REGIONAL APPLICATION OF AN INTEGRATED BIOGEOCHEMICAL MODEL TO NORTHERN NEW ENGLAND AND MAINE
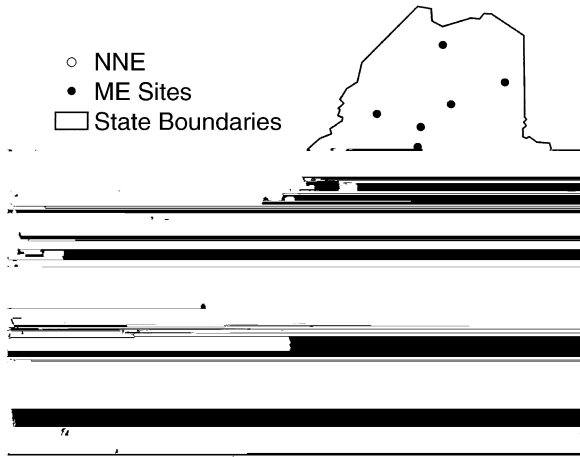
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FIG. 1. Locations of Direct/Delay Response Program (DDRP) sites in Northern New England (NNE) and Maine (ME) subregions.

Tugbawa et al. 2001). The model links a C, N and water model (PnET; Aber et al. 1997) with a soil biogeochemical model (BGC; Gbondo-Tugbawa et al. 2001) to simulate cycling of major elements within northern forests and reactions in soil and surface waters. The 60 lake watersheds located in NNE and ME were statistically selected by the Direct/Delayed Response Program (DDRP) initiated by the U.S. EPA to represent the acid-sensitive watersheds in the northeastern United States (Fig. 1; Church et al. 1989). Chemistry of these lakes was surveyed during National Surface Water Survey in 1984 (Linthurst et al. 1986, Landers et al. 1988). To evaluate the response of these lakes to recent changes in atmospheric deposition, these lakes were resurveyed in the summer of 2001 and results from these two surveys during different time periods were compared (R. A. F. Warby, C. E. Johnson, and C. T. Driscoll, *unpublished manuscript*). Here, we present a synthetic analysis of the results from the model application to these lake watersheds, including S dynamics, N status, changes in base cations, and surface water ANC, as well as other critical indicators in response to past emissions and future control scenarios.

S Dynamics

Base-case simulation

Atmospheric S deposition.—Across the northeastern United States, wet S deposition generally decreases from the west to the east, with dry deposition decreasing from more urb13ap699(N)61n -0 9U.S. 5ilate(-)]T0Linthur0t e0ypse-293(northing)r319(eositionveys)-9(e

coll 2004). Therefore, dry-to-wet deposition ratios may vary over time. At HBEF, relative changes in dry S deposition appear to be different than bulk deposition (Palmer et al. 2004). The observed relationship between air SO_2 concentrations and dry-to-wet deposition ratios as well as relationship between air SO_2 concentrations and SO_2 emissions were used to reconstruct the temporal patterns in dry S deposition. The reconstructed temporal patterns in dry S deposition were again converted into scalars and applied to all the DDRP sites

TABLE 1. Correlations between residuals and watershed characteristics for base-case simulations and simulations with modifications in watershed characteristics and RMSE and Eff values for the simulations.

