PREDICTING RECOVERY FROM ACIDIC DEPOSITION: APPLYING A MODIFIED TAF (TRACKING AND ANALYSIS FRAMEWORK) MODEL TO MAINE (USA) HIGH ELEVATION LAKES

JOSHUA J. LAWLER[1,*], JONATHAN RUBIN[2], BERNARD J. COSBY[3],
IVAN J. FERNANDEZ[4], JEFFREY S. KAHL[5] and STEPHEN A. NORTON[6]

[1]*Margaret Chase Smith Center for Public Policy and the Department of Wildlife Ecology, University of Maine, Orono, ME 04469;* [2]*Margaret Chase Smith Center for Public Policy, University of Maine, Orono, ME 04469;* [3]*Environmental Sciences, University of Virginia, Charlottesville, VA 22903;* [4]*Department of Plant Soil, and Environmental Sciences, University of Maine, Orono, ME 04469;* [5]*Senator George J. Mitchell Center for Environmental & Watershed Research, University of Maine, Orono, ME 04469;* [6]*Department of Earth Sciences, University of Maine, Orono, ME 04469-5790* (*author for correspondence: current address: Department of Zoology, Oregon State University, C/o U.S. Environmental Protection Agency, 200 SW 35th St. Corvallis, OR 97333 USA, e-mail: lawler.joshua@epa.gov; Tel: (541) 754-4834; Fax: (541) 754-4799)*

Abstract. We adapted a reduced-form model, built to predict the aquatic effects of alternative nitrogen and sulfur emissions scenarios on Adirondack lakes, New York, for use on high elevation lakes of Maine (HELM), USA. The Tracking and Analysis Framework (TAF) model was originally designed to evaluate the biotic, economic, and health effects of acid deposition. The TAF model developed in our study was used to assess the biotic effects of different levels of sulfate deposition resulting from alternative emissions scenarios. The aquatic portion of the model is based on a lumped-parameter watershed chemistry model, MAGIC (Model of Acidification of Groundwater in Catchments). The original TAF model was built by calibrating MAGIC to 33 lakes in the Adirondack Mountains. We calibrated MAGIC to 78 HELM lakes, and built reduced-form models from these MAGIC predictions. We evaluated TAF predictions of acid neutralizing capacity (ANC), a fish acid stress index (ASI), and the probability of fish presence in 2030 for four different SO_2 emissions-reduction scenarios. The most dramatic emissions reduction scenario produced only modest increases in mean ANC (16.8 μeq/L $\pm$ 7.9 μeq/L) and slight increases in mean predicted probability of presence of acid-sensitive fish (0.07 $\pm$ 0.09) across all lakes. However, a small number of lakes were predicted to have more substantial increases in ANC and improvements in other conditions for acid-sensitive fish. Our results reflect the reality that many of the high elevation lakes of Maine historically had low ANC and that some were even acidic in pre-industrial times. Thus, 'recovery' for most of the high elevation lakes of Maine will be modest under any scenario of reduced acidic deposition.

Keywords: acid deposition, fish, lake acidification, lake recovery, Maine, reduced-form modeling, watershed modeling

1. Introduction

Some surface waters in the northeastern United States have experienced adverse effects of acid deposition including acidification, aluminum mobilization, and the loss of biota (Driscoll *et al.*, 2001). The Clean Air Act Amendments of 1990 were

proposed, in part, to address these and other effects of acid deposition. The Acid Deposition Control Program, established by Title IV of the 1990 amendments called for a reduction in SO_2 emissions to approximately 40% of 1980 levels by 2010. These regulations necessitated accompanying monitoring programs to evaluate the status of surface waters in response to the legislation.

The Direct/Delayed Response Project (DDRP) was initiated to assess the effects of sulfur deposition on the lakes of the Adirondack Mountains in New York and the streams of the Blue Ridge Province and the Middle Appalachian region of the U.S. (Church *et al.*, 1989, 1992). Following the DDRP, the Tracking and Analysis Framework (TAF) model was developed as part of a program funded by the U.S. Department of Energy to evaluate the potential effects of different pollutant-emissions scenarios on human health, visibility, soils, and aquatic systems (Bloyd *et al.*, 1996). TAF integrates models of emissions, atmospheric transport, watershed chemistry, fish biology, economics, health, and visibility. The model has several modules, each of which contains a set of individual models.

The soil and aquatic modules in TAF are composed of a series of linked reduced-form models (Sinha *et al.*, 1998). Reduced-form modeling employs relatively simple models to approximate the results of more complex "full-form" process models. By using mechanistic rule sets or statistical relationships, a relatively simple reduced-form model can produce results similar to those generated by a more complex model but in significantly shorter time frames that require fewer computer resources. The reduced-form models in the soil and aquatic modules of TAF use simple statistically fit functions to approximate the results of the more complex Model of Acidification of Groundwater in Catchments (MAGIC, Cosby *et al.*, 1985a,b).

