in further analyses. This method was employed to identify and thereby account for the covariance among chemical rate parameters, and the interaction between chemistry and meteorology. Given the difficulty of undertaking a more rigorous analysis of these factors (e.g., using a detailed long-range transport model), this approach was adopted to obtain a physically reasonable and computationally feasible approximation for an integrated analysis.

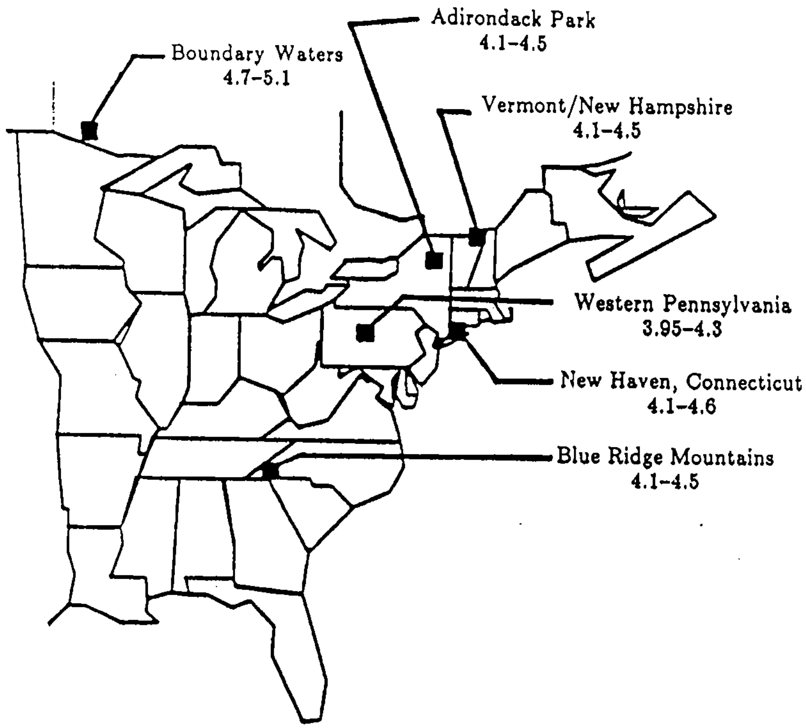

Figure 14.5. Acceptance criteria for predicted annual average precipitation acidity (pH ranges for six receptor regions).

Table 14.2 displays the range of parameter values initially used. Meteorological parameters were assumed to be independent, with "best estimate" values and variabilities characterized from historical data and published studies.

Normal distributions were assumed for the horizontal and vertical components of annual average wind speed, horizontal diffusivity, and annual average rainfall. Ranges of values for four chemical rate constants, governing the conversion of SO_2 and SO_4 in wet and dry form, were obtained from a survey of the literature. A log-uniform distribution was used to characterize each of the chemical rate constants in the model. A series of Monte Carlo runs was conducted in which values of the chemical rate parameters were chosen independently in conjunction with a value for each of the meteorological parameters of the model. A subset of these runs (100 cases) was determined to be "acceptable," based on the ability of the model to predict precipitation pH within the ranges shown in *Figure 14.5* for all receptor locations.[6]

The runs that produced unacceptable results illustrated the covariance effects noted earlier. For example, *Figure 14.6* shows scatter plots of the chemical rate constant for primary conversion of SO_2 to $SO_4(\tau_c)$ versus the rate constant for wet disposition of $SO_2(\tau_{wp})$. Since high values of both parameters are not consistent with the observed (allowable) ranges of deposition [7], such combinations are not seen in the final set of parameter values. For subsequent analysis, the 100 sets of acceptable combinations of chemical parameter values were used to represent the uncertainty distributions of the input parameters, with any of the 100 sets assumed to be equally probable. Note that this implicitly incorporates the correlation structure of the acceptable parameter ranges. Meteorological parameters continued to be taken as independent.

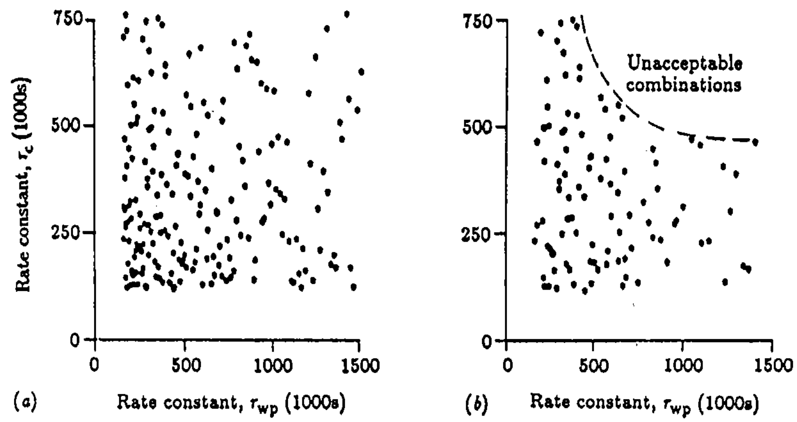

Figure 14.6. Scatter plots of initial (*a*) and final (*b*) values of two rate constants.

14.3.5. Aquatic chemistry

The response of aquatic systems to acid deposition also has been, and remains, a subject of extensive research. Predictive models for aquatic acidification span a range of scientific and computational complexity, in large part driven by the intended use of each model. Empirical equilibrium models provide a simple representation well suited to regional predictions; however, they lack a fundamental physical basis and do not provide dynamic predictions. Dynamic, mechanistic models predict changes over time using basic chemical equilibrium and kinetic relationships, but are difficult to apply to a region, with its many lakes. These two approaches are briefly described below, followed by a discussion of the model used in the present study – a dynamic Direct Distribution model, which combines features of the empirical equilibrium and dynamic mechanistic approaches.

Empirical equilibrium models predict the steady state alkalinity and corresponding pH in a lake by assuming that a given fraction of the acid deposition to a basin is neutralized by chemical weathering of soil and rocks in the watershed. The fraction of incoming acid that is neutralized, referred to as the "weathering factor," is empirically determined for each lake. The weathering factor can be adjusted to account for the full, complex range of chemical and biological processes in a watershed, including indirect effects of forests and soils on surface-water chemistry and in-lake processes, such as biological reduction of sulfate. In the Henriksen–Wright model (Henriksen, 1979; Wright and Henriksen, 1983), the weathering factor is assumed equal to a constant value, independent of the level of acid deposition. In the Schnoor Trickle-Down model (Schnoor and Stumm, 1984; Schnoor *et al.*, 1986a), the weathering factor varies with the level of acid deposition, based on observed mechanisms of mineral hydrolysis. Equilibrium models are relatively easy to apply to many lakes in a region, though difficulties arise when attempting to characterize the variation in weathering factors from lake to lake.

