Simulation of soil chemistry and nutrient availability in a forested ecosystem of southern Quebec. Part II. Application of the SAFE model

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Abstract

The dynamic soil model SAFE was calibrated and validated in a small hardwood forest of southern Quebec as a function of its ability to reproduce current soil chemistry and similar pre-industrial soil conditions despite the difference in forest history. SAFE was relatively accurate for reproducing soil chemistry, but comparison of pre-industrial soil conditions between unburned and burned stands casts doubt as to its applicability at sites where specific processes may be involved in nutrient cycling, e.g. the immobilization of N by microbes. Simulated soil chemistry in the unburned zone reinforced the conclusions of a few historical studies which support the hypothesis that acid-sensitive forest sites of northeastern USA underwent significant acidification when major inputs of acidity from the atmosphere occurred, i.e. during the 1930–1980 time span. Model projections in the mineral soil suggest that a new steady-state should be reached in the 21st century assuming no harvest, but that this equilibrium is broken if timber harvesting is done. Model output also suggests that cation nutrient deficiencies could occur in the long-term, but future Al phytotoxic responses are unlikely to occur due to a relatively high projected pH. Finally, it was demonstrated that the time-series files of nutrient cycling should be prepared with care as they can be the source of some abnormalities in model calibration. © 2002 Elsevier Science Ltd. All rights reserved.

Keywords: Dynamic modelling; Soil acidification; Fire disturbance; Harvest operations; N mobility

Software availability

Program title: SAFE
Developers: Per Warfvinge et al.
Contact address: Per Warfvinge, Research Group in Biogeochemistry, Department of chemical engineering II, Lund University, Sweden, email: per.warfvinge@chemeng.lth.se
First available: 1993
Hardware required: Apple Macintosh with math-coprocessor or PowerMac; PC with 486 or Pentium
Software required: Any text editor
Program language: Fortran

1. Introduction

Freshwater acidification caused by the deposition of acidic compounds has been observed in eastern North America and northern Europe (Nilsson, 1985; Reuss et al., 1987; Schindler, 1988; Kirchner et al., 1992; Likens et al., 1996). In contrast, the evidence that links acidic deposition with changes in the acid–base status of the soil is not convincing: some believe that forest soils have acidified due to many decades of acid deposition (Reuss and Johnson, 1986; Johnston et al., 1986; Ulrich, 1989; Likens, 1989; Falkengren-Gerup and Eriksson, 1990; Johnson et al., 1991; Shortle and Bondietti, 1992; John-
son et al., 1994), while others suggest that acidic forest soils may have been the product of natural acidification processes, e.g. uptake of base cations by trees, dissociation of organic acids, and nitrification [Krug and Frink, 1983; Van Miegroet and Cole (1985); Turner and Lambert (1988)].

Both natural and anthropogenic sources of H ions to the soil solution acidify the soil as follows. In the forest floor, the H ions replace base cations adsorbed on the soil exchange complex. In the mineral soil, a decrease in soil solution pH can mobilize Al ions which can subsequently displace base cations from exchange sites (Reuss and Johnson, 1986; Hendershot et al., 1991; Lotse, 1999). As a result, levels of base cations in the soil solution increase initially with increasing H ion activity, and are removed from the soil in leachate. If the base cations lost through leaching are not replaced by mineral weathering or atmospheric inputs, the long-term soil response will be a decrease in base cation concentrations in the soil solution, and then a decrease in base saturation and the loss of soil fertility.

In eastern North America and Europe, soil acidification is suspected as one factor contributing to forest decline. First, a decrease in base saturation means that base cations become less available to plants and thus, can lead to nutrient deficiencies in forest stands (Hendershot and Jones, 1989; Ulrich, 1989; Ouimet and Camiré, 1995; de Vries et al., 1995). A second problem originates from inorganic Al phytotoxicity since pH values below 4.2 may lead to high concentrations of Al ions in the soil solution (Ulrich, 1983). Despite the uncertainty in the available data on inorganic Al toxicity to trees, a (Ca+Mg+K)/Al molar ratio less than 1.0 is usually considered to inhibit plant growth (Sverdrup and Warfvinge, 1993; Cronan and Grigal, 1995).

Sulfate and base cation deposition has decreased over the last 30 years in North America (Hedin et al., 1994; Driscoll et al., 1995) and is likely to continue decreasing if the goals of international agreements are reached. The latest N emission estimates for the 21st century are much less encouraging. Galloway (1995) showed that N emissions increased in most parts of the world since 1950 and they expect further increases within the next 50 years, particularly in developing countries such as China (Galloway et al., 1996). An excess of N in forest soils and increased nitrification could mean higher levels of H ions in the solution and further removal of base cations from the soil environment through leaching losses (Van Miegroet et al., 1992). Northern Europe has witnessed over the last 30 years substantial increases in tree growth due to increased N availability (Binkley and Högberg, 1997). A similar response has yet to be demonstrated in hardwoods of northeastern North America (Aber et al., 1998). However, rates of depletion of the soil base cation pool would increase if the rates of tree growth increase, especially if base cation emission continues to decline. It is therefore increasingly important to assess the effects of acid deposition on soil chemistry in order to set adequate air quality standards that will assure the health and vigour of forest stands.

Identifying the processes governing soil acidity in forest soils is difficult since a forest is a highly complex interactive system in which processes act as combinations. This has led scientists to develop models that allow the examination of acidification and buffering processes as a whole instead of quantifying each process individually. MAGIC (Cosby et al., 1985), SMART (de Vries et al., 1989) and SAFE (Warfvinge et al., 1993) are examples of such dynamic simulation soil models. These models require historic information of atmospheric deposition and nutrient cycling as input data.

An approach to estimate/validate pre-industrial soil conditions is to calibrate the computer simulation models for sites where factors driving soil formation, e.g. climate, parent material, and topography, are similar but forest disturbance history is different. Historical soil conditions calculated by dynamic models should theoretically be similar for sites where factors of soil formation and historical forest disturbance history have been the same. However, a change in stand dynamics and species composition after harvest or fire disturbance should have an influence on nutrient cycling, which would then have an influence on the acid–base status of the soil. We have sufficient information to fulfil model requirements for burned and unburned stands at the Hermine experimental watershed (HEW), a typical maple-dominated forest of southern Quebec. Thus, the data at HEW offer the possibility of estimating/validating historical (or pre-industrial) soil conditions in that fashion. The objectives of this paper were to: (1) test whether SAFE reproduces homogeneous pre-industrial soil conditions for unburned and burned stands, despite the major change in tree species composition and growth rates triggered by fire in 1925, (2) evaluate the performance of the dynamic biogeochemical model SAFE to reproduce current soil chemistry at the stand level, (3) identify the major processes responsible for soil acidification, (4) forecast soil chemistry as a function of timber harvesting operations, and (5) assess the change in model calibration associated with various sets of time-series files of nutrient cycling.