The TAF model was originally applied to 33 lakes in the Adirondack Mountains in New York, USA. This was a population of low acid neutralizing capacity (ANC) lakes known to be relatively sensitive to acid deposition (Church and Van Sickle, 1999; Stoddard *et al.*, 1998). Earlier modeling efforts demonstrated the potential for increases in lake-water ANC given implementation of Title IV amendments to the Clean Air Act. Although fish populations in a number of Adirondack lakes were predicted to improve (Rubin *et al.*, 1992), the magnitude of the initial changes was not expected to be large (Sinha *et al.*, 1998).

A subset of small Maine lakes above 600 m in elevation has been shown to be as acidic as lakes in the Adirondacks (Kahl *et al.*, 1991). Both the water chemistry and the biota of these small headwater lakes are potentially dependent on past and future emissions regulations. We investigated the effects of four alternative future SO_2 emissions scenarios on the fish of these high elevation lakes using the TAF model. We compared the potential effects of four different emissions scenarios outlined in the original TAF model (Palmer and Burtraw, 1997; Burtraw *et al.*, 1998). These comparisons differed with respect to: the implementation of Title IV, the life of power plants, and the growth in electricity demand. We calibrated MAGIC to 78 high elevation lakes, and then refit TAF's reduced-form models for

soil and water chemistry with the MAGIC model predictions. We related changes in lake water ANC to predicted acid stress for fish using laboratory-derived effects models (Baker *et al.*, 1990).

2. Methods

2.1. MODELS

Our application of the TAF model concentrated on the soil and aquatic effects module. This module contains several sub-modules that predict water chemistry characteristics including ANC, pH, and concentrations of calcium (Ca) and aluminum (Al), soil base saturation, and a few simple characteristics of lake fish communities. The reduced-form models that comprise these sub-modules were originally developed and fit by Sinha *et al.* (1998) for predicting the effects of different acid deposition levels on the lakes of the Adirondacks. Applying the TAF model to the high elevation lakes of Maine required running the MAGIC model and refitting these reduced-form models.

MAGIC (Cosby *et al.*, 1985a,b) is a model of intermediate complexity that tracks the movement of ions through a watershed, estimating the changes in stream or lake water chemistry in response to atmospheric deposition. MAGIC simulates reactions involving sulfur adsorption/desorption, Al dissolution/precipitation/speciation, inorganic carbon dissolution/speciation, and cation weathering and ion exchange. The model uses mass balance equations to track the flux of major ions to and from the soils in response to weathering, deposition, and runoff. Several improvements have been made to the original model (Cosby *et al.*, 1985a,b) including an alternative model for predicting Al concentrations (Sullivan and Cosby, 1998) and an alternative module for simulating nitrogen dynamics in soils (Cosby *et al.*, 2001). We used the MAGIC version described by Cosby *et al.* (2001), but did not use the dynamic nitrogen module. For these simulations nitrogen deposition was held constant and there were assumed to be no changes in future nitrogen retention characteristics of the catchment soils.

Before MAGIC can be used to make predictions about a given system, it must be calibrated to observed data (Cosby *et al.*, 1985a,b). The calibration process involves first parameterizing the model with a set of known input oriented watershed attributes, including water and soil chemistry. The model is then run with a set of observed atmospheric and hydrological inputs. The simulated output variables are compared to a corresponding set of empirically measured variables. If the simulated output fails to match the measured variables, a second set of MAGIC model parameters (e.g., cation weathering rates) are adjusted through an iterative optimization process to improve the model predictions. After MAGIC is calibrated, it can be used to make predictions with new atmospheric and hydrological inputs.

TAF's reduced-form models were used to approximate the output from MAGIC model simulations. We were able to use the same basic reduced-form models developed by Sinha *et al.* (1998) for Ca, soil base saturation, and Al, but made slight adjustments to the model for ANC and the ANC-pH relationship for the application to the Maine lakes. These two adjustments are discussed in the section on TAF model calibration. Full descriptions of the reduced-form models summarized below can be found in Sinha *et al.* (1998).