Dynamic, mechanistic models incorporate a variety of hydrologic, mass balance, and chemical equilibrium relationships to predict the evolution of lake chemistry resulting from a dynamic profile of acid deposition. Examples include the ILWAS (Integrated Lake Watershed Acidification Study) model (Chen *et al.*, 1982; Gherini *et al.*, 1985); the Birkenes model (Christophersen *et al.*, 1982); the MAGIC (Modeling Acidification of Groundwater in Catchments) model (Cosby *et al.*, 1985a, 1985b); and the RAINS (Regional Acidification INformation and Simulation) model (Kämäri and Posch, 1987). These models generally require estimates for a number of parameters for each lake, and the dynamic simulations are computationally intensive. Thus, they are difficult to apply in regional assessments. While significant progress has been made in the development of methods for regionalizing these models (Cosby and Wright, 1987; Kämäri and Posch, 1987), the computational requirements still preclude their inclusion in an integrated model, such as ADAM, where evaluations of multiple scenarios are required for policy and uncertainty analysis.

To address these shortcomings, this study uses the newly developed dynamic Direct Distribution model of Small and co-workers (Small and Sutton, 1986a, 1986b; Small *et al.*, 1987). The model uses the Henriksen–Wright equilibrium model to predict the acid-neutralizating capacity (*ANC*) of a watershed [8], but applies the model to the probability distribution functions of *ANC* and pH for a *region*. Lake-to-lake variations in the weathering factor are explicitly

incorporated by the identification and use of the mean weathering factor, F, together with the variance of the weathering factor, F_{var}, for the region. A dynamic version of the model was developed by assuming that all lakes in a region approach a new equilibrium value of *ANC* in a manner described by an exponential equation with a characteristic time constant, t_{alk}. The resulting pH–distribution is derived from the *ANC* distribution at each time step, and is integrated with a pH–fish presence–absence relationship to determine the fraction of lakes able to support a fish species. The development of this relationship is described later in this chapter.

The parameters of the dynamic Direct Distribution model can be estimated from a regional mechanistic model, such as the RAINS or MAGIC model. This allows the Direct Distribution model to provide a nearly equivalent representation of a more complex mechanistic model, while maintaining the computational efficiency of an empirical equilibrium model. This procedure is illustrated in Chapter 10. Such work, however, remains for future applications. In the examples that follow, we have estimated the parameters of the Direct Distribution model from judgmental assessments based on previous studies of the regions of interest. Uncertainties in the assumed parameter values are assigned to reflect the empirical nature of this assessment.

An important question in the application of regional aquatic acidification models is whether available lake chemistry data provide a representative sample of the overall population of lakes in a region. To address this problem, the US EPA conducted the National Surface Water Survey (NSWS), in which lakes were selected and monitored from stratified samples of target sensitive regions (Environmental Protection Agency, 1986). The resulting data sets provide the most representative characterization currently available of the overall distribution of US regional lake chemistry. The parameters of these distributions defined in the Direct Distribution model include three parameters of the log-normal distribution for *ANC* (Θ, ξ, and ϕ^2), and two parameters of the pH–*ANC* relationship (p_1 and p_2). Parameter values for the NSWS subregions corresponding to the Adirondack Park and Boundary Waters (northeastern Minnesota) were evaluated by Small *et al.* (1988). Results are presented in *Table 14.4*, along with the other parameters determined for the Direct Distribution model. Comparisons of the fitted *ANC* distributions and pH–*ANC* relationships with observed values of current lake chemistry are presented in *Figure 14.7*.

The most critical parameter shown in *Table 14.4* for determining the response of lakes to changing acid deposition is the mean weathering factor, F. Specification of the weathering factor even for a single lake is a difficult approximation (Johnson *et al.*, 1985). F likely varies with the acidity of the deposition (Schnoor and Stumm, 1984), and, as suggested by our analysis of the RAINS and MAGIC models (Small *et al.*, Chapter 10; Labieniec, 1988), also varies over time, even with constant deposition. It is important to note that our use of the weathering parameters considers all sources of acid neutralization in the watershed, not only base cation generation from soil weathering. Our analysis of the RAINS model applied to southernmost Finland resulted in an average value of F equal

Table 14.4. Aquatic chemistry model parameters and uncertainty.

	Adirondacks Park		Boundary Water	
Parameter[a]	*Mean*[a]	*Range or* σ[b]	*Mean*[a]	*Range or* σ[b]
ANC distribution, parameter, θ	−41	(−51/−31)	23	(13/33)
ANC distribution, parameter, ξ	496	(0.9)	5.13	(0.08)
ANC distribution, parameter, ϕ^2	1.145	(0.13)	0.941	(0.11)
Flow through ratio, FTR	0.45	(0.35/0.55)	0.35	(0.25/0.45)
Mean weathering factor, F	0.6	(0.4/0.8)	0.7	(0.5/0.9)
Weathering factor variance F_{var}	0.047	(0.027/0.067)	0.047	(0.027/0.067)
Alkalinity/weathering factor cor., ρ	0.5	(0.1/0.9)	0.5	(0.1/0.9)
Alkalinity char. time, τ_{alk}	8	(1/14)	8	(1/14)
Alkalinity/pH trans. factor, ρ_1	5.24	(5.14/5.34)	5.4	(5.2/5.6)
Alkalinity/pH trans. factor, rhp_2	11.02	(1.02/21.02)	10.00	(−2.42/17.58)

[a]Based on Small and Sutton (1986b) and Small *et al.* (1987).
[b]Ranges assume a uniform distribution. A single value is the standard deviation of a normal distribution.

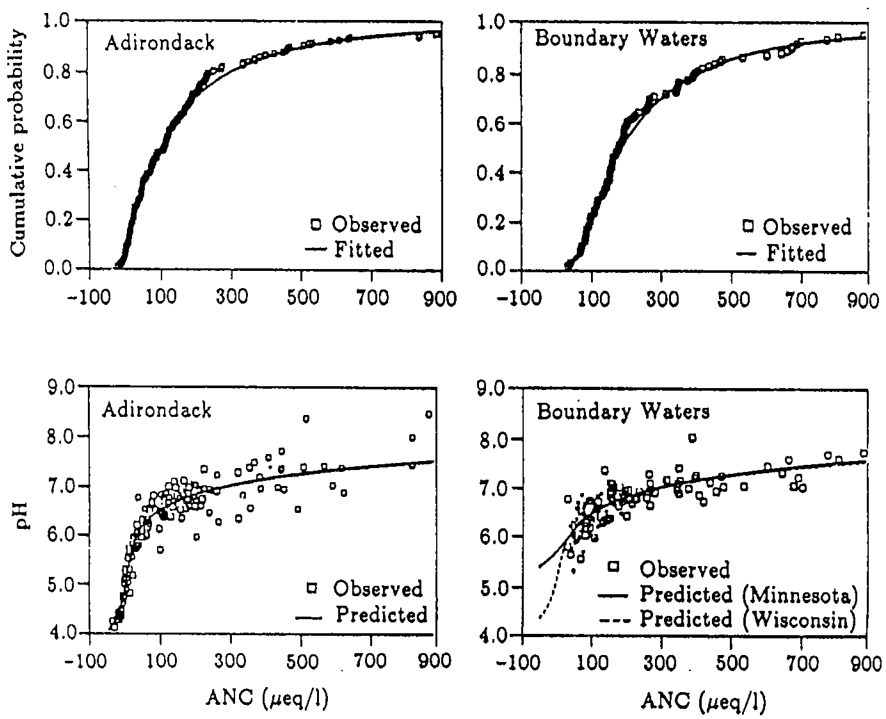

Figure 14.7. Lake *ANC* distribution and pH–*ANC* relationship for two receptor regions. (Source: Small *et al.*, 1988.)

