PROTOCOL FOR ASSESSMENT AND MAPPING OF FOREST SENSITIVITY TO ATMOSPHERIC S AND N DEPOSITION

New England Governors/Eastern Canadian Premiers

Acid Rain Action Plan

2001

Action Item 4: Forest Mapping Research Project

Prepared by: NEG/ECP Forest Mapping Group

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PREFACE

In June 1997, the Conference of the New England Governors and Eastern Canadian Premiers (NEG/ECP) recognized that acidic deposition is "*a joint concern for which a regional approach on research and strategic action is required*" and that "*state and provincial monitoring efforts and analysis remain a high priority within their respective programs*". The Conference charged its Committee on the Environment to present specific policy recommendations at their next meeting, June 1998. A draft framework for the Acid Rain Action Plan was subsequently developed by representatives of the New England states and Eastern Canadian provinces. This draft was refined following the NEG/ECP Workshop on Acid Rain and Mercury in February 1997 in Portland, Maine, and the final work plan was approved in October 1999.

The New England/Eastern Canadian Acid Rain Action Plan identifies steps to address those aspects of the acid rain problem in northeastern North America that are within the region's control. Specifically, the action plan includes:

- a comprehensive and coordinated plan for further reducing emissions of sulphur dioxide and oxides of nitrogen which contribute to the problem of long-range transport of air pollutants, acidic deposition, and nutrient enrichment of marine waters in the region
- a research and monitoring agenda targeted at both improving the state-of-the-science for this environmental problem, and increasing regional cooperation in sharing research and data in order to better understand the impact of acidic deposition on the region and analyze the effectiveness of current control programs on sensitive ecosystems
- a public education and outreach agenda to ensure that the public continues to be educated and mobilized towards the overall goal of protecting the natural environment.

The action plan contains 22 recommendations for specific actions that the provinces and states can undertake to ensure that significant progress is realized in reducing the effects of acidic deposition on ecosystems.

The NEG/ECP Committee on the Environment has appointed a steering committee to coordinate and prioritize the implementation of the action items, and a forest mapping group to carry out Action Item 4 on the forest mapping research project. Action Item 4 concerns the mapping of forest sensitivity to acidifying sulphur (S) and nitrogen (N) pollutants for upland forests in northeastern North America.

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1. PROJECT GOALS

1.1. Problem statement

As a result of SO_2 abatement legislation, sulphur emissions have decreased across North America. However, current emissions of both sulphur (S) and nitrogen (N) compounds are expected to have continuing negative impacts on forest soils, and forest health and productivity (Driscoll et al. 2001). Upland forests, as opposed to marshy wetlands, are expected to be impacted most: acid buffering capacities of upland soils are generally low compared to down-slope locations, and such soils typically do not receive acid-buffering seepage water from higher ground or from upwelling subsurface flows. Over time, the ability or the potential of the upland *forest soil/vegetation complex* to buffer acidic deposition is expected to decrease, and soil nutrient supplies for sustainable tree growth are expected to become depleted and/or imbalanced. The most sensitive forest ecosystems are likely to be in mountainous regions where glacial till and soils are thinnest, and where atmospheric deposition rates are highest (Miller et al. 1993).

In general, assessing forest sensitivity to acidic deposition is a complex task, and doing so in a reliable, yet practical manner requires a scientifically acceptable protocol to:

- Identify, quantify and map those parameters that best capture the ability or potential of each major upland soil-vegetation combination to buffer against increasing soil acidification
- Develop a criterion for determining the level of acidic deposition above which upland forest soils are no longer protected against increasing soil acidification and consequent base cation depletion (*sustainable acidic deposition*)
- Determine whether current and projected atmospheric S and N deposition rates exceed *sustainable acidic deposition*.

1.2. Objectives

The overall goal of this project is to generate maps of eastern Canada and the northeastern United States that identify those forest areas that are most sensitive to acidic deposition. Sensitive areas are those where current or projected acidic deposition potentially exceeds *sustainable acidic deposition rates*. Sustainable acidic deposition rates are those that would maintain forest ecosystem health and related productivity indefinitely based on S and N deposition inputs. Sustainable acidic deposition rates can be determined from analyzing existing information on geology, soils, vegetation, and land-use history. Therefore, the specific objectives of this project are to:

1. Estimate sustainable acidic deposition rates and exceedances for upland forests representative of the New England States and of the Eastern Canadian Provinces, using site-specific data, and ecozone subdivisions
2. Produce maps of sustainable acidic deposition rates and associated exceedances
3. Relate potential exceedance of sustainable acidic deposition to forest productivity and health, based on existing information.

The purpose of this document is to outline the procedures and methodologies that will be used to calculate sustainable acidic deposition rates and the related exceedances. Specifically:

- Section 2 (*Methods for estimating sustainable acidic deposition rate*) defines equations and assumptions used to estimate sustainable acidic deposition using a steady-state mass balance approach.
- Section 3 (*Data requirements for model application and validation*) lists data required to make site-specific calculations and to compare the site-specific to the ecological unit approach.
- Section 4 (*Mapping Methodologies*) outlines database development for the ecological unit approach and the related mapping methodologies.
- Section 5 (*Data Management Protocol*) describes the approach for standardizing data for the ensuing mapping effort.
- Appendices addressing: sustainable acidic deposition, soil base saturation, soil weathering, soil N accumulation, nutrient uptake, atmospheric deposition, impacts of exceedance of sustainable deposition on forest health, and project administration.

2. METHODS FOR ESTIMATING THE SUSTAINABLE ACIDIC DEPOSITION RATE

2.1. Background on sustainable deposition and critical loads

The procedures to be used generally follow the steady-state mass balance approach (SMB), as documented by Posch et al. (1995), the Mapping Manual (UBA, 1996), and Posch et al. (1999) for the calculations and mapping of critical soil acidification loads in Europe. In that context, a critical load was defined as "*a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge*" (Nilsson and Grennfelt 1988). As noted by Posch et al. 1999:

"*The first critical loads to be calculated were for acidity, and - in the negotiations for the 1994 Sulphur Protocol - a "sulphur fraction" was used to derive a critical deposition of sulphur from the acid critical load (Downing et al., 1993; Hettelingh et al., 1995). In preparations for the negotiations for a "multi-pollutant, multi-effect" protocol, nitrogen became the focus, and thus critical loads of N had to be defined as well. This led to a revision of the Mapping Manual (UBA, 1996)*".

The approach described in this protocol, however, differs from the European Critical Load concept by defining a *sustainable acidic deposition rate* that maintains or enhances the current level of soil base saturation such that soil reserves of plant nutrients can be maintained under given forest management practices and/or natural disturbance regimes, for the foreseeable future (e.g., several forest rotations). In contrast, the European critical load concept includes several chemical threshold concentrations above or below which damage to the functioning of organisms and ecosystems is thought to occur. One chemical criterion is the concentration of soil aluminum (Al), which is solubolized by acid, with threshold values for water-soluble Al in the soil solution ranging from 6 to 25 mg L^{-1} (or 0.1 to 0.4 meq L^{-1}; de Vries 1991, Posch et al. 1995). Another soil acidification criterion is the molar ratio of base cations to aluminum (*BC:Al*) in the soil solution. This criterion was selected because excessive amounts of Al ions in soil solution interfere with plant uptake of Ca, Mg and/or K ions (Shortle and Smith 1988). For the BC:Al criterion, a molar ratio of 1 is generally used, because that is at or near the threshold for inducing nutritional Ca, Mg or K deficiencies in tree seedlings (Warfvinge and Sverdrup 1995). These specifications, however, become problematic in terms of scaling up from observed tree seedling responses under controlled laboratory or greenhouse conditions (from which the physiological thresholds values have been derived) to the complexity of conditions in the forest. Soils rarely provide a uniform rooting medium and span a continuum of soil properties varying both laterally and vertically at the scale of a single tree. Often, the standard deviation of any soil property is as large as the mean value of that property (Arp 1984, Arp and Krause 1984). Therefore, a factor of 10 has been suggested as a safety factor to ensure that the soil conditions do not deteriorate under varying conditions and episodic acidification events (Arp et al. 1996). Many questions arise when scaling greenhouse results to the field, including: Is the critical concentration threshold for soluble Al that causes physiological damage to tree roots in the field the same as that in the greenhouse or hydroponic solution? Given the substantial vertical variation in solution Al, to which part of the rooting space should the threshold be assigned? Do tree roots simply adapt by avoiding soil pockets or individual soil layers with high soluble Al? To what extent do soil organic matter and dissolved organic matter render water soluble Al non-toxic via complexation?

Adopting the concept of a sustainable acidic deposition rate to protect against soil acidification and subsequent base cation depletion eliminates the difficulties of setting and accepting physiological thresholds and the related uncertainties as part of the forest sensitivity assessment. Moreover, focusing on the base status of soils is helpful in ensuring that the overall viability of forests and forest soils is maintained via sustainable nutrient capitals. Such an analysis will demonstrate whether nutrient pools can be maintained by primary means (e.g. soil weathering, atmospheric deposition), or by artificial means (e.g. forest fertilization). This protocol also addresses actual and anticipated disturbance regimes which can affect the sustainable acidic deposition rate. Uncertainties in the calculation of sustainable acidic deposition are caused by difficulties in quantifying:

- Inputs of primary nutrients into the soil (atmospheric deposition, soil weathering)
- Extent of nutrient retention within the soil and by the vegetation
- Effective depth of rooting zone
- Changes in individual nutrient pools within soil and vegetation, caused by changes in climate, forest management, or land use

The proposed methodology will allow for sensitivity assessments that specifically deal with these uncertainties.

In Appendix 1, the term *sustainable acidic deposition* - hereafter referred to as *Sustainable Deposition* or *SD* - is defined in terms of a simple mass balance calculation, for (i) the soil alone, (ii) the soil-vegetation complex (the forest), and (iii) the soil-vegetation-atmosphere complex (the ecosystem). It is important to note that the SD is calculated based on inputs of S and N deposition and does not include deposition of other pollutants such as ozone, or the effects of other factors detrimental to forest health, such as pests. Therefore the SD should reflect the level of S and N deposition an upland forest can tolerate in the absence of other impacts. The SD must be considered the *potentially* sustainable level of S and N deposition when other pollutants or forest health impacts are significant. The overall intent of this protocol is to provide a conservative estimate of the SD – the highest value of SD that will maintain the overall soil base status within the context of current or expected acidic deposition loads and related exceedance.

An important objective of this project is to relate the exceedance of SD to forest production and health. In an assessment of critical acidification loads in the Turkey Lake Watershed of northeastern Ontario, preliminary evaluations of data have revealed that many permanent sample plots that were located in regions with high acid exceedances showed symptoms of decline: trees have reduced growth, visible signs of damage, and increased canopy transparencies (Moayeri et al. *in press*). Declines of red spruce at high elevations have been linked to imbalances in soil nutrients and high concentration of acid-mobilized aluminum in the soil (Environment Canada

1997, Schaberg et al. 2000). Another study has shown strong correlations between forest decline symptoms and local acidification exceedances (Ouimet et al. 2000). There is increasing evidence that essential nutrients, such as calcium and magnesium, are being lost from soils exposed to acidic deposition (Environment Canada 1997, Alewell et al. 2000, Driscoll et al. 2001). In addition, air pollution damage to trees influences ecological processes such that the trees become more susceptible to other stressors such as insects and diseases, both directly and indirectly (Hall et al. 1998).

2.2. The steady-state mass balance (SMB)

The steady-state mass-balance approach uses a simplified, steady-state input/output description of the most important biogeochemical processes that affect soil acidification.

Ecosystem inputs include:

1. Atmospheric deposition of S, N, Ca, Mg, K
2. Soil base cation weathering rate (Ca. Mg, K, Na, P)
3. N fixation, where significant.

Ecosystem retention and output processes lead to:

1. Net nutrient accumulation in the soil (N, S, Ca, Mg, K, P)
2. Net nutrient storage in above-ground biomass by uptake (N, S, Ca, Mg, K, P)
3. Net removal of nutrients by forest harvesting or other disturbance (N, S, Ca, Mg, K, P)
4. Nutrient loss through soil leaching (N, S, Ca, Mg, K)
5. Denitrification (N).

Based on the above ecosystem processes, a mass-balance framework is used to calculate sustainable rates of acidifying sulphur and nitrogen deposition for upland forest soils, in order to maintain or enhance the current level of soil base saturation. The background and computational framework dealing with maintaining soil base saturation of ecosystems receiving acidic deposition is presented in Appendix 2.

2.3. Maximum sustainable deposition of acidifying sulphur

The maximum sustainable sulphur deposition rate to maintain the current soil base cation status is given by:

$$SD_{max}(S) = BC_{dep} + BC_w - BC_u - ANC_{le}(SD) \qquad (1)$$

where:

BC_{dep} - sum of Ca + Mg + Na+ K deposition rate (eq ha^{-1} yr^{-1})

BC_w - soil weathering rate of Ca + Mg + K + Na (eq ha^{-1} yr^{-1})

BC_u - net Ca + Mg + K uptake rate (eq ha^{-1} yr^{-1}) ultimately removed by harvest or disturbance

$ANC_{le}(SD)$ - sustainable acid neutralizing capacity (ANC) leaching rate (eq ha^{-1} yr^{-1}).

Atmospherically deposited base cations are acid-neutralizing. The soil weathering process whereby Ca, Mg, K, and Na ions are released into the soil solution is also acid neutralizing. In contrast, base cation uptake is considered acidifying (because plants release H^+ ions during base cation uptake to maintain charge neutrality on either side of the soil-root interface). In these calculations, Na can be neglected except for the Na component of soil weathering. Most of the incoming Na is lost from the ecosystem by leaching, because of low Na retention.

The relationship between soil acidity and base cation status is based on the ion exchange equilibrium by way of the BC:Al ratios of ion exchange sites and in soil solution (Appendix 2). In particular, the maintenance of the soil base saturation is closely linked with what constitutes a sustainable base cation leaching rate. This rate can be calculated from

$$ANC_{le}(SD) = -1.5\frac{BC_{dep} + BC_w - BC_u}{(BC/Al)_{SD}} - Q^{2/3}\left(1.5\frac{BC_{dep} + BC_w - BC_u}{(BC/Al)_{SD}K_{gibb}}\right)^{1/3} \qquad (2)$$

where:

$(BC:Al)_{SD}$ - ratio of base cations to Al (eq/eq) in the soil percolate which would be consistent with maintaining a particular base saturation level

K_{gibb} - gibbsite dissolution constant that controls Al solubility (m^6 eq^{-2}) the multiplication factor 1.5 arises from the conversion from moles to equivalents

Q - rate of soil percolation (combined lateral and downward), which can be assumed equal to streamwater flux (m yr^{-1}).

For background on setting $(BC:Al)_{SD}$, see Appendix 2. For background on determining soil weathering rates, see Appendix 3.

2.4. Minimum sustainable deposition of acidifying nitrogen

As long as the deposition of N stays below the minimum sustainable acidic deposition rate of nitrogen, i.e.,

$$N_{dep} \le N_a + N_u + N_{de} = SD_{\min}(N) \qquad (3)$$

where:

N_{dep} - atmospheric N deposition rate (eq ha^{-1} yr^{-1})

N_a - net N accumulation rate in the soil (eq ha^{-1} yr^{-1})

N_u - net N uptake rate (i.e., increment of nutrient in biomass; eq ha^{-1} yr^{-1})

N_{de} - soil denitrification rate (eq ha^{-1} yr^{-1})

then all deposited N is consumed by N sinks within the ecosystem (N accumulation in soil, N uptake by the vegetation) or lost via denitrification. In this case, SD_{max} *(S)* alone determines the maximum sustainable acidic deposition rate.

All of the above fluxes of N are expressed as net annual quantities, but net soil accumulation of N may vary significantly from location to location, because it is affected by long-term site history (old growth, intensively managed, fire, and other natural disturbances). Current estimates for net N accumulation vary from 0 to 5 kg ha^{-1} yr^{-1} (0 to 350 eq ha^{-1} yr^{-1}). For further background, see Appendix 4.

Equation 3 accounts not only for N_a and N_u, but also for denitrification. However, for upland forest soils, denitrification rates are small to negligible (Binkley et al. 1995; Appendix 1), hence N_{de} is set to 0. Assuming denitrification to be negligible gives a conservative estimate of SD (i.e., SD would be higher if denitrification were assumed to be greater than 0).

2.5. Maximum sustainable deposition of acidifying nitrogen

The maximum allowable sustainable acidic deposition rate of N (for the case of no S deposition) is given by:

$$SD_{\max}(N) = SD_{\min}(N) + SD_{\max}(S) \qquad (4)$$

In this equation the SD_{max} for N is the sum of the sinks for N in the ecosystem and the maximum deposition rate.

2.6. Sustainable deposition of nutrient nitrogen

In addition to the acidifying effect of nitrogen deposition, excess N deposition can cause water quality problems including eutrophication of surface water (the prolific growth of unwanted nitrophilic species in otherwise N-limited ecosystems), and deterioration of drinking water supplies and subsequent human health problems (Fisher et al. 1988, Nilsson and Grennfelt 1988, Skeffington and Wilson 1988). More significantly, excess N deposition can lead to plant nutrient imbalances and forest health decline (Agren and Bosatta 1988, Aber et al. 1998,

Dehayes et al. 1999, Schaberg et al. 2000). Upland forests initially respond positively to the fertilizing effect of additional N deposition until they reach N saturation (Aber et al. 1989). Once a forest reaches N saturation, acidification from N deposition increases, nitrate leaching increases, and plant nutrient imbalances may occur. When there is excess available nitrogen, other nutrient elements such as Ca, Mg, K and P become growth limiting (Schulze 1989). The nitrogen leaching rate, N_{le}, is the eutrophication limit for surface waters or the maximum acceptable leaching rate (the maximum leaching rate for an ecosystem that is not at N saturation). This leaching rate is given by

$$N_{le[acc]} = Q[N]_{eut} \tag{5}$$

where:

$N_{le[acc]}$ - acceptable leaching of N

$[N]_{eut}$ - that N concentration in the soil solution above which it would be considered detrimental to ecosystem or soil

The sustainable deposition rate for nitrogen with respect to ecosystem eutrophication can then be expressed as

$$SD_{nut}(N) = N_a + N_u + N_{de} + N_{le} \tag{6}$$

In view of both the acidification and eutrophication issues, the maximum allowable N deposition can then be obtained from $\min\left[SD_{\max}(N), SD_{nut}(N)\right]$.

2.7. Sustainable acidic deposition (SD)

Since both S and N deposition contribute to acidity, they are both included in the calculation for the sustainable acidic deposition rate. For a given forest, it is therefore possible to determine those combinations of S and N deposition that will not exceed the sustainable acidic deposition rate. The various combinations of S and N deposition that do add up to the maximum sustainable acidic deposition rate therefore delineate the **sustainable acidic deposition region within the S_{dep} - N_{dep} deposition continuum** (i.e., the shaded area in Figure 1).