Sinha *et al.* (1998) modeled ANC in TAF as a linear function of the ANC values predicted by MAGIC assuming an exponential approach to equilibrium ANC in the soil water. Equilibrium ANC was approximated using the fraction of the acid deposition that was neutralized in the soil before entering the lake (Small *et al.*, 1995).

$$\text{ANC}_t = (1 - d_f)\,\text{ANC}_{s,t} - d_f\,D_t \tag{1}$$

$$\text{ANC}_{s,t+\delta t} = \text{ANC}_{s,t} + [\text{ANC}_{se,t+\delta t} - \text{ANC}_{s,t}]\left(1 - e^{-\frac{\delta t}{\tau}}\right) \tag{2}$$

$$\text{ANC}_{se,t} = \text{ANC}_{s,0} - (1 - N_{F,t})(D_t - D_0) \tag{3}$$

In these equations d_f is the fraction of direct deposition to the lake, the acid which bypasses the soil; $\text{ANC}_{s,t}$ is the soil water ANC at time t; D_t is the acid deposition at time t; $\text{ANC}_{se,t}$ is equilibrium ANC at time t; τ is the lag time with which ANC approaches equilibrium; $N_{F,t}$ is the fraction of acid deposition that is neutralized in the watershed at time t; and $\text{ANC}_{s,0}$ is the initial soil water ANC concentration (at time 0). Sinha *et al.* (1998) modeled the neutralization fraction ($N_{F,t}$) as being dynamic (Equations 4 and 5).

$$\Delta N_{F,t} = m(\text{Dep}_t - \text{DNF}) \tag{4}$$

$$N_{F,t} = N_{F,0} + m\left(\left(\sum \text{Dep}_t - tp\right)\text{DNF}\right) \tag{5}$$

In these two equations, m is a slope factor; Dep_t is the average deposition over one time interval; DNF is the nominal deposition level, at which there is no change in N_F; and tp is the number of time periods elapsed. TAF uses a time period of five years.

The TAF model predicts pH as a function of ANC using the pH-ANC relationship of Small and Sutton (1986),

$$\text{pH} = a + b\left(\text{arcsinh}\left(\frac{\text{ANC} - d}{c}\right)\right) \tag{6}$$

in which the parameters a, b, c, and d were all fit as part of the model calibration process. TAF uses a reduced-form model for Ca that is very similar to the model used

to approximate ANC. As with ANC, lake water calcium approaches equilibrium with a lag, τ,

$$Ca_{t+\delta t} = Ca_t + [Ca_{e,t+\delta t} - Ca_t]\left(1 - e^{\frac{-\delta t}{\tau}}\right) \tag{7}$$

in which $Ca_{e,t+\delta t}$ is Ca at a future time step and Ca_t is Ca at time t. Equilibrium Ca at time t is given by,

$$Ca_{e,t} = m(Dep_t) + b \tag{8}$$

in which m and b are the slope and intercept of the linear relationship fit in the calibration process. Inorganic Al is modeled as a function of pH using the empirical relationship of Sullivan and Cosby (1998),

$$pAl_t = [pH_t - pH_{2000}] + pAl_{2000} \tag{9}$$

that assumes an equivalent change in Al and pH since the base year (2000 in our analysis). In this equation, pAl_t and pH_t are the concentration of Al and the pH at time t and pAl_{2000} and pH_{2000} are the concentration of Al and the pH in the base year. Soil base saturation is modeled as an empirical linear relationship between the change in base saturation in the surface soil in a one-year period and the change in sulfate deposition from the base year (Sinha $et\ al.$, 1998),

$$\Delta BS_t = m(D_t - D_{2000}) + b \tag{10}$$
$$BS_t = BS_{t-1} + \Delta BS_t \tag{11}$$

where BS_t is soil base saturation at time t and D_t and D_{2000} are deposition at time t and deposition in the base year respectively. The parameters m and b are the slope and the intercept of the linear relationship fit in the model calibration process.

Assessments of the effects of acid deposition on fish are made through the calculation of an acid stress index (ASI), an experimentally derived logistic function of Al, Ca, and pH (Baker $et\ al.$, 1990),

$$ASI = \frac{100}{1 + e^x} \tag{12}$$

in which the parameter x is specific to fish species with different levels of acid tolerance. Separate models were used to predict effects for fish that were generally tolerant of acid stress (e.g., brook trout $Salvelinus\ fontinalis$ and white suckers $Catostomus\ commersoni$),

$$x_{\text{tol}} = -23.49 + 5.35pH + 2.97 \times 10^{-3}Ca - 1.93 \times 10^{-3}Al \tag{13}$$

and fish that were generally sensitive to acid stress (e.g., Flathead Minnows *Pimephales promelas* and blacknose dace *Rhinichthys atratulus*),

$$x_{sens} = -9.90 + 1.56\text{pH} + 4.08 \times 10^{-3}\text{Ca} - 7.04 \times 10^{-3}\text{Al} \tag{14}$$

where Al is inorganic lake Al in μg/L and the Ca is lake Ca in μeq/L. ASI takes values from 0 to 100, with larger values indicating higher levels of acid stress.