to about 0.8 (Chapter 10). Somewhat lower values of about 0.2–0.6 were obtained from MAGIC for southernmost Norway (Labieniec, 1988). This is consistent with previous estimates for North America (Small and Sutton, 1986b; Johnson *et al.*, 1985). The assumption in *Table 14.4* of somewhat greater weathering in the Boundary Waters region relative to Adirondack Park (mean F of 0.7 relative to 0.6) is consistent with geological assessments of the respective regions (Eilers *et al.*, 1988; Brakke *et al.*, 1988). The estimation of F and its variability (F_{var}) in *Table 14.4*, however, remain highly judgmental at the present time, although, as indicated above, there is considerable opportunity to reduce the uncertainties in these parameters through additional research using mechanistic models.

The uncertainty in each parameter of the Direct Distribution model was assumed to be either normally or uniformly distributed across the range of values shown. Monte Carlo sampling from these distributions was used to characterize the overall uncertainty in the distribution of regional lake alkalinities. Because covariance effects cannot yet be rigorously assessed for lake chemistry models, we conservatively assumed that all model parameters were independent. While this generally tends to overstate the calculated estimates of uncertainty, a lack of knowledge regarding the precise nature of current alkalinity distributions for large collections of lakes mitigates in favor of this approach.

14.3.6. Fish viability

The final link in the chain from emissions to acid deposition effects is the biological effect on aquatic life associated with changes in lake acidity. A number of laboratory and field studies have been undertaken; see Haines (1981), Johnson (1982), and Muniz *et al.* (1984). However, it has been difficult to generalize these results into mechanistic models that incorporate effects at different stages of the life cycle, and to aggregate such effects to predict population dynamics. Thus, current models and measures of acid deposition effects on fish populations remain relatively rudimentary.

One measure of fish viability, developed in Canada, is the number of different species found as a function of lake acidity. Correlations of this type have been reported by Minns (in Chapter 5) for one Canadian region. However, such data on species diversity are not generally available for other locations in North America.

Studies of US fish populations have characterized fish viability in terms of the presence or absence of a given species in lakes of a given pH. A relatively simple model (*Table 14.5*) by Reckhow *et al.* (1987) used observed fish preserve data to describe the probability of finding a given fish species in lakes of a given pH. Reckhow (1987) also reported the estimation error in the two model parameters, and subsequently characterized their correlation structure. *Figure 14.8* shows the resulting distributions of fish viability for two common recreational fishing species: brook trout and lake trout. Nominal values are shown together with an 80% confidence interval calculated from the data in *Table 14.5*. While fish viability is known to depend as well on other parameters, such as calcium,

Table 14.5. Parameter values and uncertainty estimates for fish viability model.[a]

		Brook trout		*Lake trout*	
Parameter	*Definition*	*Mean*	*Variation*[b]	*Mean*	*Variation*[b]
α	Viability intercept	6.20	1.72/–0.989	5.77	2.83/–0.991
β	Viability slope	–1.05	0.28/–0.989	–0.92	0.45/–0.991

[a]Based on Reckhow *et al.* (1987). Survival probability for fish species, i, is given by :

$$P_i = \frac{1}{1 + \exp\left[\alpha_i + \beta_i(\text{pH}_l)\right]} \quad \text{where } \text{pH}_l = \text{lake pH}.$$

[b]Standard deviation of a normal distribution and the slope/intercept correlation coefficient.

magnesium, and aluminum ion concentrations, validated models of this sort are not yet available; nor are aquatic chemistry models yet able to provide valid estimates of metal ion concentrations that could serve as inputs to improve biological models. These factors contribute to the empirical uncertainty in fish response to changes in lake pH indicated in *Figure 14.8*.

14.4. Results

We turn next to a summary showing the regional acidification impacts of SO_2 emissions from the USA and Canada. The computer simulation model (ADAM) was used to link each of the previously described model components. Uncertainties were combined and propagated by sampling randomly from the specified distributions for each stochastic variable (or sets of acceptable values, in the case of atmospheric chemistry parameters), using a total of 100 iterations to obtain experimental distributions for the values of model output parameters. Examples of these results are shown in *Figure 14.9*.

These curves show the cumulative probability distributions of selected model outputs for the years 1980 to 2010 for the acid rain control scenario. Results of this type were obtained at five-year intervals for both the Adirondack Park and Boundary Waters receptors. The median (expected) value of any result corresponds to a probability of 0.50. An 80% confidence interval is bounded by cumulative probabilities of 0.10 and 0.90. This is the confidence interval illustrated in all later results. The slightly irregular shape of several distribution functions in *Figure 14.9* is a result of the finite number of simulations used to approximate the true distribution.[9]

The output quantities shown in *Figure 14.9*, together with emission control costs (not shown in that figure), represent the key results of the integrated analysis. In the discussion that follows, we summarize the principal findings for the parameters and illustrate their temporal trends over the 30-year simulation period.

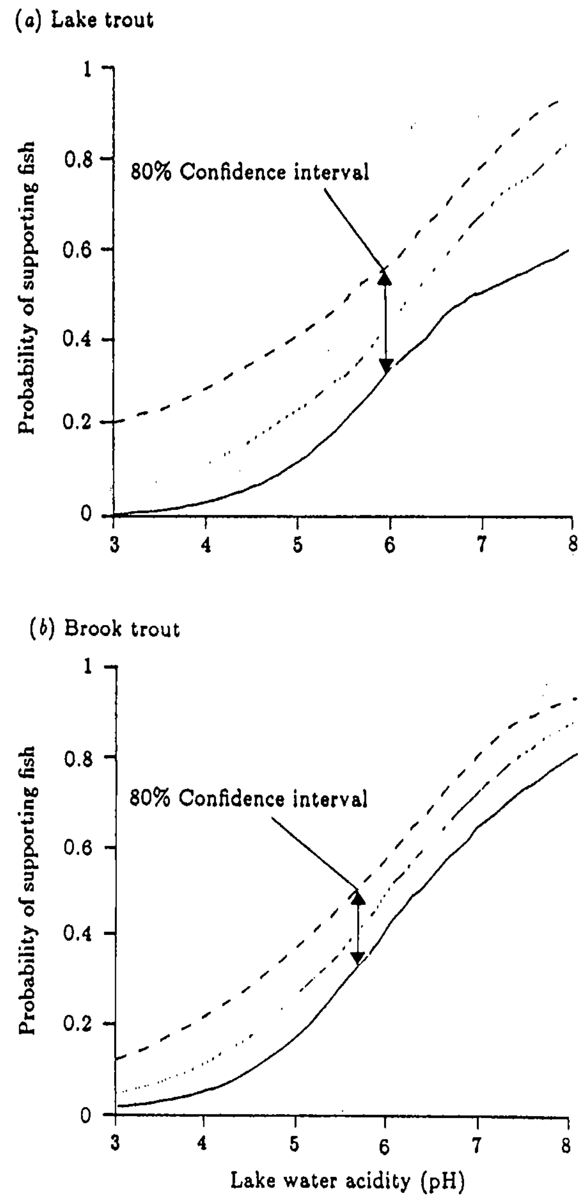

Figure 14.8. Fish viability functions for lake trout (*a*) and brook (*b*).

