Integrated Assessment of Acid-Deposition Effects on Lake Acidification

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Abstract: An integrated assessment model is used to estimate SO_2 emission effects on regional lake acidification and fish viability in two regions of North America (Adirondack Park, New York, and the Boundary Waters region of northern Minnesota). An uncertainty analysis is employed to estimate the likely range of possible impacts. Based on emission projections for the United States and Canada, lake acidification in these two regions appears likely to improve slowly over the next two decades. An acid-rain control program will accelerate the recovery of acidic lakes at Adirondack Park, with a projected decrease over the no-control case of approximately 2–11% in the number of lakes below pH 5.5 and a 0.4–6% increase in the number of lakes potentially able to support brook or lake trout by the year 2010. For Boundary Waters, the expected improvements are negligible since deposition levels are relatively low. Our analysis does demonstrate a potential for larger or smaller improvements in these two regions, with lower probabilities of occurrence. Uncertainties in regional lake chemistry and aquatic biology dominate the overall uncertainty in acidification effects estimated for these two regions, within the limitations of the analysis.

Introduction

Research on the problem of acid rain has been focused primarily along disciplinary lines related to the study of precursor emissions, atmospheric transport and transformation, wet and dry deposition processes, and the chemical and biological effects of acidic deposition on aquatic systems, materials, forests, crops, and human health. The general aim of scientific research has been to characterize the nature and severity of the acid-rain problem, and to develop methods to predict the response of environmental systems to changes in parameters governing their physical, chemical, or biological behavior. More limited research efforts in the United States and Europe also have sought methods to systematically integrate research findings across different disciplines in order to assess measures proposed to control acid deposition (Balson and North 1982; Rubin et al. 1986; Alcamo et al. 1987). A key feature of these integrated models is the ability to estimate and prioritize uncertainties in the predicted impacts of emission-abatement policies (Rubin 1990).

In this paper, we describe an integrated stochastic simulation framework used to model the links between emissions of acid-deposition precursors and the consequent effects on regional lake acidification and potential fish habitat. We offer preliminary estimates of the magnitudes and sources of

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uncertainties in the principal components of the problem for two regions of eastern North America: the Adirondack Park area of upstate New York, and the Boundary Waters region of northern Minnesota. Uncertainties not amenable to quantification are discussed in the context of ongoing research programs and their relevance to acid-rain control.

ANALYSIS FRAMEWORK

A generalized modeling framework called the atmospheric deposition assessment model (ADAM) has been developed to analyze acid-rain issues in North America. 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. For the present analysis, the stochastic simulation framework links analytical models of: (1) Precursor emissions at the state or subprovince level; (2) atmospheric transport and transformation; (3) deposition at selected receptors; (4) regional aquatic chemistry; and (5) fish-species viability. Uncertainties in model results are expressed as probability distributions reflecting the combined effect of all uncertain input parameters, which are also expressed probabilistically.

The results shown in this paper consider only the emissions and effects of sulfur dioxide and its reaction products. While other acidic species, such as nitrogen oxides, affect precipitation chemistry, their effects on aquatic systems are often indirect via terrestrial systems or related more to episodic events; sulfur deposition is considered to be the principal input affecting long-term average lake acidity (Galloway et al. 1983; Henriksen and Brakke 1988; Driscoll and Schafran 1988; Wright 1988). Changes in the deposition rates of base cations can also influence the acidity of precipitation and surface waters (Driscoll et al. 1989), but are not considered in the current model.

Emission Trajectories

The emission source regions included in the ADAM framework are identified in Fig. 1. The model contains up to 66 source regions, including the 48 lower 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.