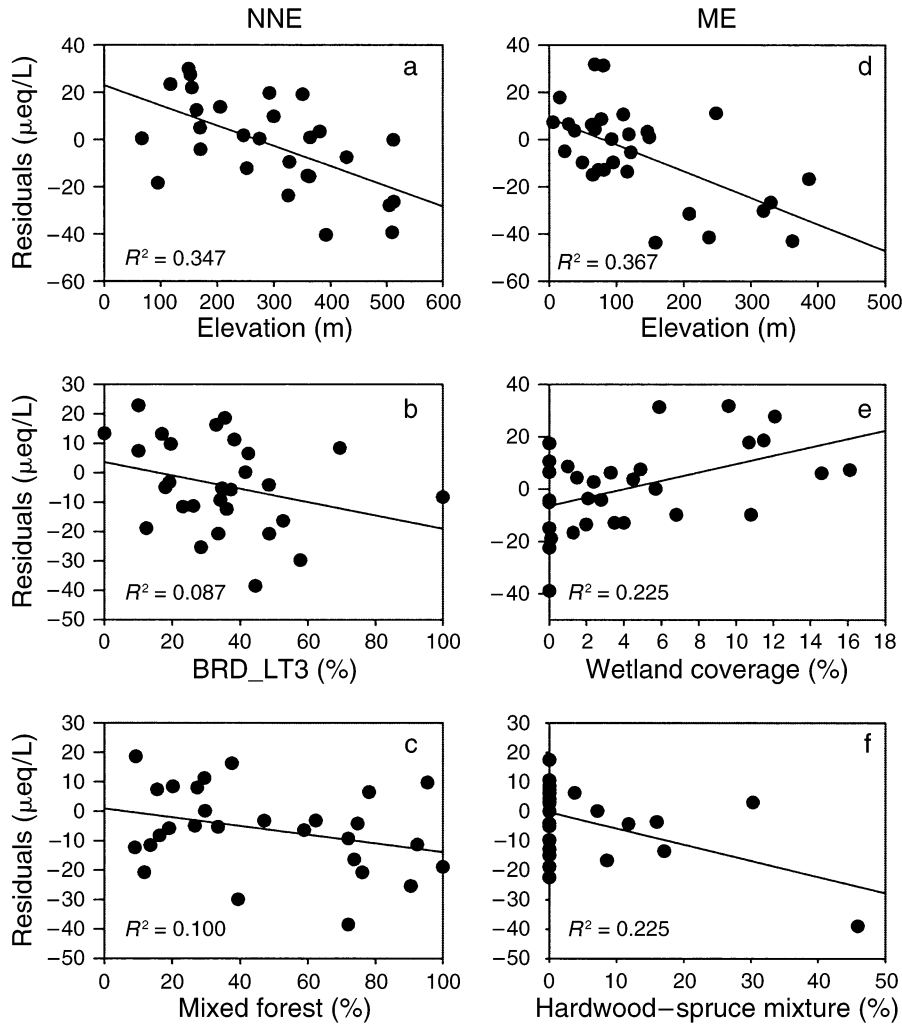|	Subregion							
	NNE simulations				ME simulations			
Parameter	1	2	3	4	1	2	3	4
Latitude	0.15	0.12	0.12	0.15	0.45	0.09	0.16	0.08
Longitude	0.15	0.05	0.09	0.11	0.05	0.18	0.18	0.07
Elevation	**0.59**	0.15	0.25	0.21	**0.61**	0.14	0.22	0.15
Wetland (%)	0.03	0.02	0.08	0.10	0.38	**0.47**	0.31	0.23
Wetland area	0.08	0.32	0.31					

FIG. 2. Relationships between residuals and the watershed characteristics for NNE sites (a–c) and ME sites (d–f). BRD_LT3 is the percentage of watershed with depth to bedrock less than 1 m.