The probability of presence for both tolerant and sensitive fish species was calculated as,

$$P = \frac{1}{1 + e^y} \tag{15}$$

where P is the probability of presence and y is a parameter specific to the fish species being modeled (Baker *et al.*, 1990). The y parameter for acid-tolerant fish was calculated as,

$$y_{tol} = -16.94 + 18.34\frac{\text{pCa}}{\text{pH}} \tag{16}$$

whereas the y parameter for acid-sensitive fish was calculated as,

$$y_{sens} = -9.28 + 12.16\frac{\text{pCa}}{\text{pH}} \tag{17}$$

2.2. DATA

Our study included lakes in Maine above 600 m that were at least 0.4 ha in area (hereafter the high elevation lakes of Maine, HELM). The majority of the 90 HELM lakes sit on granitic or quartzo-feldspathic schist and gneiss bedrock and are located in watersheds dominated by red spruce (*Picea rubens*) and balsam fir (*Abies balsamea*). Seventy-eight of the 90 lakes had sufficient time-series data from both the 1980s and 1990s to be included in this project.

Water chemistry data used in the parameterization of MAGIC were collected from the HELM lakes in 1986–1988 (Kahl and Scott, 1988; Kahl *et al.*, 1991) and 1998–2000. These data included concentrations of H^+, ANC, Ca^{2+}, Mg^{2+}, Na^+, K^+, NH_4^+, NO_3^-, Cl^-, SO_4^{2-}, and HCO_3^-. We used time series data for these water chemistry metrics in both the 1980's and 1990's for the calibration of the MAGIC model.

Lake water chemistry as predicted by MAGIC is affected by a combination of deposition to the lake surface and to water that has passed through the soil. Soil chemistry data including cation exchange capacity, base saturation (Ca^{2+}, Mg^{2+}, Na^+, K^+), pH, and total monomeric Al, were collected for 19 of the 78 watersheds in the study (Fernandez unpublished data). The 19 watersheds sampled were carefully

selected to represent a stratified sub-sample of the population of 78 watersheds based on *a priori* knowledge of water chemistry, topography, and vegetation. The high elevation lakes of Maine share the same general bedrock geology, elevation, climate, and vegetation. Thus, the 19 watersheds are likely to be highly representative of the range of values found in the population of 78 watersheds. The soil data from these 19 watersheds were averaged and the means and variation of those data were used in the MAGIC model for each of the 78 lakes in the HELM dataset. The variation in the soils data was explicitly modeled in the multiple MAGIC runs (Wright and Cosby, 2003).

Wet deposition inputs to both TAF and MAGIC were based on observed deposition at National Atmospheric Deposition Program/National Trends Network (NADP/NTN) monitoring stations (http://nadp.sws.uiuc.edu/). The spatial extrapolation model of Grimm and Lynch (1997) was used to derive a spatially interpolated value of wet deposition at each site, correcting for changes in precipitation volume with elevation. Additional deposition from aerosols, clouds, and dry sources was estimated using a mass-balance calculation based on measured and estimated lake inputs and outputs (Sullivan *et al.*, 2004).

2.3. TAF MODEL CALIBRATION

We used MAGIC to make predictions of Ca, sulfur (S), ANC, and soil base saturation concentrations at one-year intervals from 2000 to 2030. We first calibrated MAGIC for the 78 HELM lakes using the measured water, soil, and deposition characteristics. We then ran MAGIC for a total of eight different sulfate deposition scenarios (increasing and decreasing deposition by 20, 40, 60, and 80% above and below measured year 2000 levels). Because MAGIC has a stochastic component to accommodate uncertainty and variability in soil chemistry, we ran 100 MAGIC simulations for each deposition scenario for each of the 78 lakes and used the median results from these runs to parameterize the TAF modules with values for S, Ca, base saturation, and ANC. Preliminary analyses with the MAGIC model indicated that 100 simulation runs provide stable mean and variance estimates for the model outputs.

We then fit the reduced-form model parameters for both ANC and Ca, and the ANC-pH and pH-Al relationships in TAF using predictions made by the MAGIC model. We used least squares to fit the TAF reduced-form models to the MAGIC output using SPLUS (Mathsoft, version 4.5, 1998). As in the original TAF model, we used coefficients of variation from the multiple MAGIC runs to provide TAF with an estimate of variability in MAGIC projections.

As mentioned above, we made slight changes to two of the reduced-form models of Sinha *et al.* (1998). We were able to create better fitting reduced-form models for the HELM lakes by altering the model for ANC and the ANC-pH relationship. In contrast to the original reduced-form model for ANC, we fit separate functions to increasing and decreasing deposition scenarios, respectively. Thus we

produced two sets of lake slope factors, effective nominal deposition values, and lake lag factors for Sinha *et al.*'s (1998) ANC models (Equations 1–5). We also slightly altered the procedure for fitting the pH-ANC relationship in TAF. Sinha *et al.* (1998) fit two separate functions for Equation 6 to relate pH to ANC, one for lakes with ANC concentrations below $50 \, \mu$eq/L, and one for those with concentrations above $50 \, \mu$eq/L. Because we found that two separate functions fit the data no better than a single function, we used the same parameters for all HELM lakes.