Canadian emission estimates together with the range of U.S. projections shown in Fig. 2 are used to characterize the uncertainty in future baseline SO_2 emissions, i.e., without U.S. initiatives to abate acid deposition. The range of emissions in Fig. 2 reflects recent projections by parties with an interest in the acid-rain debate. Differences for the U.S. utililty sector are principally due to assumptions about the future demand for electricity, the retirement age of coal-fired power plants, and the relative use of coal, nuclear, and other fuels for power generation in the future (McGowin et al. 1986; Placet et al. 1986; Braine and Stuebi 1987; "NAPAP's" 1987). For the nonutility sectors, future SO_2 emissions depend principally on the use of coal in the industrial sector and the extent of emissions control from smelters. U.S. studies project relatively constant emissions from the industrial sector over the next two decades (Braine and Stuebi 1987; Placet et al. 1986). Projected emissions for Canada, which are dominated by smelters, reflect planned SO_2 reductions announced by the Canadian government (*The Canadian* 1987).

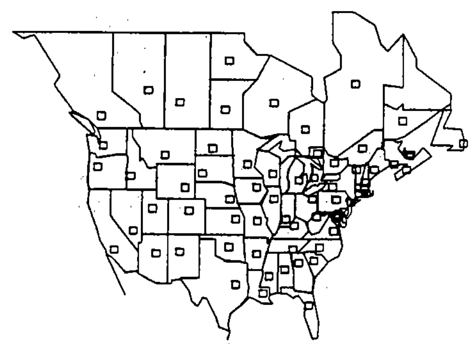

FIG. 1. Model Source Regions (Squares Indicate Source Centroids)

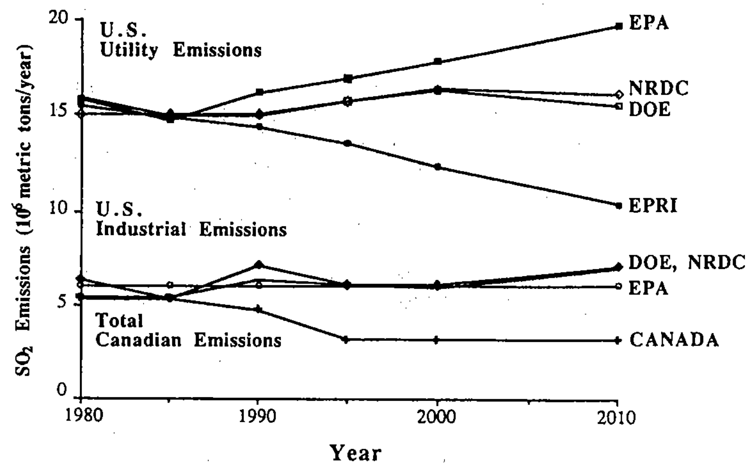

FIG. 2. SO_2 Emission Projections for United States and Canada (Base-Case Scenario)

To characterize uncertainty, a uniform probability distribution is assumed for future U.S. utility emissions in the range encompassed by the high and low scenarios. For the nonutility sectors, future U.S. emissions are assumed constant at the average of available estimates, with uncertainty taken to be normally distributed. The combined emissions for the United States and Canada then defines the base-case scenario, with all emissions disaggregated to the state or subprovince level (66 source regions).

A second scenario assumes an acid-rain control program for the United States in which SO_2 emissions in 1995 are reduced by 9,100,000 metric tons per year (9.1 mtpy) below 1980 levels, with emission reductions allocated to individual states based on 1980 emissions exceeding 520 ng/j (1.2 lb SO_2/10^6 Btu). This scenario reflects the emission reduction called for in most congressional proposals of the late 1980s. (The Clean Air Act amendments, adopted in 1990, also call for an overall 9.1-mtpy reduction, but delay full implementation until the year 2000. Thus, the resulting improvements in aquatic systems modeled in this paper would occur later in time. The emissions trading provisions of the new Clean Air Act also may result in a different pattern of emission reductions among various states, leading to some quantitative differences in the magnitude of predicted effects for specific regions.) To assess the maximum effect of this reduction on lake chemistry and potential fish viability, emissions uncertainty for this scenario reflects only the estimate of current emissions (i.e., no additional uncertainties are included to reflect possible delays in full compliance with emission-reduction requirements).

