

MASS BALANCE RELATED SUSTAINABILITY OF FOREST  
BIOMASS PRODUCTION: CONCEPTS AND APPLICATIONS

by

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## ABSTRACT

The purpose of this thesis is to provide a computational mass balance framework for evaluating the long-term sustainability of potential biomass growth rates (productivity). For this purpose, a forest sustainability model (ForSust) was developed to address: (i) potential forest biomass productivity; (ii) constraints on biomass production due to nutrient limitation and soil acidification; (iii) likely impacts of various forest disturbance scenarios on forest biomass production; (iv) soil nutrient availability and soil acidification as net balances of atmospheric inputs, plant nutrient uptake and mineral soil weathering; (v) effects of soil acidification on forest biomass production indicators such as % canopy transparency.

The model performance was evaluated with data that belong to the six well-studied jack pine and sugar maple sites. Specifically, the model performance tests were carried out by comparing model-simulated and field-estimated values for annual biomass growth, for Ca, Mg, K, and N net uptake and leaching rates. The resulting agreements between the various model simulations and the available field observations suggest that potential biomass production rates are numerically consistent with the cumulative accumulations of nutrients by way of soil weathering (mainly Ca, Mg, and K), and atmospheric deposition (mainly N).

The model was applied for 125 forest plots that belong to the Acid Rain National Early Warning System of the Canadian Forest Service (ARNEWS). For many of these plots, there was a positive relationship between % canopy transparency and the exceedance of critical acid deposition loads, adding further evidence that forest growth (and forest health) in acid sensitive soils may be negatively affected by high rates of atmospheric S and N deposition. The results further indicated that the biomass productivity values might be susceptible to decline where a deficit of base cation supply may occur even without biomass removal. Whole-tree harvesting may likely cause serious Ca, Mg, and K deficits and may therefore reduce forest biomass for most studied jack pine and sugar maple sites. With stem-only harvesting, the overall input/output balances can be maintained.

In summary, the model and the procedures that are developed in this thesis can potentially be used to identify: (i) sites where intensified harvesting levels may not be advisable in terms of maintaining the expected biomass production rates; (ii) sites where intensified harvest levels are likely sustainable; (iii) sites where high rates of atmospheric deposition may be a problem in view of enhanced soil nutrient leaching and related effects on sustainable soil nutrient supplies and subsequent forest health.

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**Key words:** atmospheric deposition, biomass growth, forest harvesting, forest health, leaching, mass balance, nutrient availability, sustainable productivity, uptake, weathering.

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# CHAPTER 1

## GENERAL INTRODUCTION

### BACKGROUND AND AIM

There is continued concern about the sustainability of nutrient supply and forest biomass productivity in light of increased forest harvesting pressures, and actual as well as perceived threats to forest ecosystem health on account of acid rain and climate change (Kimmins 1996). The reasons for this include:

1. Accelerated nutrient removal from forest ecosystem due to increased forest biomass production and increased rates of timber harvesting. For example, fast-growing plantations increase nutrient consumption (uptake) and, in turn, increase nutrient export from site due to short rotation. Also, intensive timber harvesting (clear-cut, whole-tree harvest) will increase nutrient export from site by high rates of biomass removal and by accelerated post-harvest leaching.
2. Accelerated deposition of atmospheric acidity due to increased  $\text{NO}_x$  and  $\text{SO}_x$  emissions from upwind urban and industrial centers adds to site related nutrient losses by way of acid-induced leaching of base cations such as  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$  and  $\text{K}^+$ .
3. Increased soil acidification, in turn, may lead to a gradual decrease of soil rooting space: mobilized Al ions from the topsoil accumulate in the subsoil where these accumulations may interfere with fine root growth and nutrient uptake. In this respect, individual forest sites are considered to each have a so-called critical soil acidification load (CL). Below these loads, soil acidification is not a factor because the overall acid load is within the acid buffering capacity of the site. Above that load (exceedances of

CL), is considered to lead to gradual site deterioration on account of unsustainable base cation losses, and a strong possibility of negative effects of Al mobilization on forest growth and soil quality (Sverdrup and Warfvinge 1994).