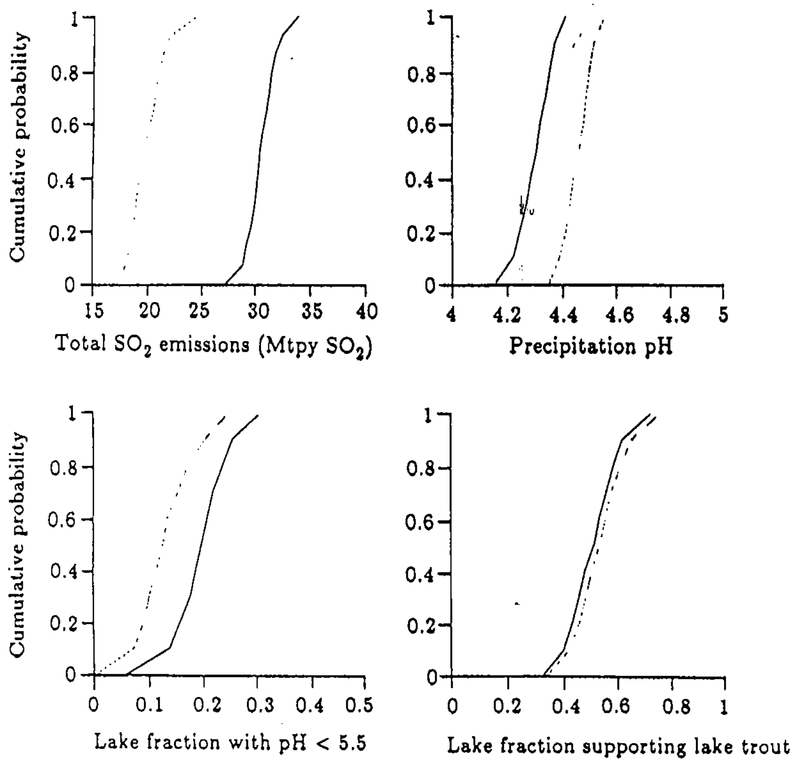

Figure 14.9. Model results for the 10 Mtpy reduction scenario, the Adirondack Park receptor (cumulative probability distributions for selected output parameters). Solid lines indicate actual, 1980 results; dashed lines indicate predicted, 2010 results.

14.4.1. Projected trends and uncertainties

The assumed trends in total SO_2 emissions from US and Canadian sources are again shown in *Figure 14.10* for the base case scenario (no change in current policy) and the acid rain control scenario (reducing US emissions to 10 Mtpy below 1980 levels). In contrast to the scenario ranges shown in *Figure 14.4*, an 80% confidence interval about the mean is shown here based on the frequency distributions illustrated in *Figure 14.9.*

For the emission reduction scenario, *Figure 14.11* shows the corresponding range of SO_2 control costs that would be incurred if all US emission reductions were achieved at coal-fired power plants.[10] The cost ranges cover four cases. The first assumes that only coal switching, coal cleaning, and current flue gas desulfurization (FGD) processes are available for retrofit applications. A second case assumes that two dry SO_2 removal processes now under development for

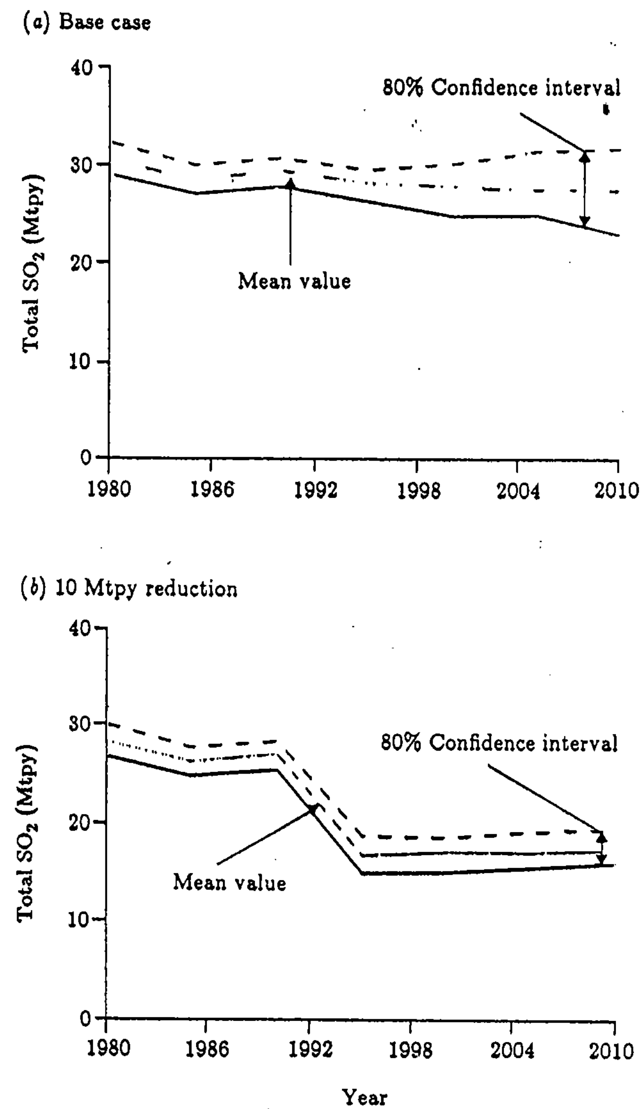

Figure 14.10. Total SO_2 emissions for the base case and acid rain control scenarios (mean values with 80% confidence intervals).