Atmospheric Transport and Deposition

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 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 meterological variations over large geographic areas; and the role of nitrogen oxides (NO_x), volatile organic compounds (VOCs) and other species in atmospheric deposition chemistry. While NO_x and VOCs are known to affect the formation of photochemical oxidants, which, in turn, may affect sulfur chemistry, the precise nature of such interactions and the magnitude of the resulting nonlinearity are subjects of ongoing research (Chang et al. 1987; Venkatram et al. 1990). Preliminary results indicate that nonlinearities are most important for wet sulfur deposition, which is oxidant-limited, but that this importance is primarily for space and time scales that are small relative to the major environmental effects of concern. Thus, linear atmospheric transport models are appropriate for modeling annual average sulfur deposition, and provide the basis for most evaluations of emission-control strategies [e.g., Ellis (1988)].

Two linear modeling methods are used to represent source-receptor relationships in the integrated modeling framework. 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 emission at each source region. Total annual averages are then calculated using superposition. The ADAM data base contains default transfer matrices based on the Advanced Statistical Trajectory Regional Air Pollution (ASTRAP) model (Shannon 1981), which has been widely used for acid-deposition analysis. A second approach directly incorporates the atmospheric transport model developed by Fay et al. (1985, 1986). This model uses relatively simple formulations of atmospheric chemistry and meteorology calibrated on data for wet sulfate deposition at 109 sites in North America for a recent three-year period. Predictions from this model have been found to be comparable to those from more complex Lagrangian models (Matthias and Lo 1986). The model of Fay et al. (1985) is used for the results shown in this paper, with uncertainties ascribed to both meteorological and chemical rate parameters (Table 1).

In this study, acid-deposition effects on aquatic systems are related to precipitation acidity, which is determined by the annual average hydrogen ion concentration. An extensive study of the correlation between precipitation acidity and the wet deposition of sulfate for various regions of the United States has been undertaken by Hales (1982). The regression relationship shown in Table 2 is used with parameter values for the Adirondack Park and Boundary Waters receptor regions. Mean annual rainfall values

TABLE 1. Source-Receptor Model Parameter and Prior Uncertainties

Parameter[a] (1)	Base value[a] (2)	Standard deviation[b] or range[c] (3)
(a) Meteorological Factors		
Wind speed (zonal, west to east)	4.0 m/s	0.74 m/s
Wind speed (meridional, south to north)	5.9 m/s	0.74 m/s
Horizontal diffusivity	4.3×10^6 m²/s	0.43×10^6 m²/s
(b) Chemical Factors—Time Constants		
Conversion ($SO_2 \rightarrow SO_4$), τ_c	1.9×10^5 s	$(1.0–7.5) \times 10^5$ s
Wet SO_2 deposition, τ_{wp}	11.3×10^5 s	$(1.5–15.0) \times 10^5$ s
Dry SO_4 deposition, τ_{ds}	12.5×10^5 s	$(0.25–25.0) \times 10^5$ s
Wet SO_4 deposition, τ_{ws}	0.6×10^5 s	$(0.15–2.5) \times 10^5$ s

[a]Based on Fay et al. (1985, 1986).
[b]Standard deviations for meteorological factors based on historical data (Small et al. 1989).
[c]Ranges for chemical factors based on literature values.

TABLE 2. Parameter Values and Prior Uncertainty Estimates for Precipitation Acidity Model[a]

Parameter (1)	Definition (2)	Adirondacks Park Mean (3)	Adirondacks Park Standard deviation (4)	Boundary Waters Mean (5)	Boundary Waters Standard deviation (6)
a_i	regression coefficient	16.5×10^{-6}	1.65×10^{-6}	2.3×10^{-6}	0.23×10^{-6}
b_i	regression coefficient	1.618	0.162	1.573	0.157
R	annual rainfall (mm)	1,045	91	680	68

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

and variances for each region are based on historical data (U.S. 1982), while the uncertainty in regression coefficients has been estimated from the work of Hales and others (McNaughton 1981; Chen et al. 1987). The present approach implicitly assumes that atmospheric levels of photochemical oxidants, base cations, and other species that may affect deposition chemistry over the simulation period do not change appreciably from recent historic values. This is another limitation of our approach.