Various supply factors relating to CO<sub>2</sub>, light, moisture, temperature, and nutrient availability control the potential biomass productivity of each forest site (Binkley 1986). Among these, nutrient availability becomes a key factor when focusing on the long-term sustainability of biomass production for any given site because forest growth eventually becomes limited by the availability of Ca, Mg, K, N, or P (Ågren 1985). Moreover, site related nutrient availability is the most susceptible to change due to impacts of natural and/or anthropogenic disturbances (e.g., forest fire, harvesting, acid rain, etc.).

Numerous processes determine the availability of nutrients for plant growth. Important processes are mineral soil weathering, atmospheric deposition, soil organic matter mineralization, ion adsorption/desorption reactions such as ion exchange, nitrification/denitrification reactions, extent of nutrients already taken up from the available nutrient pools, and ion leaching. The importance of each process depends on the temporal scale of the overall forest sustainability assessment. For example, processes such as ion exchange, soil organic matter mineralization, and nutrient uptake play an important role when focussing on availability of soil nutrients from one day, week, month or year to the next (Van der Salm 1999). For long-term considerations (several decades and/or stand rotations), sustainability assessments of nutrient availability focus on primary nutrient supply and loss processes (inputs and outputs). Primary nutrient inputs refer to atmospheric deposition and mineral soil weathering. Primary nutrient outputs refer to soil leaching losses, and export losses due to harvesting and fire.

This thesis focuses on the long-term primary nutrient input/output balances and the net implications of these balances on sustainability of potential forest biomass production and related expectations. The general aim of this thesis is to provide a computational framework for evaluating the primary net input/output balances, and for calculating the rate of biomass production that is numerically (and logically) consistent with these balances, at the stand level. Focusing on does all of these:

1. A means to calculate weathering rates of mineral soils based on simple considerations of substrate weatherability.
2. A means to examine atmospheric inputs into forest stands based on existing data regarding atmospheric deposition.
3. A means to calculate potential biomass growth rates that are based on nutrient availability as calculated from primary input rates.
4. A means to evaluate potential nutrient losses from site due to harvesting or fire, as function of harvest or fire intensity.
5. A means to formulate the critical soil acidification load in the context of sustainable forest biomass production.
6. A means to evaluate the resulting concepts with site-specific data from forest plots that belong to the Acid Rain National Early Warning System of the Canadian Forest Service (ARNEWS).

The thesis is presented in articles format because it is my intention to submit some of the chapters to journals for publication. Subject matter and concepts relevant to this thesis are introduced as follows:

Chapter 2: describes the concept of forest sustainability in the context of relating forest biomass production to primary nutrient supply and input/output nutrient balances.

Chapter 3: reviews literature with a generalized characterization of primary nutrient inputs and outputs with regard to forest ecosystems, i.e., atmospheric deposition, soil mineral weathering, nutrient losses due to forest harvesting, and nutrient losses due to soil leaching.

Chapter 4: deals specifically with means to estimate mineral soil weathering rates by a mechanistic geo-chemical model.

Chapter 5: introduces, describes, discusses and evaluates a model (ForSust) to estimate potential forest biomass production based on primary nutrient inputs into the forest.

Chapters 6, 7, 8: deal with ARNEWS data (building of database and analyzing the data regarding atmospheric deposition, soil and vegetation).

Chapter 9: deals with applying the ForSust model to evaluate the critical soil acidification loads and related exceedances of the ARNEWS sites.

Chapter 10: deals with relating the critical load and exceedance calculations to ARNEWS forest health indicators such as % canopy transparency.

Chapters 11 and 12: examine sustainable forest biomass production for a subset of ARNEWS sites, namely jack pine and sugar maple.

Chapter 13: provides a general summary, statement of original contributions, and recommendations for further work.

## GLOSSARY

**Acid rain.** Atmospheric precipitation with pH values less than about 5.6, the acidity being due to inorganic acid such as nitric and sulfuric that are formed when oxides of nitrogen and sulfur are emitted into the atmosphere.

**Adsorption.** The attraction of ions or compounds to the surface of a solid. Soil colloids adsorb large amounts of ions and water.