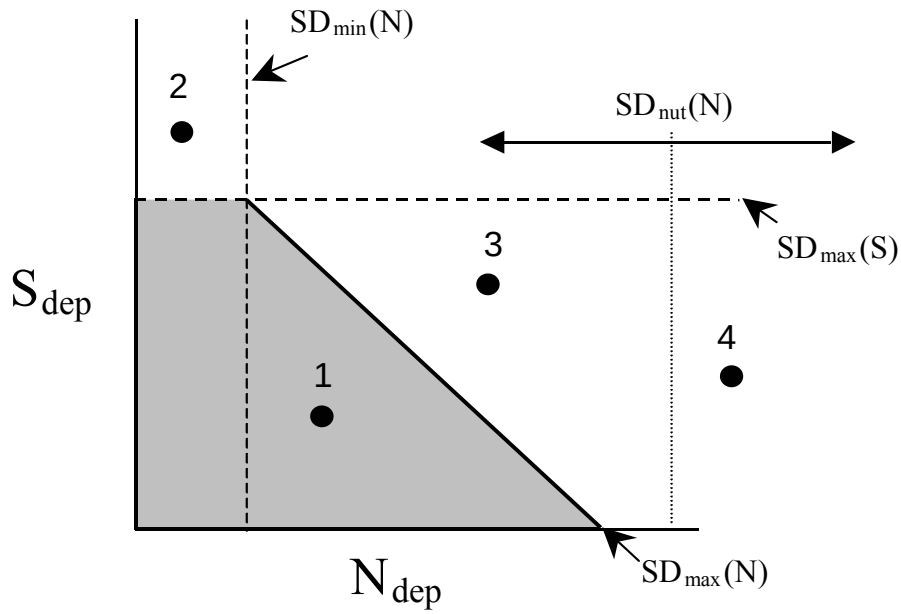

Figure 1. Relationship between atmospheric S and N deposition and the sustainable acidic deposition rate SD(S) + SD(N) for upland forest soils. For each point lying in the shaded area (e.g., Point 1), there is no exceedance of the sustainable acidic deposition rate. Points lying outside the shaded area exceed the sustainable deposition rate. For Point 2, S deposition is larger than the maximum acceptable rate for S deposition, and N deposition is less than the amount that the forest ecosystem can retain [$N_{dep} < SD_{min}(N)$]. This means that the system would not be saturated with respect to N, and, in this case, there would be an exceedance of sustainable S acidification but no exceedance of N acidification. For Points 3 and 4, there would be sustainable acidification exceedances due to the combined effects of S and N deposition. For Point 4, there would be a soil acidification exceedance as well as a N nutrient exceedance (the associated vertical line can be moved to the right or the left depending on one's choice about $[N]_{eut}$). Notes: see text for $SD_{max}(S)$, $SD_{max}(N)$, and $SD_{min}(N)$. The slope of the shaded area is -1 for the case of upland forests, when denitrification is considered negligible.

2.8. Forest biomass production and N uptake

In order to determine the sustainable deposition rate, mean annual forest biomass production rate or mean annual increment (MAI) is calculated. The MAI that can be sustained is a function of: (1) atmospheric deposition, (2) soil weathering, (3) the maintenance of current soil base status, (4) existing or anticipated nutrient availabilities, and (5) forest disturbance regimes, including harvesting and fire. The sustainable MAI can be derived once sustainable N concentrations in the above-ground forest biomass (leaves, branches, bark, stemwood) are specified using steady-state conditions for N accumulation and biomass.

Since

$$[N]_{biomass} = \text{N content } \textit{(kg N)} \text{ / Biomass } \textit{(tonne)} = N_u \text{ / MAI} \qquad (8)$$

one obtains the MAI using N uptake (defined below) and N concentration in biomass

$$\text{MAI} = N_u \text{ / } [N]_{biomass} \qquad (9)$$

where:

Biomass - above-ground forest biomass

N content - accumulation of N in biomass

MAI - mean annual increment, is equivalent to above-ground biomass divided by rotation length (years), or average period of recurring disturbance regime

$[N]_{biomass}$ - wheighted/whole-tree concentrations of N (in kg N /tonne).

The MAI-based calculations are used to estimate the sink for nutrients in vegetation over a forest rotation. Coarse and fine roots usually remain on site. Thus, mean annual increment calculations and related N_u estimates can be restricted to the increment in above-ground vegetative components which will be removed by harvesting.

N uptake for mean annual leaf, branch, bark, and stemwood production is computed from

$$N_u = \text{season_length_factor} * \min \{X \text{ supply } [N]_{biomass} / [X]_{biomass} \text{ , N availability}\} \qquad (10)$$

where:

season_length_factor - the fraction of the year during which nutrients are absorbed by the soil-vegetation complex (i.e., the growing season)

X_{supply} - the long-term mean annual supply rate of Ca, Mg, K, and P. Square brackets denote weighted, whole-tree concentrations of X for each major tree type.

$N_{availability}$ is the net sum of inputs from deposition and N fixation and losses to soil N accumulation

$$N_{availability} = N_{dep} + N_{fix} - N_a, \quad (11)$$

and X_{supply} is the sum of inputs from deposition and mineral weathering

$$X_{supply} = X_{dep} + X_w \quad (12)$$

Note that equation 10 makes use the limitation of X and N availability to calculate biomass growth and therefore uptake. Long-term N uptake is considered Ca, Mg, K, or P limited if

$$N_{availability} \text{ (in eq)} > X_{supply} \cdot [N]_{biomass} / [X]_{biomass} \quad (13)$$

Otherwise, N uptake is only limited by N availability. Ecosystems for which biomass growth is X-limited are at N saturation. In contrast, ecosystems for which biomass growth is N-limited have not reached N saturation (Aber et al. 1998). Note that because $[N]_{biomass} / [X]_{biomass}$ is fixed, sustainable deposition rates are calculated for conditions under which nutrient imbalances are not expected to occur.

In the above equations, S uptake does not contribute to the calculated sustainable acidic deposition rate, because S:N ratios in vegetative tissues are approximately 0.1. If S uptake were included in the calculation, then SD would increase very slightly.

The overall accuracy of this approach is a function of the accuracy of the N and X uptake estimates, and on the presumed constancy of the N:X ratios. While the nutrient ratios are considered constant, they are plastic in nature, thereby requiring ratio-specific sensitivity analysis of the model-calculated SD and related exceedance values. For details regarding X_u and N_u and related species-specific concentration values and nutrient ratios in foliage, branches, bark, and stemwood, see Appendix 5.

2.9. Exceedances of sustainable acidic deposition

Sustainable acidic deposition rates, once defined and calculated, can be compared with current or expected rates of acidic deposition (wet + dry). An exceedance of sustainable deposition can then be defined as follows (Appendix 2):

$$\text{Exceedance} = S_{dep} + \max\{N_{dep} - SD_{min}(N), 0\} - SD_{max}(S) \quad (14)$$

For background and details regarding the estimation of local S, N, Ca, Mg, and K deposition rates, see Appendix 6. For background on observed or expected effects of exceedance on forest growth, see Appendix 7.

3. DATA REQUIREMENTS FOR MODEL APPLICATION AND VALIDATION

The goal of this project is to calculate and map SD based on detailed, site-specific data from locations throughout eastern Canada and the northeastern U.S. An approach similar to the one proposed here has already been used in a preliminary assessment of exceedance of critical loads in parts of Eastern Canada (Arp et al. 1996, Ouimet et al. 2000, Moayeri et al. *in press*), but that approach has not yet been applied to large regions. The overall intent of that assessment was to calibrate and validate the SMB model by comparing field observations of forest biomass increment, nutrient uptake, and soil ion leaching with corresponding model calculations. The data can also be used to evaluate underlying model assumptions. For example, it will be important to evaluate the sensitivity of the sustainable acidic deposition rate to uncertainty and variability related to tree nutrient uptake, soil N accumulation, N fixation, soil base weathering, etc., as affected by species composition, site conditions, harvest regimes, natural site disturbances (including fire), stand age, and biomass.

3.1. Site selection

Site selection for model calibration and validation is based on the distribution of stand types in the region and the ability to meet the data requirements for the SD calculations. There are three categories of sites:

Level 1 site: all necessary data are available

Level 2 site: some data are unavailable, but missing data can be modeled or extrapolated with confidence (e.g., atmospheric deposition)

Level 3 site: many data are unavailable and must be modeled or extrapolated.

3.2. Basic information requirements

Basic information of reference sites consists of:

- Political jurisdiction
- Number of the site (running by jurisdiction)
- Name of the site
- Geographical coordinates (degrees, minutes, seconds)

- Elevation
- Size of the site (ha)
- Long-term average temperature (ºC)
- Average length of vegetation period (day yr^{-1}). This represents the mean air temperature > 5 ºC for 5 consecutive days
- Land-use history.

3.3. Atmospheric deposition

Atmospheric deposition is made up of dry deposition (gaseous and particulate compounds) and wet deposition (snow and water deposition). The interception of mountain cloud water and coastal fog may also contribute to total deposition. In general, information on the chemistry of wet deposition is available from specialized meteorological networks. The precipitation rate is measured at a large number of locations. Dry deposition is more difficult to evaluate. Air concentrations of dry depositing species are available from a relatively small number of observing stations. Deposition velocity estimates can be obtained from deposition models (or from throughfall sampling for S only). Cloud and fog water deposition estimates require fog collectors and deposition models.

Data for the five most recent years should be reported for deposition chemistry. The mandatory parameters for deposition chemistry are:

- Specification of deposition origin (1-Wet deposition only, 2-Bulk deposition, 3-Wet + dry + fog deposition)
- Specification of deposition measurements (1- Samplers, 2-Models (or maps), 3- Samplers and models).
- Long term average precipitation amount (mm)
- Long term runoff (mm)
- Average percentage of precipitation as snow (%)
- pH
- Specific conductivity (m S m^{-1})
- SO_4^{2-}-S (sulphate as sulphur mmol S L^{-1} or mol ha^{-1})
- NO_3^--N (nitrate as nitrogen mmol N L^{-1} or mol N ha^{-1})
- NH_4^+-N (ammonium as nitrogen mmol N L^{-1} or mol N ha^{-1})
- Cl (Chloride mmol Cl L^{-1} or mol Cl ha^{-1})
- Na (Sodium mmol L^{-1} or mol Na ha^{-1})
- K (potassium mmol L^{-1} or mol K ha^{-1})
- Ca (calcium mmol L^{-1} or mol Ca ha^{-1})
- Mg (magnesium mmol L^{-1} or mol Mg ha^{-1})

- Dates for which the chemistry data are averaged (e.g. 1988-1993).

For Level 2 and 3 sites, the data can be estimated from spatial data coverages for acidic deposition available from NATChem (National Atmospheric Chemistry Database and Analysis System, Atmospheric Environment Service of Environment Canada) or the HRDM (High Resolution Deposition Model, Ecosystems Research Group, Ltd.).

3.4. Soil characteristics

The soil parameters required provide information about soil quality and acidification impacts of S and N deposition and the potential nutrient imbalance/eutrophication impact of N deposition. Soil information should be reported to the bottom of the B horizon, which is considered to be the extent of the rooting zone.

- Soil type (according to the American (Soil Survey ref) or Canadian Soil Classification System (Soil Classification Working Group, 1988))
- Soil profile description (this information may be found using site soil sampling, pedological, or ecological surveys)
- Humus form (Mor, Moder, Mull)
- Forest floor thickness (cm)
- Horizon designation and depth (cm)
- Rooting depth (cm)
- Parent material type. This information can be found from soil sampling or from pedological, geomorphological or ecological surveys.
- Mineralogy and/or total element content (% P_2O_5, K_2O, CaO, MgO, Na_2O, Al_2O_3, Fe_2O_3, SiO_2) to the bottom of the B horizon, or for soil substrate. This information can be found from total soil analysis ($HClO_4$-HNO_3 digestion), from pedological surveys, or from geological surveys. See Appendix 3.
- Properties describing the acid and N status of the soil, e.g. pH, cation exchange capacity (CEC; cmol (+) kg^{-1}), percent base saturation (BS), N (%), and exchangeable base cation concentrations (K, Ca, and Mg in ppm). This information can be obtained from site soil sampling or pedological surveys.
- Background properties that determine soil acidity and nutrient status, i.e. organic matter content (%) and particle size distribution (texture: proportion of sand, silt, and clay in % of weight of combined soil minerals =2 mm in size). This information can be found from site soil sampling or pedological surveys.

- Parameters necessary for calculating soil chemistry pools, i.e. soil bulk density (<2 mm soil fraction, g cm^{-3}) and coarse fragment content (>2 mm soil fraction, %). This information can be found from site soil sampling or pedological surveys.
- Dates of sampling (yr).

3.5. Stand characteristics

This information is used for estimation of nutrient uptake or increment by the stand:

- Stand type (e.g. deciduous, coniferous)
- Species forming the stand and stand table:
  For each tree species:
  - number of stems (number ha^{-1})
  - average diameter at breast height (cm)
  - total volume (m^3 ha^{-1} and/or – preferably – total biomass when dry (tonnes ha^{-1})
  - wood density of dry wood (kg m^{-3})
  - stand age (years)
- If available, N, K, Ca, and Mg concentrations in the wood, bark, branches, and leaves for each major tree species (ppm).

3.6 Data for model verification and forest health impact assessment

In addition to the input parameters, some parameters will be used for model verification including: ion leaching loss, mean annual increment, denitrification rates, nitrogen fixation rates, and ion concentrations in the soil solution. Additional information may be used to evaluate forest health including: foliage transparency, crown density, dieback, incidence of insects and disease, areas of forest decline, diversity and abundance of understory vegetation and lichens.

4. MAPPING METHODOLOGIES

4.1. Overview

At least two methodologies will be used for SD mapping (Arp et al. *in press*). For the first approach -- termed the ***site-specific approach*** -- as many sites as possible will be considered. However, not all sites will have all required information for the calculation. Sites that do have all required information are designated *Level 1* sites. There are presently about 20 Canadian sites and 15 American Level 1 sites. As already stated, sites with few missing variables are termed *Level 2* sites. The ARNEWS sites (108 sites) are Level 2 sites. These sites have the potential to become Level 1 sites with little effort, or have enough data that can be used to generate the missing information. Sites that have some useful data but fall short of having sufficient data are termed *Level 3* sites. The NAMP sites (147 sites), EMAN sites (38 sites) and FIA sites fall into this category. In all, there are about 325 sites within the study region for establishing a basic site-based mapping grid for the region of interest, i.e. all eastern provinces (Ontario east to Newfoundland) and the seven northeastern states. Level 3 sites can also be used to evaluate how the missing data for these sites can best be determined, either through examining and analyzing existing data bases for these sites or, if necessary, by further data collection at the sites.

The second approach--termed the ***ecological unit approach***--uses established terrestrial ecozone maps for the region of interest. In Canada, there are fifteen ecozones altogether. These units are subdivided into ecoregions and ecodistricts. Each ecodistrict has its own combination of physical environmental parameters (i.e. soils, water, and climate) and forest type. The ecological unit approach, therefore, is a means for ecosystem-level SD mapping, at the ecodistrict level (Wiken et al. 1996).

The two approaches will be compared in order to assess uncertainties associated with SD mapping. Comparisons will be made between the values in the site specific and ecological unit approaches for (1) input parameters such as deposition, (2) SD estimates, and (3) indicator parameters, such as ANC of streamwater nitrate, at sensitive sites.

4.2. Building the required databases

For the site-specific approach, Level 1, 2, and 3 sites will be considered: The ecological unit approach will make use of the following geospatial databases:

Canada

- Acid deposition coverage – (from NATChem, MSC)
- Climate data coverage (from CFS)
- Actual Evapotranspiration (AET) - (calculated)
- National Forest Inventory - 1:20,000,000 (from CFS)
- Provincial Forest Inventories – 1 km grid based on 1:12,000 (from Nfld)
- Soil Landscape of Canada - 1:1,000,000 (from CANSIS, AAFC)
- Geological Map of Canada - 1:5,000,000 (from NRCan)
- National Topographic Database 1:250,000 (from NRCan)
- Digital Elevation Model - 1 km grid (from NRCAN)
- Ecodistricts 1:2,000,000 (from AAFC)
- VMap Level 0 (Digital Chart of the World) – 1:1,000,000 (from NIMA)

United States

- Acid deposition coverages - (from NATChem and HRDM, Ecosystems Research Group, Ltd.)
- Climate data coverage - (from NE Regional Climate Center, and UNH model)
- Actual Evapotranspiration - (calculated from NE Regional Climate Center and UNH climate model, and VT Reference Evapotranspiration model)
- Forest Inventory and Assessment - (state level FIA) 1:250,000
- Soil Landscape - (from STATSGO, NRI) 1:250,000
- Geological map - (from VGS and VMC) 1:250,000; (regional map from USGS) 1:500,000
- Source Till map - (calculated from geologic map information)
- Topographic database - (from USGS digital elevation model) 1:250,000
- Ecological Units-Subsection - (from USFS) 1:1,000,000

The ecological unit approach will examine all upland areas within each ecodistrict to establish upland polygon areas for the model. Each upland polygon, which is to be based on existing soil drainage mapping, will be given a unique identifier and associated ecodistrict class. All input data will be cross-referenced to these polygon numbers. Tree species data, soil data, acidic deposition, and climate data will be determined for each upland polygon area from the existing spatial data sets and by way of GIS analysis techniques. Tree nutrient data will be estimated based on data gathered from the ARNEWS program and from intensive research sites in the U.S.A. The geo-referenced data will be compiled as a series of spreadsheets, which – in turn – will be accessed by the SMB model, one eco-unit at a time (each eco-unit is represented by one unit-referenced row of data).

The following steps outline the mapping method:

1. Acquire all spatial data sets and prepare the data for analysis
2. Develop upland forest polygons, by dominant tree type
3. Estimate atmospheric deposition and climate date, including actual evapotranspiration (AET), for each upland polygon
4. Estimate soil weathering rates for each upland polygon
5. Input data and run the SMB model for likely forest disturbance scenarios (including old-growth condition and repeat harvesting)
6. Use model to estimate nutrient uptake
7. Use model to estimate tree biomass production (mean annual increments) and cross check with field estimates
8. Use model to estimate soil leaching rates and cross check with field estimates
9. Reparameterize model if needed
10. Map forest sensitivity to acidic deposition on both a site and ecodistrict basis

For Eastern Canada, ecological and environmental characteristics of the landscape will be mapped at the ecodistrict scale, polygon by polygon. For the New England states, ecological and environmental characteristics of the landscape will be mapped in a raster GIS at scales of 90 to 30 meter ground resolution. This mapping scheme is chosen to represent the conditions of the complex, mountainous topography of the New England states in the most explicit manner. High-resolution mapping is necessary to capture the full range of conditions (soils, forest type, atmospheric deposition), at the resolution at which these conditions change. Sustainable deposition rates, as calculated for individual grid cells can - in turn - be aggregated by ecological unit to provide map representations more directly comparable with the results from the Canadian provinces.

4.3 Comparison of approaches

Both the site-specific and the ecological unit approach have advantages and disadvantages. Since the site-specific approach uses actual data, there are fewer assumptions associated with model input. However, regional interpretations are limited by the relatively small number of sites with the required information in comparison to the size of the study area. In addition, site data may not adequately represent all major regional and local soil, climate, and forest combinations. In contrast, the ecological unit approach utilizes existing spatial data, thus basing the resulting sustainable deposition calculations on known soil, climate, and forest distributions. Its main limitation is that the input data per mapping unit are numerically derived from other data, i.e., they are not measured *per se*. This approach also differs from the European approach since all

input data are keyed and mapped deterministically on an ecodistrict or high-resolution grid cell basis rather than statistically on a coarse grid-cell basis (Posch et al. 1999).

Ideally, the combination of both approaches will facilitate a self-correcting mapping approach, such that the site-specific mapping effort can be used to evaluate and re-parameterize the ecologically-based mapping methodology. Alternatively, the ecologically-based mapping methodology can be used as a means to infer the spatial representativeness of the site-specific analyses. By using both approaches, the work group will be able to compare results and will have independent quality control as well.