To characterize the combined uncertainties in sulfur deposition and precipitation acidity, the sensitivity method of Hornberger and Cosby (1985) and Hornberger et al. (1986) was applied to the model predictions. The method is implemented by assuming the prior distributions for uncertain model inputs shown in Tables 1 and 2, then determining which combinations of these inputs given acceptable results for precipitation pH, consistent with observed wet deposition data. This allows estimation of the posterior uncertainty distributions of the model input and output. Only those simulations that resulted in values of annual average precipitation pH consistent with the range of observed data at six geographically dispersed locations in eastern North America (Adirondack Park, New York, pH = 4.1–4.5; Blue Ridge Mountains, Tennessee/North Carolina, 4.1–4.5; Boundary Waters, 4.7–5.1; New Haven, Connecticut, 4.1–4.6; Vermont/New Hampshire, 4.1–4.5; and western Pennsylvania, 3.95–4.3) were accepted for use in the subsequent analyses.

Aquatic Chemistry

Predictive models for aquatic acidification span a range of scientific and computational complexity, in large part driven by the intended use of the model (Schecher and Driscoll 1988). Empirical equilibrium models are based on fundamental concepts of charge balance and provide a simple representation well suited to regional predictions; however, they require the selection of an empirical neutralization factor, and they do not provide dynamic predictions (Henriksen 1979; Wright and Henriksen 1983; Schnoor et al. 1986). Such models predict the steady-state acid-neutralizing capacity (ANC) and corresponding pH in a lake by assuming that a given fraction of the acid deposition to a basin is neutralized by cation exchange and chemical weathering of soil and rocks in the watershed. The fraction of incoming acid, which is neutralized is computed by subtracting the current ANC from an estimate of the predeposition (or zero-deposition) ANC ($= ANC_0$) and comparing this change to the quantity of acid deposition. The neutralization fraction is thus $N_F = 1 - (ANC_0 - ANC)/D$, where D is the incoming acid deposition adjusted for evapotranspiration. This approach represents the full range of chemical and biological processes in a watershed and lake system with a single constant, which must be empirically determined for each lake. The equilibrium models are relatively easy to apply to many lakes in a region, though difficulties arise when attempting to characterize the variation in neutralization fraction from one lake to another.

Dynamic, mechanistic models incorporate a variety of hydrologic, mass balance, and chemical equilibrium and kinetic relationships to predict the evolution of lake chemistry resulting from a dynamic profile of acid deposition. Examples include the Integrated Lake Watershed Acidification Study (ILWAS) model (Gherini et al. 1985), the Birkenes model (Christophersen et al. 1982), the Modeling Acidification of Ground Water in Catchments (MAGIC) model (Cosby et al. 1985a, 1985b), and the model of Lung (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, although progress has been made in this direction (Cosby et al. 1989; Posch and Kämäri 1990).

The present study employs a dynamic direct distribution model that combines features of the empirical equilibrium and dynamic mechanistic approaches (Small and Sutton 1986; Small et al. 1987). The model uses a modified form of the Henriksen-Wright equilibrium model to predict the equilibrium ANC of a watershed, but applies the model to the probability distribution functions of ANC and pH for a region. Lake-to-lake variations in the neutralization fraction are explicitly incorporated by the identification and use of the mean neutralization fraction, $\bar{N}_F$, together with the standard deviation of the neutralization fraction, σ_{N_F}, for the region. A correlation

factor, ρ, is incorporated to account for the observation that watersheds with higher initial ANC should provide a higher degree of neutralization of subsequent acidic deposition (e.g., Marmorek et al. 1990).