2. Model description

The SAFE model is a dynamic, process-oriented soil model that simulates soil and soil solution chemical composition based on a schematic description of nutrient cycling and site-specific parameters. The schematic description of the SAFE model includes: (1) atmospheric deposition of Ca, Mg, K, Na, NO₃, NH₄, SO₄, and Cl, (2) mineralization of Ca, Mg, K, and N in litter, (3) canopy
exchange of Ca, Mg, K, and N. (4) plant uptake of Ca, Mg, K, and N, and 5) net mineralization of Ca, Mg, K, and N (Fig. 1a). All of these processes are specified in the SAFE model as time-series files. SAFE includes mathematical process-oriented descriptions of cation exchange reactions, chemical weathering of minerals, leaching and accumulation of dissolved chemical components, and solution equilibrium reactions involving CO₂, organic acids and Al-species (Fig. 1b). The soil profile can be divided into a maximum of six layers which correspond to the natural soil horizons. Each horizon is assumed to be chemically and physically homogeneous. The processes included in the SAFE model interact only via the soil solution. A full description of the SAFE model can be found in Warfvinge et al. (1993) but a brief description is given here.

Some chemical reactions were simplified or omitted. For example, base cations (Ca, Mg and K) are lumped together into one divalent cation component BC. The effects of BC on exchange reactions of K are limited because SAFE was formulated with the Gapon equation so that it views the exchanger as a monovalent continuum (Jönsson et al., 1995). Sodium is ignored as an exchangeable species as well as a nutrient since it behaves as a conservative element. At HEW, Na and K account for less than 5 and 10% of cations on the exchange complex of the forest floor and mineral soil, respectively (Belanger et al., submitted) and thus, they do not have a large control on the exchange and movement of cations through the soils. Also, SO₄ adsorption is not modelled. Adsorption of SO₄ on the soil exchange complex can result in the release of OH ions into the soil solution and can increase cation exchange capacity (Rajan, 1978; Cuoto et al., 1979). This chemical reaction can potentially decrease the transport of base cations, H, and Al to surface waters (Fuller et al., 1985). But most hardwoods and coniferous forests of northeastern North America growing on young podzolic soils (i.e. glacialized during the Quaternary) have relatively low rates of SO₄ adsorption [see Fuller et al. (1985) and Mitchell et al. (1992) for New England states and Ontario, and Courchesne et al. (2001) at HEW]. Neglecting this chemical reaction is therefore expected to provide acceptable simulated results.

Changes in soil acidity are expressed as acid neutralizing capacity (ANC) and cation exchange capacity (CEC) and are modelled by calculating the mass balance of the various processes contributing or consuming acidity. The following differential mass balance equation constitutes the framework of the SAFE model:

\[
d[\text{ANC}] / dt = Q / (\theta \cdot z) \cdot (\text{ANC})_{\text{in}} - (\text{ANC}) + R_W
\]

\[-R_X - R_{BC} + R_N\]

where no subscript refers to conditions in the soil layer as well as in the leachate of that soil layer, subscript "in" denotes concentrations in the inflow to the compartment, \(z\) is the physical height of the soil layer (m), \(\theta\) is the volumetric water content (m³ m⁻³), \(Q\) is the flow intensity through the layer (m³ m⁻² y⁻¹), \(R_W\) is silicate weathering, \(R_X\) is cation exchange, \(R_{BC}\) is base cation uptake, and \(R_N\) is nitrogen uptake (all \(R\) expressed by the rate at which they supply or withdraw ANC in mmol m⁻³ y⁻¹). If there is yearly variation in hydrological conditions, changes in soil solution concentrations will occur due to dilution or concentration. Mass balances are also used to determine BC, NO₃ and NH₄ in the soil solution.

The buffering of the liquid phase is controlled by the carbonate equilibrium reactions, the dissolution–precipitation reactions of a solid gibbsite phase which produce various charged Al–OH complexes, and the acid–base reactions of a monovalent organic acid (RH):

\[\text{[ANC]} = [\text{OH}^-] + [\text{R}] + [\text{HCO}_3^-] + 2[\text{CO}_3^{2-}]\]

\[-[\text{H}^+] - 3[\text{Al}^{3+}] - 2[\text{Al(OH)}^{2+}] - [\text{Al(OH)}^{3+}]\]

Ammonium, SO₄ and BC are neglected as they are not believed to be involved in the acid–base reactions in the liquid phase. In equilibrium, species from the right-hand side of the equation can be estimated based on the solution H ion concentration. Soil solution ANC is therefore a function of pH, and vice-versa. Species of the CO₂
The system is calculated with the equilibrium equations given by Henry’s law for $H_2CO_3$, using soil $P_{CO_2}$ as input. The dissociation of RH is not modeled by the rate of mineralization of organic matter but rather specified as input data using the concentration of dissolved organic carbon (DOC) in the soil solutions. It should be noted that the complexation of Al by organic ligands is not included in the SAFE model. Gibbsite (Al(OH)$_3$) is used to produce inorganic species only. The gibbsite solubility constant relates the concentration of free Al ions to pH using the relationship $[Al^{3+}] = Kg [H^+]^3$. Instead of being an input to the model, weathering rates are calculated from soil properties. The surface reactions of minerals follow kinetic rate laws. The total weathering rate ($R_w$) equals the sum of all dissolution rates of fourteen minerals. The dissolution of minerals can be increased by augmenting H ion activity, concentrations of organic acids, and elevated CO$_2$ partial pressure. Mineral weathering can be decreased after the increase in AI and base cations. The soil moisture content determines the exposed mineral surface in contact with water. Weathering is also affected by the soil temperature which is modeled using an Arrhenius equation.