During the pilot phase of the project, Newfoundland, Nova Scotia, and Vermont will be used to develop and test mapping methodology for the ecological unit and site-specific approach. Part of the pilot phase of this project is to compare the two approaches and determine why results differ. The lessons learned will then be applied to the region-wide mapping effort to allow policy and decision makers to make informed decisions about the spatial extent of the acid rain problem in the northeastern part of North America.

5. DATA MANAGEMENT PROTOCOL

A serious hurdle in making sustainable deposition calculations for a large region with many jurisdictions is the compilation and standardization of the data for cross-boundary mapping purposes. The purpose of the data standardization and management protocol is three-fold, to:

1. ensure that data compiled from each jurisdiction represent the same measures in the same way
2. minimize differences in data handling by different individuals by creating a roadmap which will guide the data-handler through the series of assumptions necessary to convert the data to the form used to calculate SD
3. ensure proper documentation of the data, data origin, and assumptions made at each step of the data handling.

While data relating to meteorology and atmospheric deposition are easily standardized, the same is not true for extensive datasets of soil and forest cover type. In addition, for model validation, soil leaching data are relatively sparse, and may include considerable variability based on collection method.

In all cases, decisions will be made about how existing information from various organizations can be transformed into common model input. These decisions ultimately depend on what data will actually be available and on the modeling objectives. For example, sustainable deposition rates can be mapped for given areas by species (e.g., balsam fir, sugar maple, red spruce), by group of species (e.g. fir, maple, spruce), or by forest cover type (e.g. softwood, mixed, hardwood).

While the data management protocol will necessarily be lengthy and the process of documenting assumptions time-consuming, the documentation produced will be invaluable both in evaluating the methods (especially in comparing the two approaches) and interpreting the results.

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Appendix 1. Sustainable acidic deposition: definitions

P. A. Arp, E. K. Miller

The sustainable acidic deposition rate is the total amount of acidic deposition that can be received by a forest ecosystem without depletion of plant-available nutrient cations. The term *acidic deposition* refers to atmospheric deposition of the acid-generating compounds H_2SO_4 and HNO_3. In order to determine the ecosystem ANC, one not only needs to consider the acid buffering capacity of the soil, but all of the ecosystem processes that react with incoming acidity. Hydrogen ion activity is strongly buffered by reaction with solid phase and dissolved organic matter in soil and soil water. Some of the incoming H_2SO_4 and HNO_3 acidity is neutralized by way of plant (root) uptake, because plants exchange NO_3^- and SO_4^{2-} with HCO_3^- ions. Similarly, soil adsorption of SO_4^{2-} ions is acid neutralizing because this process also releases OH^- or HCO_3^- ions from the soil mineral surface into the soil solution. Nitrate ions, in contrast to SO_4^{2-} ions, are, however, not readily retained but percolate quite freely through most forest soils (Reuss and Johnson 1986). The overall rate of soil acidification due to atmospheric H_2SO_4 and HNO_3 deposition, therefore, depends, in part, on whether SO_4^{2-} and NO_3^- ions are retained by the ecosystem. Lack of SO_4^{2-} and NO_3^- retention means that some of the SO_4^{2-} and NO_3^- ions are exported from the ecosystem, thereby causing loss of base cations from soil through cation exchange reactions and subsequent leaching.

Our definition of forest sustainability encompasses much more than prevention of soil acidification *per se*. Although we discuss (and calculate) sustainable acid deposition rates in terms of ecosystem ANC, these calculations also allow us to keep track of the inputs and outputs of the important plant nutrient base cations Ca^{2+}, Mg^{2+} and K^+. Maintaining a soil environment which provides forest vegetation with adequate access to these nutrient cations without undue energy costs is central to our concept of forest sustainability with respect to S and N deposition. A substantial body of research suggests that plant stress indicators, growth rate reductions, and susceptibility to mortality-producing injuries are related to poor Ca, Mg or K nutrition or the interaction of nutrient limitation with other direct and indirect effects of acid deposition (Schaberg et al. 2000, see discussion in Appendix 4). Since N deposition has the potential to stimulate forest growth when N is growth limiting, our definition of sustainability considers the effect of enhanced productivity on long-term nutrient demands as well.

In this analysis, N in the form of NH_3 or NH_4^+ requires special consideration (Kämäri et al. 1992). In principle, the atmosphere originally receives these forms of N by way of upwind NH_3 volatilization. Within the atmosphere, NH_3 is then transformed to NH_4^+ by taking H^+ ions from HNO_3, H_2SO_4, or H_2CO_3. Hence, the addition of NH_3 into the atmosphere does not, by itself,

introduce new H^+ ions. Nevertheless, the deposition of NH_4^+ is also considered acidifying (Nihlgard 1985) because of

1. the release of H^+ as a result of vegetation NH_4^+ uptake, and
2. the microbially induced nitrification reaction given by

$$\mathbf{NH_4^+ + 2\ O_2 \rightarrow NO_3^- + 2\ H^+ + H_2} \qquad \text{(A1.1)}$$

For long-term considerations, NH_4^+ uptake can be considered inconsequential in terms of net soil acidification because the uptake-induced H^+ release is neutralized again once the N taken up is returned to the soil via litter and is re-mineralized to NH_4^+. Nitrification, in contrast, is expected to add to the long-term soil acidification stress when NO_3^- is leached from the ecosystem. In comparison, leaching losses of NH_4^+ from soils are generally inconsequential because of (i) high rates of soil retention (via cation exchange), and (ii) high rates of plant uptake of NH_4^+.

Another N input at the ecosystem level is biological N_2 fixation. Nitrogen that enters the system by this process also becomes acidifying if it eventually becomes NO_3^-. The electron-mass balance reaction for N_2 fixation is given by (Ledgard and Giller 1995):

$$\mathbf{N_2 + 4\ H_2O + 2\ C + 2\ H^+ \rightarrow 2\ NH_4^+ + 2\ CO_2 + H_2} \qquad \text{(A1.2)}$$

The subsequent conversion of NH_4^+ ions to NO_3^- will result in net soil acidification even though the fixation process is H^+ consuming (Dixon and Wheeler 1983). As an additional complication, one must consider that biological N transformations depend, to a large extent, on the state of soil aeration. For well-drained soils under well-aerated conditions, N losses from soil are mainly restricted to N uptake and leaching. In soils with anaerobic conditions, denitrification and volatilization of soil N by way of N_2 and NH_3 volatilization is de-acidifying. For well-drained forest soils, gaseous N losses are: "typically slight; NH_3 volatilization is very low, and denitrification losses do not appear to be important" (Binkley et al. 1995). In this project we focus our attention on upland soils, which will be most sensitive to acidification.

During the initial phase of deposition-driven soil acidification, leached NO_3^- and SO_4^{2-} ions would be accompanied mainly by base cations in the soil exchange complex. With increasing soil acidification (i.e. a decrease in the proportion of base cations on soil exchange sites), the mix of accompanying cations will gradually change from mainly base cations (Ca^{2+}, Mg^{2+}, K^+) to mainly acid cations (Al^{3+}, Fe^{3+} and H^+). In terms of sustainable acid deposition evaluation, it is important to consider how much of the exchangeable, and therefore plant-available, base cation

pool would be lost from the soil due to this anion leaching and through removal of BC in biomass by harvesting.

A1.1. Acid neutralizing capacity

We now define sustainable acidic deposition in terms of several components of the acid neutralizing capacity (ANC) of a forest ecosystem:

1. ANC generation by the soil due to the weathering of primary minerals**,** $\mathbf{ANC_{(soil)}}$
2. net ANC due to growth and harvest removal of forest vegetation, $\mathbf{ANC_{(forest)}}$
3. ANC due to atmospheric deposition of base cations, $\mathbf{ANC_{(BCdep)}}$

The total amount of acidic deposition that can be sustained by a forest ecosystem without a depletion of plant-available nutrient cations is the sum of these components of the system ANC:

$$\mathbf{SD_{(ecosystem)} = ANC_{(soil)} + ANC_{(forest)} + ANC_{(BCdep)}} \qquad \text{(A1.3)}$$

$\mathbf{ANC_{(soil)}}$ is based on:

1. Weatherability of primary minerals in the soil (Appendix 3)
2. Acceptable ANC leaching loss from the system and, by extension, an acceptable base saturation of the soil exchange complex (see Appendix 2)

Hence,

$$\mathbf{ANC_{(soil)} = BC_{(weathering)} - \text{acceptable ANC leaching from soil.}} \qquad \text{(A1.4)}$$

ANC(forest) is defined to include the annual rate of soil acid neutralization or acidification due to net vegetation nutrient uptake, N immobilization in the soil, and N_2 fixation, as follows:

$$\mathbf{ANC_{(forest)} = N_{(uptake)} + N_{(accum)} - N_{(fixation)} - BC_{(uptake)}} \qquad \text{(A1.5)}$$

where $\mathbf{BC_{(uptake)}}$ and $\mathbf{N_{(uptake)}}$ are net rates of base cation and N uptake; $\mathbf{N_{(accum)}}$ is the net accumulation of N in soil organic matter, which has a positive effect on $\mathbf{ANC_{(forest)}}$ when positive; $\mathbf{N_{(fixation)}}$ is the annual net gain of N through biological N_2 fixation. $\mathbf{N_{(fixation)}}$ is negative because N fixation leads to net soil acidification by way of subsequent soil nitrification. Note that vegetation affects soil acid buffering capacity in two contrasting ways:

1. Enhanced N uptake raises the acid buffering potential of soil by preventing NO_3^- leaching;
2. Enhanced base cation uptake decreases soil acid buffering capacity

Nitrate-N and NH_4^+-N species both contribute to soil acidification, NH_4^+ through uptake, and NO_3^- through leaching. Plant uptake of NH_4^+-N releases H^+ ions while plant uptake of NO_3^--N releases bicarbonate ions. In contrast, NH_4^+-N leaching losses are often small to negligible, while acidifying NO_3^--N leaching losses can be substantial.

$\mathbf{N_{(accum)}}$ needs to be evaluated for specific stand conditions, with prior land use history and soil organic matter C:N ratios as primary indicators of the net N immobilization potential of the soil (see Appendix 4). Net-N_2 fixation is assumed to be proportional to the rate of soil weathering, i.e.,

$$\mathbf{N_2\ fixation = N_fix_factor^* BC\ weathering,} \qquad \text{(A1.6)}$$

on the assumption that there would be little or no net N_2 fixation if long-term base supply to the N fixing micro-organisms is also low or absent. Low soil base weathering would also mean low soil pH buffering, and related decreases in soil pH are known to affect N_2 fixation negatively (Dixon and Wheeler 1983). In the equation above, **N_fix_factor** is a calibration constant that can be obtained from sites for which the rates of weathering and N_2 fixation are known.

Atmospheric base cation deposition must be part of the evaluation of sustainable acid deposition rates at the ecosystem level. Base cation deposition contributes to the overall ability of the system to neutralize acidity, hence

$$\mathbf{ANC_{(BCdep)} = Base\ cation\ deposition} \qquad \text{(A1.7)}$$

$\mathbf{SD_{(ecosystem)}}$, therefore, combines the effects of soil weathering, acceptable ANC leaching, acceptable exchangeable base cation reserves, vegetative uptake, net N gain via N accumulation in soil humus, N_2 fixation, and atmospheric base cation deposition.

Sulphur uptake by vegetation is not accounted for in any of the above definitions, because S uptake does not contribute significantly to sustainable acid deposition. S:N ratios in vegetative tissues are approximately 0.1 or less. If S uptake were to be considered, it would increase ANC(forest) and SD(ecosystem) very slightly. Soil adsorption and desorption of S, however, may need to be considered in soils that have an appreciable anion exchange capacity. For some forest soil layers, this is not the case. For podsolized soil layers, S absorption increases from practically 0 above pH 7 to maximum levels at pH 4. Below pH 4, S adsorption decreases again (Arp and Oja 1997). For most forest soils, anion exchange capacities are low.

A1.2. Exceedance of sustainable acidic deposition

For convenience, we refer to $\mathbf{SD_{(ecosystem)}}$ as simply **SD.** Once **SD** is defined and calculated for a given locality in terms of $SD_{min}(N)$ and $SD_{max}(S)$, it can then be compared with the expected, current and/or past rates of acidic deposition at that location [see the protocol body for definitions of SD(S) and SD(N)]. A **sustainable deposition exceedance** occurs when given atmospheric S and N deposition rates exceed **SD** as follows:

$$\mathbf{Exceedance = S_{(dep)} + max\ \{N_{(dep)} - SD_{min}(N), 0\} - SD_{max}(S),} \qquad \text{(A1.8)}$$

This equation can then be used to show which localities are likely to experience exceedance of sustainable rates of acid deposition (*positive exceedance*), and which localities have an acid neutralization capacity that exceeds that of the incoming rate of acid deposition (*negative exceedance*).

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Appendix 2: Sustaining the soil base saturation (Author: P. A. Arp and R. Ouimet)

The concept of "sustainable acidic deposition" is based on protecting an ecosystem from indefinite soil acidification. Indefinite soil acidification, would lead to: (i) a continuing decline of soil pH, (ii) a continuing increase in terms of the mobilization of soil Al (and eventually of soil Fe), and (iii) a continuing depletion of available base cations such as Ca, Mg and K. Two of the most relevant criteria for keeping soil acidification rates in check are:

1. not to exceed a certain threshold level of Al^{3+} concentrations in the soil solution or in the soil leachate;
2. not to further decrease the soil base saturation due to increased base cation leaching.

Typically, the acceptable level for soil acidification have been set as follows:

acceptable Al concentrations <= 0.2 meq/l
acceptable BC:Al concentration ratio level =>1
acceptable pH level =>4.2

These levels were selected based on forest tree seedlings growing in controlled hydroponic media have shown negative responses to solutions with Al concentrations > 0.2 meq/l and BC:Al leaching level <1 UNITS (REF). Negative responses include reduced shoot growth, root swelling, shutting down of water uptake, tie-up of P in the rhizosphere, and reduced uptake of base cations (REF). Further examinations revealed that Al toxicity is only a factor when Al is in its water soluble Al3+ state. When Al is complexed by organic matter, toxicity levels are low although Al complexation may facilitate Al uptake by the plants. Key articles summarizing these threshold for major tree species were compiled by Sverdrup and Warfvinge (1993).

Several factors must be taken into consideration in estimating sustainable acidic deposition, we now refer to the following considerations:

1. Soils are highly variable in chemical and physical properties. Hence, setting sustainable levels at values where there are known growth and mortality responses cannot be considered safe: some areas will exceed these limits, while others will not. Therefore, a safety factor of approximately 10 should be invoked for the Al and Al:BC levels to maintain healthy forest soil conditions at the ecosystem level. Therefore:

Critical Al^{3+}/ base cation ratio in soil solution < =0.15 eq/eq or BC:Al => 10 (M/M)

2. Maintaining Al^{3+} concentration less than 0.02 meq/l ensures that soil pH levels remain greater than 4.2. In contrast, accepting $[Al^{3+}]$ = 0.2 meq/l allows the targeted soil pH to drop below 4.2 from topsoil to subsoil. Considering that Al oxide/hydroxides tend to accumulate from the top B layer to the subsoil, and given that these oxides/hydroxide tend to dissolve at pH 4.2, accepting an $[Al^{3+}]$ of 0.2 meq/l would lead to substantial Al mobilization throughout the soil profile.
3. The organic complexation of Al adds to the overall mobilization of Al, and the organic component of this complex is also subject to decomposition, with the extent of decomposition depending on the nature of the complexing molecules (Arp and D'Avignon 1989). Hence, the state of the Al speciation in the soil solution depends on the stability of the Al complexing molecules.
4. Under forest soil conditions, most of the water-soluble Al fraction is, in fact, organically complexed. Typical total Al:BC ratios exceed 1 (UNITS) often by an order of magnitude in the organic forest floor, and in the top and mid portions of the soil (where most or all of the tree roots are located). The total Al:BC ratio in the subsoil, however, drops to values <0.1 or lower. The reason for this drop is the generally increasing soil pH with increasing soil depth. As a result, much of the water soluble Al precipitates into essentially inorganic Al oxide/hydroxide minerals; some may precipitate as jurbanite and/or alunite.
5. A considerable portion of soil Al exists on the cation exchange sites as exchangeable Al. The exchangeable portion is easily displaced into the soil solution by adding a neutral salt solution such as NaCl or NH4Cl.

Calculating the sustainable Al:BC ratio for the sustainable deposition calculations presents a further complication. Generally, this ratio can be calculated as Al leaching rate / base cation leaching rate for the steady-state sustainable deposition condition for which it is assumed that the mass balance for base cations is given by :

$$BC_{le} = BC_{dep} + Bc_w - BC_u \qquad \text{(A2.1)}$$

where BC_{le} : net bsae cation (K+Ca+Mg) leaching rate (eq ha^{-1} yr^{-1}) out of the soil
BC_{dep} : net base cation (K+Ca+Mg) deposition rate (eq ha^{-1} yr^{-1}) from the atmosphere
Bc_w : base cation (K+Ca+Mg) weathering rate (eq ha^{-1} yr^{-1}) of the soil
BC_u : base cation (K+Ca+Mg) uptake rate (eq ha^{-1} yr^{-1}) by the vegetation.

The Al leaching rate can then be calculated from the base cation leaching rate by using a generally acceptable multiplication factor. Defining that factor in order to ensure that the soils do not acidify further is however difficult. Soil base cations and soil acid cations participate in an ion exchange equilibrium that directly connects acid/base ratio in the soil solution with the acid/base ratio on the ion exchange complex of the soil, following the Gaines-Thomas exchange equilibrium model :

$$\frac{f_{Al}^{\ 2}}{f_{BC}^{\ 3}} = K_{exch} \frac{[Al]_{SD}^{2}}{[BC]_{SD}^{3}} \qquad \text{(A2.2)}$$

where f_{Al} : proportion of soil CEC occupied by Al;

f_{bc} : proportion of soil CEC occupied by base cations (soil base saturation);

K_{exch} : Gaines-Thomas selectivity coefficient (L mol^{-1});

$[Al]_{SD}$: inorganic aluminum concentration in soil solution at sustainable deposition (mol L^{-1})

$[BC]_{SD}$: base cations concentration in soil solution at sustainable deposition (mol L^{-1})

The SMB model being a one-layer soil model, [BC] in soil leachate corresponds to [BC] in the soil solution. A detailed case study for a tolerant hardwood site at Turkey Lakes (TL, Ontario; REF) has that cation leaching was closely related to SO_4^{2-} and NO_3^- leaching, as follows:

1. total cation leaching losses exceed total cation inputs by about 500 equivalents per ha per year; this difference plus additional base cation uptake by the forest vegetation would have to be matched by base cation weathering;
2. total anion leaching losses exceed total anion inputs by about 350 equivalents per ha per year; this number would likely be similar to the net total base cation losses if bicarbonate ions and organic anions were also part of the leachate analysis;
3. ion fluxes into the site and into the subsoil were generally correlated with one another, to some extent. For example, the atmospheric deposition of base cations was well correlated with S and N deposition, i.e.,

   BC dep = 0.30 * (SO_4^{2-} dep + NO_3^- dep) (r2 = 0.662)

   Base cation leaching was well correlated with SO_4^{2-} and NO_3^- deposition :

   C leaching = 1.81 * (SO_4^{2-} dep + NO_3^- dep) (r2= 0.573).

   BC leaching was also be related to acid cation deposition, i.e.,

   BC leaching = 2.2 * (H^+ dep + NH_4^+ dep) (r2 = 0.449).

   BC leaching was most correlated with SO_4^{2-} and NO_3^- leaching, i.e.,

   BC leaching = 1.07 * (SO_4^{2-} leaching + NO_3^- leach) (r2 = .882).