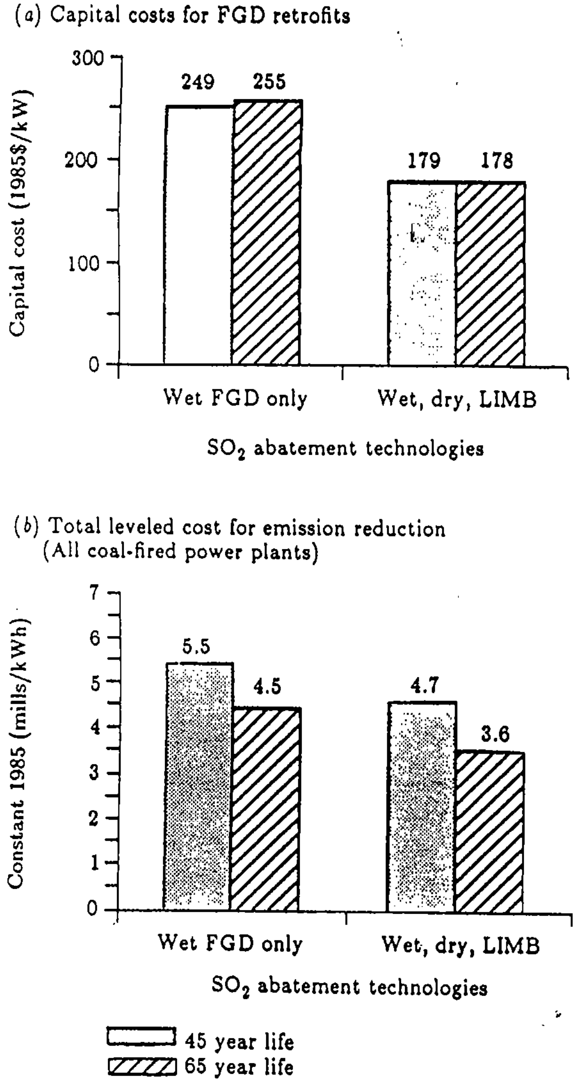

Figure 14.11. Cost for the 10 Mtpy acid rain control scenario.

medium- and high-sulfur coal applications (lime spray dryers and furnace limestone injection, or LIMB) also will be commercially available by the mid-1990s. For both cases, two power plant lifetimes are analyzed: a nominal (historical) case of 45 years, and an extended life of 65 years for all but the smallest units. The average increase in levelized electricity cost for these four cases varies from 3.6 to 5.5 mills/kWh (in constant 1985 dollars). This is roughly 10% to 14% of current generation costs for coal-fired plants in the eastern USA. As noted

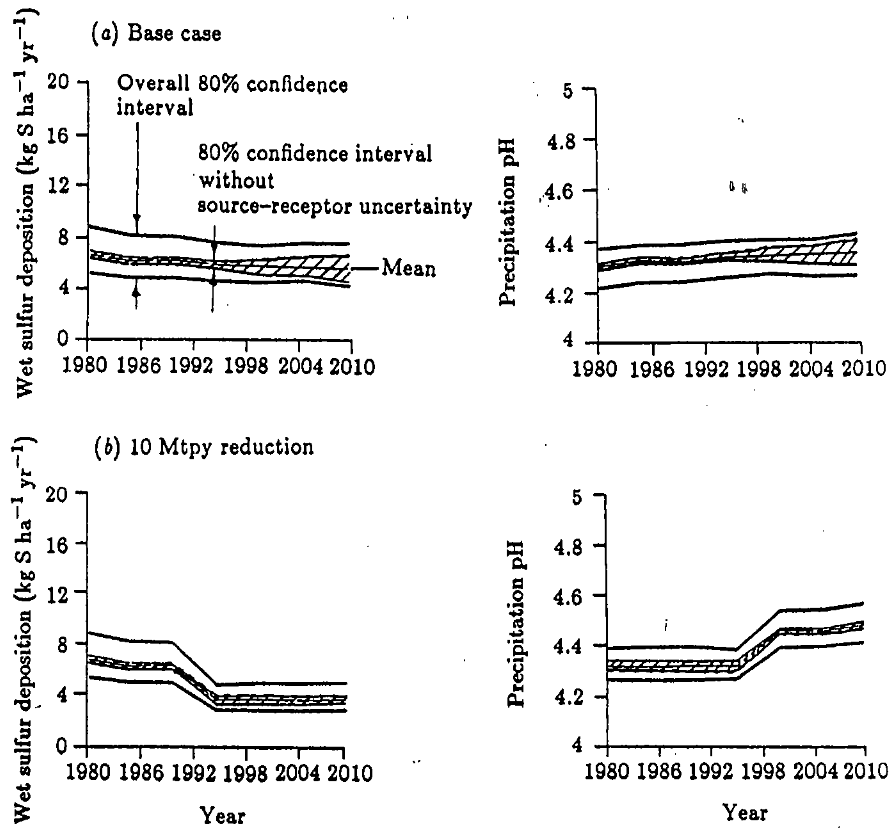

Figure 14.12. **Atmospheric deposition quantities for Adirondack Park (expected values with 80% confidence intervals).**

earlier, other technical and economic factors could introduce greater uncertainty in the cost of emission reductions (Rubin *et al.*, 1986b).

Figure 14.12 next shows the predicted values of wet sulfur deposition and precipitation pH computed at the Adirondack Park receptor. An 80% confidence interval is show first for an uncertainty only in SO_2 emissions, then for the combined uncertainty in both SO_2 emissions and source–receptor relationships. For the base case, the uncertainty in future emissions increasingly dominates the total uncertainty as time goes on; while for the emission reduction scenario, the atmospheric transport uncertainties are generally more significant. In the year 2010, the overall 80% confidence interval encompasses wet sulfur deposition rates 4.1 to 7.5 kg S ha^{-1} yr^{-1} in the base case and 2.8 and 4.9 kg S ha^{-1} yr^{-1} for the 10 Mtpy reduction scenario.

For both scenarios, the generally decreasing trend in sulfur deposition corresponds to the downward trend in average emissions across the range of scenarios considered. The 80% confidence interval for the base case encompasses a range of wet sulfur deposition rates similar to that found at Adirondack Park with the ASTRAP model when both annual emissions and meteorology were varied over a recent six-year period (Niemann, 1986a). A 90% to 95% confidence interval would encompass a larger range of deposition rates (see *Figure 14.9*). An analysis of the calculated source contributions to wet sulfur deposition at the Adirondack Park and Boundary Waters receptors showed that both regions were affected principally by emissions from midwestern states along the Ohio River Valley in the USA, and by sources in southern Ontario in Canada.

The changes in wet deposition lead to changes in precipitation pH (*Figure 14.12*). The 80% confidence interval for Adirondack Park encompasses a range of approximately ±0.14 pH units. A similar pattern of change was found at the Boundary Waters receptor (not shown), where the pH profile remained about 0.5 pH units above that predicted for Adirondack Park throughout the simulation period.