The direct distribution model computes the mean and variance of the equilibrium ANC distribution in a region as a function of the acid-deposition input rate (Small and Sutton 1986). However, lakes are not expected to respond instantaneously to changes in the rate of acid-deposition input. To account for delays in the response, a dynamic version of the direct distribution model was developed by assuming that the lakes in a region approach a new equilibrium value of ANC in an exponential manner with a characteristic time constant τ_{ANC} (Small et al. 1987). This time constant is used to empirically approximate system lags resulting from lake detention times or watershed delay processes, such as sulfate adsorption on soils. At each time step in the model, the moments of the regional ANC distribution that would be in equilibrium with the given acid-deposition rate are first computed. These moments are modified to account for the fact that only a partial shift between the previous time step and the predicted equilibrium is assumed to occur, as controlled by τ_{ANC}. The moments of the predicted ANC distribution are used to determine the parameters of the lognormal ANC distribution for the region, from which the parameters of the regional distribution of pH are derived. A full description of the methods and equations needed to implement the direct distribution model is found in Small and Sutton (1986) and Small et al. (1987).

Parameters of the direct distribution model are estimated from data for each region of interest. An important issue is whether the available lake chemistry data can provide a representative sample of the overall population of lakes in a region. To address this problem, the Environmental Protection Agency (EPA) conducted the National Surface Water Survey (NSWS), in which lakes were selected and monitored from stratified samples of target sensitive regions ("Characteristics" 1986; Landers et al. 1988). The resulting data sets provide the most representative characterization currently available of the overall distribution of regional lake chemistry in the United States.

Data for the NSWS subregions corresponding to the Adirondack Park and Boundary Waters (northeastern Minnesota) were used to evaluate parameters of the direct distribution model for these regions, with uncertainty assumed to be either normal or uniformly distributed (Table 3). The population weights of the NSWS were used in the estimation of the distribution parameters, as described by Small et al. (1988). The stratified samples upon which the EPA lake survey data were based represent approximately 1,500 lakes in each region with surface areas of 4–2,000 ha ("Characteristics" 1986). However, both study regions have a total population of lakes in this size range that is nearly twice as large as the EPA sample. The inclusion of lakes smaller than 4 ha would lead to another doubling of the population estimates including more lakes with low ANC (Johnson et al. 1989). The distributions are thus only partially representative of the overall set of lakes, and exclude the very small lakes, which tend to be more sensitive and acidic. The lake chemistry distributions determined from the NSWS provide the initial conditions for the simulation period, which begins in the year 1980.

Due to the lack of historical data on regional lake chemistry, the neutralization fraction distributions are estimated based on judgmental assessments for each region that reflect the effects of both dry and wet deposition. The mean neutralization fraction is a critical parameter in the model that determines the magnitude of the regional lake-chemistry response to changes in deposition. The smaller the value of $\bar{N}_F$, the greater the predicted response

TABLE 3. Aquatic Chemistry Model Parameters and Uncertainty

Parameter[a] (1)	Adirondacks Mean[a] (2)	Adirondacks Range or σ^b (2)	Boundary Waters Mean[a] (4)	Boundary Waters Range or σ^b (5)
ANC distribution parameter, θ (μeq/L)	−41	(−51/−31)	23	(13/33)
ANC distribution parameter, ξ	4.96	(0.09)	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 neutralization fraction, $\bar{N}_F$	0.4	(0.0/0.8)	0.5	(0.1/0.9)
Neutralization fraction variance, σ^2_{NF}	0.047	(0.027/0.067)	0.047	(0.027/0.067)
ANC/neutralization fraction correlation, ρ	0.5	(0.1/0.9)	0.5	(0.1/0.9)
ANC characteristic time, τ_{ANC} (year)	8	(1/15)	8	(1/15)
pH-ANC tranformation factor, p_1	5.24	(5.14/5.34)	5.4	(5.2/5.6)
pH-ANC tranformation factor, p_2 (μeq/L)	9.37	(fixed)	12.77	(fixed)
pH-ANC transformation factor, p_3 (μeq/L)	11.02	(1.02/21.02)	7.58	(−2.4/17.6)

[a]Based on Small and Sutton (1986) and Small et al. (1988). The probability density function for ANC is given by: $f(\text{ANC}) = [1/(\text{ANC-}\theta)\ \phi\sqrt{2\pi}]\exp\{-[\ln(\text{ANC}-\theta) - \xi]^2/2\phi^2\}$; ANC $\geq \theta$. The pH-ANC relationship is given by: pH $= p_1 + 0.4343 \text{ arcsinh } [(\text{ANC} - p_3)/(p_2)]$.