1. The above regression coefficients (0.30, 1.81,1.07, and 2.2) indicate that:

- only 30 % of wet SO_4^{2-} and NO_3^- deposition are accompanied with the atmospheric base cation deposition,
- factors other than SO_4^{2-} deposition + NO_3^- deposition contribute to base cation leaching,
- H^+ deposition + NH_4^+ deposition likely contribute to base cation leaching due to cation exchange reactions,

- BC leaching through the soil was essentially accompanied by a one-on-one equivalent leaching of SO_4^{2-} and NO_3^- ions.

For several soil profiles, including the soil profile for the Turkey Lakes study site, it was found that (Meyer and Arp, unpublished manuscript; Yanni 1994):

1. The equilibrium "constant" K_{exch} remains essentially independent of the resulting soil solution pH with increasing amounts of acidity added to the soil (at low acid additions, considerable portions of Al in solution are organically complexed).
2. The K_{exch} values increase from the top soils (<1 for the organic soil layers) to an average of 15 in subsoils, in relation to decreasing organic matter content.
3. Maintaining constant base saturation for each soil layer implies that acid cation leaching is directly proportional to base cation leaching.
4. Decreasing K_{exch} values imply that acid cation leaching must drop in relation to base cation leaching with increasing soil depth in order to maintain the base saturation level of each soil layer.

Further analysis of exchangeable base cations in soils reveals that base saturation generally increases with soil depth. On average, non-calcareous forest subsoils have base saturation levels at about 20 to 25%. At this level, the amount of tolerable acid cation leaching into, through and from subsoils amounts to 0.1 times the base cation leaching rate. For the upper soil layers, this factor is much higher.

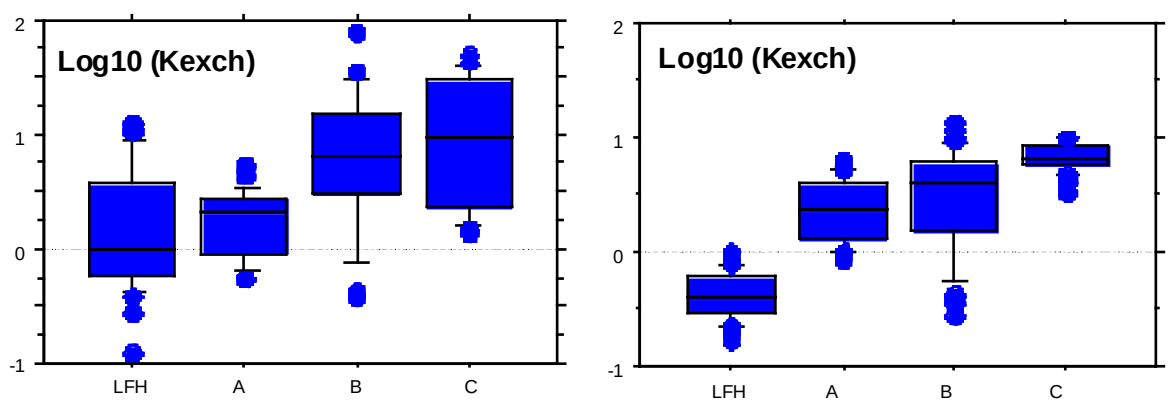

Figure A2.1. Box plots for the acid/base cation exchange constant in upland forest soils at Kejimkujik National Park (left), and select forest soils, one from Ontario and Quebec, and two from New Brunswick (right) ; LFH: organic surface layers, A: all A horizons, B: all midsoil or B layers; C: all subsoils, including transitional BC layers. For subsoils, Kexch is about 10.

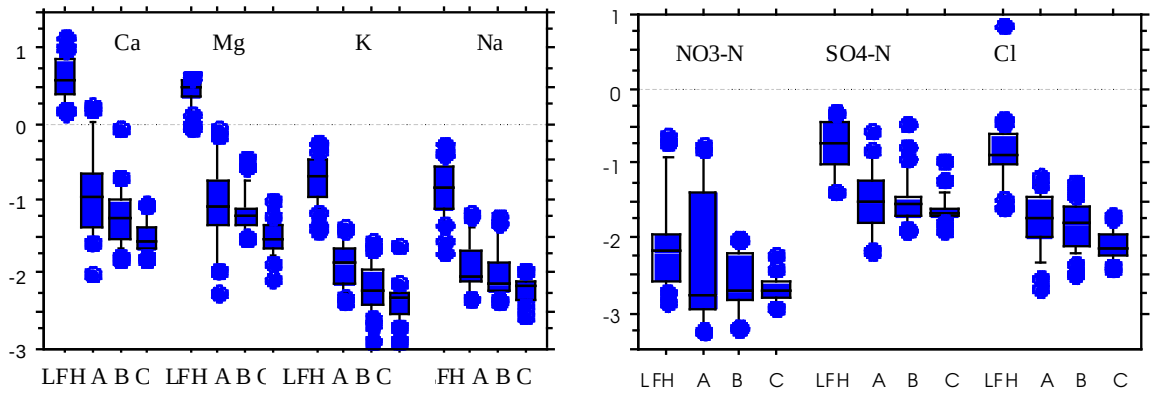

Figure A2.2. Water-soluble Ca, Mg K, Na, NO3-N, SO4-S and Cl ion concentrations in upland forest soils at Kejimkujik National Park, in relation to soil type; LFH: organic surface layers, A: all A horizons, B: all midsoil or B layers; C: all subsoils, including transitional BC layers. Note systematic drop of ion concentrations with increasing soil depth.

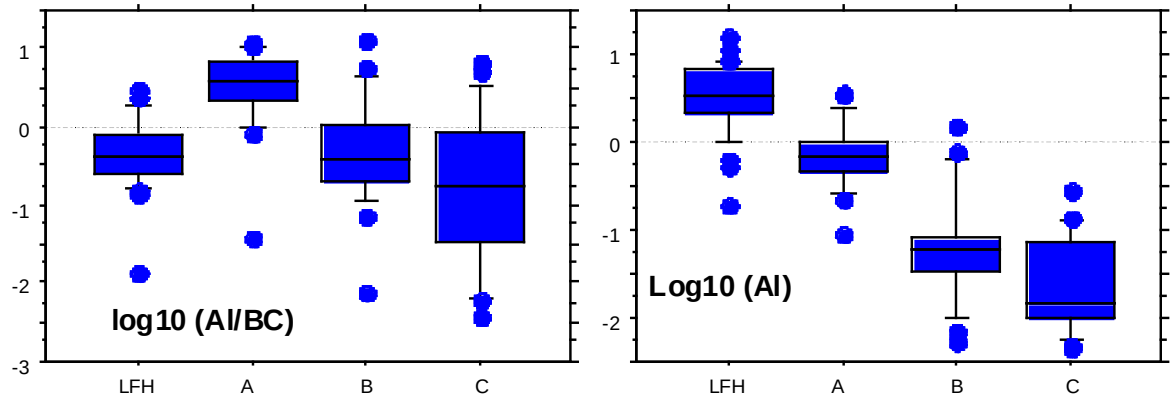

Figure A2.3. Al:BC ratio and water-soluble Al (total) in upland forest soils at Kejimkujik National Park (left), and select forest soils, one from Ontario and Quebec, and two from New Brunswick (right), by soil type; LFH: organic surface layers, A: all A horizons, B: all midsoil or B layers; C: all subsoils, including transitional BC layers. Note the drop of the Al:BC ratio with increasing depth in the mineral soil. In the C layer, Al:BC ratios are generally 0.1 or lower.

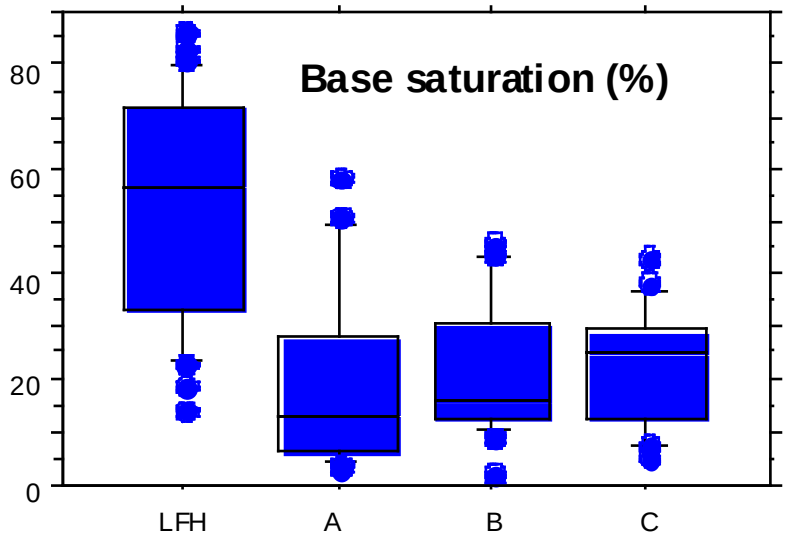

Figure A2.4. Base saturation of upland forest soils at Kejimkujik National Park, in relation to soil type; LFH: organic surface layers, A: all A horizons, B: all midsoil or B layers; C: all subsoils, including transitional BC layers. Note that subsoil base saturation are about 20% (+/-).

Based on these observations, we now conclude that:

1. Al increasingly exists in mineral form with increasing soil depth,
2. Al toxicity likely increases with soil depth
3. Al:BC ratios have little effect on plant growth in heavily rooted soil layers where there is ample supply of organic matter to complex with and render Al non-toxic
4. Existing data for Al:BC ratios in soil solutions provide an important means to assess the acid/base cation ion exchange status in all soil layers, and what is required to maintain same base saturation status
5. Determining these ratios is essential for quantifying Al and base cation fluxes into, through and from each soil layer
6. Base cations flowing into the subsoil from above constitute a loss of available base cations from the above-lying rooting space
7. This base cation loss is coupled with Al leaching as well.
8. The acid-base cation exchange equilibrium in the subsoil determines to what extent the incoming bases can be retained in close proximity to the rooting space as a potential nutrient reserve, or must get leached further, with Al taking their place on the ion exchange sites instead.
9. Maintaining the base saturation of the subsoil is a reasonable goal, and the related flux ratio of base cations versus acid cations into, through and from the subsoils is a good and readily quantified indicator of forest soil stability against all sources of soil acidification.

10. For the rooted soil layers, total Al/base cation ratios are high, and actual Al3+/base cation ratios are not easily determined, and are likely too low to produce toxic effects. What is important, however, is the potential loss of base cations from the rooting space, and that – at the same time – Al levels remain in the rooting space so that the subsoil and the seepage through that subsoil remains protected against high levels of acid cation leaching.

Determining the BC:Al ratio that maintains soil pH and base saturation

From the above acid-base equilibrium equation A2.2 , and the following law of mass conservation :

$$f_{Al} + f_{BC} = 1 \qquad \text{(A2.3)}$$

soil base saturation (f_{BC}) can be calculated with the following relationship ;

$$\frac{1 - f_{BC}}{f_{BC}^{3/2}} = \frac{\sqrt{K_{exch} / [BC]_{SD}}}{\left(BC/Al\right)_{SD}} \qquad \text{(A2.4)}$$

where $(BC/Al)_{SD}$ is the BC:Al ratio (M/M) in soil solution at sustainable deposition. For clarification, note that

$(BC/Al)_{SD}$ (M/M) = 1.5/(Al/BC) (eq/eq)

If $[BC]_{SD} = [Al]_{SD} \times \left(BC/Al\right)_{SD}$ (A2.5)

and $[Al] = K_{Gib} \times [H]^3 = K_{Gib} \times 10^{-3\,pH_{SD}}$ (A2.6)

where Kgib : Gibbsite dissolution constant (L mol^{-2})

$[H]$: Hydrogen ion concentration in soil solution (mol L^{-1}),

pH_{SD} : soil pH at sustainable deposition,

then, combining eq. A2.4, A2.5, and A2.6 gives :

$$\frac{1 - f_{BC}}{f_{BC}^{3/2}} = \frac{\sqrt{K_{exch}}}{\sqrt{K_{Gib} \times 10^{-3\,pH_{SD}}} \times \left(BC/Al\right)_{SD}^{3/2}} \qquad \text{(A2.7)}$$

Rearranging eq. A2.7 to isolate soil pH gives :

$$pH_{SD} = \frac{\left(3\log_{10}\left(BC/Al\right)_{SD} + \log_{10}\left(K_{Gib}\right) + 2\log_{10}\left(\frac{1-f_{BC}}{f_{BC}^{3/2}}\right) - \log_{10}\left(K_{exch}\right)\right)}{3} \quad \text{(A2.8)}$$

With this equation, we simulated the relationship between soil pH and base saturation for BC : Al ratio = 10, i.e. Al:BC (eq/eq) values of 0.15, K_{Gib} ($\log_{10}$ value = 8) and K_{exch} ($\log_{10}$ value = 1). As expected from equ. A2.8, the higher the soil pH at a given sustainable acid deposition value, the lower soil base saturation needs to be set to respect sustainable load (Figure A2.5). Of course, forest soils do not respect this theoretical relationship because of the arbitrary value we assigned to the BC:Al ratio (=10).

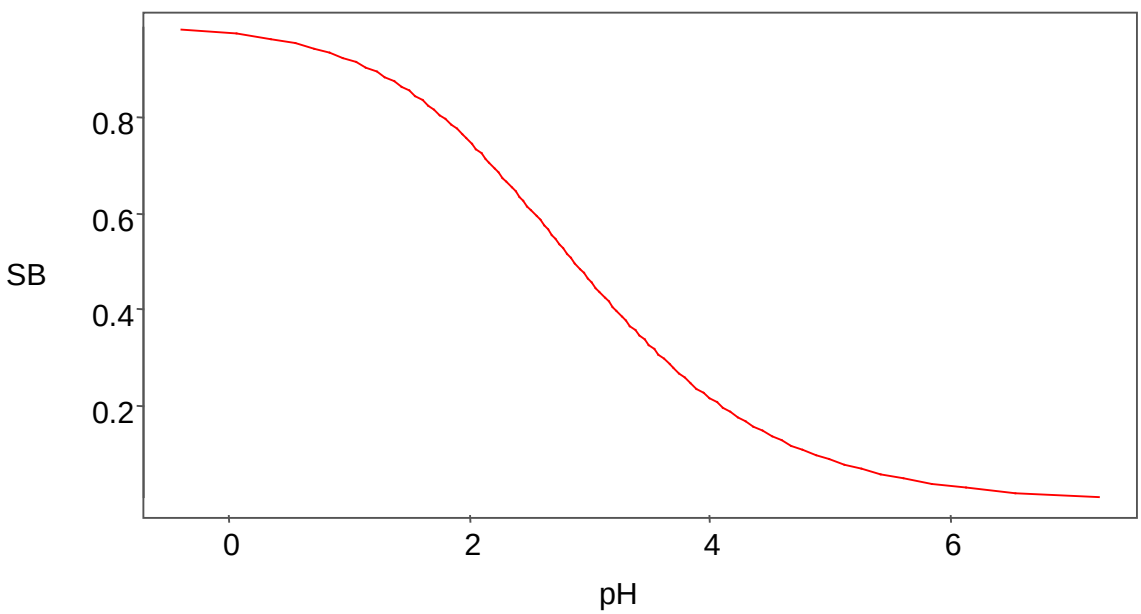

Figure A2.5. Theoretical relationship between soil pH and soil base saturation (SB) at sustainable acid deposition.

A way to model the relationship between soil pH and BS, one shall use equ. A2.2 in conjunction with A2.3, A2.6, and insuring charge neutrality of the soil solution, considering $[A^-] >> [H_2CO_3^-]$ (Bosatta 1993):

$$[Ca^{2+}] + [Al^{3+}] + [H^+] - [A^-] = 0 \quad \text{(A2.9)}$$

where $[A^-]$ represents all acid anions, including organic anions.

Assuming $[A^-]$ proportional to organic matter concentration in soils (i.e. $[A^-] = K_A*MO$), then soil BS is related to pH by the relationship (Bosatta 1993):

$$pH = (1/6)\log_{10}(K_{exch}) + (1/3)\log_{10}(K_{Gig}) + (1/2)\log_{10}(f_{BC}) - (1/3)\log_{10}(1 - f_{BC}) - (1/2)\log_{10}(K_A * MO)$$

(A2.10)

Data from 516 organic soil horizons and 2154 mineral soil horizons from the Québec province Forest Ecosystem Monitoring Network (RESEF) were used to adjust the K_A parameter. The following values were also used: 10^1 for K_{exch}, and 10^8 for K_{Gib}, The results indicated that K_A =0.11 and 0.0014 for organic and mineral soils, respectively. Figure A2.6 shows the fitted relationship between soil pH and base saturation for organic and mineral soils (considering their average OM contents of 72 and 3.7 %, respectively).

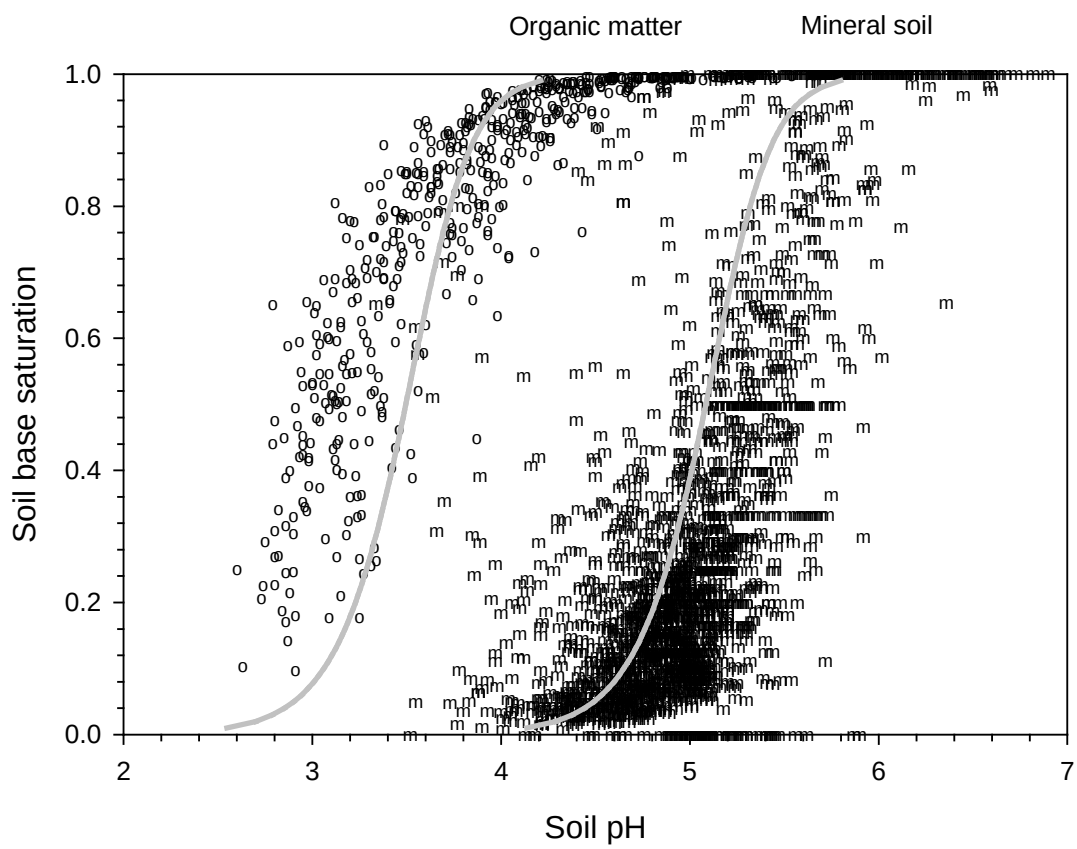

Figure A2.6. Relationship between soil pH and base saturation according to eq. A2.10. Overlaid are observations for forest organic and mineral soil horizons in Québec.