< 20%). Using a shallower soil depth did not improve model simulations at watersheds with shallow surficial deposits, indicating the underprediction at these sites is probably due to other factors. The use of a deeper soil depth for watersheds with thick surficial deposits, however, significantly improved model results. Thus we corrected for the influence of thickness of surficial deposits by changing soil depth to depth to bedrock at sites with thick surficial deposits (BRD_LT3 < 20%).
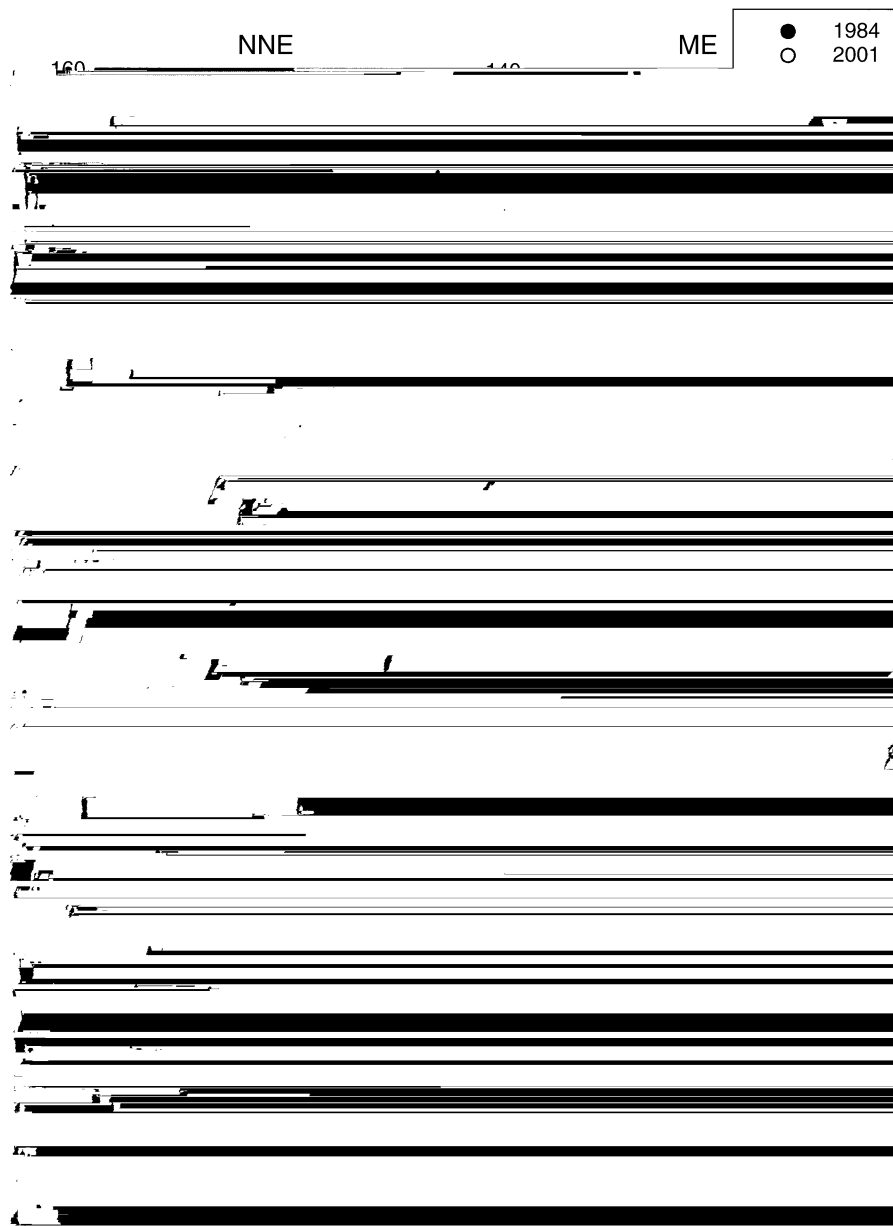
FIG. 3. Predicted concentrations of SO_4^{2-}, Ca^{2+}, Mg^{2+}, and ANC in 1984 (solid circles) and 2001 (open circles) compared to concentrations obtained from surveys for (a–d) NNE lakes and (e–h) ME lakes.

efficient for lakes with large wetland coverage in the watershed ($\geq4\%$) and had overpredictions of lake SO_4^{2-}. The resulting simulations significantly decreased RMSE, but decreased Eff slightly (Table 1).