[b]Ranges assume a uniform distribution. A single value is the standard deviation of a normal distribution.

to a change in deposition. Previous studies have determined values of the neutralization fraction ranging from 0. to 0.4 (Henriksen and Brakke 1988; Schindler et al. 1989). Others suggest that these values are appropriate only for sensitive systems, and report average values of $\bar{N}_F$ ranging from 0.45 to 0.75 (Johnson et al. 1985). As shown in Table 3, our uncertainty in $\bar{N}_F$ allows for values ranging from 0 to 0.8 for the Adirondack region and 0.1 to 0.9 for northeastern Minnesota. More recent paleolimnological studies of inferred historical changes in Adirondack lake chemistry suggest that significant ANC changes have occurred only in lakes with low ANC values, below 25 μeq/L, and that even these changes have been small in magnitude (Sullivan 1990). If the paleolimnological studies are correct and representative, then regional values of N_F closer to one would be more appropriate for our model. The magnitude of the future changes in lake chemistry resulting from a reduction in deposition would then be skewed towards the low end of the range of predicted values determined in this study.

An additional uncertainty in the structure of the current lake-chemistry model is the assumption that the neutralization fractions in a region remain constant over time. Applications of regional mechanistic models suggest that these fractions may be reduced over time, even under constant deposition, due to the gradual loss of the base saturation of the soils (Labieniec et al. 1989). This model uncertainly has implications for the future sensitivity and response of lakes to changes in deposition, and is an important priority for continued study.

Fish Viability

Although basic research efforts are providing improved understanding of fish response to environmental factors (Schindler et al. 1985, 1989; Baker et al. 1990), current models describing the biological effects on aquatic life associated with changes in regional water chemistry remain mostly empirical in nature. A relatively simple logistic regression model (Table 4) has been applied by Reckhow et al. (1987) to describe the probability of finding a given fish species in lakes of a given pH. The model assumes that fish survival is related to the long-term average acidity of the lake. While episodic events do have important impacts on fish populations, the average acidity of a lake provides the baseline value upon which episodic depressions are superimposed (Eshleman 1988; Schaefer et al. 1990). Average and minimum pH values are thus, highly correlated; the long-term average lake acidity may thus be used to provide a first estimate of those lakes where fish populations are lost due to both chronic and acute impacts.

Parameters other than pH, such as calcium, magnesium, and aluminum ion concentrations are also known to affect fish survival (Reckhow et al. 1987; Baker et al. 1990). Many of these other parameters are highly correlated with pH, so the presence-absence relationship based on pH does indirectly incorporate their effects. However, possible changes in these correlations with changing lake chemistry could alter future predictions. As such, incorporation of base cations and aluminum in the regional lake chemistry distribution models is another important priority for future research.

The relationship between pH and the fish-survival probability in Table 4 implicitly accounts for the effects of nonchemical habitat factors, such as lake depth and temperature, which also impact the probability of fish survival. This is reflected in the behavior of the logistics curve at high values of pH. Using the mean parameter values in Table 4, the predicted brook trout and lake trout survival fractions at a pH of 7.0 are 0.76 and 0.66, respectively. Thus, even without acidification, only a fraction of the lakes are predicted to support fish populations. The pH survival function in Table 4 is assumed to be applicable to all lakes in the region, and the fraction of lakes potentially able to support a given fish species is calculated as the expected value of the function integrated over the entire pH distribution for the region. Alternative methods for combining the effects of lake-acidification chemistry and baseline habitat are discussed by Baker et al. (1990).

Estimates of Acidification Effects

Parameter uncertainties in each component model, expressed as probability distributions, were simulated using the Latin hypercube sampling

TABLE 4. Parameter Values and Uncertainty Estimates for Fish Viability Model[a]

Parameter (1)	Definition (2)	Brook Trout Mean (3)	Brook Trout Variation[b] (4)	Lake Trout Mean (5)	Lake Trout Variation[b] (6)
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: $P_i = 1/\{1 + \exp[\alpha_i + \beta_i(\text{pH}_l)]\}$, where pH$_l$ = lake pH.