From this relationship and equ. A2.6 and A2.2, relationships between soil solution BC:Al ratio and soil pH and soil base saturation can be derived (Figure A 2.7). The model indicated that for a soil solution BC:Al ratio = 10, soil pH and base saturation of organic soils would be 3.5 and 45 %, respectively. For mineral soils, soil pH and base saturation would be 4.5 and 5.5 %, respectively. From this modelling exercise, it appears that setting long term soil solution BC:Al ratio to 10 would represent a minimum for sustaining soil pH and base saturation, at least for mineral soils.

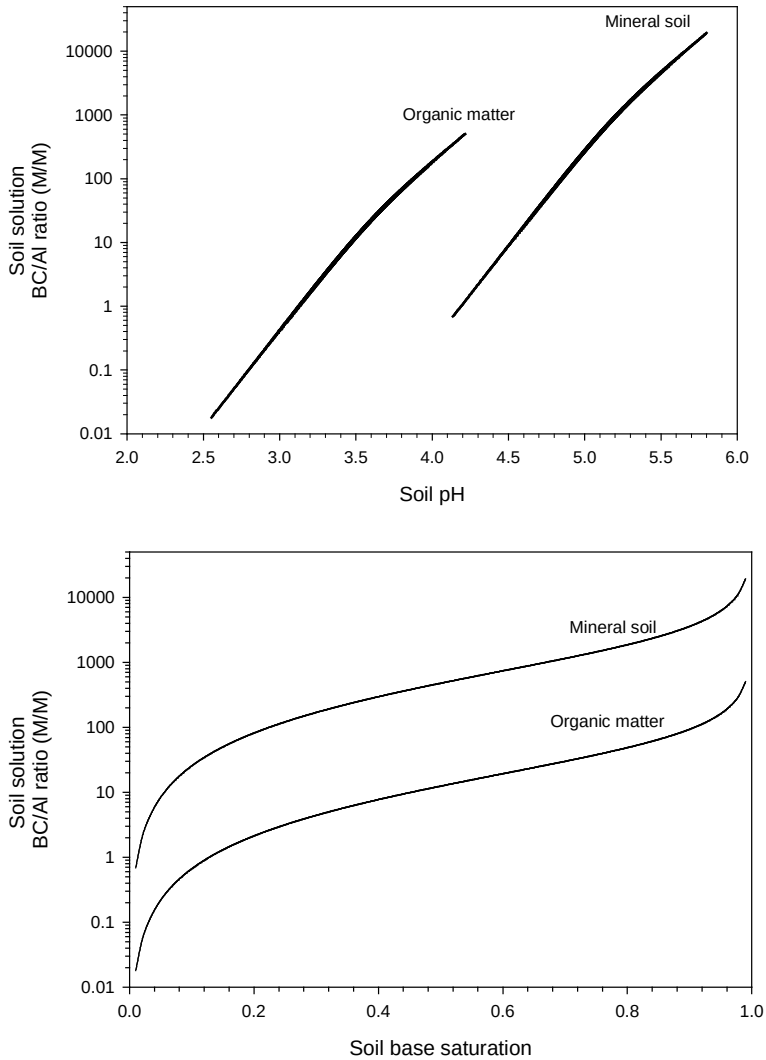

Figure A2.7. Modelled relationships between soil pH and base saturation and the soil solution BC/Al ratio for organic (72 % OM) and mineral (3.7 % OM) soil horizons. Values used were 10^1 for K_{exch}, and 10^8 for K_{Gib}.

Calculation of the sustainable acid neutralizing capacity (ANC) leaching rate

The maximum sustainable sulphur deposition rate to maintain the current soil base cation status is given by

$$SD_{max}(S) = BC_{dep} + BC_w - BC_u - ANC_{le}(SD) \quad \text{(A2.9)}$$

where BC_{dep} is the sum of Ca + Mg + K + Na + Cl deposition rate (eq ha^{-1} yr^{-1}),

BC_w is the soil weathering rate of Ca + Mg + K + Na (eq ha^{-1} yr^{-1}),

BC_u is the net Ca + Mg + K uptake rate (eq ha^{-1} yr^{-1}) ultimately removed by harvest or disturbance, and

$ANC_{le}(SD)$ is the sustainable acid neutralizing capacity (ANC) leaching rate (eq ha^{-1} yr^{-1}).

$$ANC_{le}(SD) = -Al_{le\,(SD)} - H_{le\,(SD)} \quad \text{(A2.10)}$$

where $Al_{le(SD)}$: aluminum leaching rate (eq ha^{-1} yr^{-1}) at sustainable deposition

$H_{le(SD)}$: hydrogen ion leaching rate (eq ha^{-1} yr^{-1}) at sustainable deposition.

As $$Al_{le(SD)} = BC_{le(SD)} \times \left(Al/BC \right) \text{ (eq/eq)} \quad \text{(A2.11)}$$

and $$H_{le(SD)} = Q^{2/3} \times \left(Al_{le(SD)} / K_{Gib} \right)^{1/3}, \quad \text{(A2.12)}$$

combining eq. A2.1, A2.9, A2.10, A2.11, and A2.12 gives the maximum sustainable sulphur deposition rate as a function of the Al:BC ratio and K_{Gib} values that are selected to maintain soil base saturation (f_{BC}) and soil pH (from eq. A2.8 and fig. A2.5) :

$$SD_{max}(S) = BC_{dep} + BC_w - BC_u + \left(\left(BC_{dep} + Bc_w - BC_u \right) \times \left(Al/BC \right) \right) + Q^{2/3} \left(\left(BC_{dep} + Bc_w - BC_u \right) \times \left(Al/BC \right) / K_{Gib} \right)^{1/3} \quad \text{(A2.13)}$$

Reference

Bosatta, E. 1993. One equation to analyse pH of soil systems. J. Soil Sci. 44: 299-306.

Appendix 3: Estimating soil weathering rates

E. Miller

Estimation of the rate of release of base cations from the weathering of primary minerals is at the center of sustainable acidic deposition calculations. Methods for field measurement and modeling of the primary mineral weathering rate have been the subject of research for several decades, but as of yet, no clear consensus has emerged on a definitive measurement or modeling methodology. Thus from the outset, we must accept that there will be considerable uncertainty associated with our estimates of base cation supply via mineral weathering. Our goal is to quantify the uncertainty of our estimates to the greatest extent possible and to select a method of calculation that will produce logically consistent estimates across the study region. Such an approach will yield a good first approximation of forest sensitivity to acidic deposition at any one site and an even better comparative evaluation of the relative sensitivity of different parts of the landscape.

A3.1. Sensitivity of forest sustainability index to the primary mineral weathering rate

Our conceptual model of a forest sustainability index with respect to the deposition of acidifying substances (S and N) is based on a steady-state mass balance (SMB) of ANC (acid neutralizing) and BNC (base neutralizing capacity) in forest soils. The sustainability index is expressed as the amount of BNC that can be tolerated by the system without depleting the soil of plant available (exchangeable) Ca, Mg, K, and Na and maintaining at least the current production of ANC (this follows from the stipulation that *at least* the current base saturation of the soil should be maintained). The steady-state mass balance of ANC and BNC in acid forest soils can be written as follows:

Sustainability Index (keq/ha/y) =

BC atmospheric deposition (Ca^{2+} + Mg^{2+} + K^{+} + Na^{+})
+ BC weathering (Ca^{2+} + Mg^{2+} + K^{+} + Na^{+})
- BC uptake to vegetation that is ultimately removed from the site by harvest
- ANC reserve (ANC leaching with present soil base saturation)
- Cl^{-} atmospheric deposition
- OA^{-} leaching (organic anions associated with DOC leaching)
- NO_3^{-} leaching

where BC refers to the base cations (Ca, Mg, K, and Na) and OA^{-} refers to anionic charge associated with DOC (organic acids). Calculated in this way, the sustainability index

corresponds to the maximum level of S and N deposition that can be tolerated by a forest without further decline of soil exchangeable base cation pools, after the effects of N deposition have been assessed with respect to the potential for nitrate leaching.

An example of the calculation for a typical mixed northern hardwood forest of the region (data for Huntington Forest, NY from Johnson and Lindberg 1992) is as follows:

BC deposition	**=**	**0.196 keq/ha/y**
BC weathering	**=**	**unknown**
BC harvest	**=**	**0.542 keq/ha/y (assuming a 100 y rotation, WTH)**
ANC reserve	**=**	**0.020 keq/ha/y**
Cl- deposition	**=**	**0.044 keq/ha/y**
OA- leaching	**=**	**0.000 keq/ha/y**
NO3- leaching	**=**	**0.071 keq/ha/y**

The sustainability index exclusive of weathering equals -0.437 keq/ha/y. This example illustrates the following points:

1. Even with modest rotation lengths, BC extraction due to harvest is an important determinant of forest sensitivity to acidic deposition. In the absence of any acidic deposition, this forest requires a weathering rate of 0.437 keq/ha/y to sustain a 100-y rotation of whole tree harvesting[1]. In the absence of sulfur deposition, selective harvest or longer rotations could be used to achieve sustainability if the weathering rate were less than 0.437 keq/ha/y.
2. The requirement of an ANC leaching reserve, while important to our definition of forest sustainability, does not contribute significantly to the index.
3. Uncertainty in measurements or estimates of the BC deposition, Cl- deposition, and OA- leaching are likely to have a minor impact on the aggregate uncertainty of the index.
4. Uncertainty in NO_3^- leaching could be significant (see Appendix 4).
5. Considering the above points, BC release from mineral weathering is the primary determinant of the ability of a forest to sustain a given forest management regime under a given level of acidic deposition.

A3.2. Alternative approaches to estimating primary mineral weathering rates under field conditions

[1] Our formal analysis considers greater complexity than this simple example. In particular, a weathering rate greater than 0.437 keq/ha/y would actually be required because harvest selectively removes predominantly Ca, and some of the weathering total is accounted for by Na.

Several approaches have been employed to estimate the field weathering rates of primary minerals (Langan et al. 1996, Kolka et al. 1996). Unfortunately, few of the existing field measurement approaches appear to be appropriate for our intended application as described below. *One of the key constraints imposed by our analysis is that we need to determine the weathering rate in the soil to the maximum depth of root penetration.* This is the primary part of the soil that can contribute BC to plant growth and, via biocycling, maintain the plant available supply of nutrients on the soil exchange complex in upland soils. As mentioned in the protocol body, lowland areas receiving drainage water from upslope, as well as upland areas where soil water upwells through the mineral soil in response to local topographic and hydrologic conditions, should be well protected against nutrient cation depletion. In this analysis we are not concerned with these wetland and seep environments subject to fluctuating water tables. We ignore segments of the landscape in which the root-zone exchange complex is frequently resupplied by hydrologic means, because they are generally well protected with respect to acidic deposition. Our analysis focuses instead on the well-drained portions of upland landscapes that are more likely to be sensitive to acid deposition. We recognize that upland landscapes will include a mixture of well-drained and seep environments varying at the scale of meters. Therefore, our analysis provides an assessment of the condition of the well-drained portions of upland landscapes. Within the areas we identify as potentially sensitive to acid deposition, there will likely be microenvironments that are not sensitive due to local topographic and hydrologic conditions.

Most approaches to weathering rate estimation consider chemical reactions over the entire soil profile and/or geologic materials in the ground water zone, thus providing estimates of mineral weathering which are not strictly applicable to the rooting zone. Some possible approaches to weathering rate estimation, and their limitations for our application, are described below.

A3.2.1. Various forms of solute mass-balance computations using stream water chemistry

Stream water chemistry generally reflects some contribution from deep till and bedrock weathering. By limiting our estimate of mineral weathering to the rooting zone, we rule out any weathering rate estimates based on stream water measurements or watershed mass balance approaches. As discussed below, trace calcite and biotite in deep till and bedrock are very important to base cation release rates as measured in stream water. However, all calcite is expected to be depleted from the rooting zone except in soils developed on highly calcareous materials. Much or all of the biotite is expected to have been removed by prior weathering from the rooting zones of the majority of soils in the study region. Additionally, solute mass-balance calculations of field weathering rates suffer from potential problems due to the short-term dynamic release of base cations from soil exchange sites often being misinterpreted as a mineral

weathering flux. Few systems subject to acidic deposition, natural disturbance, or forest management are likely to be at steady state with respect to inputs and exports from the soil exchangeable cation pool. Because of these complications, watershed mass balance studies do not generally provide direct information on the weathering of primary minerals in the rooting zone of the soil (Bouchard and Jolicoeur 2000). Fitting a weathering rate using the MAGIC model and stream water chemistry will also not provide an estimate directly applicable to the rooting zone.

A3.2.2. Natural Cl or Na as tracer

Base cation release by soil mineral weathering can also be assessed by determining the change in the BC:Na or BC:Cl ratio in water flowing from the atmosphere through the soil. Likens et al. (1996) used the change in the Ca:Na ratio in soil and water flowing through the soil to infer mineral weathering rates. Chloride has also been used as a tracer in a similar way (Lockwood et al. 1995). One advantage of these tracer methods is that they use only annual atmospheric deposition and soil water leaching analyses for the estimation of the weathering rate. These methods will be applicable to the rooting zone when lysimeter measurements of Cl or Na flux are obtained from lysimeters at the base of the rooting zone.

A3.2.3. Sr isotope mass-balance

The strontium isotope mass-balance method (Aberg et al. 1989, Miller et al. 1993) is generally too time consuming and expensive to apply. The approach is also severely complicated by multiple mineral end members with varying strontium isotopic composition, different weathering rates, and varying BC:Sr ratios (Clow et al. 1997) in most geologic settings. As with traditional solute mass-balance methods, the approach is not applicable to our focus on the rooting zone unless lysimeter measurements of soil solution Sr and BC fluxes are available.

A3.2.4. Correlation analysis and parent material tables

Although statistical correlations or tables relating weathering rates to basic mineralogy descriptions were employed in early European work on critical loads of acidic deposition (de Vries method), more refined methods have since been developed and adopted in Europe (Posch et al. 1995). While correlations and tables might be obvious choices for use as a first approximation in North America, the methods have been criticized by some as arbitrary (Hodsen and Langan 1999). They are also not directly applicable to our analysis because the site-specific weathering rates on which the correlations have been established or from which the tables have been prepared frequently used weathering rate measurements based on watershed mass balance. Thus the base cation release rates estimated by these approaches would be inappropriate for our focus on the rooting zone.

A3.2.5. Element depletion analysis

Element depletion analysis (EDA) involves the calculation of BC lost due to primary mineral weathering over long periods of time (10^4 to 10^6 y) by determining the ratio of BC to a very stable mineral (or element contained in such a mineral, e.g. Ti, Zr) in unweathered parent material and in the weathered region of a soil profile (April et al. 1986, Kirkwood and Nesbitt 1991, Jersak et al. 1995). Depletion of BC in the surface horizons (estimated from till bulk density and the BC:Zr ratio for example) divided by the age of the landscape yields the depth-averaged and time-averaged weathering release of BC over the age of the landscape. For our study region, landscape ages on upland sites (flood plains are excluded for the hydrologic reasons given above) range from ~10,000 to ~14,000 years and are primarily determined by the final retreat of continental glaciers. The difficulty with interpreting EDA in terms of the present day BC release rate is that weathering rates in glaciated terrain are known to decrease with age of the surface (White et al. 1996, Taylor and Blum 1995, Bain et al. 1993). Simply stated, the weathering rate provided by EDA is an average of relatively higher weathering rates immediately following glacial recession and relatively lower rates under present conditions. It is not completely clear how to generalize the results of studies such as Taylor and Blum (1995) and White et al. (1996) in order to estimate the modern-day BC release rates from the time-averaged value obtained via EDA. However, we can use EDA-based weathering rates as a conservative upper limit to the weathering rate at a given site, if we are careful to limit the calculations to the depth of rooting. While EDA is a viable approach to obtaining weathering information for a specific site, there is no direct way to generalize estimates obtained by this method to the entire study region.

A3.2.6. Modeling field weathering rates using laboratory derived kinetics

The most promising approach for a logically consistent estimation of the present-day weathering rate over broad regions is the application of models that predict the weathering rate from first principles given detailed measurements of the soil environment and laboratory-derived rate constants for specific mineral weathering reactions. Despite the advantages of this approach, there are many limitations including the volume of detailed information required as model input, uncertainty in the laboratory-derived rate constants, and uncertainty in relating the reactive mineral surface area of the laboratory experiments to that in the field. Furthermore, we have already pointed out how difficult (if not impossible) it is to accurately measure the present day weathering rate in the rooting zone, therefore, it is nearly impossible to properly validate a kinetic model against field measurements. Nevertheless, kinetic models have had considerable success in producing weathering rate estimates within the range of field measurements at a number of sites (Sverdrup et al. 1998, Kolka et al. 1996). Although some researchers have

suggested that this agreement is fortuitous (Hodsen et al. 1997) or have demonstrated that site-specific information input to such models is non-unique for a given weathering rate at a specific site (Zak and Beven 1999), these concerns can be adequately addressed by judicious model parameterization and careful interpretation of model results (Malmström et al. 2000). Even with their limitations, kinetic models provide important insight into the key soil and landscape properties that control regional variation in the weathering rate.

A3.3. Selecting the most appropriate method for site-specific analyses

Selection of the most appropriate method for estimation of the weathering rate must be made on a site-by-site basis. Each site will present different conditions that either prevent or allow a given method to be used to estimate the base cation release within the active rooting zone. Consideration should be given to several factors:

- depth of rooting
- depth of the weathering front
- depth of soil / surfical deposits
- surficial and bedrock geology
- size of the plot or watershed
- spatial variability in soil and soil water measurements
- temporal variability in atmospheric deposition, soil water, and stream water measurements

The discussion in the preceding sections makes clear that the EDA and kinetic modeling approaches will be the most widely applicable methods. However, certain specific sites may present the right conditions for successful application of one or more of the other methods. Reference should be made to the works cited in this appendix for detailed instructions on application of each of the methods discussed.

A3.4. First-order estimates of root-zone weathering rates

By combining the strengths of several of the approaches described above we construct a process for weathering rate estimation in the ecological unit mapping of the ECP/NEG FMI study area which is logically consistent across study sites and which allows for assessment of uncertainty. As noted previously, kinetic models provide important insight into the soil and landscape properties that control the weathering rate. Below we employ an extensively evaluated steady-state kinetic model of primary mineral weathering, PROFILE (Sverdrup and Warfvinge 1993), to illustrate the key soil and landscape variables responsible for regional variations in weathering rates. Based on considerations of weathering rate sensitivity to specific input parameters, as well as our ability to quantify those parameters in a uniform way across the study region, we develop a procedure to constrain the range of likely weathering rates at a given location on the landscape.

A3.4.1 The dominant influence of parent material mineralogy

The first-order control of BC release due to weathering is the constituent mineralogy of the soil parent material. This is based on the well understood orders of magnitude variation in dissolution rates of primary minerals (Table A3.1, see also White et al. 1996). Recent research indicates that it is less important to know the detailed mineralogy of soil than it is to know of the existence and abundance of several highly reactive phases (White et al. 1999, Blum et al. 1998, Clow et al. 1997, Bain et al. 1995). The BC release rate will be controlled by calcite, biotite, hornblende, apatite, and plagioclase in approximately that order. Several field studies have demonstrated that even trace amounts of calcite present in igneous rocks will dominate the BC release rate until this phase is exhausted (White et al. 1999). Other studies have shown that biotite is also highly reactive and rapidly depleted from the soil following glaciation (Blum et al. 1994, Taylor and Blum 1995, White et al. 1996). April et al. (1986) described how variation in heavy mineral content (particularly hornblende or other Ca-bearing heavy minerals) significantly influences BC release rate.