The effect of changing deposition rates on regional aquatic systems is reflected by a shift in the distribution of lake *ANC*. This is illustrated in *Figure 14.13* for the Adirondack Park receptor. The shift toward higher values of *ANC* implies a reduction in the fraction of lakes with a pH below some given value. Trends in the fraction of lakes with a pH of 5.5 or less are shown in *Figure 14.14*. This pH value was chosen to represent lakes where effects on fish life are likely to be pronounced. For the base case, the expected value of lakes with pH less than

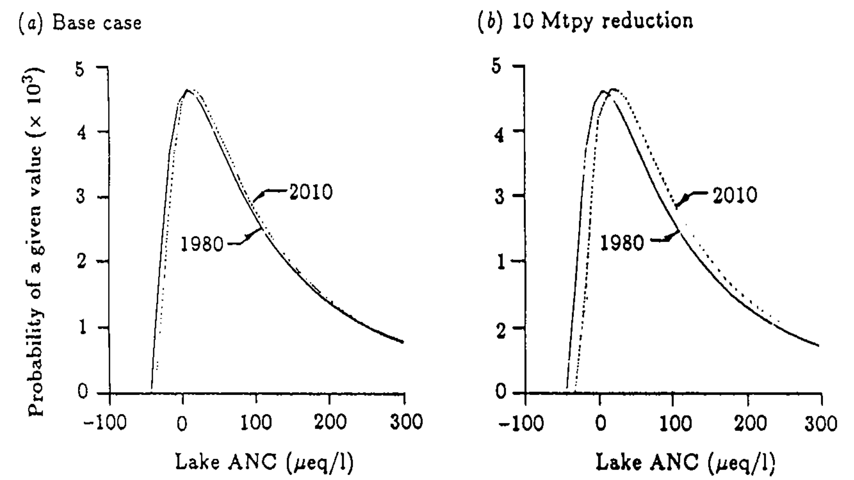

Figure 14.13. Current and projected distributions of lake alkalinity for Adirondack Park.

5.5 falls from about 20% in 1980 to 17% in 2010. For the SO_2 reduction scenario, this fraction falls to approximately 13% in 2010.

The confidence intervals in *Figure 14.14* first show the cumulative effect of uncertainties in all model components prior to the aquatic chemistry model (i.e.,

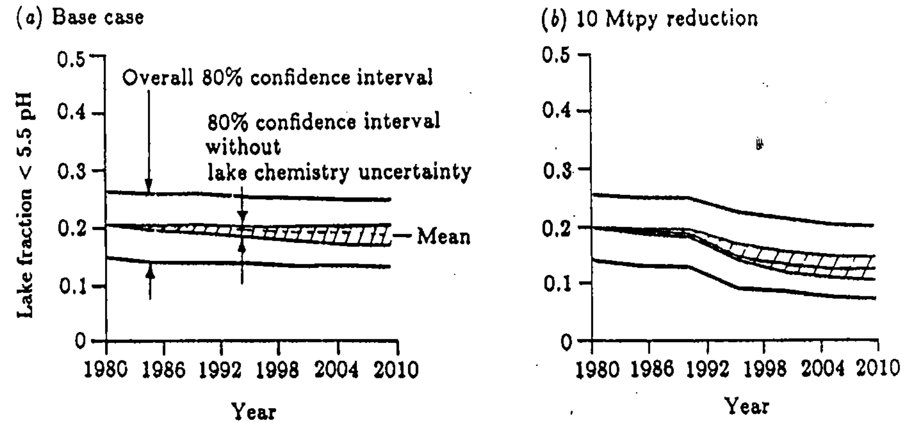

Figure 14.14. Fraction of Adirondack Park lakes with pH below 5.5 (80% confidence bands).

the emissions and atmospheric transport models), then the combined effect of the additional uncertainty in aquatic chemistry. This shows that the incremental effect of aquatic uncertainties dominates overall result. Because the frequency distribution of the fraction of lakes with pH less than 5.5 is not symmetric with respect to the median value (*Figure 14.9*), the 80% confidence interval also is not symmetric.

For both scenarios, the overall uncertainty interval about the mean remains relatively constant with time, reflecting primarily the uncertainties in atmospheric transport and lake chemistry. The effect of the 1995 emission reduction scenario, however, is clearly seen in the trend lines for each confidence level, which show a declining fraction of acidic lakes. This suggests that, although there is considerable uncertainty as to the actual number of acidic lakes (even today, reflecting uncertainty in the current resource inventory), the likelihood of recovery as a result of emission reductions is more robust. For example, over the 25 years from 1985 to 2010, the fraction of lakes with pH less than 5.5 decreases by 5% to 6% for all confidence levels. For the base case scenario, the expected change is a decrease of 1% to 2% for the assumed range of future SO_2 emissions.[11]

Results for the Boundary Waters receptor indicated much smaller effects. A negligible fraction of the lakes there currently has a pH as low as 5.5, and only a third has a pH below 6.7 (Environmental Protection Agency, 1986). By 2010, this fraction is projected to fall by only 0.5% in the base case and by 1% in the emission reduction case. The 80% confidence bands encompass values from approximately 20% to 50% of the lakes with pH below 6.7.

The effect of changing lake acidity on the expected fraction of the lakes in the two study regions able to support fish life is shown in *Figure 14.15*, based on the presence/absence relationships described earlier. For brook trout and lake trout at Adirondack Park, there is a small increase of 0.6% from 1985 to 2010 for the base case scenario, with a larger increase of 2.4% for the emission reduction scenario. These changes are similar for the 10% and 90% confidence levels. Thus, the results indicate that the number of Adirondacks lakes capable of supporting trout will most probably remain stable and even improve slightly in the *absence* of acid rain controls (given the assumed range of projected emissions), but would improve more significantly with an emissions reduction policy. A further *decline* in fishable lakes would be expected, however, if the increasing emission levels reflected in the EPA assumptions were judged to be the most likely case. *Figure 14.15* shows the overall 80% confidence intervals extending to approximately ±7% of the median values for lakes able to support brook trout and ±11% for lakes able to support lake trout at Adirondack Park. For Boundary Waters, the comparable uncertainty intervals are ±8% for brook trout and ±14% for lake trout.

Perhaps the most appropriate way of characterizing the impact of an acid rain control program on future lake acidification is to compare the expected