For sites in Maine, the residuals were found to be positively correlated with the percentage of pine forest coverage in the watershed and negatively correlated with the percentage of hardwood–spruce mixture forests, suggesting an overprediction in lake SO_4^{2-} under pine forests and an underprediction under mixed hardwood and spruce forests. It is not clear why the model overpredicted lake SO_4^{2-} in watersheds with pine for-

ests. However, mixed hardwood and spruce forests could influence watershed S dynamics through enhanced collection of dry S deposition. Thus similar to NNE sites, a dry deposition enhancements factor of 1.75 was applied to sites with mixed hardwood–spruce coverage. This correction further reduced the RMSE of the simulations to 10.1 μeq/L.

By considering these factors, the model was able to explain about 75–78% of the variation in lake SO_4^{2-} concentrations (Fig. 3). The predicted spatial pattern of lake SO_4^{2-} generally agreed with the pattern exhibited in the 1984 survey.
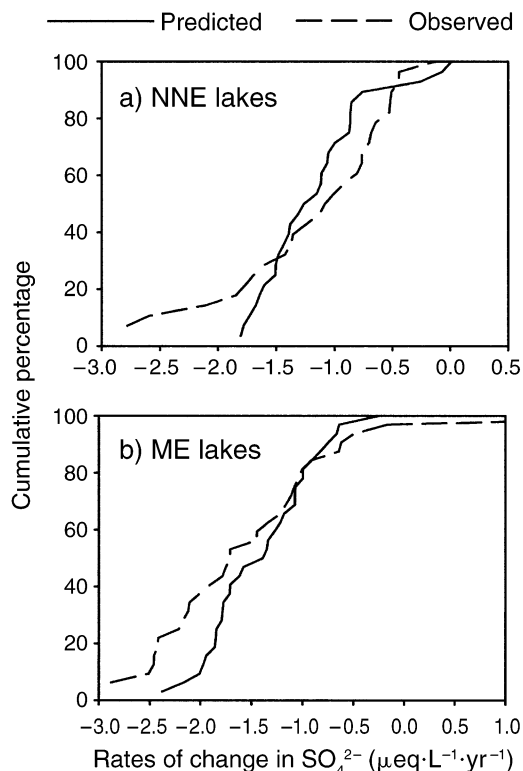
FIG. 4. Distribution of predicted rates of change in SO_4^2 compared to rates derived from surveys for (a) NNE lakes and (b) ME lakes for 1984–2001.

Changes in SO_4^2 concentrations in response to changes in atmospheric deposition

In response to declines in atmospheric S deposition in recent years, the model predicted decreases in lake SO_4^2 concentrations in both subregions. The predicted lake SO_4^2 concentrations in 2001 compared well with concentrations measured in the survey during summer 2001 (regression $R^2 = 0.71$ for NNE sites and 0.65 for ME sites; Fig. 3). The predicted median rates of change in lake SO_4^2 of -1.80 $\mu eq \cdot L^{-1} \cdot yr^{-1}$ for NNE and -1.74 $\mu eq \cdot L^{-1} \cdot yr^{-1}$ for ME during 1984–2001 compared well with rates observed at long-term monitoring (LTM) sites in New England (
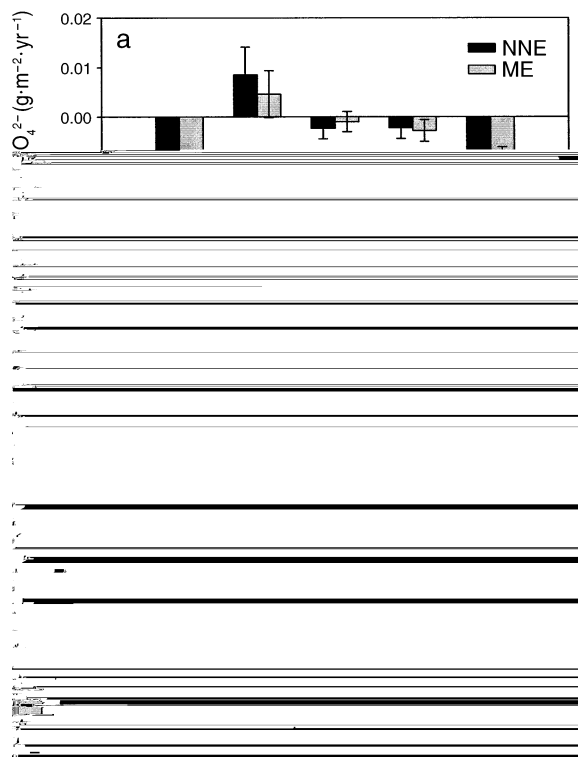