In addition to the reduced-form models, TAF uses a watershed deposition adjustment factor (WDF) to adjust regional deposition estimates generated by its atmospheric pathways module to watershed specific deposition levels. We calculated these values by dividing the lake-specific deposition estimates by average annual regional deposition levels.

2.4. ALTERNATIVE EMISSIONS SCENARIOS

TAF uses emission projections in five-year intervals beginning in 1990 and continuing through 2030. These are provided at the state level for the U.S., the provincial level for Canada and national level for Mexico. These simulations reveal, according to the assumptions in the model, the likely effects of changes in average plant retirement age, differences in actual demand growth, and the implementation of Title IV emissions controls. For the illustrative analysis presented here, which is designed to reveal capabilities of the model and be consistent and comparable with past work (Palmer and Burtraw, 1997; Burtraw *et al.*, 1998), we use four of the existing emissions scenarios. The scenarios differ according to their assumptions about power plant lifetimes (60 versus 70 years), growth in electricity demand to 2030 (3%, termed "high growth," and 1%, termed "low growth"), and whether or not Title IV controls are in place.

The Title IV emissions controls were scheduled to be implemented in phases. The first phase of SO_2 reductions, implemented in 1995, required average emission rates of about 2.5 lb of sulfur per million BTU heat input. This rate applied to 431 actual generating units nationwide, including the so-called "substitution and compensation" units that were voluntarily brought into Phase I to ease the cost of compliance on average. The second phase, which took effect in 2000, lowered average emission rates to about 1.2 lb of sulfur per million BTU heat input, affecting over 2,000 units nationwide.

For our analyses, we compared two scenarios in which the Title IV provisions had been implemented with two corresponding scenarios in which no Title IV controls were in place. We selected the scenarios to capture some of the largest and smallest changes in emissions modeled in the original TAF model. The largest changes in emissions were predicted by the "60-low" scenario which modeled Title IV controls under low population growth corresponding to a 1% increase in electricity demand per year and scheduled plant closings after 60 years. The "60-low-baseline" scenario

also assumed low population growth and plant closings after 60 years, but no Title IV controls. In contrast, substantially smaller emissions reductions were predicted by the "70-high" scenario in which population growth was predicted to produce a 3% increase in electricity demand per year and plants are retired after 70 years. The "70-low-baseline" scenario assumed these same conditions but did not include Title IV controls.

3. Results

3.1. MODEL CALIBRATION

In general, the TAF model parameters were consistent with our knowledge of the HELM watersheds (Table I). Soil water ANC response times (the time it took soil water ANC to reach equilibrium) ranged from about 20 to 207 years with a mean of roughly 88 years. This range is narrower and the mean higher than predictions for the Adirondack lakes (2–315 years, mean = 33 years) (Sinha *et al.*, 1998). As in the Adirondack analysis, we found that Ca response times were shorter, ranging from 10–18 years, mean = 12.5. These were comparable to the Adirondack response times (range 4–19 years, mean = 11 years).

There was less variation in the predictions of ANC and calcium from the 100 MAGIC runs for the HELM lakes than for the Adirondack lakes over the 30-year

TABLE I

Reduced-form model parameters summarized for the watersheds of 78 high elevation lakes in Maine, U.S.A.

Model parameter*	Equation #	Units	Min	Max	Mean	SD
N_F DNF$_I$	5	μeq/L	−24.21	269.8	67.22	44.45
N_F DNF$_D$	5	μeq/L	−130.80	2462	71.62	275.50
N_F slope$_I$	5	none	−3.04e-005	3.66e-006	−1.15e-005	7.60e-006
N_F slope$_D$	5	none	−5.86e-004	3.33e-004	−1.16e-004	1.82e-004
ANC lag factor$_I$	2	years	19.9	207.1	81.8	40.2
ANC lag factor$_D$	2	years	5.1	170.5	36.6	32.8
Equilibrium Ca slope	8	none	0.02	0.54	0.23	0.13
Equilibrium Ca intercept	8	μeq/L	10.33	180.40	71.65	36.90
Lake Ca lag factor	7	years	10.1	18.4	12.5	1.9
Δ Base Saturation slope	10	%/years × L/μeq	−0.023	−0.002	−0.010	0.005
Δ Base Saturation intercept	10	%	−1.116	0.460	−0.040	0.379
WDF	none	none	0.31	0.95	0.58	0.14

Note. Subscripts indicate parameters for increasing (*I*) or decreasing emissions scenarios (*D*).

* DNF = Nominal deposition level; N_F = neutralization fraction; ANC = acid neutralization capacity; WDF = watershed deposition factor.