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

method. The Latin hypercube method employs stratified sampling from each of the uncertain model inputs and allows a representative sample to be obtained with a smaller sample size than would be required if random, Monte Carlo sampling methods were employed (e.g., McKay et al. 1979; Jaffe and Ferrara 1984). The results shown herein are based on 100 simulations of the model with predictions of precipitation pH that fall within observed, accepted ranges for the year 1980. The model results of interest include probability distributions for SO_2 emissions, sulfur deposition, precipitation pH, lake ANC and potential trout viability for each year between 1980 and 2010. Fig. 3 shows results for the first and last year of the simulation at Adirondack Park for the acid-rain control scenario. The 90% confidence interval for wet sulfur deposition is seen to range from 4.8 to 9.3 kg S/ha-year for the base year, and from 2.6 to 5.1 kg S/ha-year in 2010. The average reduction in deposition is 46%, compared to 47% predicted in a separate analysis using the ASTRAP model. Decreases in the fraction of acidic lakes below pH 5.5, and increases in the fraction of lakes supporting brook and lake trout also are seen in Fig. 3.

A stochastic sampling procedure also was developed to obtain distributions for the overall change in results from 1980 to 2010 for each scenario, and for the net difference in 2010 between the base case and the emission-reduction scenario when parameters common to both cases were sampled in an identical manner. We use these comparative results, shown in Figs. 4 and 5, to depict the projected effects on lake acidification.

Results for lake chemistry are summarized as the fraction of lakes below a given pH. The set of lakes considered is limited to those targeted in the EPA Eastern Lake Survey ("Characteristics" 1986); i.e., lakes with surface areas between 4 and 2,000 ha. A pH of 5.5 is chosen to represent acid levels where effects on fish life are likely to be pronounced. For Adirondack Park, the 90% confidence interval for the net effect of the acid-rain control scenario is an expected decrease from 1980 to 2010 of approximately 4–17%

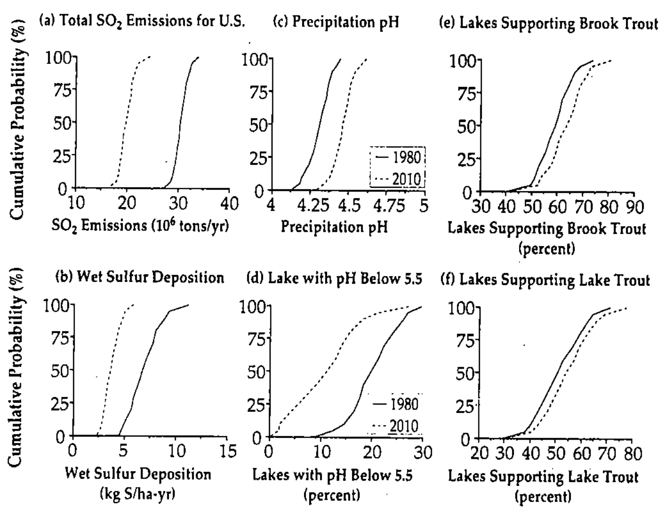

FIG. 3. Probabilistic Results for SO_2 Emission-Reduction Scenario at Adirondack Park Receptor

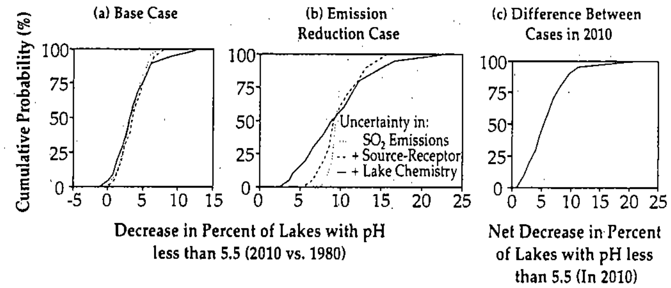

FIG. 4. Change in Fraction of Adirondack Lakes with pH below 5.5 (1980–2010)