FIG. 5. (a) Estimated rates of change in atmospheric SO_4^2 deposition in relation to simulated changes in desorption, mineralization, uptake, and stream loss of SO_4^2 and (b) the relationship between changes in S deposition and lake watershed loss of SO_4^2 for Northern New England (NNE) and Maine (ME) over the period 1984–2001. Values in (a) are means ± SD.

average desorption rates were 0.028 _ 0.030 g S·m $^{-2}$·yr $^{-1}$ for NNE sites and 0.038 _ 0.020 g S·m $^{-2}$·yr $^{-1}$ for ME sites. Note that predicted rates of net SO_4^{2-} desorption exhibited an increasing trend of 0.009_ 0.006 g S·m $^{-2}$·yr $^{-1}$ for NNE and 0.005_ 0.005 g S·m $^{-2}$·yr $^{-1}$ for ME during this period. The rates of net SO_4^{2-} desorption agreed with the rates previously estimated for the HBEF by Gbondo-Tugbawa et al. (2002). Model predictions indicate smaller effects of mineralization and vegetation uptake on S retention during this relatively short period (Fig. 5). The model predicted slight decreases in net S mineralization. These fluxes were almost completely balanced by decreases in net plant uptake of S (Fig. 5).

N Dynamics

Model inputs related to N dynamics

N deposition.—Similar to S deposition, N deposition also exhibits a distinct gradient across the northeastern United States, decreasing from southwest to northeast (Ollinger et al. 1993). There have also been considerable changes in NO_x emission and N deposition over the last 100 years. However unlike S, NO_x emissions and N deposition have remained relatively constant during the last 20 years. Similar to S, the spatial pattern of atmospheric N deposition was derived from ion concentrations in precipitation and precipitation quantity estimated from regional models of Ollinger et al. (1993, 1995). The temporal pattern of N deposition was also scaled from the reconstructed time series at the representative NADP site in each subregion.

Vegetation type.—Vegetation type has a strong influence on N dynamics in forested ecosystems (e.g., Aber et al. 2003, Lovett et al. 2002). In our model simulations, inputs for the dominant vegetation type were obtained from the original DDRP data (Church et al. 1989).

Land-disturbance history.—Past land disturbances have considerable impact on N cycling within forest ecosystems (Aber et al. 1997, Goodale et al. 2000). Forests of New England and Maine have experienced severe disturbances and generally followed a pattern of decreases in forest area through the late 1700s, a peak in agricultural lands from 1830–1890, and rapid reforestation from farmland abandonment through the late 19th and early 20th centuries (Foster 1995). A major hurricane in 1938 impacted forests in central and northern New England. Site-specific land-disturbance history is difficult to obtain for our study sites. For these watersheds, two major classes of disturbance patterns were assumed: (1) continuous forest (primary forests that have never been cleared), and (2) post agriculture (clearing for agriculture and reforestation on the abandoned field; Goodale et al. 2002). A large percentage of sites in New England and Maine contain open lands (pasture or abandoned farmland) and agricultural lands, suggesting possible agriculture activ-

ities in the past. Thus for sites with open lands or agriculture lands, we assumed the post agriculture land use pattern. While for sites without open land or agriculture land, continuous forest land use pattern was assumed. After Goodale et al. (2002), we assumed forest clearing in 1750 followed by continuous removal of 5% biomass through 1880 for post-agriculture land use and an 80-yr harvest cycling for continuous forest land use.