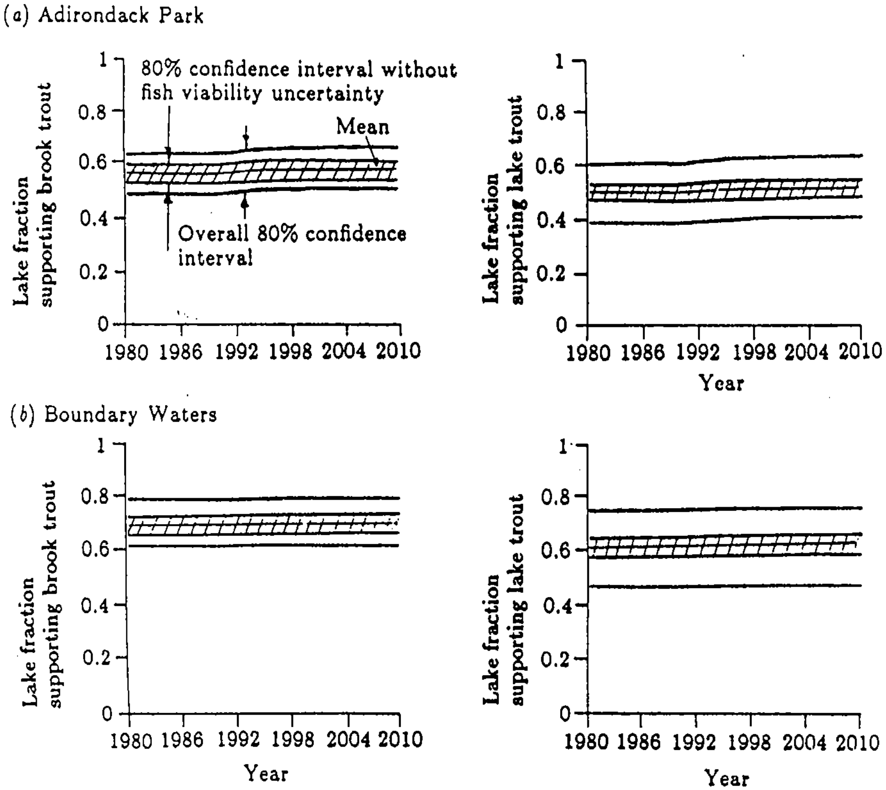

Figure 14.15. Fraction of lakes able to support brook trout and lake trout (80% confidence intervals for 10 Mtpy reduction scenario).

effects in some future year relative with the base case. For 2010, this yields an expected improvement of 1.7% in the number of Adirondack Park lakes able to support either brook or lake trout. For Boundary Waters, the analogous figure is 0.3%.

To put these numbers into perspective, the stratified samples upon which the NAPAP lake survey data were based represented approximately 1,500 lakes in both the Adirondack Park and Boundary Waters region. The total lake population in these two regions, however, is nearly twice as large, although the distribution of *ANC* among the total population may not necessarily reflect that of the subset chosen for the survey. Based on the total population, however, the 1.7% difference in the Adirondack Park lakes supporting brook and lake trout in 2010 corresponds to approximately an additional 50 lakes in that region that would be expected to recover as a result of the 10 Mtpy emission reduction. For Boundary Waters, an increase of six lakes supporting trout would be expected.

The implication of these figures is that the effects of an acid rain control program on fish viability could well be masked by the significant uncertainty that exists as a result of inadequate data, incomplete scientific understanding, and the natural variability of the atmosphere and ecosystems. In view of these results, it is useful to explore further the factors that contribute most to the overall uncertainty. In doing this, we introduce another measure for characterizing model results – namely, the probability distribution of *changes* in key parameters from 1980 to 2010.

14.4.2. Major sources of uncertainty

The procedure used in this study to model the effects of uncertainty is a stochastic simulation in which a 30-year trajectory linking emissions to aquatic effects is repeated 100 times, choosing different values of key model parameters from specified probability distributions. While the preceding discussions have emphasized the distribution of results predicted for any given year, one can also look at the overall *change* between the first and last years predicted by each of these 100 iterations. This is done in *Figure 14.16*, which shows the probability distribution function of the change from 1980 to 2010 in the percentage of the Adirondack Park lakes supporting lake trout. Both the base case and 10 Mtpy SO_2 reduction scenarios are shown.

Figure 14.16 also illustrates how the overall change is influenced by uncertainties in each model component. The cumulative distribution function is decomposed to show first the effect of uncertainty only in the projected SO_2 emissions, with no uncertainty in other model components. Then, uncertainty in atmospheric transport–transformation is added, followed by uncertainty in precipitation pH and so on.

For the base case scenario, the expected change is less than 1% in lakes able to support lake trout, as discussed earlier. The computer uncertainty distribution, however, indicates a 10% chance the changes may be as low as zero or as high as 2%. The negative range corresponds to a decrease in the number of fishable lakes (which has a lower probability of occurrence).

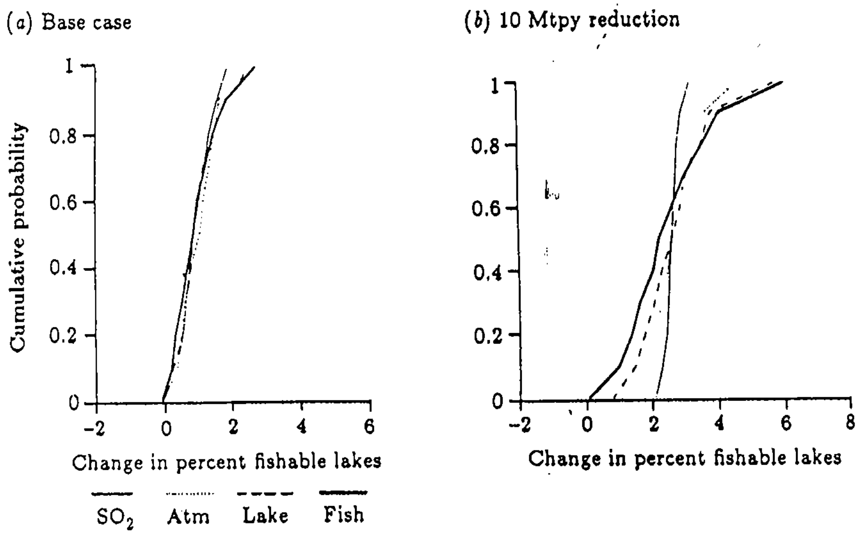

Figure 14.16. Change from 1980 to 2010 in fraction of the Adirondack Park lakes able to support lake trout: cumulative contribution of component uncertainties.

The overall distribution of values is much wider for the acid rain control scenario. An 80% confidence interval includes changes of 1% to 4%, compared with the expected value of 2.3%. This range corresponds to an increase of roughly 30 to 120 additional lakes able to support trout fishing. The range would be still larger for higher degrees of confidence.

The decomposition of uncertainty by component shows that the most significant factors in predicting the change in lakes able to support fish life are the uncertainties in lake chemistry and fish viability functions. By comparison, uncertainties in SO_2 emissions and atmospheric transport have notably smaller impacts, particularly for the emission reduction scenario (where full compliance with the reduction requirement is assumed).

If only the degree of lake acidification is of interest, rather than the biological effects on fish, the dominant uncertainties lie in the modeling of lake chemistry, as seen in *Figure 14.17.* Thus, our results suggest that a better understanding of fish biology together with improved models of regional lake response are most essential to reduce the uncertainties in predicted impacts of emission reduce on aquatic resources.