The exchange reactions between H ions and BC are modeled as reversible chemical reactions. Cation exchange is proportional to the difference in BC concentrations between the exchange complex and the soil solution (i.e. soil solution pH does not affect mass transfer). Further, SAFE relies on input data to specify the magnitude of BC and N uptake by trees in the various soil horizons. All N is assumed to be deposited as HNO$_3$, which means that the addition of NH$_4$ in atmospheric deposition contributes acidity through oxidation. Uptake of BC by trees is a source of acidity as roots release H ions in the soil solution in exchange for base cations, whereas NO$_3$ uptake contributes alkalinity from the release of OH ions.

To assure that the simulation reflects changes caused by variations in external and internal loads, the SAFE model must be started prior to industrialization so that the soil is under a steady-state, i.e. all sources of acidity are balanced with all sources of alkalinity. The steady-state conditions are calculated with the PROFILE model which includes the same chemistry as SAFE but solves the algebraic equations that result from setting all differentials to 0. The model is calibrated on base saturation for a specified input year. This presents some advantages at HEW because: (1) all indicates that changes in base saturation have not been very dynamic, and (2) monitoring of base saturation started only recently, i.e. 1994. SAFE adjusts the base saturation to a level at the initial year that results in maximum agreement between recent and modeled base saturation. Each layer is calibrated separately, but the chemistry of the above layer affects the lower layer. The calibration in SAFE is somewhat less complicated that with other soil acidification models which are generally calibrated on two or more parameters (e.g. MAGIC).

Uncertainties in the SAFE model output associated with the imprecision of most soil properties were assessed in Warfvinge and Sandén (1992). Soil bulk density and specific surface area were found to be the most sensitive parameters on simulated ANC, followed by soil moisture content and temperature, and then by CO$_2$ partial pressure and CEC. For example, increasing mineral surface area and CEC by 25% increased solution ANC in the mineral soil by about 50 and 2%, respectively.

3. Site description, field study, and laboratory analysis

The Hermine Experimental Watershed is located at the Station de Biologie des Laurentides de l’Université de Montréal near Saint-Hippolyte, Quebec, in the Lower Laurentians. Briefly, two out of three cohorts of trees studied at HEW are maple-dominated, whereas the other is rich in aspen and birch trees. The aspen and birch cohort is about 75 years old and is thought to have originated from fire around the mid-1920s. The other two cohorts are about 100 years and were likely regenerated after timber harvesting at the turn of the 20th century. See Bélanger et al., (2002) for more information on the study zones and forest history.

In 1993, three zones reflecting the cohorts of trees were selected. Three 300 m$^2$ circular plots were delimited within the three study zones. Samples from the surface organic horizons were collected in four permanent sub-plots of 5 m$^2$ selected randomly in each plot. Soil samples from the sub-plots were pooled together. The discontinuous eluvial A horizon was not sampled. In 1993, samples were collected in September, October, and November. Soil samples were collected once a month from May to November of 1994 and 1995. In order to decrease the impact on the plots, however, samples were collected three times in 1996 and 1997 (i.e. spring, summer, and fall). Zero-tension lysimeters (Hendershot and Courchesne, 1991a) were installed in each plot in September of 1993 at depths of 0 cm (below the LFH layer) and 50 cm (in the lower podzolic horizon). From 1994 to 1997, soil solutions were collected every two weeks from May to November, and every month from December to April. The surface area covered by the sampling design is approximately 10% of the total surface of the southern hillslope.

Soil pH in water and exchangeable cations were determined on all soil samples collected between 1993 and 1997. Soil pH in water was measured using a soil to water ratio of 1:2, whereas exchangeable cations were determined using an unbuffered BaCl$_2$ extraction (Hendershot et al., 1993). Levels of cations in the BaCl$_2$ extracts were determined by atomic absorption spectro-
Reconstruction of nutrient cycling and deposition was done assuming a cleared condition back in 1700. Conventional harvesting was modelled in 1900 in both zones (i.e. stemwood or lopping operations at the stem’s base), whereas fire disturbance was modelled in 1925 in zone no.3 only assuming destruction of all above-ground standing forest. Simulations with SAFE were initiated in 1700 but simulated results are presented starting in 1800 when calculated soil chemistry reached a pre-industrial steady-state. Soil chemistry was forecasted until the year 2100; 1. no forest disturbance beyond the year 2100; 2. conventional harvesting in 2020; and 3. partial cutting in 2020 where one half of the forest biomass was exported. Thereafter, the forest was assumed to regenerate in sugar maple each time. The soil chemical, physical, and hydrological properties used to run the SAFE model are presented in Table 1 for zones no.2 and no.3. Units in Table 1 are those needed for running the SAFE model.

4.1. Soil properties

In order to fit the soil and soil solution chemistry data at HEW (1993–1997) with the model output, the soil profile was divided into two layers, i.e. the forest floor (10 cm) and the podzolic B horizons (60 cm). Cation exchange capacity, DOC, soil density and soil texture in the mineral soil were not calculated as weighted means according to the thickness of the podzolic B horizons because each had approximately the same thickness. Specific surface area was calculated from the mean soil density and texture with a simple algorithm as described by Jönsson et al. (1995):

\[ A_w = ((100 - \% CF)/100) * (0.003 * \% sand \\
+ 0.022 * \% silt + 0.08 * \% clay) * SD * 1000 \]  

(5)

where \( A_w \) is the exposed mineral surface area, SD is soil density, CF represents coarse fragments, and the sum of clay, silt, sand, and coarse fragments equals 1. Coarse fragments were assessed in the field on all the soil profiles using the Munsell chart diagrams and were unaccounted in the measurement of soil density because their contribution to the total surface area was included in Eq. (5). Soil density specified in SAFE may be slightly underestimated (relative to the bulk mineral soil). However, this is not expected to produce a bias in the model output because soil density and surface area are used to generate mineral weathering but not other processes. Mineralogy of the mineral soil of both zones was assumed to be homogeneous (Table 2). Mineral weathering in the forest floor was unaccounted for by setting density and specific surface area at zero.