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TABLE 2. Predicted rates of change in the sum of base cation concentrations in relation to changes in strong acid anion concentrations for acid-sensitive watersheds (ANC < 50 μeq/L) in NNE and ME.
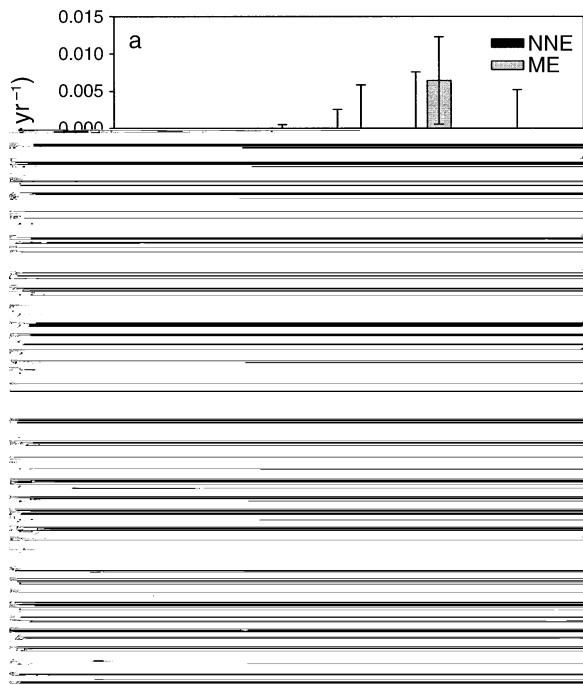
	Site name

FIG. 6. Estimated rates of change in base cation deposition (DEP) in relation to rates of change in mineral weathering (WEA), net cation exchange (CEX), net effect of biotic processes (BIO), and stream loss of base cations (STR) for (a) all sites and (b) acid-sensitive sites (ANC < 50 μeq/L) in northern New England (NNE) and Maine (ME) for the period 1984–2001. Values are mean ⌴ SD.

TABLE 3. Summary of the three future scenarios used in model simulations.

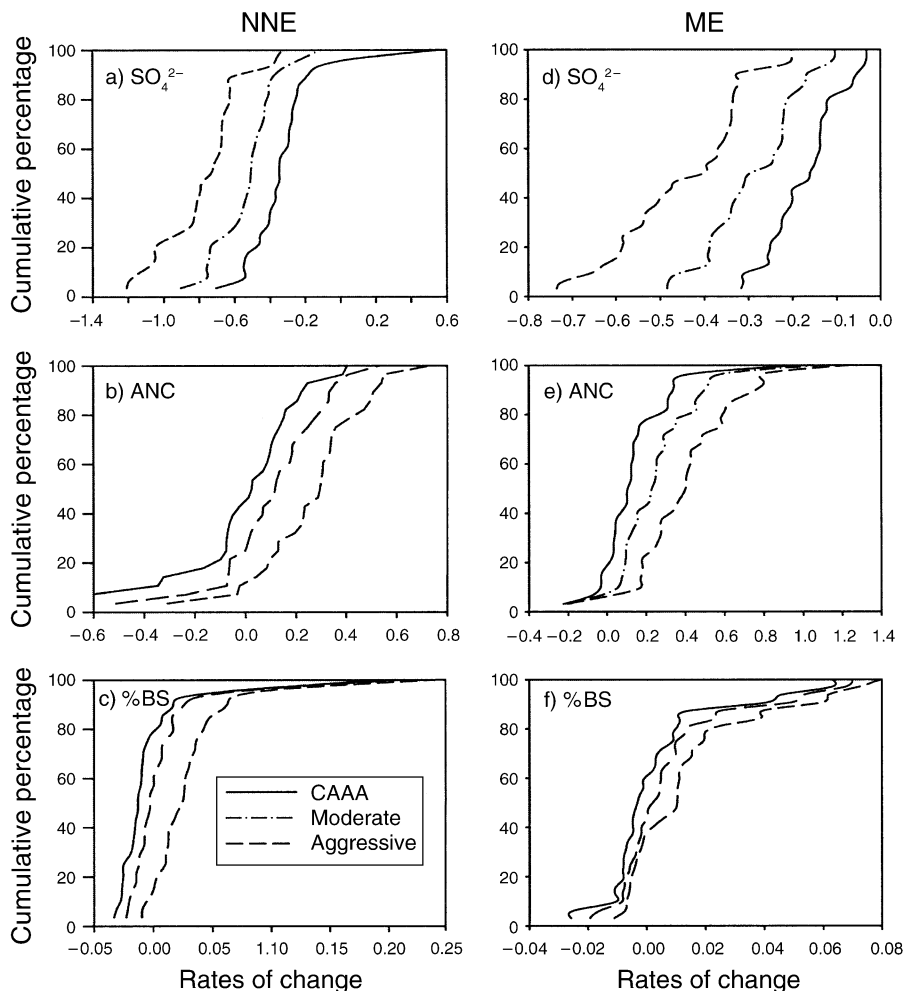| Model	Summary
1990 CAAA (base case)	SO_2 emissions from utility capped at 8.95 million short tons (8.12⌴ 10^6 Mg). Non-utility SO_2