**Anion.** Negatively charged ion; during electrolysis it is attracted to the positively charged anode.

**Available nutrient.** That portion of any element or compound in the soil that can be readily absorbed and assimilated by growing plants.

**Biomass.** Refer to total weight of organic material in an ecosystem in a given area at any point in time.

**Buffering capacity.** The ability of a soil to resist change in pH. Commonly determined by presence of clay, humus, and other colloidal materials.

**Bulk density, soil.** The mass of dry soil per unit of bulk volume, including the air space. The bulk volume is determined before drying to constant weight at 105 °C.

**Cation.** A positively charged ion; during electrolysis it is attracted to the negatively charged cathode.

**Deciduous tree.** A tree that sheds all its leaves every year at a certain season.

**Denitrification.** The biochemical reduction of nitrate or nitrite to gaseous nitrogen, either as molecular nitrogen or as an oxide of nitrogen.

**Desorption.** The removal of sorbed material from surfaces.

**Empirical model.** The models are based on empirical relationships between dependent and independent variables to predict specific variables. These models can be used to represent existing information about the relevant processes, and are useful for interpolating and summarizing data.

**Evapotranspiration.** The combined loss of water from a given area, and during a specified period of time, by evaporation from the soil surface and by transpiration from plants.

**Exchangeable cations.** The cations adsorbed by colloids are subject to replacement by other cation through a process called cation exchanges, the replaced cations are called exchangeable cations.

**Fixation, nitrogen.** Process by which gaseous elemental nitrogen is chemically combined with hydrogen to form ammonia.

**Immobilization.** The conversion of an element from the inorganic to the organic form in microbial tissues or in plant tissues, thus rendering the element not readily available to other organisms or to plants.

**Ions.** Atoms, group of atoms, or compounds that are electrically charged as a result of the loss of electrons (cations) or the gain of electrons (anions).

**Leaching.** The removal of materials in solution from the soil by percolating waters.

**Liebig's law.** The growth and reproduction of an organism are determined by the nutrient substance (oxygen, carbon dioxide, calcium, etc.) that is available in minimum quantity, the *limiting factor*.

**Mechanistic models.** The models are constructed from theoretical principles which are suitable for making general predictions.

**Mineralization.** The conversion of an element from an organic form to an inorganic state as a result of microbial decomposition.

**Model validation.** The demonstration of the model that does not contain known or detectable flaws and is internally consistent.

**Net uptake.** The nutrient remained in living biomass in unit time. It is calculated by:  
net uptake = total uptake – litter nutrient fall

**Nitrification.** The biochemical oxidation of ammonium to nitrate, predominantly by autotrophic bacteria.

**Nutrient cycling.** The sequences of biogeochemical changes undergone by nutrient elements as they move from the atmosphere into water, soil, and living organisms is recycled through a part of or all of the entire process.

**Nutrient biogeochemical cycling.** Nutrient circulation between the soil and plants within a forest ecosystem.

**Nutrient geochemical cycling.** Nutrient input to and output from a forest ecosystem.

**Nutrient storage.** The nutrient stored in the soil organic matter.

**Parent material.** The unconsolidated and more or less chemically weathered mineral or organic matter from which the column of soils is developed by pedogenic processes.

**Percolation, soil water.** The downward movement of water through soil. Especially, the downward flow of water in saturated or nearly saturated soil at hydraulic gradients of the order of 1.0 or less.

**Primary mineral.** A mineral that has not been altered chemically since deposition and crystallization from molten lava.

**Process model.** The models are based on theoretical principles with process-based parameters between dependent and independent variables to predict specific variables.

**Production.** The increase in total weight (biomass) or quantity of organic material on a given area over a defined period.

**Productivity, soil.** The capacity of a soil for producing a specified plant or sequence of plants under a specified system of management. Productivity emphasizes the capacity of soil to produce crops and should be expressed in terms of yields.

**Runoff.** The portion of the precipitation on an area that is discharged from the area through stream channels.

**Soil.** A dynamic natural body composed of mineral and organic materials and living forms in which plants grow.

**Soil texture.** The relative proportions of the various soil separates in a soil.

**Stand.** A homogeneous, geographically contiguous parcel of land.

**Weathering.** All physical and chemical changes produced in rock, at or near the Earth's surface, by atmospheric agents.