Soil CO\(_2\) partial pressure in acidic forest soils in Shenandoah National Park, Virginia (Castelle and Galloway, 1990), was used for input. In SAFE, CO\(_2\) is involved in

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**4. Model parameterization**

In respect to the times-series files of nutrient cycling and deposition created with MMAKEDEP at HEW (Belanger et al., 2002), simulations with SAFE were conducted for zone no.2 (unburned) and zone no.3 (burned) with scenario no.3 and scenario no.6, respectively. A simulation with the time-series files produced in the unburned zone with the original schematic description of the MAKEDEP model (scenario no.4) was also performed to show some of the potential problems associated with the assumption that N constraints on tree growth were high prior to the onset of industrial air pollution (see Belanger et al., 2002 for more details).
Table 1
Soil chemical, physical and hydrological properties used for the SAFE simulation in zone no.2 (unburned) and zone no.3 (burned) at the Hermine experimental watershed

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Zone no.2 (Unburned)</th>
<th>Zone no.3 (Burned)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth/thickness</td>
<td>cm</td>
<td>10-0</td>
<td>0-60</td>
</tr>
<tr>
<td>Forest floor</td>
<td></td>
<td>10-0</td>
<td>0-60</td>
</tr>
<tr>
<td>Mineral soil</td>
<td></td>
<td>10-0</td>
<td>0-60</td>
</tr>
<tr>
<td>Base saturation</td>
<td>%</td>
<td>76</td>
<td>80</td>
</tr>
<tr>
<td>Cation exchange capacity</td>
<td>kmol kg⁻¹</td>
<td>2.69E-04</td>
<td>3.00E-05</td>
</tr>
<tr>
<td>Moisture content</td>
<td>m⁻³ m⁻³</td>
<td>0.50</td>
<td>0.30</td>
</tr>
<tr>
<td>Soil bulk density</td>
<td>kg m⁻³</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Surface area</td>
<td>m² m⁻³</td>
<td>0</td>
<td>1.70E+06</td>
</tr>
<tr>
<td>CO₂ pressure</td>
<td>x ambient</td>
<td>5</td>
<td>5</td>
</tr>
<tr>
<td>Inflow</td>
<td>% of 100</td>
<td>100</td>
<td>80</td>
</tr>
<tr>
<td>Outflow</td>
<td>precipitation</td>
<td>80</td>
<td>50</td>
</tr>
<tr>
<td>Dissolved organic carbon</td>
<td>mg L⁻¹</td>
<td>26.3</td>
<td>2.2</td>
</tr>
<tr>
<td>Gibbsite equilibrium constant</td>
<td>kmol² m⁻⁶</td>
<td>5.5</td>
<td>8.5</td>
</tr>
<tr>
<td>Ca+Mg+K uptake</td>
<td>% of max.</td>
<td>30</td>
<td>70</td>
</tr>
<tr>
<td>N uptake</td>
<td>uptake</td>
<td>60</td>
<td>40</td>
</tr>
</tbody>
</table>

Table 2
Soil mineralogical properties used for the SAFE simulation in zone no.2 (unburned) and zone no.3 (burned) at the Hermine experimental watershed

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Zone no.2 (Unburned) &amp; Zone no.3 (Burned)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quartz</td>
<td>45</td>
<td></td>
</tr>
<tr>
<td>K-feldspars</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Plagioclase</td>
<td>15</td>
<td></td>
</tr>
<tr>
<td>Albite</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Hornblende</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Pyroxene</td>
<td>% of</td>
<td></td>
</tr>
<tr>
<td>Epidote</td>
<td>total</td>
<td></td>
</tr>
<tr>
<td>Garnet</td>
<td>mineral</td>
<td></td>
</tr>
<tr>
<td>Biotite</td>
<td>matrix</td>
<td>1.5</td>
</tr>
<tr>
<td>Muscovite</td>
<td>10</td>
<td></td>
</tr>
<tr>
<td>Vermiculite</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Chlorite</td>
<td>19</td>
<td></td>
</tr>
<tr>
<td>Apatite</td>
<td>0.1</td>
<td></td>
</tr>
<tr>
<td>Gibbsite</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Kaolinite</td>
<td>0</td>
<td></td>
</tr>
</tbody>
</table>

weathering reactions and is a component of the buffering system of the soil solution. The influence of CO₂ partial pressure on simulated ANC is however small when the soil pH is below 5.0 because carbonic acid fails to dissociate and Al buffering dominates (Warfvinge and Sandén, 1992). Values of pH found in the first meter of the soil profiles at HEW are therefore too acidic for carbonic acid to dissociate and contribute significantly to mineral weathering and buffering the solution.

Bloom (1983) doubts gibbsite controls Al concentrations in the soil solution since it has very slow kinetics. Also, the wide range of gibbsite equilibrium constant found in field measurements (see Alveteg et al., 1995) casts doubt on the possibility of using gibbsite to predict Al³⁺ activity in the soil solution. Gibbsite has not been found in the Hermine watershed or to our knowledge elsewhere in southern Quebec. Data near the study site indicate however the presence of hydroxy-interlayered vermiculite which may control Al solubility at values close to those of gibbsite (Courchesne and Hendershot, 1989; Hendershot and Courchesne, 1991b). Therefore, the specified inputs reflect the median value of the apparent solubility constant measured in the forest floor and the podzolic B horizons near HEW (Hendershot and Courchesne, 1991b).

4.2. Nutrient uptake

Au (1999) measured at HEW that N induced respiration rates (NIR) in the forest floor were about 40 times greater than NIR rates in the mineral soil. Using an average depth and density for each soil layers, we determined from the NIR data that 65% of the total N mineralized in the soil profile occurs in the forest floor, whereas 35% occurs in the first 25 cm of the mineral soil. Considering the relatively tight N cycle, 70% of its total uptake was allocated in the forest floor and the remaining 30% in the mineral soil. Conversely, sources of base cations are not restricted to the forest floor and are supplied through weathering in the mineral soil. Lajeunesse (1990) observed that about 70% of fine root mass at HEW was located in the first 25 cm of the soil (forest floor and upper podzolic horizon). Therefore, uptake of BC was allocated throughout the soil profile as a function of root mass and layer depth, resulting in 30% of uptake in the forest floor and 70% of uptake in the first 60 cm of the mineral soil.