In soils derived from sedimentary rocks or local tills composed of sedimentary or meta-sedimentary rocks, the key determinant of the BC release rate in the rooting zone will be the extent of calcite in these materials (Bain et al. 1995). Trace calcite cements could possibly be exhausted, but massive calcite vein fillings or matrix could possibly still be present in the rooting zone. These types of soils can be broken into two classes: Class I – calcite-rich, sedimentary (high BC release rates); Class II – calcite-free, sedimentary (very low BC release rates). Class I would also include clay rich, calcareous marine or lacustrine sediments such as are present in the St. Lawrence and Champlain Valleys.

In soils dominated by till derived from granitic or metamorphic rocks (excluding calcareous meta-sediments) and greater than 10,000 y old, it is a reasonable expectation that all calcite and biotite has been exhausted from the rooting zone (Taylor and Blum 1995, White et al. 1996). In such soils, the weathering release of BC in the rooting zone will depend primarily on the heavy mineral and plagioclase content (and the anorthite content of the plagioclase). We will refer to these soils as class III soils (low to moderate BC release rates).

Table A3.1 Mean lifetime of a 1 mm crystal of common primary minerals at 25°C and pH 5.0. From compilation by Lasaga et al. (1994).

Mineral	Mean Lifetime of a 1 mm crystal (years)
Anorthite	112

Nepheline	211
Diopside	6,800
Enstatite	10,100
Gibbsite	276,000
Sanidine	291,000
Albite	575,000
Microcline	921,000
Epidote	923,000
Muscovite	2,600,000
Kaolinite	6,000,000
Quartz	34,000,000

Soils formed on mixtures of class I, II and III type tills are designated as class IV, and their weathering rate depends on the proportion of calcite in the mixture, but will generally provide moderate to high BC release rates.

Thus we begin our procedure with classification of parent materials as described above (Table A3.2). Classification of parent material at a specific site is accomplished by mineralogical analysis and/or chemical analysis with normative calculations to reveal the mineralogy. Classification of soils and parent materials for the regional mapping project relies on the combination of available spatial data. Where mineralogy or chemistry is available from soil survey information, it can be employed in a similar fashion to the site-specific data. Where this information is lacking, a soil parent material classification can be derived using the till-source area concepts of Bailey and others (Bailey and Hornbeck 1992, Hornbeck et al. 1997). Briefly, till mineralogy is estimated by distance-area weighted means of bedrock mineralogy within a 60° arc of 32 km radius in the upstream (glacial flow) direction. Bedrock mineralogy and glacial motions are determined from maps provided by the geologic survey. This procedure has been demonstrated to produce reliable estimates of till composition and chemistry at a test area in New Hampshire, USA (Bailey and Hornbeck 1992).

For a first-order estimate of the weathering rate, it is reasonable to determine the percent of key reactive mineral phases present or to assign a representative mineralogy based on the soil parent material classes identified above. The inherent weathering rate predicted by a kinetic model for each parent material class can then be adjusted for landscape and climatic factors also known to regulate base cation release rates.

Table A3.2 Classification of soil parent materials by key primary mineral phases and representative relative ranges of base cation release rates due to weathering. The relative base cation release rates were calculated with PROFILE (Sverdrup and Warfvinge 1993) using the following environmental conditions: soil depth = 50 cm, temperature = 12°C, soil moisture content = 0.2 m^3 m^{-3}, bulk density = 1000 kg m^{-3}, reactive surface area = 1.5×10^6 m^2 g^{-1}.

CLASS	Key Primary Minerals	Weathering Rate Range (keq ha^{-1} y^{-1})	Basis of Range	Representative Soil Series
I	calcite (>1%)	0.860 - 7.88	1 - 10% calcite	
II	chlorite, vermiculite, garnet	0.311	2.8%, 17.7%, 5.1%	Houghtonville
III	hornblende, plagioclase	0.380 - 0.634	0.5 - 6% hornblende	Adams/Becket
IV	calcite, hornblende, plagioclase	0.311 - >5.0	class I - III	

A3.4.2. Influence of landscape and climatic factors

The effects of several landscape and climate factors on the weathering rate are well constrained. Weathering kinetics scale with temperature according to an Arrhenius law (Sverdrup and Warfvinge 1993, Lasaga et al. 1994, Malmström et al. 2000). Over the range of average soil temperatures expected for the study region, weathering rates for the same parent material can be expected to vary by a factor of two (Figure A3.1). For much of the study area, the mean annual soil temperature is known to be equivalent to the mean annual air temperature and can be established with reasonable certainty (±5%) for any position on the landscape. In northern Canada, this assumption may not be correct, but a model to calculate soil temperature from air temperature is available (Yin and Arp 1993). Weathering rates are also affected by water availability as reflected in the degree of saturation of the soil. Weathering reactions can only occur if the mineral surface is wet with an adequately thick film of water to allow free progress of the reaction. Water flow through the soil is required to supply reactants and remove reaction products. The degree of soil moisture content and water flow are functions of both climate (precipitation, evaporation and the time between precipitation events) and soil (water retention capacity, hydraulic conductivity). Over most of the study region, the range of actual evapotranspiration rates and frequency of precipitation are such that soil water conditions are frequently near field capacity and infrequently near the permanent wilting point. This observation is used to constrain the average soil moisture content based on field capacity and permanent wilting point for a given soil (see below).

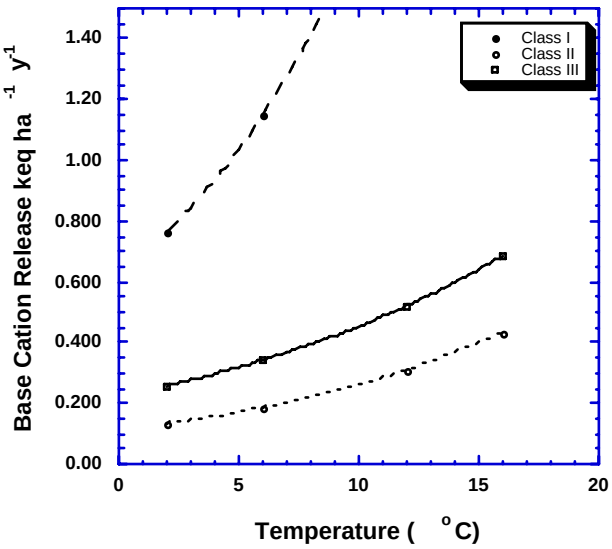

Figure A3.1 Influence of soil temperature on base cation release during weathering of class I, II, and III soils. Reference conditions: bulk density = 1000 kg m^{-3}; soil moisture content = 0.2 m^3 m^{-3} , reactive surface area = $1.5x10^6$ m^2 g^{-1}.

A3.4.3 Influence of local soil properties (bulk density, texture)

Field capacity and permanent wilting point can be estimated from soil texture (Dunne and Leopold 1978, Dingman 1994). For soils derived from class III and IV parent materials we expect textures of fine sandy loam or coarser. This limits the range of possible water contents to 0.05 to 0.25 (see Figure 6-4 in Dingman 1994). On an annual average basis we would expect moisture contents to range from 0.1 for fine sand to 0.2 for a loam. This range of average water contents could potentially produce a ±50% difference in base cation release rates (Figure A3.2). Field capacity water content could also be scaled by an index such as actual evapotranspiration:potential evapotranspiration to include climate related variance in the soil moisture estimate.

Bulk density also affects the weathering rate through its effect on pore space volume and mineral mass per volume of soil. The effect is linear and only about ±10% over the range of bulk densities expected for the rooting zone (Figure A3.3). Textural class and bulk density can be obtained from soil survey data and used to reduce the aggregate uncertainty due to these parameters to perhaps as low as ±10% for a given soil.

For class I soils, weathering rates are inherently high due to calcite. Even if moisture content is at the lower limit for these soils (~0.2), a very low forest sensitivity to acidic deposition will still be calculated. Therefore, there is less need to be concerned with the accuracy of soil moisture estimates for this soil class. Forests occupying class I soils will generally be well protected against acidic deposition.

For class II soils, inherent weathering rates are so low, that even at the upper end of expected water content of these soils (~0.25), we will likely calculate high sensitivities to acidic deposition. Texture and bulk density data from the soil survey will reasonably constrain the uncertainty.

A3.4.4. Reactive Surface Area

Perhaps the most difficult parameter to determine for the application of a kinetic model is the extent of mineral surface area that is available to participate in mineral weathering reactions (Malmström et al. 2000). Measurements of BET-derived surface area are time consuming and moderately expensive (White et al. 1996, Hodsen et al. 1998). They will not be available except for a small number of sites. There are published relationships of surface area with soil texture (Sverdrup and Warfvinge 1993, Hodsen et al. 1998) that can be used to estimate total mineral surface area.

The total surface area (either BET or texture based) must then be apportioned into reactive and non-reactive surface area attributable to each mineral phase. Non-reactive surface area (in terms of primary mineral weathering) includes secondary (clay and amorphous) mineral phases that are not undergoing dissolution in the present

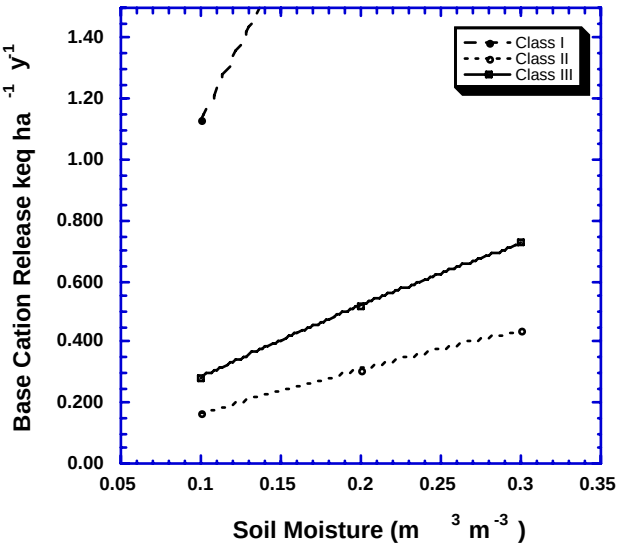

Figure A3.2 Influence of soil moisture content on base cation release during weathering of class I, II, and III soils. Reference conditions: temperature = 12 ^{o}C, bulk density = 1000 kg m^{-3}, reactive surface area = 1.5x10^{6} m^{2} g^{-1}.

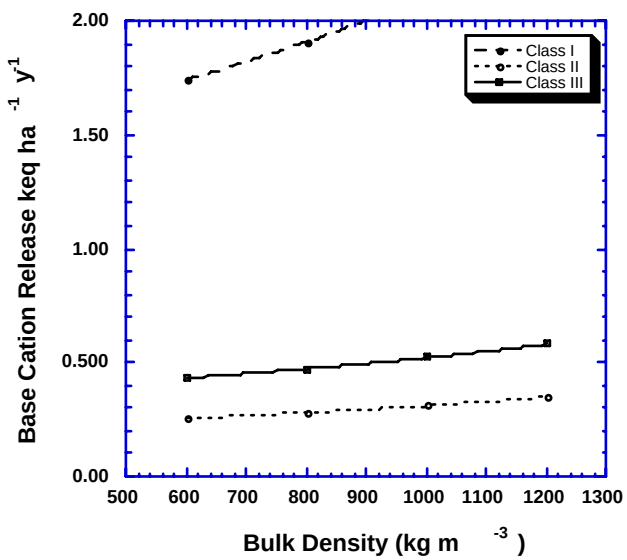

Figure A3.3 Influence of bulk density on base cation release during weathering of class I, II, and III soils. Reference conditions: temperature = 12 ^{o}C, soil moisture content = 0.2 m^{3} m^{-3}, reactive surface area = 1.5x10^{6} m^{2} g^{-1}.

environment (or the kinetics of which are so slow as to be insignificant). Amorphous minerals form coatings on the surfaces of primary minerals, further reducing reactive surface area (White et al. 1996). Because we need to estimate the reactive surface area of ***primary*** minerals within the active rooting zone, we must take care to correct for the influence of weathering products (secondary minerals) on the texture measured within the rooting zone. The clay size fraction is much larger in B horizons of soils due to the formation and accumulation of secondary clays and amorphous minerals than in the parent material or C horizon. The best estimate for the texture of the residual primary mineral constituents of the rooting zone will be the texture of the C horizon. Even though the C horizon will not contain many authegenic minerals, it may contain a greater proportion of small primary mineral grains than currently exist in the more highly weathered upper soil layers. For this reason, we will adjust C horizon clay contents to reflect the likelihood that small primary mineral grains have been lost from the rooting zone due to previous mineral weathering:

1. For class I parent material, reactive area may be relatively high, but even if we underestimate it, the estimated weathering rate will remain very high due to calcite. In class I we can safely assume that only 50% of clay size particles are reactive.
2. For class II, we estimate that 100% of clay size particles are reactive, and clay minerals such as chlorite will be important contributors to the very low weathering rates in these soils.
3. In class III and IV soils, we expect essentially none of the clay size fraction to be reactive. Fine particle sizes produced by glacial abrasion are rapidly lost during post-glaciation (Taylor and Blum 1995). Most of the clay size particles currently present in the rooting zone of these soils will be stable secondary phases (White et al. 1996). To be somewhat conservative, we will allow 10% of the clay size fraction to be reactive.

Operationally, we adjust the observed C-horizon texture data by the amounts discussed above, and use Sverdrup and Warfvinge's (1993) surface area – texture correlations to predict reactive surface area. As these adjustments are arbitrary (even if defensible), we will conduct a sensitivity analysis with respect to surface area to constrain the possible range of weathering rates. Model estimates of base cation release scale linearly with reactive surface area. The scaling is slightly less than 1:1. Therefore our estimates of base cation release will be very sensitive to this parameter (Figure A3.4). It must be emphasized that studies show that substantially less than 100% of the BET surface area is reactive (White et al. 1996) and it is critical to take this factor into account.

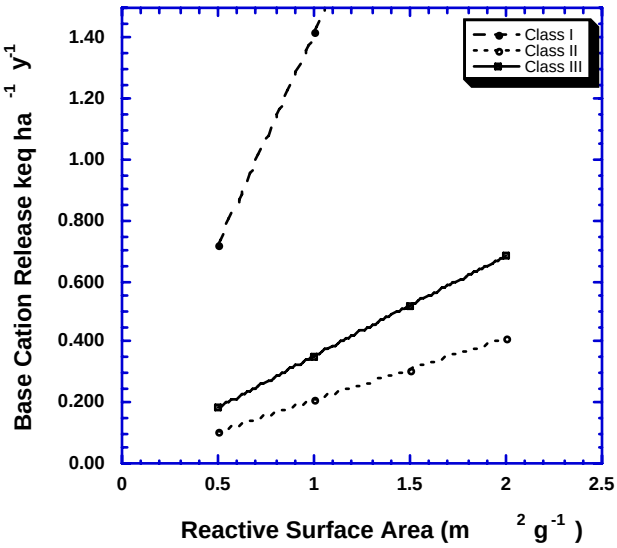

Figure A3.4 Influence of reactive surface area on base cation release during weathering of class I, II, and III soils. Reference conditions: temperature = 12 ^{o}C, bulk density = 1000 kg m^{-3}, soil moisture content = 0.2 m^{3} m^{-3}.

A3.5 Summary of Required Information

Our process for estimating the regional variation in root-zone weathering rates requires estimates of the following parameters:

1. Presence and/or abundance of key reactive mineral phases
2. Depth of the active rooting zone
3. Depth of soil and surfical deposits
4. Mean annual soil temperature
5. Potential and actual evapotranspiration
6. Soil Texture and bulk density.

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Appendix 4: Net soil N accumulation

L. Pardo, E. Miller

Determining net soil N accumulation provides a challenge because of the large spatial heterogeneity in soil properties and the long-lasting influence that disturbance can have on those properties. Because the total soil N pool is large, even a small change in the soil C:N ratio could cause a significant change in the total soil N pool. There is, however, little quantitative information about such changes or about the stability of N in humus (Gundersen 1992). In considering the net soil N accumulation rate, the timeframe for the accumulation is significant: the current, actual rate of soil N accumulation could be significantly higher than the long-term rate of accumulation over several rotations. The purpose of this appendix is to provide background on soil nitrogen accumulation in forest ecosystems. First, we define soil N accumulation, second we present previous thinking and methods of determining acceptable soil N accumulation, third we discuss the way N accumulation is handled is the SMB method, and finally, we present complicating factors and other potential approaches.

A4.1. Definition of soil N accumulation

The flux N_a (Eq. 3) refers to the acceptable, annual net accumulation rate of N in the soil that is sustainable over the forest rotation. Forests are capable of accumulating N in soil at much higher rates than are used here, however, forests can maintain those high rates of N accumulation only until they reach N saturation, at which point elevated nitrification leads to elevated nitrate leaching (Aber et al. 1989). Thus, the acceptable net soil N accumulation rate is intended to restrict nitrate leaching levels over the forest rotation to acceptable levels. Typically, N accumulation and N loss vary with stand age, land-use history, species composition, soil type, and stand dynamics. Soil N accumulation provides a sink for N deposition and is therefore included in the calculation of $SD_{min}(N)$ (Eq.3). The larger the acceptable soil N accumulation, the larger the sustainable deposition. Soil N accumulation is difficult to measure, therefore a number of different approaches for estimating acceptable soil N are presented in this section.

A4.2. Previous approaches for determining soil N accumulation

One proposed approach to treating N accumulation in soil via the SMB was to assume that there is no net long-term accumulation. Thus, at steady state, N inputs equal N outputs. There are three problems with this approach. First, we cannot require the soil N pool size to remain the same over long time periods with more precision than we can measure it. For example, Huntington et al. (1988) demonstrate that there is a large variability in the soil N pool, such that changes of less than 12-20 % are not detectable even with intensive sampling, precluding simple assessment of changes in the size of the pool. Secondly, and more importantly, it is clear that ecosystems which continue to be

productive and healthy have accumulated finite quantities of nitrogen since glaciation. Some of the long-term accumulation of N appears to be in unreactive, recalcitrant forms and represents a sequestration of N that is not plant-available (Aber et al 1991). Thirdly, if we assume all N input is translated into an N output as NO_3, we may seriously overestimate the acid leaching potential of a site. In both unpolluted (Hedin et al. 1995) and polluted forests (Friedland and Miller 1999), substantial amounts of N are exported from mature upland forests as dissolved organic N.

At the Lökeberg Critical Loads for Nitrogen Workshop (1992), a number of approaches for estimating N accumulation were considered based on the following assumptions (Grennfelt and Thörnelöf 1992).

1. That soil N increases 5% over 100 years. This results in a change of 0.05% per year. The change over 100 years could be chosen to be the smallest detectable change.
2. That the C:N ratio in soil remains constant.
3. That the rate of accumulation of N must not exceed the rate since glaciation (determined for a site from soil N divided by the number of years since glaciation).
4. That the acceptable N accumulation rate depends on land management .