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## CHAPTER 2

### NUTRIENT CYCLING: LINKS WITH PRODUCTIVITY

#### INTRODUCTION

Linking forest productivity in nutrient availability requires knowledge about nutrient cycling, i.e., nutrient pools, flows, and controlling factors. There is, in general, ample anecdotal information about nutrient pools and flows for many forest ecosystems, but little is known about how these pools and flows are controlled naturally, and how these flows and controls integrate over the course of stand development, and from one stand rotation to another. Therefore, uncertainties in predicting nutrient cycling and forest productivity are largely related to uncertainties in understanding the controls of the transfers (MacLean *et al.* 1983). In particular, forest productivity is directly related to nutrient supplies, transfers and flows, nutrient pool sizes, nutrient availability, and the nutrient demands of the growing forest biomass stock.

The aim of this chapter is to briefly review forest nutrient systems with respect to biomass production potentials, and to develop from this a sustainability concept that would be relevant for assessing long-term forest biomass production rates and related expectations. The following key questions are asked:

1. How are nutrients distributed across the various forest ecosystem nutrient pools and how do these pools change with stand age?
2. How large is the plant-available soil nutrient pool versus the total soil nutrient capital?
3. How nutrients flow within and between forest ecosystems.

4. What nutrient cycling components are most important for assessing long-term forest biomass growth expectations?
5. How can the sustainability concepts be set up in the context of nutrient availability and nutrient input/output balances?

## NUTRIENT DISTRIBUTION AND CHANGES

Nutrient elements in a forest ecosystem are distributed across various nutrient pools such as the mineral soil layers, the organic soil layers, the understory and the overstory layers. The amounts of nutrients stored in each of these pools depend on pool-specific nutrient concentrations and mass.

In general, nutrient concentrations in the mineral soil are relatively low, but mineral soils contain the largest nutrient pools in most forest ecosystems because of their large mass. Organic soil layers contain the second largest pools. The tree layer contains the third largest nutrient pool. Although ground vegetation tends to have the highest nutrient concentration among the various forest ecosystem components, it has the smallest nutrient pool due to its small mass. An example of total nutrient contents in various nutrient pools is shown in Table 2.1 for a 30-yr-old jack pine (*Pinus banksiana* Lamb.) ecosystem in Ontario (Foster and Morrison 1976).

**Table 2.1** Distribution of the total nutrients in 30-yr-old jack pine ecosystem.

Components	N	P	K	Ca	Mg
	(Kg ha <sup>-1</sup> )				
Trees	165	14	82	112	18
Ground vegetation	6	<1	3	2	<1
Organic soil	328	43	524	319	116
Mineral soil	3729	6100	165354	98708	53839

Although the mineral soil tends to have the largest nutrient pool, only a small amount of soil nutrients are actually available for plant uptake. For example, Foster and Morrison (1976) found that only 29 kg ha<sup>-1</sup> nitrogen was available in the organic and mineral soil layers of a 30-yr-old jack pine stand in Ontario. A study in New Brunswick (MacLean and Wein 1977) showed that organic layers (F + H) of either jack pine or hardwood stands contained only 2.8 kg ha<sup>-1</sup> available nitrogen (on average). The mineral soils below contained 35 kg ha<sup>-1</sup> and 40 kg ha<sup>-1</sup>, respectively. A study of nutrient cycling in Scots pine (*Pinus sylvestries* L.) stands in eastern Finland (Helmisaari 1995) showed that, of soil total pool (30 cm soil depth), only 0.1-1.0% of N, a few percent of P and Mg, 4-14% of Ca, and 8-20% of K was actually plant-available.