TABLE II

Comparison of errors of fit of the reduced form models across 78 HELM watersheds and 33 Adirondack watersheds

Model parameter	HELM watersheds		Adirondack watersheds	
	CV MAGIC	MSE	CV MAGIC	MSE
Lake ANC (μeq/L)	9.9 ± 11.9	9.9 ± 8.1, 12.3 ± 9.4[a]	40.3 ± 42.7	0.76 ± 0.89
Lake pH	–	0.4377	–	0.16, 0.23[b]
Lake Calcium (μeq/L)	4.0 ± 1.8	1.00 ± 1.20	94.2 ± 96.5	1.30 ± 1.60
Δ Soil Base Saturation (%)	–	0.05 ± 0.04	–	0.01 ± 0.01

Note. Adirondack errors are taken from the TAF model (1998). Values are means plus or minus one standard deviation.

[a] Values are for increasing and decreasing deposition scenarios, respectively.

[b] Values are for lakes with ANC < 50 μeq/L and lakes with ANC > 50 μeq/L, respectively.

prediction period (Table II). In contrast, the fit of the reduced form models to the MAGIC output was better for the Adirondacks with respect to ANC and similar for both sets of lakes for Ca. The reduced-form models for pH and the change in soil base saturation fit the MAGIC output better for the Adirondack lakes (Table II).

3.2. SOIL AND WATER CHEMISTRY PREDICTIONS

As expected, the largest changes in water and soil chemistry occurred under the 60-low emissions scenario. In general, ANC values increased from 2000 to 2030 under all four scenarios (Table III). Under the 60-low scenario, changes in ANC ranged from −0.6 to 39.0 μeq/L. In contrast, the baseline scenario for these conditions predicted changes ranging from −0.8 to 16.3 μeq/L. Changes predicted by the 70-high scenario ranged from −1.0 to 17.0 μeq/L compared to a range of −0.5 to 9.5 μeq/L under the 70-high-baseline scenario.

Calcium concentrations generally decreased in all HELM lakes between 2000 and 2030. These changes ranged from −22.0 to 1.0 μeq/L. There was little difference in the change in Ca concentrations across the four scenarios (ANOVA $F_{3,308} = 1.47$, $P = 0.22$), (Table III). Soil base saturation increased only slightly over the 30-year period. The largest change predicted by the 60-low scenario was 3.5%. Al decreased across most lakes under all scenarios from 2000 to 2030. Predicted changes in Al ranged from −6.0 to 0.0 μeq/L (−161.9 to 0.0 μg/L) under the 60-low scenario compared to −4.5 to 0.0 μeq/L (−121.4 to 0.0 μg/L) under the corresponding baseline scenario. Predicted changes under the 70-high scenario ranged from −3.3 to 0.0 μeq/L (−89.0 to 0.0 μg/L) compared to −1.8 to 0.0 μeq/L (−48.6 to 0.0 μg/L) under the 70-high-baseline scenario.

TABLE III

Predicted mean change in water chemistry, soil chemistry, and biotic stresses from 2000 to 2030 in 78 high elevation lakes in Maine

Parameter	Higher Emissions		Lower Emissions	
	70-high-baseline	70-high	60-low-baseline	60-low
Deposition (μeq/L)	-3.3 ± 0.6	-2.5 ± 0.4	-10.2 ± 1.7	-8.4 ± 1.4
Lake ANC (μeq/L)	4.0 ± 1.6	8.1 ± 2.8	9.7 ± 3.3	16.8 ± 7.9
Lake Ca (μeq/L)	-5.2 ± 4.5	-6.2 ± 4.8	-6.2 ± 4.8	-6.8 ± 5.1
Lake Al (μeq/L)	$-0.3 + 0.4$	$-0.5 + 0.7$	-0.8 ± 1.1	-1.1 ± 1.9
Soil Base-Saturation (%)	0.3 ± 1.4	0.8 ± 1.3	0.6 ± 1.3	1.0 ± 1.3
ASI (tolerant)	-0.2 ± 0.5	-0.2 ± 0.6	-0.5 ± 1.7	-0.3 ± 1.0
ASI (sensitive)	-1.4 ± 9.4	-5.8 ± 19.3	-8.0 ± 23.0	-9.5 ± 23.7
Probability of presence (tolerant)	0.01 ± 0.02	0.02 ± 0.04	0.04 ± 0.06	0.04 ± 0.09
Probability of presence (sensitive)	0.01 ± 0.02	0.03 ± 0.04	0.05 ± 0.06	0.07 ± 0.09

Note. Values are means $\pm$ 1SD. Results are presented for four emissions scenarios, respectively from left to right, baseline emissions at high population growth levels with plant closings in 70 years; Title IV emissions controls at high population growth levels and 70-year plant closings; baseline emissions with low population growth and 60-year plant closings, and Title IV controls with low growth and 60-year closings. The calculations for the acid stress index (ASI) and the probability of fish presences are found in the text.