4.3. Precipitation and hydrology

Precipitation data at the Station de Biologie de l’Université de Montréal is available starting in 1964. A linear
regression was performed between Montreal data and the biology station data to see whether the Montreal data could be used prior to 1964. The McGill University station was selected for the Montreal data since it was one of the few in existence by the turn of the 20th century. The correlation between the biology station and McGill University data between 1964 and 1997 was marginally significant ($R^2=0.30, P<0.12$). Out of necessity, precipitation records of the McGill University station were taken for the period 1900–1964. Those at the biology station were taken thereafter until present day and was also used to provide the input prior to 1900 and after 1997. Precipitation volumes were corrected for throughfall using the least-square regression model in Bélanger et al. (2002), which predicts that about 10% of precipitation water is intercepted by the canopy.

Biron et al. (1999) calculated input–output budgets of precipitation at HEW between 1994 and 1997 and found that between 50 and 60% of the precipitation water did not exit in stream runoff. All major water fluxes within the soil profile were simulated with the FORHYM model (Arp and Yin, 1992), a model designed for estimating hydrological processes in upland forest soils from monthly climate records and descriptive site information. FORHYM was calibrated in accordance with the precipitation budget calculated at HEW. The model simulated that 13, 15 and 30% of the water was intercepted or evaporated by/in the forest canopy, forest floor, and mineral soil, respectively. FORHYM also predicted about 3% of surface flow. The output was slightly adjusted so that the input to the SAFE model reflected that: (1) 10% of precipitation was intercepted by the canopy (as predicted with the linear regression model in Bélanger et al. (2002), and (2) surface runoff was eventually intercepted after infiltrating in the forest floor. In this respect, 20 and 30% of throughfall was assumed to be intercepted by the forest floor and mineral soil, respectively.

5. Results

5.1. Simulated vs measured soil and soil solution chemistry

Comparison of model output at HEW with means measured in the field between 1994 and 1997 is presented in Table 3. Measured mean inorganic SO$_4$ concentrations in the forest floor and the mineral soil were underestimated by less than 25%. Simulated base cation concentrations in the soil solutions were overestimated in the forest floor and mineral soil by 19 and 43% in the unburned zone and 13 and 22% in the burned zone, respectively. Simulated inorganic N concentrations in the solution of the unburned zone was optimal in the forest floor and overestimates of 85% in the mineral soil. Simulated results in the burned zone exceeded measured inorganic N concentrations by 82 and 73% in the forest floor and mineral soil, respectively. The logarithmic transformation of pH values showed that H ion concentrations in the burned zone were overestimated by 15 and 37% in the forest floor and mineral soil, respectively. Measured values in the unburned zone were overestimated by 34% in the forest floor and by 65% in the mineral soil. The SAFE model is calibrated on base saturation so that the agreement between the simulated and measured values in the forest floor and mineral soil in both zones was optimal for both zones.

5.2. Changes in ion concentrations in soil solution over time

Concentrations of base cations in the forest floor were relatively high and stable between 1800 to 1900, declined after the harvest in 1900 in both zones, increased thereafter to reach maximum concentrations around 1970, and then declined again until present day to about 80% of maximum concentrations (Fig. 2). Patterns of base cations in the solutions of the mineral soil in both zones were similar: simulated concentrations were low but stable from 1800 to 1900, increased sharply in 1900 after the harvest until 1970, and decreased by about 50% of maximum concentrations until present day (Fig. 3). Sharp increases in base cation concentrations in the solution of both soil depths were simulated after the harvest in 1900 and after the fire in 1925 in zone no.3.

Simulated inorganic N concentrations in the solution of the forest floor was fairly stable during most of the simulation period, except that sharp increases occurred for about 10 years after the harvest and 5 years after fire disturbance in zone no.3 (Fig. 2). In the mineral soil, leaching of inorganic N occurred mainly after the harvest (1900), in the early stages of stand development of the second stand (until about 1920), after fire disturbance in the burned zone (1925), and at present time (Fig. 3). Patterns of SO$_4$ were similar in both zones and at both soil depths. Concentrations increased at rates similar to SO$_4$ deposition until the late-1960s (see Bélanger et al. (2002) for reconstructed history of SO$_4$ deposition) and decreased thereafter (Figs. 2 and 3). A temporary decrease in SO$_4$ concentrations in both the solutions of the forest floor and mineral soil occurred in the burned zone after timber harvesting, but also after fire disturbance.

5.3. Pre-industrial soil pH, base saturation, soil solution acid neutralizing capacity (ANC) and changes over time

The simulated forest floor pre-industrial pH in the unburned zone was 5.81, whereas simulated forest floor pre-industrial pH in the burned zone was 5.22 (Fig. 4).
Table 3
Mean soil and soil solution chemistry in the forest floor and mineral soil of zone no.2 (unburned) and zone no.3 (burned) at the Hermine experimental watershed and simulated chemistry between the same years using SAFE. Means were calculated from data collected between 1994 and 1997 from the three plots constituting a zone. Base cations, SO$_4$, and N concentrations are in µmol l$^{-1}$. Base saturation is in %

<table>
<thead>
<tr>
<th>Soil parameter</th>
<th>Soil layer</th>
<th>Measured Zone no.2 (Unburned)</th>
<th>Simulated</th>
<th>Measured Zone no.3 (Burned)</th>
<th>Simulated</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>Forest floor</td>
<td>4.28</td>
<td>4.46</td>
<td>4.21</td>
<td>4.28</td>
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<td></td>
<td>Mineral soil</td>
<td>4.98</td>
<td>5.44</td>
<td>5.02</td>
<td>5.30</td>
</tr>
<tr>
<td>Base saturation</td>
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<td>76</td>
<td>76</td>
<td>80</td>
<td>80</td>
</tr>
<tr>
<td></td>
<td>Mineral soil</td>
<td>37</td>
<td>37</td>
<td>41</td>
<td>41</td>
</tr>
<tr>
<td>Ca, Mg, K</td>
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<td>213</td>
<td>262</td>
<td>214</td>
<td>246</td>
</tr>
<tr>
<td></td>
<td>Mineral soil</td>
<td>110</td>
<td>194</td>
<td>100</td>
<td>138</td>
</tr>
<tr>
<td>NO$_3$ and NH$_4$</td>
<td>Forest floor</td>
<td>100</td>
<td>100</td>
<td>14</td>
<td>78</td>
</tr>
<tr>
<td></td>
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<td>2.5</td>
<td>9.4</td>
</tr>
<tr>
<td>SO$_4$</td>
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<td>89</td>
<td>69</td>
</tr>
<tr>
<td></td>
<td>Mineral soil</td>
<td>116</td>
<td>109</td>
<td>115</td>
<td>106</td>
</tr>
</tbody>
</table>