A study of 12 jack pine stands and 11 mixed hardwood stands (all post-fire) of varying ages on similar sites revealed that nutrient concentrations in trees are relatively constant with stand maturation (MacLean and Wein 1977). The overall accumulation of nutrient elements (N, P, K, Ca, and Mg), in turn, followed the biomass accumulation trend in both series of stands. More specifically, nutrients in trees accumulated slowly during the first 15 years, and more rapidly between age 15-30 years. Gradual increases

and final stabilization occurred after 40 years. From the same study, it was found that the understory nutrient pools for the pine and hardwood stands increased quickly during a few years after the fire, and peaked and stabilized at age 15-20 years. No trends were reported for changes in the soil nutrient pools with stand maturation.

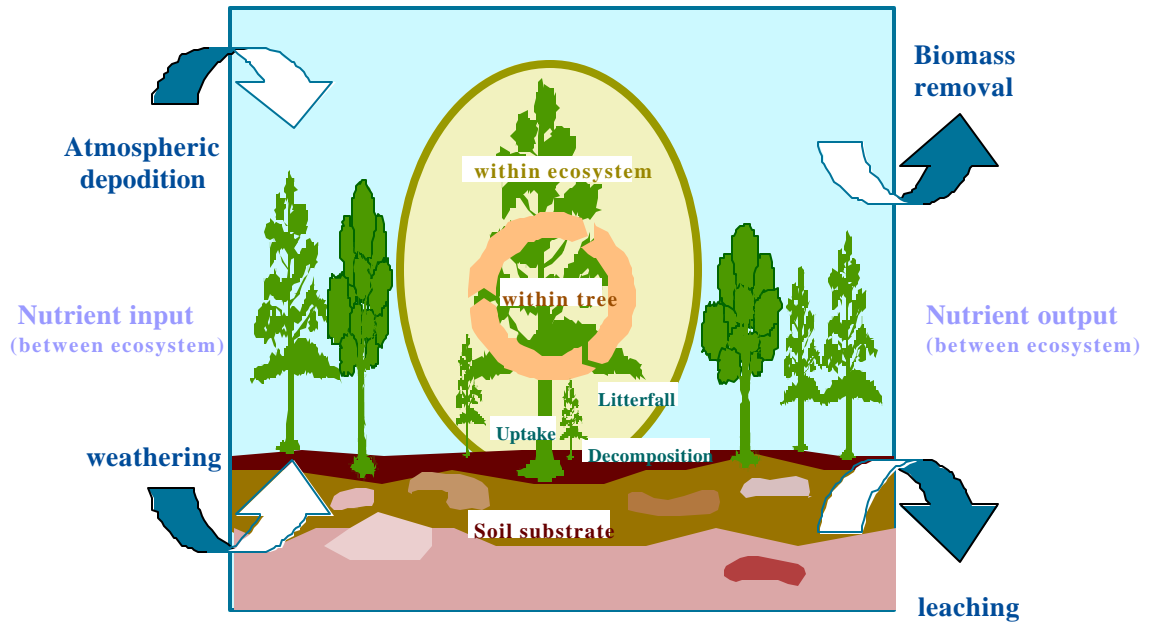
## NUTRIENT CYCLING

There are 3 scales to nutrient cycling: biochemical, biogeochemical, and geochemical (Kimmins 1987, see Figure 2.1).

### Biochemical cycle

The redistribution of nutrients within individual plants (e.g., nutrient translocation from senescent foliage before abscission) is part of the biochemical cycle. This tree-internal cycling of nutrients is important for easily translocated nutrients such as nitrogen and phosphorus, particularly in areas where the external supply is low. Retranslocation of nutrients becomes significant in older forests, and may provide the majority of the nutrients needed to support new tissue production (Kimmins 1994).

A study on three Scots pine stands in eastern Finland (sapling, pole, and mature stages) (Helmisaari 1995) showed that P and N were mostly transferred from leaves to wood through retranslocation in the youngest stands. Retranslocation supplied 30%, 40%, and 30% of the annual tree requirements in the sapling, pole, and mature stage, respectively. In mature stands, more N went into litter-fall than into retranslocation.



**Figure 2.1** The three major types of nutrient cycle: geochemical (between ecosystems), biogeochemical (within an ecosystem), and biochemical (within tree) (re-drawn from Kimmins, 1987).