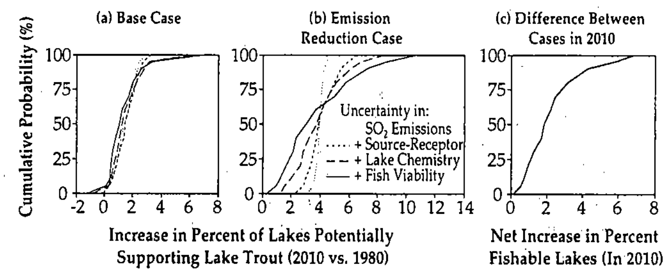

FIG. 5. Change in Fraction of Adirondack Lakes Potentially Able to Support Lake Trout (1980–2010)

in the number of lakes with pH $\leq$ 5.5 [Fig. 4(*b*)], compared to 0–9% decrease in the base case [Fig. 4(*a*)]. The net difference between the two scenarios in 2010 [Fig. 4(*c*)] is a median increase of 5.5% in the number of lakes crossing the pH 5.5 threshold as a result of the U.S. emission reductions. The 90% confidence interval is 1.5–11%. If lake-survey data are extrapolated to the total population of approximately 2,500 lakes, the 90% confidence interval represents approximately 40–300 additional lakes crossing pH 5.5, though all acidic lakes recover to some degree. Larger numbers would result if small lakes (less than 4 ha) were included in the analysis. Results for the Boundary Waters region shows insignificant changes in lake chemistry since deposition values in that region are relatively low. Less than 1% of lakes in the region currently have pH below 5.5, and less than a third have pH below 6.7 ("Characteristics" 1986).

Applying the distribution of lake ANC to the fish presence-absence models for brook trout and lake trout yields a measure of the potential change in fish habitat. Fig. 5 shows results for lake trout at the Adirondack Park. The net benefit of the SO_2 emission reductions in 2010 [Fig. 5(*c*)] is an expected improvement of 2% in the number of lakes potentially able to support lake trout. The 90% confidence interval ranges from 0.4–5%. Based on the total lake population for the Adirondack Park, this would represent an increase of about 10–130 lakes potentially able to support lake trout relative to the base case. Comparable results for brook trout show an increase in the percentage able to support the fish population of from 0.6–6%, corresponding to 15–150 lakes. For the Boundary Waters region, our model predicts that fewer than 20 additional lakes (out of approximately 3,000) would potentially support brook or lake trout given the relatively low deposition and acidity levels that already exist.

Figs. 4 and 5 also illustrate how the results for lake acidity and potential fish viability are influenced by uncertainties in each of the component models. Uncertainties in parameters of the fish presence-absence and lake-chemistry models have the greatest influence on the overall result, while uncertainties in SO_2 emissions and atmospheric transport parameters provide smaller contributions.

Caveats

Many scientific uncertainties potentially relevant to an assessment of acid-deposition effects are not currently subject to quantification. These include the magnitude of potential errors in annual average source-receptor relationships, the effects of dry deposition on changes in watershed chemistry, the effects of changes in base cation deposition rates, temporal changes in neutralization fractions in the lake acidification model, the relative import of episodic versus average baseline lake chemistry, and the impact on fish of lake habitat and chemical constituents other than pH. Different quantitative results also would be expected for other emission-reduction scenarios or base-case emission projections. Similarly, our results for the Adirondack Park and Boundary Waters regions cannot be applied to other regions of North America (e.g., southern U.S. watershed systems, which, because of sulfate adsorption in soils, appear to have significantly longer time lags in their response to atmospheric deposition than the northern lake systems modeled here). Also, judgments concerning the magnitude of the predicted impacts must consider the number of lakes included in other regions, such as the 700,000 lakes in eastern Canada.

Additional research and data collection should continue as planned emission-reduction strategies are implemented, helping to reduce key uncertainties and improve the predictive power of future integrated models. Futhermore, the techniques developed in this study for linking multiple submodels of varying complexity and evaluating overall uncertainties in model results should be useful for the evaluation of other large-scale environmental problems where a linked, integrated assessment is required.

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Appendix. References

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