Fig. 2. Simulated concentrations of base cations (Ca, Mg and K), N (NO$_3$ and NH$_4$) and SO$_4$ in the soil solution of the forest floor of zone no.2 (unburned) and zone no.3 (burned) at the Hermine experimental watershed with scenarios no.3 and no.6, respectively.
Simulated pre-industrial pH of the mineral soil was 6.00 in the unburned zone and 5.90 in the burned zone. Forest floor pH in both zones declined slowly from 1800 to 1900, but decreased drastically within 2–3 years after the harvest in 1900 (Fig. 4). Until present day, a net increase in forest floor pH in both zones was simulated since harvest and a small and temporary increase was seen after fire disturbance in zone no.3. Simulated mineral soil pH was unstable throughout the simulation period: a gradual decrease of about 0.60 unit of pH occurred within the 200 years in both zones (Fig. 5). Both the simulated harvest and fire contributed to a decrease in mineral soil pH for a period of about 10 years.

Pre-industrial base saturation simulated in the forest floor was 96 and 92% in the unburned zone and the burned zone, respectively. In the mineral soil, initial base saturation was 64% in the unburned zone and 68% in the burned zone. Simulated base saturation in the mineral soil was stable until 1900, but decreased in both zones by more than half after the harvest (Fig. 4). Simulated base saturation in the forest floor then increased gradually until present day to reach 76% in the unburned zone and 80% in the burned zone. A small and temporary increase in base saturation in the forest floor occurred in zone no.3 after fire disturbance. In contrast, simulated base saturation in the mineral soil decreased from 1800 to present day by 42% in the unburned zone and 39% in the burned zone (Fig. 5). Small increases in simulated base saturation occurred after harvest in both zones, but the effect of simulated fire disturbance on base saturation was hardly noticeable.

Simulated pre-industrial soil solution ANC in the forest floor was 193 µmol L⁻¹ in both zones, whereas soil solution ANC in the mineral soil was 185 µmol L⁻¹ in the unburned zone and 142 µmol L⁻¹ in the burned zone (Figs. 4 and 5). Soil solution ANC followed the same
trends as solution pH. However, the decrease in solution ANC in the mineral soil of the burned zone was more pronounced than solution pH (Fig. 5).

6. Discussion

6.1. Harvest and fire simulation

The forest fire in zone no.3 was simulated as if all base cations contained in the trees were released to the soil solutions. Also, thirty percent of N in tree components was assumed to be lost in the fire through volatilization [as estimated by Knight (1966)]. The harvest was simulated as if all base cations and N contained in leaves and branches were released to the soil. The increase in base cation concentrations after harvest or fire disturbance is thought to create a new equilibrium between the solution and the liquid phase where more acidity is displaced from the soil complex in exchange for base cations and thus, the increase in soil ANC generally favors increases in soil pH and base saturation (Grier, 1975; Kutiel and Shaviv, 1992; Olson et al., 1996). Similarly, the simulated harvest contributed to a significant increase in pH and base saturation in the forest floor, whereas the net positive effect of fire on simulated forest floor fertility in zone no.3 were small. However, both the simulated harvest and fire acidified the mineral soil. The decline in solution ANC in the mineral soil was likely due to the increased uptake of base cations by the regenerating forests and perhaps to the release of NO₃ in the soil solution from the decomposing slash. The high base cation demand of aspen stands is well documented (Alban, 1982; Paré and Bergeron, 1995).
of NO$_3$ in stream runoff after whole-tree clear-cutting in New Hampshire and in the boreal forest after fire have also been reported (Dahlgren and Driscoll, 1994; Bayley et al., 1992). Part of the decrease in solution ANC in the mineral soil after harvest or fire may also have been caused by the simulated decrease of base cation input from dry deposition because of the lack of the canopy to filter dry particulates from the air (Belanger et al., 2002).

6.2. Forest regrowth

The trends in simulated post-fire pH and base saturation in the forest floor suggest that faster growing trees in the burned zone (aspen and birch) did not acidify the soil at a greater rate than maple trees in the unburned zone. Rather, greater uptake of base cations in the forest floor of the burned zone may have been overshadowed by a higher base cation flux, such as the high quality litter of aspen. Simulated base saturation in the forest floor of the burned zone increased by an absolute 49% from a low shortly after the fire until 1997, whereas simulated base saturation in the unburned zone increased by an absolute 31% from a low 20 years after harvest until 1997. Similarly, Belanger et al., (submitted) observed that the aspen- and birch-dominated plots at HEW promoted the enrichment of the forest floor in base cations, particularly K. Simulated post-fire pH and base saturation in the mineral soil were lower in the burned zone than in the unburned zone throughout the simulation period. However, rates of decrease of soil pH and base saturation from the year following the harvest until 1997 were not higher in the aspen- and birch-dominated zone (burned) than in the sugar maple-dominated zone (unburned). Base saturation declined from 63 to 41% in
the burned zone and from 60 to 37% in the unburned zone, whereas pH decreased from 5.55 to 5.30 in the burned zone and from 5.74 to 5.44 in the unburned zone. Similarly to the forest floor, the simulated trends in pH and base saturation in the mineral soil therefore suggest that aspen and birch regrowth in the burned zone has not contributed to faster acidification than the adjacent mature maple zone [as proposed by the model of Alban (1982) in forests of Minnesota, where base cation sequestration in the tree components in exchange for H ions was found to be greater under aspen than spruce and pine]. This is in agreement with Bélanger et al. (submitted) who found no evidence of change in the chemistry of the mineral soil at HEW after 75 years of influence of aspen and birch in the burned plots.

6.3. Pre-industrial soil conditions

Simulated rates of mineral weathering in the burned zone were lower than those in the unburned zone (Fig. 6). Differences in weathering rates between the zones were due to lower specified bulk density, mineral surface area and soil moisture content as mineralogy was assumed to be homogeneous between the zones (see Table 1). It is therefore not surprising that pH, ANC and base saturation calculated with SAFE in the mineral soil of the burned zone were lower than those simulated in the unburned zone under steady-state. For comparison, the simulated release of base cations from weathering in the unburned zone is about 12% less than what was estimated by Courchesne (unpublished) at HEW with the mass balance approach. Estimated release of base cations by weathering using the same approach at the Hubbard Brook Experimental Forest (HBEF) is about 22% more than the simulated release of base cations with SAFE at HEW [from Likens and Bormann (1995)]. The comparison of weathering rates between HBEF and HEW is appropriate because of the similarities in climate, species composition, and parent material.