3.3. FISH ASI AND PROBABILITY OF PRESENCE

On average, lakes showed little change in acid stress or predicted fish presence under any scenario (Table III). The four policies showed no difference in ASI or probability of presence for acid-tolerant fish. Despite these seemingly equivocal results, the probability of presence of sensitive fish species was predicted to increase in a small number of lakes. Under the 60-low scenario, the probability of presence of sensitive fish species was predicted to increase by at least .10 in 22 lakes. On average, the probability of presence increased in these lakes by 0.21. In contrast, the probability or presence was predicted to rise by more than 0.10 in only 14 lakes under the 60-low-baseline scenario. The average probability of presence increased by 0.15 in these lakes.

In general, the largest changes in the probability of presence of sensitive fish were found in the lakes with the lowest ANC values (Figure 1). This relationship was also generally true for the largest predicted changes in Ca, Al, and ANC. The three lakes that did not fit this relationship had extremely low lake Ca values. Not surprisingly, the lake with the largest predicted change in the probability of presence of sensitive fish under the 60-low scenario (0.33) had a large predicted increase in
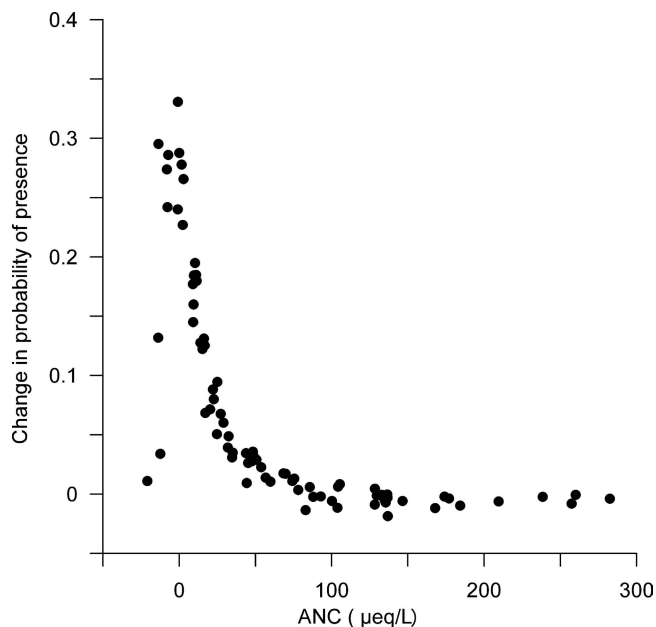
Figure 1. The relationship between the predicted change in the probability of presence of acid-sensitive fish from 2000 to 2030 and measured ANC values in 78 high elevation lakes in Maine, U.S.A. Predictions were made using the Tracking and Analysis Framework model under the assumptions of low population growth, 60-year plant closings, and the implementation of Title IV controls.

ANC (33.7 μeq/L), a relatively large decrease in Al (-5.3 μeq/L [143.0 μg/L]), and only a small decrease in Ca (0.2 μeq/L) (Figure 2).

4. Discussion

In general, applying relatively drastic emissions-reduction scenarios over a thirty-year period would be expected to stimulate acid recovery in lakes, resulting in substantial increases in ANC and decreases in acid stress for fish. The relatively small changes in both lake chemistry and the predicted presence of fish in the 78 HELM lakes, likely reflect the fact that many of these lakes were marginally acidic even in pre-historic times (Kahl *et al.*, in press, ES&T; Stoddard *et al.*, 2003). Thus, 'full' recovery of acidic HELM lakes will likely yield pH values less than 6.5.