FIG. 7.   Distribution of predicted rates of changes in surface water SO_4^{2-} concentrations ($\mu eq \cdot L^{-1} \cdot yr^{-1}$), ANC ($\mu eq \cdot L^{-1} \cdot yr^{-1}$), and soil %BS (percent base saturation, %/yr) during 2001–2050 under the three scenarios of base case (1990 CAAA), moderate control, and aggressive control at (a–c) northern New England (NNE) sites and (d–f) Maine (ME) sites.

inorganic monomeric Al concentrations   2 $\mu mol/L$, which indicate that aquatic biota are at risk from surface water acidification due to acidic deposition (McAvoy and Bulger 1995).

Predicted soil solution Ca/Al ratios in 2050 are generally higher than the critical values of 1 for sites at both subregions. Soil %BS at 80% of the NNE sites and 60% of the ME sites are generally less than critical value of 20%. Median background %BS is 26.8% at ME and 22.2% at NNE.

Since most of the DDRP sites in these two subregions had ANC values greater than 50 $\mu eq/L$ in 1984, predicted surface water pH and ANC in 2050 are generally above the critical values, and for inorganic monomeric Al concentrations below the critical values. However, about 20% of the sites will have pH less than 6, and ANC of around 25% of the watersheds will remain under 50 $\mu eq/L$ in 2050.

CONCLUSIONS

We summarize the main findings from this study:

1) For both northern New England and Maine, elevation appeared to be an important factor influencing inputs of atmospheric S deposition to watersheds. Including elevation as a component in estimating S deposition inputs significantly improved model predictions in lake SO_4^{2-} concentrations.

2) Changes in lake SO_4^{2-} concentrations were related to changes in atmospheric deposition, although desorption of soil SO_4^{2-} delayed the response of lake water to decreases in atmospheric S deposition.