Pre-industrial base saturation simulated in the forest floor was higher in the unburned zone than in the burned zone although simulated and measured base saturation in the unburned zone for the years 1994 to 1997 was lower than base saturation in the burned zone. Moreover, pre-industrial solution pH simulated in the forest floor were, after logarithmic transformation, 34% higher in the unburned zone than in the burned zone, despite that the measured difference between 1994 and 1997 was 15%. Simulated pre-industrial solution ANC in the forest floor also seemed to follow the trend in pH. Three explanations for the divergence in pre-industrial solution pH and ANC are possible, the first being that N concentrations in the solution of the forest floor vary considerably over time and that annual means used for comparison with the model output are not weighted according to the absolute amounts in each sample.

A second explanation is the poorly simulated N concentrations in the forest floor of the burned zone. Nitrification produces two moles of H ions for one mole of NH₄ nitrited, whereas for every mole of NO₃ and NH₄ absorbed by vegetation, one mole of OH ions and H ions is released by the roots into the soil solution, respectively. In this respect, assuming total nitrification in the soil does not produce more acidity if all NO₃ was absorbed by vegetation in the forest floor. This is not the case at HEW because NH₄ leaching occurs at rates similar to those of NO₃. Warfvinge et al. (1998) found that the significant overestimation of nitrification and NO₃ concentrations in the forest floor of a spruce stand of southern Bavaria, Germany, created lower soil solution ANC and pH. This also seemed true in the forest floor of the burned zone at HEW.

The cause for the lower measured inorganic N concentrations in the soil solutions of the burned zone has yet to be clarified but one hypothesis may be brought forward. Tietema (1998) suggested that under N-limiting conditions, fungal biomass constitutes the largest fraction of the total microbial community, whereas at N saturation, the microbial biomass is dominated by bacteria. Similarly, Aber et al. (1998) hypothesized that in con-

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Fig. 6. Simulated release of Ca, Mg and K (BC) by weathering in the mineral soil of zone no.2 (unburned) and zone no.3 (burned) at the Hermine experimental watershed with scenarios no.3 and no.6, respectively.
ditions of N limitation, mycorrhizae can immobilize large amounts of mineral N that lead to inhibition of soil nitrification. In contrast, temperate forest soils that have accumulated N due to decades of atmospheric deposition would show more intense nitrification and NO₃ leaching because of a greater bacterial activity. It is possible that the relative abundance of mycorrhizae in the burned zones could have been favoured by the phase-out of the bacterial community following fire disturbance and extraction of large amounts of N through volatilization. If fungal biomass constitutes a larger fraction of the total microbial community in the burned stand, then it is possible that the activity of these microbes at HEW resulted in an efficient N immobilization and in turn, in lower N concentrations in the solutions of the forest floor.

We thought that perhaps the effects of mycorrhizae could be modelled in order to explain the low mobility of inorganic N species in the solution of the forest floor of the burned zone. However, allocating 95% of the uptake of N in the forest floor and 5% of N uptake in the mineral soil still overestimated inorganic N concentrations and pH in the solution of the forest floor by about 60%, whereas simulated N concentrations and pH in the solution of the mineral soil were unchanged. In the light of these results, the net immobilization/exudation of mineral N by mycorrhizae is not likely to be reproduced successfully with a simple NO₃ uptake–OH ion release model. Perhaps a more complex model of uptake in which the mechanisms of N immobilization by microbes are included would have produced more convergence between simulated and measured solution pH and N concentrations. Also, model output may have converge even more towards field values if the distribution of uptake was specified as a time-series rather than a steady-state input so that the possible dynamic changes in immobilization/uptake rates of N are considered.

A third explanation for the discrepancy in pre-industrial simulated base saturation and pH may be site history. It must be remembered that SAFE is calibrated on base saturation and if it cannot obtain a good fit with current (measured) base saturation using the specified input, then it modifies pre-industrial steady-state base saturation rather than modify the rates of change in base saturation (Walse et al., 1998). This perhaps indicates that our assumption that factors driving soil formation in the zones were similar prior to industrialization was incorrect. Past stands at HEW may not always have been dominated by maple trees. If a different regeneration pattern was shown to have taken place at HEW, then our approach to estimate/validate the SAFE model with pre-industrial conditions under various site history would be considered unacceptable.

### 6.4. Causes of soil acidification

The mechanisms responsible for the movement of SO₄ are difficult to validate because of some contradictory results on SO₄ sorption at HEW: (1) SO₄ input-output budget calculated by Courchesne et al. (2001) between 1994 and 1997 suggest that SO₄ is neither accumulated nor lost at the watershed scale; and (2) SO₄ desorption in the upper mineral soils was found to increase with increasing air temperature. A possible explanation for this contradiction in SO₄ sorption is that while the inorganic SO₄ mineral soil pool is depleted from desorption, other pools may increase from the accumulation of S in tree biomass and/or in surface horizons under organic forms (Dhamala and Mitchell, 1995). Sinks and sources of S are likely to have balanced each other at the watershed scale. Omitting SO₄ sorption at HEW therefore seemed to be a simple and accurate model for reproducing solution SO₄ concentrations measured in the forest floor and mineral soil of the two zones. Because of the imprecision in the simulated NO₃ concentrations in the solution of the forest floor in the burned zone, however, only patterns of acidification in the unburned zone were investigated.