14.5. Caveats and Conclusions

This chapter has attempted to illustrate how an integrated model of the acid deposition process can be used to estimate the impacts and uncertainties of

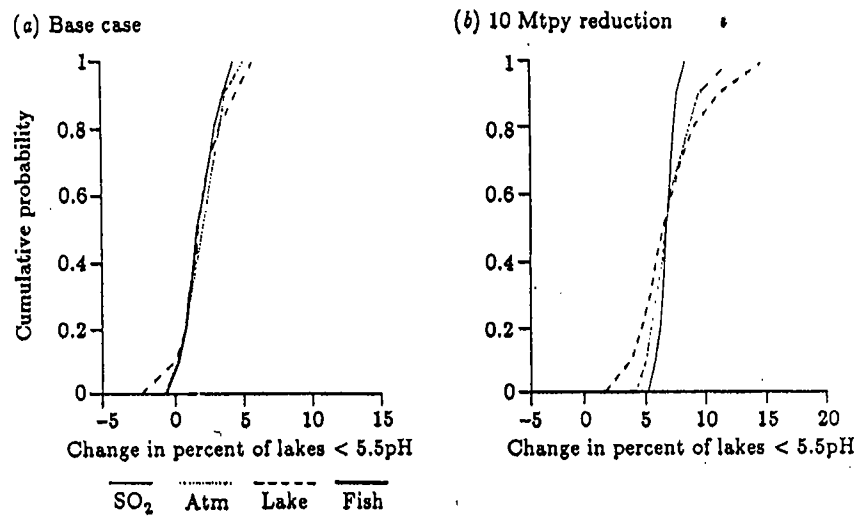

Figure 14.17. Change from 1980 to 2010 in fraction of the Adirondack Park lakes with pH below 5.5: cumulative contribution of component uncertainties.

policy measures currently under consideration to abate acid deposition in North America. We have focused on the problem of regional lake acidification and the consequent loss of fish life, which is the problem that initially brought "acid rain" to the forefront of public attention.

While many significant advances in our understanding of the acid deposition phenomenon have been derived from research over the past decade, we have seen that considerable uncertainties remain. Some of these dominate the ability to predict or discern the benefits of an acid rain control program, or the consequences of taking no action, over the next several decades. For aquatic effects, the present analysis suggests that the greatest uncertainties lie in our understanding of the biological response of different fish species to acidic waters and the processes governing the response of aquatic systems to changes in acid deposition at the regional level. These two factors – particularly the former – contribute most to the overall uncertainty in the estimated impact of emission reductions on the number of lakes potentially able to support fish life in two regions of eastern North America. The additional uncertainties estimated for atmospheric transport–transformation processes and the level of future SO_2 emissions were of less significance to the overall uncertainty for the range of cases examined.

Important caveats, of course, include the assumptions of annual average atmospheric chemistry as a linear process, the adoption of single values to characterize regional deposition levels, and the use of empirically based models as reasonable representations for this level of analysis. Such assumptions can only be refined through additional research and data acquisition, and the development of improved models in the future. Indeed, the extent to which current models may give biased or inaccurate predictions cannot be assessed until verifiably better models are available. Thus, many uncertainties simply cannot be quantified at this time.

It must also be recognized that uncertainty estimates inevitably reflect various technical or professional judgments that are an inherent part of the analysis. For example, the range of scenarios characterizing future SO_2 emission levels must be accepted as reflecting a reasonable characterization of uncertainty based on current estimates or analyses by major parties at interest. Although research *can* help improve analytic *methods*, additional research cannot significantly reduce the uncertainty in estimating SO_2 emissions 30 or 50 years from now. Rather, the power of the modeling framework described in this chapter is to allow the consequences of alternative assumptions about the future to be examined easily and, by this means, to help inform the judgments and opinions important for policy analysis and research.

In the present case, our analysis suggests that despite significant uncertainties, the process of regional lake acidification in the Boundary Waters region of the upper Midwest and the Adirondack Park region of New York appears likely to proceed slowly over the next two to three decades in the absence of any new policy initiatives. The *direction* of change is most likely to follow future SO_2 emission trends, which are highly uncertain. An acid rain control program lowering SO_2 emissions in the eastern USA to 10 Mtpy below 1980 levels would accelerate the recovery of acidic lakes in Adirondack Park, although the magnitude of the recovery (an expected increase over the "no control" case of less than 2% or about 50 lakes capable of supporting trout) could well be masked by the uncertainties and variability in atmospheric processes, lake chemistry, and biological processes. Changes at Boundary Waters would be far less discernible, according to the present analysis.

Our analysis does demonstrate a potential for more dramatic changes in regional aquatic acidification, though with lower probabilities of occurrence. Thus, the need for policy actions inevitably must be weighed against acceptable levels of risk. Further research is expected to reduce current uncertainties and to expand the geographical extent over which regional acidification effects may be quantitatively estimated. Additional research also will allow other potential effects of atmospheric deposition (on materials, forests, etc.) to be evaluated in future analyses.

Acknowledgments

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Notes

[1] Although other species also are involved in acid deposition chemistry, their effects on aquatic systems appear to be indirect via terrestrial systems or, in the case of nitrogen oxide species, related more to episodic events rather than the long-term average lake acidity modeled in this chapter. The impacts of episodic and seasonal events are implicitly reflected in the longer-term acidity and fish viability models.

[2] Some studies project emissions and effects over a time period of 50 years or more. Because of the very large uncertainty in emission estimates over such periods, however, we chose a somewhat shorter time period, believed to be more relevant for current policy decisions regarding acid rain controls.

[3] The English system of units is used in this chapter, reflecting prevalent practice in the USA. To convert short tons to metric tons, multiply by 0.9.

[4] A number of studies project emissions to the year 2030: Braine and Steubi (1987), Placet *et al.* (1986), and McGowin *et al.* (1986). In that time period, the effects of power plant lifetime disappear as all older plants are eventually retired. Emission estimates are then dominated by the assumed demand for electricity and the commercial use of "clean coal technology," offering lower emission levels than current plant designs. Current EPA, DOE, and EPRI projections beyond 2010 continue to assume electricity demand growth rates of 2% to 3% per year. Historically, however, rates several times higher than that also have been sustained (e.g., 7% per year during the 1950s and 1960) and cannot necessarily be ruled out in the future.

[5] This does not affect any subsequent part of the analysis since all emissions are aggregated in the source-receptor model.

[6] These values were judged to represent the widest range of pH that could occur at these locations with current emissions, based on current and historical deposition data.

[7] High values of τ_c imply low transformation rates of SO_2 to SO_4 (which deposits more rapidly than SO_2 in wet form), together with high values of τ_{wp} (low wet deposition rates of SO_2) and would produce very low sulfate deposition values and associated high pH values outside the acceptable range.

[8] Note that the term *ANC* is used interchangeably with alkalinity in this chapter.

[9] Computer experiments indicated that 100 iterations were sufficient to determine consistent estimates of confidence intervals of 90% or less. A larger number of iterations could be expected to yield some differences in the extreme tails of the distributions.

[10] Control costs are reported to illustrate the capabilities of the integrated assessment model. The reader certainly should not infer that these costs would be incurred solely for the purpose of mitigating aquatic effects in the two regions analyzed in this chapter.

[11] Of course, for either of the bounding scenarios – i.e., the EPA projections on the high side or the EPRI scenario on the low – the expected changes would be different from those shown in *Figure 14.9*. Because future emissions in this analysis are assumed to lie anywhere within a range (with uniform probability),

the bounding scenario values themselves are reflected primarily in the tails of the overall distributions for lake acidity shown earlier in *Figure 14.9*.