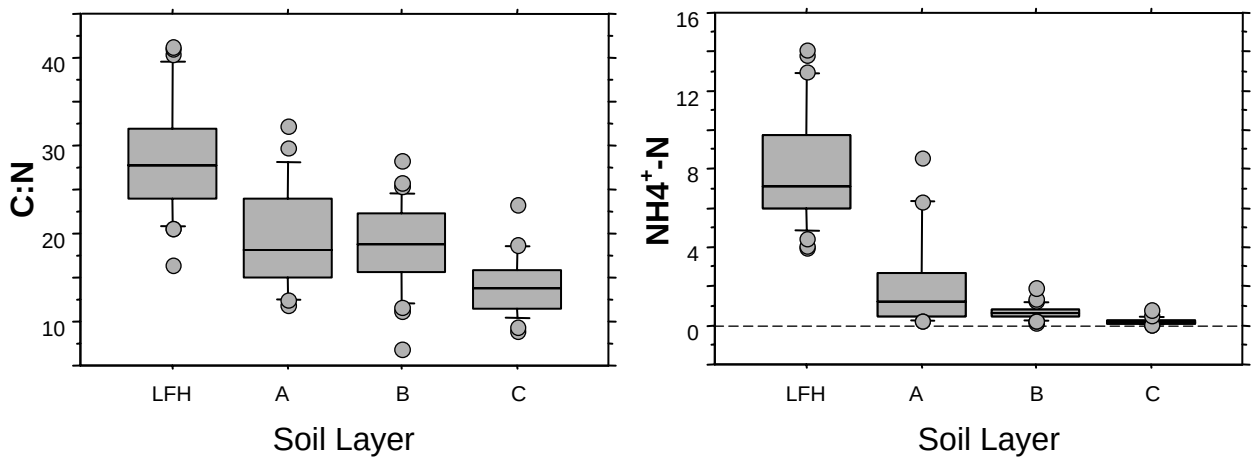
Based on approach number 3 in Sweden, model calculations showed a soil N accumulation rate of 0.2 kg N ha^{-1} yr^{-1} (Ågren and Bosatta 1988) while a chronosequence study showed a range for soil N accumulation of 0.2-0.5 kg N ha^{-1} yr^{-1} (Rosén 1992).

Approach number 4 was considered the best approximation at that time. This approach ignores any climate or forest species effects, but does incorporate a simple differentiation based on land use.

Table A4.1 Acceptable soil N accumulation for forested ecosystems.

Ecosystem	Acceptable soil N accumulation (kg/ha/yr)
Managed forests	0.5-2
Unmanaged forests	0
N-limited forests	0.5
non N-limited (N-saturated)	0

Figure A4.1. C:N ratio and water-soluble NH_4-N in upland forest soils at Kejimkujik National Park. (a) C:N ratio and (b) water-soluble NH_4-N in upland forest soils at Kejimkujik



National Park, in relation to soil layer; LFH: organic surface layers, A: all A horizons, B: all midsoil or B layers; C: all subsoils, including transitional BC layers.

A4.3. Soil N accumulation in the SMB analysis

In order to determine soil N accumulation, one must consider the C:N ratio of soil organic matter. Typical values of soil C:N in upland forests are shown in Figure A4.1. Also shown in this Figure is the amount of NH_4^+-N present in the soil solution, as a function of soil depth in the same soils.

More recently, Aber et al. (1997) presented a modeling analysis suggesting that N-cycling rates in eastern North America are correlated simply with climate variables such as growing degree days (Figure A4.2) and precipitation volume. They noted that these relationships are specific to particular forest species and may be modified by prior land-use effects on soil N pools. Theoretical maximum N-cycling rates for the northeastern US (expressed as net mineralization) were on the order of 80 to 100 kg-N ha^{-1} yr^{-1} (Aber et al. 1997). If forest growth does not consume the nitrogen made available through mineralization plus the atmospheric inputs, then the potential for nitrification and NO_3^- leaching exists. Aber et al.
(1997) observed that NO_3^- leaching begins to be significant when ecosystems cycle N at ~0.75 times the maximimum potential cycling rate for a given forest species-climate combination. This analysis provides an additional framework for addressing N accumulation in the context of the SMB.

Typically, N accumulation and N loss vary with stand dynamics: opening the forest canopy leads to net N loss through enhanced soil temperatures and related increased rates of soil organic matter decomposition (Likens et al. 1970, Hornbeck et al. 1987). Gradual closing of the forest canopy has the opposite effect. Current estimates of N accumulation in soils vary from 2 to 5 kg ha^{-1} yr^{-1} (equivalent to 140 to 350 eq ha^{-1} yr^{-1} of potential NO_3 leaching), but can be substantially

higher when forest growth rates are high in response to recent disturbance or harvest (Gundersen 1992). The net-N accumulation rate can also be lower or zero where fire disturbance cycles are important.

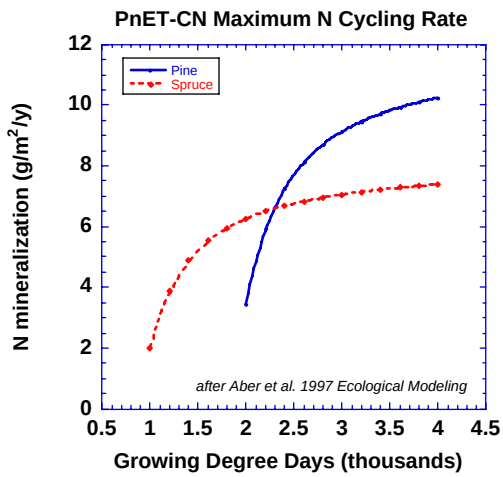


Figure A4.2. Maximum climatic potential N-cycling expressed as N-mineralization rate for white pine and red spruce forests of the northeastern US (after Aber et al. 1997).

A4.4. Provisional approach for SMB assessment of N accumulation

- Ecosystem N accumulation is driven primarily by forest growth rate and disturbance regime.
- Land-use and disturbance history, including fire, determine the rate of long-term soil N sequestration. The Lökeberg Workshop information modified using North American literature on agricultural land use effects can be used to establish appropriate categories of long-term soil N accumulation.
- Estimated soil N accumulation should be constrained by C:N ratios.
- For systems with no history of agricultural use or not subject to fire, nitrification and NO_3 leaching will occur when plant uptake of N is less than the net of: climatic potential mineralization rate + atmospheric deposition – long-term soil sequestration – DON leaching.

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Appendix 5: Estimating nutrient uptake rates
R. Ouimet, P. A. Arp

Net annual uptake rates of nitrogen and base cations into the aboveground woody biomass can be estimated based on average element concentrations in wood and by way of the nutrient limitation approach. In this approach, N uptake is restricted when growth is limited by base cation availability (Arp et al. 1996), i.e., long-term net annual N uptake is limited by the lowest long-term availability among Ca, Mg, and K from soil weathering and deposition:

$$N_u = \min\left(\frac{X_{dep} + X_w}{[X_i]/[N]}, X_i = (Ca, Mg, K) \right) \qquad \text{(A5.1)}$$

where

N_u - net nitrogen increment in the woody biomass (eq ha^{-1} y^{-1})
X_{dep} - atmospheric deposition of element X (eq ha^{-1} y^{-1})
X_w - soil weathering rate of element X (eq ha^{-1} y^{-1})
$[X_i]/[N]$ - average element : N concentration ratio in the aboveground wood.

Once N_u has been determined, net basic cation uptake can be calculated as :

$$BC_u = \sum_{i=1}^{4} \left(N_u \times {[X_i]}/{[N]} \right), X_i = (P, Ca, Mg, K) \qquad \text{(A5.2)}$$

where

BC_u - net K + Ca + Mg uptake rate (eq ha^{-1} y^{-1}).

This method implies that the minimum rates of Ca, Mg, or K deposition and weathering from the soil control the net annual accumulation of these elements and of P and N in the aboveground woody biomass, in the long term. It is assumed that the ratios of element concentrations in wood remain constant. Using this nutrient limitation approach makes basic cations, P, and N net uptake almost proportional to basic cation weathering rate.

Average ratios of element to N concentration in the aboveground forest vegetation were estimated for eight groups of American forest tree species of the Northeast (Table 1). Data comes mainly from ARNEWS (P.A. Arp unpublished data, 185 obs.), RESEF (R. Ouimet unpublished data, 67 obs.), and the compilation of nutrient concentrations in the woody biomass from selected studies by Kimmins et al. (1985). Other individual sources after 1985 are from Integrated Forest Study (IFS) plots (Johnson and Lindberg 1992), De Visser (1992), Frelich et al.

(1989), Bondietti et al. (1990), Momoshima and Bondietti (1990), McClenahen et al. (1989), Long and Davis (1989), DeWalle et al. (1991), Watmough et al. (1998), Hendrickson et al. (1987), and Ruark and Bocheim (1988). Nutrient concentrations and nutrient ratios are compiled for four tree compartments (stemwood, bark, branches, and foliage) in Tables 2 and 3, respectively. The biomass partitioning of these compartments is shown in Table 6. These values are used to model the impact of different forest harvesting scenarios on net nutrient exports from sites. For instance, conventional tree harvesting exports only bole wood and bark while whole-tree harvesting exports boles, branches, and leaves, so the rate of export reported on an annual basis is much higher for the latter harvesting method than for the former.

Table A5.1. Northeastern American forest tree species groups included in the database.

Forest type	Tree species included
Balsam fir	*Abies balsamea* (L.) Mill.
	Tsuga canadensis (L.) Carr.
Intolerant hardwood	*Populus* sp.
Jack pine	*Pinus banksiana* Lamb.
Spruces	*Picea mariana* (Mill.) BSP.
	Picea rubens Sarg.
Tolerant hardwood	*Acer saccharum* Marsh.
	Fagus grandifolia Ehrh.
	Quercus sp.
	Betula alleghaniensis Britton
	Acer rubrum L.
	Fraxinus americana L.
	Ostrya virginiana (Mill.) K. Koch
	Tilia americana L.
	Prunus serotina Ehrh.
White birch	*Betula papyrifera* Marsh.
White pine	*Pinus strobus* L.
	Pinus resinosa Ait.
	Pinus contorta var. *Latifolia* Engelm.
White spruce	*Picea glauca* (Moench) Voss.
	Picea abies L.

Table A5.2. Elemental N, Ca, Mg, K concentration (mg kg^{-1}) in stemwood, bark, branches, and foliage of select tree species.(ARNEWS and RESEF data)

	Forest type	n	N		K		Ca		Mg		P	
			Mean	SD†	Mean	SD†	Mean	SD†	Mean	SD†	Mean	SD†
Stemwood	Balsam fir	31	**494**	69	**406**	169	**753**	137	**139**	31	**21**	19
	Intol. Hardwoods*	15	**582**	178	**446**	184	**1229**	293	**167**	32	**46**	13
	Jack pine	19	**417**	60	**148**	71	**646**	67	**117**	14	**17**	8
	Spruces	42	**377**	38	**163**	81	**794**	121	**103**	14	**14**	12
	Tol. Hardwoods ‡	92	**843**	184	**508**	224	**959**	331	**157**	54	**55**	18
	White birch	20	**810**	99	**275**	89	**880**	415	**164**	26	**55**	23
	White pine	11	**453**	55	**175**	65	**594**	182	**120**	18	**20**	6
	White spruce	20	**464**	62	**156**	80	**936**	172	**88**	24	**20**	12
Bark	Balsam fir	24	**2986**	542	**1314**	270	**6702**	1956	**452**	100	**373**	83
	Intol. Hardwoods*	15	**3408**	949	**1697**	413	**12907**	5642	**694**	166	**379**	114
	Jack pine	17	**2164**	486	**655**	271	**4502**	846	**363**	86	**203**	62
	Spruces	32	**2167**	290	**940**	273	**6788**	2481	**367**	86	**279**	64
	Tol. Hardwoods‡	47	**4762**	1008	**1589**	520	**18424**	8238	**596**	230	**318**	94
	White birch	20	**3776**	668	**846**	615	**5772**	3795	**424**	235	**293**	122
	White pine	11	**2467**	700	**936**	383	**4685**	1977	**454**	194	**278**	134
	White spruce	17	**2836**	760	**1535**	349	**12390**	3247	**496**	143	**386**	138
Branches	Balsam fir	24	**5838**	922	**2077**	252	**10265**	1287	**839**	135	**392**	61
	Intol. Hardwoods*	15	**7343**	1634	**3008**	653	**15471**	3490	**1424**	281	**507**	112
	Jack pine	17	**4604**	741	**1236**	227	**7238**	746	**707**	123	**317**	46
	Spruces	32	**4269**	433	**1828**	338	**9544**	1191	**719**	119	**287**	27
	Tol. Hardwoods‡	47	**9416**	1866	**2786**	638	**18991**	3974	**1233**	392	**634**	126
	White birch	20	**8062**	1085	**2406**	501	**13406**	2105	**1047**	273	**555**	71
	White pine	11	**4965**	1192	**1627**	310	**8363**	1454	**859**	261	**336**	77
	White spruce	17	**5478**	924	**2184**	325	**12351**	1570	**897**	171	**366**	52
Foliage	Balsam fir	31	**13217**	2194	**5536**	865	**4964**	1427	**1022**	176	**1445**	287
	Intol. Hardwoods*	15	**20096**	5672	**9177**	3611	**9135**	4923	**1989**	593	**1784**	388
	Jack pine	20	**12123**	1233	**3994**	520	**2395**	542	**900**	97	**1157**	108
	Spruces	41	**9753**	1111	**5799**	1398	**3073**	950	**992**	221	**1307**	230
	Tol. Hardwoods‡	64	**22102**	4483	**8291**	2335	**9408**	3531	**1744**	624	**1617**	446
	White birch	21	**21607**	3124	**9414**	2244	**5925**	3401	**1898**	654	**2045**	645
	White pine	11	**12193**	2393	**5025**	633	**2074**	1033	**1036**	204	**1312**	237
	White spruce	21	**12302**	1461	**5586**	1070	**6461**	2708	**1018**	172	**1477**	271

†SD=Standard deviation, ‡Shade tolerant hardwoods, * Shade intolerant hardwoods

Table A5.3. Element to N ratios in the stemwood, bark, branches and foliage of select tree species (SD: standard deviation; ARNEWS and RESEF data).

	Tree type	n	P/N		K/N		Ca/N		Mg/N	
			Mean	SD	Mean	SD	Mean	SD	Mean	SD
Stemwood	Balsam fir	35	**0.060**	0.060	**0.86**	0.38	**1.63**	0.48	**0.280**	0.080
	Intol. hardwoods	21	**0.092**	0.034	**0.94**	0.51	**2.29**	0.81	**0.300**	0.090
	Jack pine	21	**0.042**	0.019	**0.38**	0.20	**1.51**	0.33	**0.270**	0.070
	Spruces	48	**0.045**	0.041	**0.45**	0.22	**2.02**	0.56	**0.260**	0.070
	Tol. Hardwoods	109	**0.071**	0.027	**0.66**	0.31	**1.23**	0.54	**0.190**	0.080
	White birch	24	**0.070**	0.024	**0.34**	0.15	**1.11**	0.56	**0.200**	0.060
	White pine	17	**0.073**	0.048	**0.41**	0.15	**1.20**	0.44	**0.260**	0.070
	White spruce	22	**0.050**	0.034	**0.34**	0.16	**2.00**	0.39	**0.190**	0.060
stembark	Balsam fir	24	**0.126**	0.022	**0.44**	0.07	**2.28**	0.71	**0.152**	0.021
	Intol. hardwoods	15	**0.113**	0.031	**0.52**	0.14	**4.03**	1.75	**0.213**	0.071
	Jack pine	17	**0.093**	0.017	**0.30**	0.10	**2.22**	0.79	**0.174**	0.049
	Spruces	32	**0.129**	0.029	**0.43**	0.11	**3.18**	1.27	**0.171**	0.040
	Tol. Hardwoods	47	**0.067**	0.019	**0.34**	0.12	**4.12**	2.40	**0.125**	0.039
	White birch	20	**0.077**	0.027	**0.22**	0.15	**1.52**	1.05	**0.107**	0.048
	White pine	11	**0.110**	0.027	**0.38**	0.10	**1.99**	1.02	**0.190**	0.081
	White spruce	17	**0.134**	0.024	**0.56**	0.14	**4.70**	1.98	**0.175**	0.030
Branches	Balsam fir	24	**0.067**	0.001	**0.36**	0.06	**1.78**	0.20	**0.150**	0.020
	Intol. hardwoods	15	**0.069**	0.003	**0.42**	0.07	**2.13**	0.31	**0.200**	0.060
	Jack pine	17	**0.069**	0.001	**0.27**	0.04	**1.59**	0.16	**0.160**	0.030
	Spruces	32	**0.067**	0.001	**0.43**	0.08	**2.25**	0.26	**0.170**	0.030
	Tol. Hardwoods	47	**0.067**	0.001	**0.30**	0.05	**2.06**	0.43	**0.130**	0.030
	White birch	20	**0.069**	0.001	**0.30**	0.07	**1.68**	0.28	**0.130**	0.030
	White pine	11	**0.068**	0.001	**0.33**	0.05	**1.72**	0.29	**0.180**	0.050
	White spruce	17	**0.067**	0.002	**0.41**	0.08	**2.29**	0.37	**0.160**	0.020
Foliage	Balsam fir	31	**0.111**	0.024	**0.43**	0.09	**0.38**	0.12	**0.080**	0.021
	Intol. hardwoods	15	**0.092**	0.017	**0.46**	0.12	**0.45**	0.20	**0.104**	0.034
	Jack pine	20	**0.096**	0.006	**0.33**	0.04	**0.20**	0.05	**0.075**	0.010
	Spruces	41	**0.135**	0.024	**0.59**	0.12	**0.32**	0.11	**0.104**	0.030
	Tol. Hardwoods	64	**0.073**	0.018	**0.38**	0.08	**0.44**	0.22	**0.081**	0.028
	White birch	21	**0.096**	0.030	**0.44**	0.10	**0.28**	0.18	**0.090**	0.036
	White pine	11	**0.109**	0.019	**0.42**	0.09	**0.16**	0.05	**0.086**	0.016
	White spruce	21	**0.121**	0.024	**0.46**	0.09	**0.52**	0.21	**0.083**	0.015

Table A5.4. Biomass partition in select tree species, in wood, bark, branch, and foliage, from P.A. Arp, unpubl. data.

Forest type	Wood	Bark	Branch	Foliage
Balsam fir	0.58	0.11	0.19	0.12
Intol. hardwoods	0.64	0.11	0.22	0.03
Jack pine	0.72	0.06	0.16	0.06
Spruces	0.61	0.08	0.21	0.10
Tol. Hardwoods	0.64	0.11	0.22	0.03
White birch	0.64	0.10	0.23	0.03
White pine	0.62	0.08	0.20	0.10
White spruce	0.58	0.08	0.22	0.12

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Appendix 6: Estimating atmospheric deposition rates

E. Miller

Atmospheric deposition of sulfur, nitrogen and base cations (Ca, Mg, K, Na) must be either measured or estimated in order to assess the sustainability of the deposition regime at a given site. We will rely on three approaches for determining atmospheric deposition rates:

1. Site specific measurements of wet or wet plus dry deposition
2. Coarse-scale wet deposition estimates interpolated from national network observations by NatChem, AES – Canada
3. Fine-scale estimates of wet plus dry plus cloud water deposition from Ecosystems Research Group's high-resolution deposition model (HRDM)

These approaches are described briefly below. It is important to note that methodologies for both local and regional deposition estimates are constantly being improved. It is expected that several new products may become available during the course of the pilot and comprehensive studies. The approaches described below will be used as a starting point for atmospheric deposition estimates. Our overall approach to assessing forest sensitivity to acidic deposition and the sustainability of deposition regimes is designed so we can easily incorporate, new or revised deposition estimates.