Simulated acidification in the mineral soil began in 1900 and accelerated during the peak of industrialization in the 1930’s until present day. The period of significant soil acidification corresponded exactly to the period when deposition contributed the most acidity to the forest ecosystem, and is reflected in the SAFE model by the deposition history recreated in Bélanger et al. (2002). Sulfate deposition increased by about 3-fold between 1930 and 1980, while NO₃ and NH₄ more than doubled. Empirical studies also suggest soil acidification during that same time frame. For example, Johnson et al. (1994) observed significant acidification and Ca losses in organic horizons of 48 mixed spruce-hardwood sites in the Adirondack region of New York during the period 1930–1984. Also, Shortle and Bondietti (1992) examined the trend of exchangeable divalent base cations in sensitive humus layers of eastern North American forests starting in the 1930s. Although the study did not allow a direct statistical analysis because of the difficulties in weighting each datum to the whole data set, measured exchangeable concentrations prior to 1950 were clearly higher than those after 1970. Evidence from tree ring chemistry also supports the hypothesis that base cation depletion and acidification in the forest soils of eastern North America occurred during approximately the same period (Bondietti et al., 1990; Momomisha and Bondietti, 1990). The forest floor was not acidified to the same extent as the mineral soil, suggesting that the soil exchange complex in the forest floor is mainly determined by organic acids which create a naturally low pH, as well as the large flux of base cations in the soil.
Despite the fact that S loading from the atmosphere has decreased over the last two decades (Driscoll et al., 1995), the mineral soil did not show any obvious sign of recovery: simulated soil solution ANC and pH have stabilized in the recent years but the rates at which base saturation decreased did not change significantly. Driscoll et al. (1989) observed that changes in stream pH at HBEF were relatively small during the 1980s, regardless of the fact that SO₄ deposition was reduced significantly starting in the 1970s. It is possible that the deposition of H₂SO₄ and HNO₃ may still be too high at HEW or that the decrease in base cation deposition may have been too large [see Hedin et al. (1994) for correlated declines of SO₄ and base cation deposition in the eastern USA] to cause a decrease in the rate of acidification.

6.5. Future nutrient availability

Lawrence et al. (1995) and Miller et al. (1993) anticipate a decrease in base saturation and soil solution pH into the 21st century as base cation inputs from the atmosphere (because of more efficient industrial dust collectors) will decrease even more, while acidifying compounds (i.e. N species) will continue to increase. Interestingly, Stoddard et al. (1999) showed that most streams and lakes in North America have not shown any evidence of recovery as strong declines in base cation concentrations in surface waters have exceeded the decrease in SO₄ deposition. Our concern is that acid deposition may already have brought the cation exchange pool in the mineral soil of some forests to a level that could necessitate centuries for the system to recover. In that sense, Likens et al. (1996) asked that
reduction plans be implemented before damage is even greater, otherwise they will fail.

Assuming no forest disturbance beyond the year 2100, projected soil chemistry in the mineral soil suggests that the balance between acidity and alkalinity is leading into the new century towards a new steady-state as no major change was simulated (Fig. 7). However, projections suggest that further soil acidification is to be expected at HEW if the forest in zone no.2 is harvested into the 21st century (Fig. 7). Soil acidification rates assuming conventional harvesting are higher than those assuming a partial cutting (Fig. 7). However, because low-intensity timber harvesting (partial cutting) seems to be jeopardizing the soil resource, the simulated equilibrium in the “no harvest” scenario is conserved due to the assumption that nutrient cycling is under steady-state, i.e. uptake of nutrients by trees was low as the stand matured.

6.6. Effects of low rates of tree growth and nutrient transfers prior to air pollution

In the original version of the MAKEDEP model, wet and dry deposition, canopy exchange, and litterfall provide N for forest growth and is assumed to be the only nutrient limiting forest growth. At HEW, the low availability of N for trees prior to the onset of air pollution therefore produced low growth rates prior to 1900 (Bélanger et al., 2002). Also, simulated nutrient transfers through the ecosystems were at a minimum because of the slow stand development. The underestimation of the
rates of ion transfer at HEW as resulted to some problems in the calibration procedure as some pre-industrial output is likely incorrect. For example, the low rates of nutrient cycling engendered low base cations concentrations in the forest floor and in turn, increased tree growth at the onset of air pollution triggered an unreasonable increase in base cation concentrations. The net effect was that pre-industrial forest floor pH, ANC and base saturation were lower than what is measured today (Fig. 8). Although this artifact does not seem as large in the mineral soil, model calibration in the mineral soil was also bound to some errors as it is affected by the above layers (in this case the forest floor). As a result, the net change is pH, ANC and base saturation simulated in the mineral soil is larger than that produced with scenario no.3, which assumes higher rates of ion transfer in the ecosystem (Fig. 9). For example, base saturation declined from 75 to 37% with scenario no.4 and from 60 to 37% with scenario no.3, whereas pH decreased from 6.15 to 5.43 with scenario no.4 and from 5.74 to 5.44 in the unburned zone. Estimating pre-industrial conditions is therefore important in order to hindcast and/or forecast rates of soil acidification and caution is required.

7. Conclusion

A few historical soil chemistry studies support the hypothesis that acid-sensitive forest sites of northeastern United States underwent a significant change that resulted in the loss of exchangeable essential base cat-
ions due to acidic inputs from the atmosphere. Simulated soil chemistry with the SAFE model reinforced the conclusions of this work indicating that the largest change in soil chemistry occurred between the time period 1930–1980 and were associated with the deposition of strong acids from the atmosphere. Model projections in the mineral soil suggest that a new steady-state should be reached in the 21st century if timber harvesting is not conducted. However, this equilibrium is likely to be broken if the plans for timber harvesting are put to execution. Although the SAFE model has proved relatively accurate to reproduce soil chemistry and useful to forecast soil chemistry as a function of harvest operations, comparison of model output between unburned and burned stands at HEW casts doubt as to its applicability for all sites because some specific processes involved in nutrient cycling are still poorly understood and described (some omitted), e.g. the immobilization of N by microbes. Furthermore, some calibration problems may be encountered because of a poor description of the site conditions, particularly prior to industrial air pollution. For example, N availability for tree growth and mineral weathering were shown to have large effects on model output at HEW.

Low Ca concentrations measured in the leaves of trees at HEW (Fyles et al., 1994) in combination with a low base saturation simulated in the mineral soil, particularly if timber harvesting is done, create uncertainty regarding the future availability of base cations in sugar maple stands of southern Quebec. Sustained low base cation availability for plant growth could adversely affect forest health and vigour, particularly in forests in the middle stages of stand development because of greater nutrient demands. Base deficient mature trees are more vulnerable to biotic agents such as insects and root rot fungi, and more sensitive to abiotic agents such as cold and drought (Côté and Ouimet, 1996). Fortunately, simulated soil chemistry assuming forest disturbance or not into the 21st century suggest that soil pH will unlikely reach values low enough (< 4.40) to render high Al levels in the soil solution that could be toxic to most tree species (Hendershot and Bélanger, 1999).

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