Our results are consistent with those produced by the original TAF model applied to the Adirondack lakes (Sinha *et al.*, 1998), and are consistent with the recent review of changes in surface water chemistry resulting from changes in deposition since 1990 (Stoddard *et al.*, 2003). In particular, Stoddard *et al.* (2003) showed that the lowest ANC lakes were recovering the fastest (highest rate of increase in ANC), suggesting that *N* is not a major controlling factor at the presently stable deposition
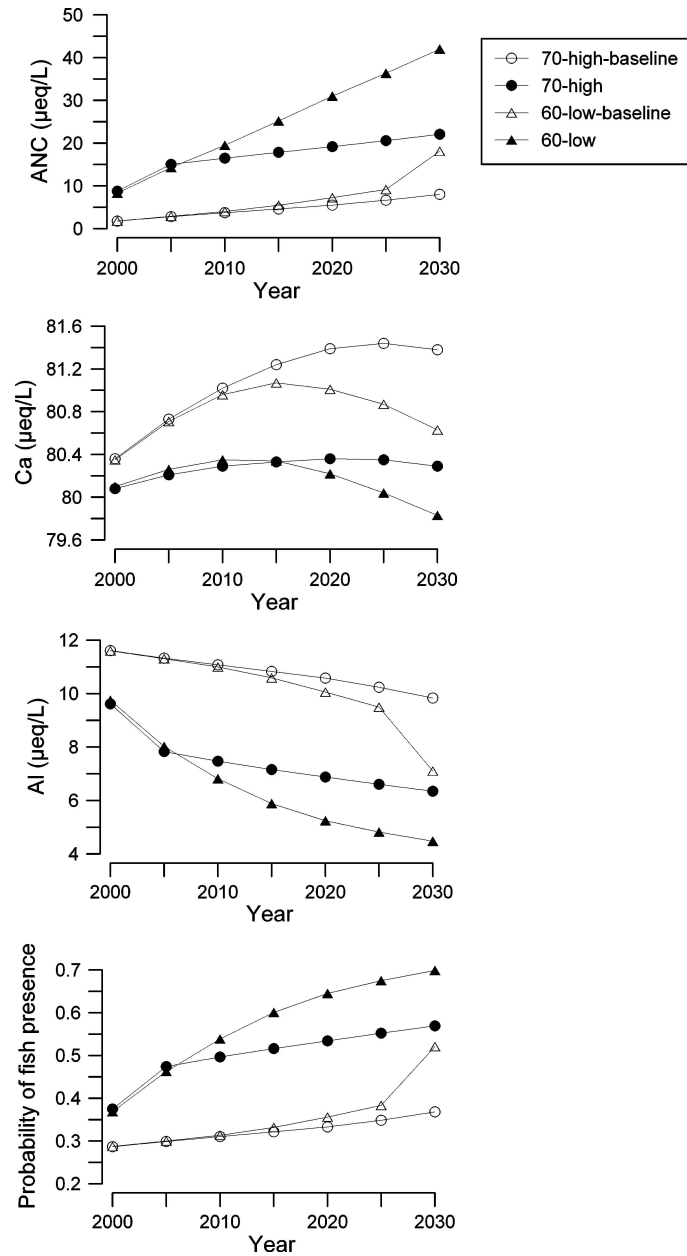
Figure 2. Tracking and Analysis Framework model predicted changes in lake ANC, lake Ca, lake Al, and the probability of presence of fish in one of the most acidic high elevation lakes in Maine, U.S.A. based on four different emissions scenarios. This lake showed some of the most drastic changes in water chemistry as predicted by the TAF model. Model predictions were made under the assumptions of high or low population growth, 60–70 year plant closings, and the implementation of Title IV emissions controls. The probability of presence was calculated for acid-sensitive fish such as flathead minnows *Pimephales promelas* and blacknose dace *Rhinichthys atratulus*.

loadings, and that soils are not regionally depleted in base cations to the extent that recovery is precluded. The relationship between changes in sulfate deposition and changes in concentrations in lakes was also shown by Stoddard *et al.* (2003) to be relatively direct and rapid, suggesting that stored sulfate will not be a major factor for delaying recovery.

There are two aspects of the TAF model predictions that warrant further explanation. First, the relatively small and consistent decreases in Ca across all future scenarios, accompanying the decreasing SO_4 concentrations, are consistent with quantitative and conceptual models of base cation response to decreasing mobile anion concentrations in soils during recovery from acidification (c.f. Reuss *et al.*, 1987; Cosby *et al.*, 1985c). Second, the relatively small increases in soil base saturation are consistent with the fact that acid deposition in the future scenarios, while decreased, is not reduced to zero, and thus the recovery of the soils might be quite slow. These relatively small increases were seen over a thirty-year period of substantial decreases in deposition. Furthermore, the model predicts that soil recovery begins in the mid 1990's. Recent studies, however, suggest that soil base cation depletion may be continuing as recently as the year 2000 based on long-term trends in stream base cations (Fernandez *et al.*, 2003). Thus, it is possible that even these small increases are over-estimates. Conceptual models of recovery from acidification (c.f. Reuss *et al.*, 1987; Cosby *et al.* 1985c) have demonstrated that residual soil acidification can delay and ultimately reverse the initial recovery of surface water ANC.

The purpose of building the reduced-form models summarizing the output of the more complex MAGIC model (Cosby *et al.*, 1985a) was to allow the results to be incorporated into the Tracking and Analysis Framework model (Bloyd *et al.*, 1996). Predicted changes in the probability of fish presence and ASI in the Maine lakes will be used in other updated modules of the TAF model to evaluate the potential effects of different emissions regulation policies on the recreational fishing economy of Maine.

We conclude that a relatively small number of high elevation lakes in Maine are likely to see significant increases in ANC which will produce only modest recovery of fishes. This conclusion is consistent with inferences that most northeastern acidic lakes were at least marginally acidic in pre-industrial times (Cumming *et al.*, 1992), and therefore recovery to 'normal' chemistry will be (and should be) modest.

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