A6.1. Direct measurements of Wet + Dry Deposition

Site-specific measurements provide the best information for analysis at the site level, but can not be directly extended to the regional analysis. Whenever possible we will obtain information on direct measurements of wet and dry deposition from researchers and institutions contributing data to the site-specific component of the study. If only wet or dry deposition measurements are available for a site, we will employ one of the methods discussed below to obtain an estimate of the unmeasured component.

A6.2. NatChem Gridded Wet deposition (40 km resolution)

The Canadian Atmospheric Environment Service Project, NatChem (http://airquality.tor.ec.gc.ca/natchem), assimilates, screens and merges data from Canadian federal, provincial and US national precipitation monitoring networks. NatChem has the ability to spatially interpolate the merged data sets using a kriging process to produce smooth, gridded estimates of wet deposition across North America. The finest resolution at which the data can be interpolated with confidence results in output with a 40x40 km grid cell dimension. This resolution is

expected to be adequate for characterization of wet deposition to large regions of the study area (predominantly in Canada) with relatively simple (low variation in relief) terrain.

Since NatChem only provides spatially interpolated wet deposition estimates at this time, dry deposition will be estimated from spatial patterns in the dry/wet deposition ratio observed at sites where both deposition modes have been measured. Lear and Schmeltz (2000) described significant spatial variation in the ratio of dry to wet N deposition at the eastern US CASTNET sites. Dry/wet ratios for N ranged from 56% to 16% from the midatlantic to far northeastern US. A similar range of dry/wet ratios was observed for S deposition at the Integrated Forest Study (IFS) sites (Johnson and Lindberg 1992). In contrast to the variability in the dry/wet ratio for S deposition and the CASTNET results, the IFS sites exhibited a nearly constant ratio of 1:1 between dry and wet N deposition. We will rely on the most recent, spatially representative observations of the dry/wet deposition ratio for our estimates (including soon to be released information from CAPMON). It is anticipated that AES could provide dry deposition estimates at approximately the same spatial resolution as the NatChem wet deposition data in the near future (Brook et al. 1999).

In large segments of the US study area, complex terrain with elevation changes of >1 km over ground distances of 2 – 10 km, and variable vegetation result in wet, dry and total deposition rates which vary by factors of 2 to 5 at local scales (Figure A6.1). These variations in deposition occur in complex spatial patterns within a 40 km grid cell. Where elevations exceed 700 m a.s.l. the interception of cloud droplets by forest vegetation adds an additional, significant component to the total atmospheric deposition rate (Miller and Friedland 1999), increasing in importance with elevation (Figure A6.1). The relatively coarse NatChem data — or any direct spatial interpolation approach which does not consider terrain variation — will provide poor local estimates of wet deposition in this situation (Ito et al. 2000). For the purposes of this project, it is important to develop representative estimates of total deposition to high-elevation and topographically complex segments of the landscape. High-elevation areas generally have poor soils, low soil temperatures and low weathering rates (see Appendix 5) and have already been identified by many researchers as potentially sensitive to acidic deposition.

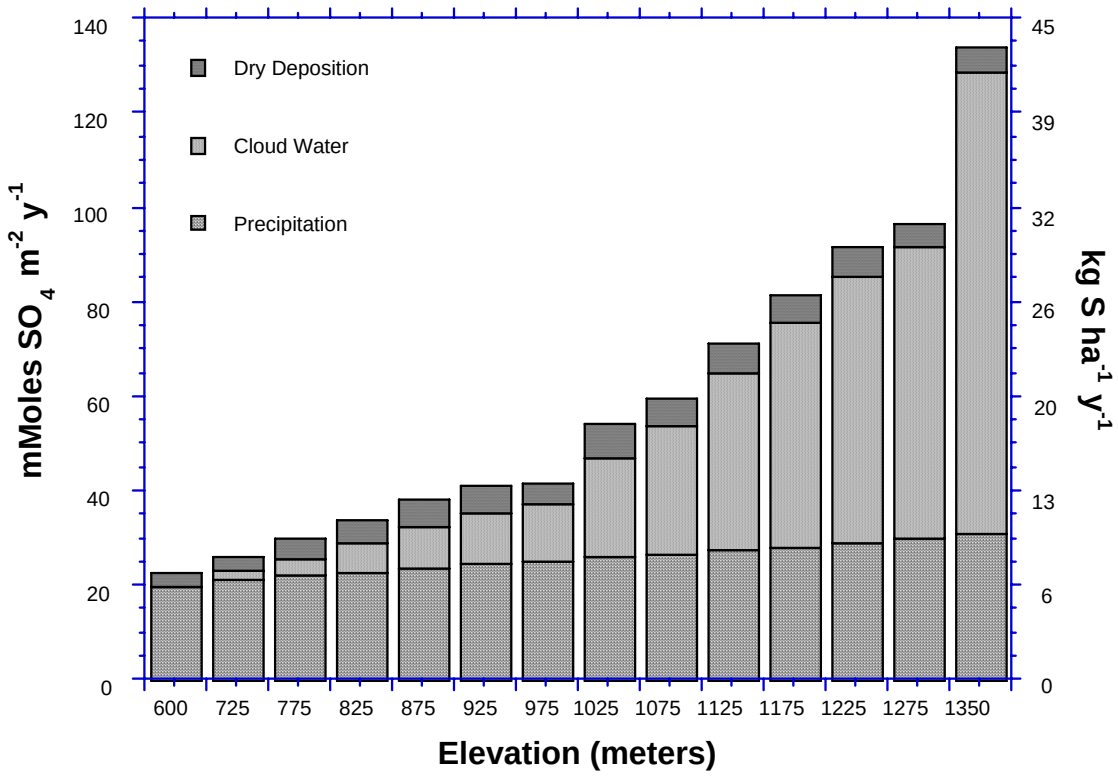

Figure A6.1. Variation in wet, dry, cloud, and total deposition over ~2km ground distance as a function of elevation on Whiteface Mt., NY, 1986-1989 (from Miller et al. 1993).

A6.3. High-Resolution Deposition Model Total Deposition Estimates (30-m resolution)

The complexity in patterns of rainfall, vegetation (dry and cloud deposition receptor surface) and deposition at sub-kilometer scales in the mountainous northeastern US states requires a high spatial-resolution approach to atmospheric deposition estimates. Miller (2000) developed a spatially-distributed modeling environment which couples detailed physical models of atmosphere-landsurface heat, mass and momentum transfer processes (Miller et al. 1993) to a high-resolution geographic information system and regional climatology for the northeastern US. The model provides estimates of wet, dry and cloud water deposition at 30- to 90-meter resolution. Key features of the HRDM include:

1. Wet, dry and cloud-water deposition processes are represented – providing a true "total deposition" regional model for mountainous landscapes.

2. Statistical modeling of regional spatial gradients is combined with surface interpolation of residual fields to obtain 10-km grid resolution estimates of atmospheric chemistry with a high degree of fidelity to network observations.
3. Wet deposition is calculated as a combination of the 10-km resolved precipitation chemistry, 1-km resolved regional precipitation field, and 90-m resolved terrain corrected precipitation amount.
4. Either point observational records or gridded meteorological model output can be spatially interpolated to 90-m resolution, corrected for local topographic and landscape positional effects and monthly regional climatology to drive the dry and cloud water deposition models.
5. The receptor surface for dry and cloud water deposition is represented at 30-m resolution. The biophysical characteristics of the receptor surface are estimated in terms of the proportion of leaf area expected to be attributable to specific plant species. Species proportions are estimated by a forest species distribution submodel with guidance from the USGS/EPA NLCD data set.
6. Deposition estimates are generated on a seasonal basis using sub-season time steps including representations of diurnal fluctuations employed in the dry deposition process model.

Figure A6.2 provides an example of the range and spatial complexity of S dry deposition estimated by the HRDM for a portion of the high-peaks region of the Adirondack Mountains, NY USA during summer of 1989. Spatial variation in dry deposition due to intermingling of forests with differing leaf areas and stomatal responses to environmental conditions is evident. Low estimated dry deposition regions occur at both low and high elevations (bare ground, fields, wetland areas, and alpine tundra). There is a moderate increase in deposition with increasing elevation over the midslope regions of the mountains in response to changes in the efficiency of turbulent exchange and leaf area as evergreens become more important and eventually dominant components of the forest at higher elevations.

HRDM estimates of atmospheric deposition for the US pilot study region (VT) and the state of Massachusetts are being conducted by Ecosystems Research Group, Ltd., Norwich, VT under a contract with the Northeast States for Coordinated Air Use Management (NESCAUM).

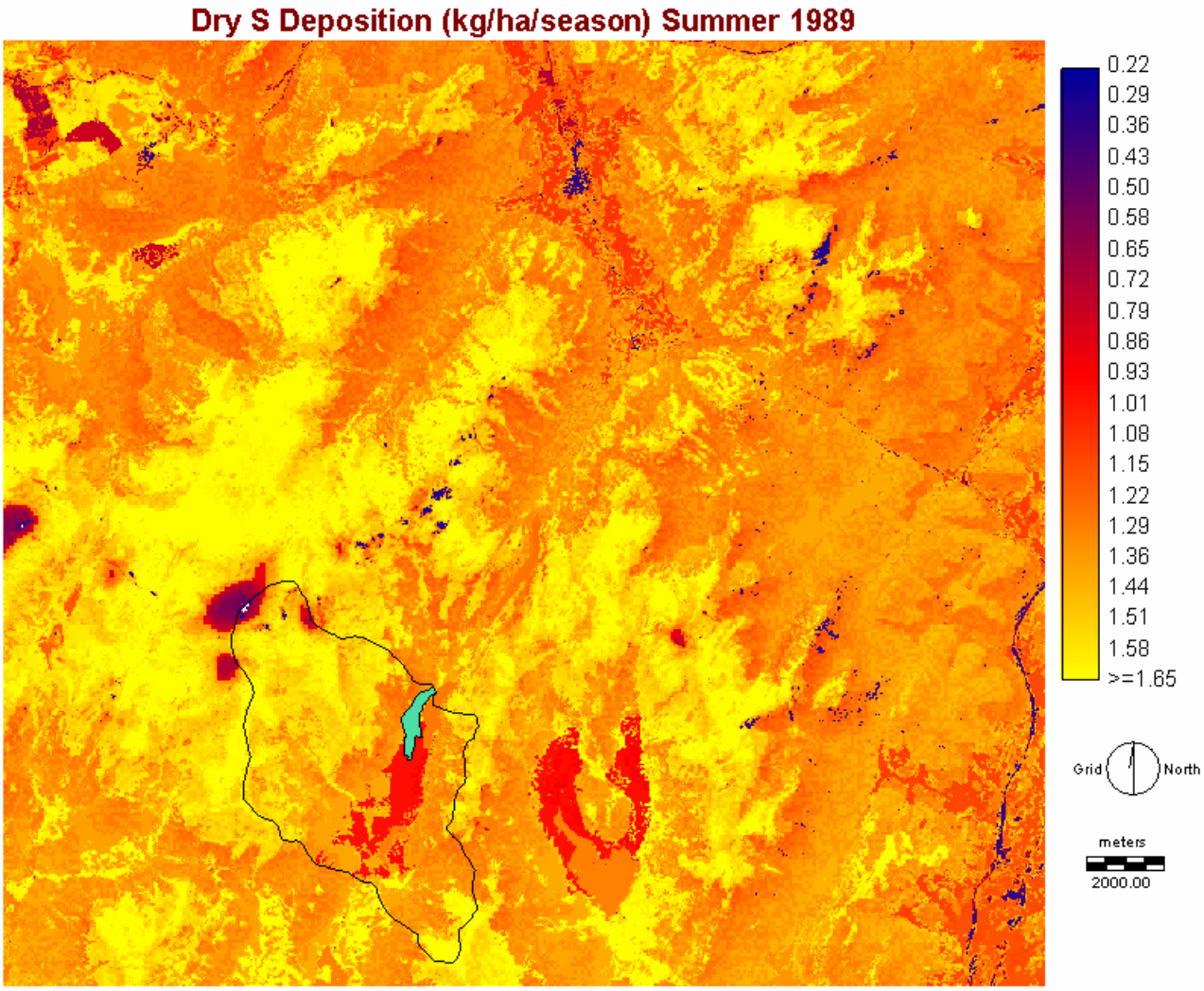

Figure A6.2. Preliminary High-Resolution Deposition Model (HDRM) estimated summer dry sulfur deposition (SO_2 plus particle SO_4) for a portion of the High-Peaks region of the Adirondack Mountains, NY, USA. Mount Marcy is on the left side of the image. Keene Valley is near the top center of the image. The black line delineates the watershed of the Upper Ausable Lake (shown in light blue). The image represents an approximately 27x28 km ground area. Dry deposition to this region would be represented by a single value when using dry/wet ratios and the 40-km resolution NatChem wet deposition grid as a basis for the estimate.

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Appendix 7: Impacts of exceedance of sustainable deposition on forest ecosystems

S. Wilmot, P. Arp, R. Ouimet, E. Miller, L. Pardo

The purpose of this appendix is fourfold. First, we present the current scientific knowledge about the relationship between elevated N and S deposition and harmful ecological effects based on research in Europe and North America. Second, we address the relationship between exceedance of sustainable deposition or critical loads, and ecosystem responses. Third, we address progression of forest decline. Finally, we identify the indicators of forest health and productivity that we will consider in our evaluation of sensitive areas.

Forest ecosystem responses to exceedance of sustainable deposition may occur in a multitude of discrete and inter-connected ways. It is probable that only the most obvious impacts will be detectable under most coarse-level investigations. For the purposes of this project, we will compile existing data on indicators of forest ecosystem health to use in comparison with the results from the sustainable deposition mapping (see Table A7.1). Relationships between areas of exceedance of sustainable deposition and indicators of declining or poor ecosystem health will be valuable in assessing current and future forest impacts (Arp et al. *in press*).

Numerous studies provide examples of plant physiological response and changes in ecosystem condition as the result of soil nutrient (Ca, Mg, or K) limitation. Increases in exchangeable metal levels (particularly Al) may also accompany decreased soil base cation nutrient pools. Elevated soil Al may contribute to diminished forest health directly or via interactions with low base cation supplies. Some examples include: increased levels of stress hormones in red spruce as a function of the soil exchangeable Al:Ca ratio (Minocha et al. 1997); reductions in net photosynthetic capacity and growth in sugar maple on base-poor sites (Wilmot et al. 1995, Liu et al. 1997); and positive growth responses to Ca-fertilization of Ca-depleted sugar maple stands (Moore et al. 2000).

In addition to impacts related to nutrient limitation, S and N deposition (and associated acidity) have been demonstrated to have direct effects on forest health. For example, fumigation studies demonstrate the potential for negative effects on photosynthesis in red spruce due to foliar assimilation of SO_2 (Meng et al. 1994). The extremely high ambient acidity of mountain cloud water in the northeastern US has been shown to reduce cold tolerance of red spruce seedlings (DeHayes et al. 1991) and mature trees (Vann et al. 1992). There is also evidence of significant interactions between soil nutrient availability and direct foliar effects on foliar nutrition and physiology of red spruce (Schaberg et al. 2000).

While our analysis of exceedance of sustainable deposition deals directly with soil nutrient availability and plant nutrient limitation, we will need to overlay information on the ambient acidity environment, SO_2 and O_3 deposition to assess the potential for direct or interacting forest health effects. This analysis is easily accomplished using information on the acidity environment, SO_2 and O_3 deposition generated by the HRDM (High-Resolution Deposition Model, see Appendix 6).

A7.1. Effects of N deposition

Elevated N deposition may cause harmful ecological effects other than those caused by direct acidity effects, discussed in the previous section. Detrimental ecological effects of N deposition fall into several general categories: plant nutrient imbalances caused by excess available N, changes in species composition, disruption of plant-mycorrhizal associations, and eutrophication of surface and coastal waters. Plant nutrient imbalances in Ca:N or Mg:N have been documented as a result of atmospheric deposition in Europe (Schulze 1989) and experimental N addition in New England, US (Schaberg et al. 1997). Declines in these BC:N ratios can lead to serious forest health impacts (Aber et al. 1998). Elevated N deposition may also lead to changes of species composition and reductions in biodiversity (Tilman 1987, Ellenberg 1988, Wedin and Tilman 1996). Finally, increased N availability may disrupt plant-mycorrhizal associations which play an important role in plant nutrition and maintaining ecosystem N retention (Wallenda et al. 2000).

A7.2. Time to reach steady-state exceedance conditions: implications for comparisons of SD with forest health

The calculation of sustainable deposition implies that forests ecosystems are in equilibrium with atmospheric deposition and forest harvesting, which would rarely be the case. One implication of this equilibrium concept for the calculations of SD is that sustainable deposition rates may be exceeded, without current damage or signs of ecosystem disturbance. This situation could result because the equilibrium condition specified in the steady-state mass balance has not been reached yet. In fact, in the case of exceedance of sustainable deposition, the pool of exchangeable base cations will buffer the extra acidity by releasing its cations, for some period of time. However, there is no direct temporal implication of the SD and exceedance calculations.

One approach for incorporating this dynamic temporal dimension in steady-state SD maps is to map, 't', the time until the effects of steady-state critical loads exceedance are observed. This concept was originally proposed by Sverdrup (U. Lund, Sweden). The ‘t’ index can be calculated by dividing the total soil available base cation pool by the steady-state exceedance of acidity. Kennedy used this approach to identify current potentially damaged areas related to critical load

exceedance for the UK (Kennedy et al. 2001). Although subject to simple assumptions, the 't' index can be a useful tool for relative comparisons of forest health indicators between areas.

Table A7.1 Forest Health Biological criteria and potential data sources.

Criteria	Indicator	Measurement	Data type
Ecosystem health and vitality	Tree health	Foliage transparency (avg.)	Plot data (Forest Health Monitoring, North American Maple Project, State Forest Health Plots)
		Crown density (avg.)	
		Dieback (avg.)	
		Incidence of insects and diseases	State-wide aerial mapping data; state plot data
		Areas of forest decline	State-wide aerial mapping data
	Vegetation	Diversity and abundance of under-story vegetation	Plot data (National Forest Health Monitoring, and other studies
		Lichen communities	Plot data (National Forest Health Monitoring, and other studies)
Productive Capacity	Timber production	Tree growth loss	Forest Inventory and Assessment Plot data
Water Resources	Biological diversity		Stream and lake data
	Chemical properties	pH	Stream and lake data
		Dissolved oxygen	
Global Carbon Cycles	Total ecosystem biomass/carbon pool	Growth rate Decomposition rate	

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Appendix 8: Project administration

- The project is coordinated by the NEG/ECP Environment Task Group which reports to the NEG/ECP Environment Committee.
- Sustainable acidic deposition and exceedance calculations and mapping will be carried out by the Forest Mapping Group whose members are nominated and supervised by the NEG/ECP Environment Task Group.
- Each jurisdiction (state, province) is responsible for assisting the Forest Mapping Group in identifying research sites, contact information for each site, and spatial data availability. Where appropriate, the jurisdiction will assist in compiling and transferring data to the Forest Mapping Group in a format described in this manual.
- The choice of reference sites should be agreed upon between the jurisdiction representatives and the NEG/ECP Environment Task Group.
- The Forest Mapping Group will compile data and metadata, check data quality, screen data for appropriate time intervals and model applications, and store in a regional database for use in this project and future regional efforts.
- The Forest Mapping Group, with the help of outside experts, will analyze the data, and report to the NEG/ECP Environment Task Group.
- The NEG/ECP Environment Task Group acts as the steering committee for the project, specifies the time-table for activities and makes progress reports to the NEG/ECP Environment Committee.
- Policy on data distribution to other groups will be developed by the NEG/ECP Environment Task Group, based on recommendations from the Forest Mapping Group.