3) The declines in SO_4^{2-} NO_3 concentrations were coupled with stoichiometric decreases in base cation concentrations, as indicated by the model predictions. As a result, changes in surface water ANC in response to changes in atmospheric deposition were minor.
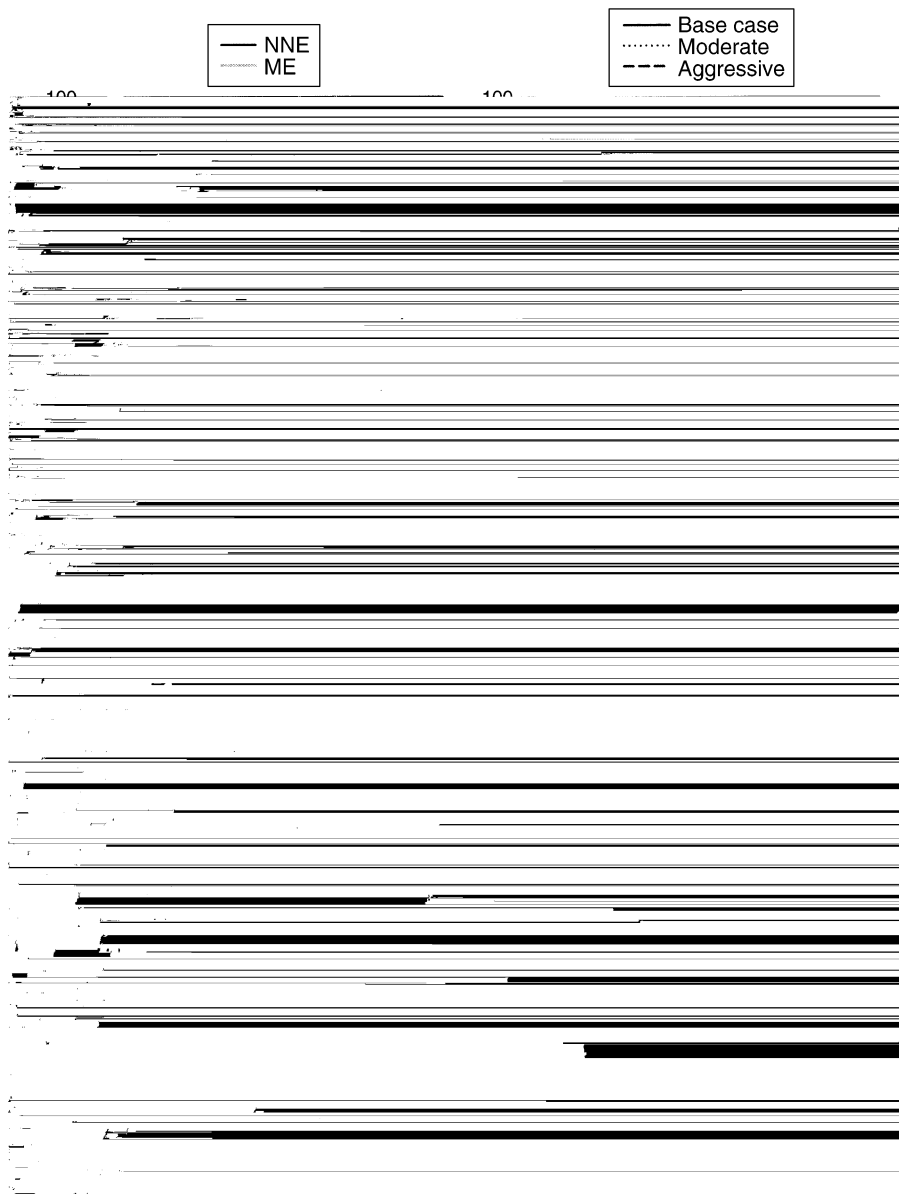
FIG. 8. Predicted surface water (a) SO_4^{2-} concentrations, (b) NO_3^- concentrations, (c) inorganic Al concentration, (d) pH, (e) ANC, and (f) soil %BS (percent base saturation) in 2050 under three scenarios for northern New England (NNE) and Maine (ME) sites.

4) Decreases in base cation deposition contributed an average of 20% and 40% to decreases in drainage loss of base cations for sites in NNE and ME, respectively. The decreases in lake base cation concentrations were largely due to declines in the net release of basic cations from the soil exchange complex.

5) Future reductions in S deposition will contribute to the recovery in soil and surface waters. Although marked improvement is expected, chemical and biological recovery to the background conditions will not be expected at all the sites in 2050.

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