### Biogeochemical cycle

Nutrient circulation between the soil and vegetation is part of the biogeochemical cycle. It is particularly important in young forests, which rely on the quantity of soil nutrients that are directly available for uptake to develop leaves, stem, and canopy biomass (Cole and Rapp 1981).

Nutrients taken up by roots of the forest vegetation are in part incorporated into the accumulating biomass, leach from leaves (by throughfall) or roots (root exudation), retranslocate before litterfall, or recycle to the soil via litterfall or root death. Most nutrients taken up by trees are returned each year to the forest floor through litterfall and leaf wash. A nutrient cycling study of a 30-yr-old jack pine stand in Ontario (Foster and Morrison 1976) showed that of the total annual nutrients uptake by trees, only 11-29% remained as biomass accumulation and 71- 89% returned to the soil. As part of this large nutrient return, 40% of K, 12% of Mg, and 7% of Ca occurred via leaf wash (leaf wash also includes dry deposition as well as leaf leaching). Leaf wash of N and P was small. It was further revealed that 80-85% of total annual nutrient uptake by the ground vegetation was returned to the soil.

The decomposition of dead tissues accumulated on forest floor and in the soil (dead roots) is an important mechanism for returning nutrients from dead plant tissues back into the available nutrient pool of the soil. The rate of litter decomposition is generally related to the local soil climatic condition and to the chemical nature of the litter. Cold soil temperatures as well as wet soil conditions tend to reduce the overall organic matter decomposition rates.

Litter production in closed canopy forests does not change significantly with differences in stand density or age (Bray and Gorham 1964). Foster and Morrison (1976) showed that the amount of organic matter accumulated in the forest floor during 35 years was < 4% of that reaching the ground as litter. This means that a steady-state condition between addition and decomposition of organic matter in the forest floor is reached in jack pine stands sometime beyond stand age 30. Damman (1971) estimated that annual root mortality in black spruce and balsam fir stands was roughly equivalent to one-half of that added in litter each year.

Nutrients released into the soil system may be taken up by roots, may be absorbed on ion exchange sites, or may be leached from the soil.

### Geochemical cycle

External nutrient inputs through atmospheric deposition and mineral weathering into a particular ecosystem, and transports of nutrients from the ecosystem via leaching, fire, and harvesting are all considered part of the geochemical cycle. Nutrients removed from the ecosystem via geochemical exports will not return to the system. Spatial and temporal scales of the geochemical cycle are generally large (at least hundreds of meters and several years, respectively).

Although annual nutrient inputs through geochemical cycle are small compared to total nutrient pools, such inputs are generally significant in terms of nutrient availability for tree growth. Geochemical inputs of base cations occur mainly through weathering, especially on young soils containing rich unweathered minerals. Seepage from uplands is another mechanism of nutrient input to lowlands.

## NUTRIENT PROCESSES IN RELATION TO SUSTAINABLE PRODUCTIVITY

Numerous processes determine the availability of nutrients for plant growth. The importance of these processes depends on the scale of study. For example, nutrient retranslocation (biochemical cycle) becomes important on the year-to-year scale. Nutrient cycling processes (biogeochemical cycle) such as soil organic matter mineralization and nutrient uptake play an important role in determining the availability of soil nutrients over short periods of time (years to decades) at the stand level. Geochemical input/output processes such as atmospheric deposition, weathering, leaching, and biomass removal are key processes when focusing on long-term assessments (several decades or stand rotations) of overall nutrient availability and forest productivity at stand or forest level.

A nutrient balance between geochemical inputs and outputs can in fact be set as the main criterion for sustainable forest biomass productivity. Kimmins (1994) stated this criterion as follows:

The nutrient capital in an ecosystem is ultimately dependent on the balance between geochemical inputs and outputs. Maintaining long-term site productivity is closely related to maintaining a positive geochemical balance of growth limiting nutrients from rotation to rotation. Nutrient losses in harvested products and all the leaching, gaseous and other losses must be compensated for by nutrient inputs to the ecosystems (atmospheric deposition, mineral weathering, etc.) over the rotation if site nutrient inventories are to be sustained.

## BASIC CONCEPTS OF THE SUSTAINABLE FOREST BIOMASS PRODUCTION

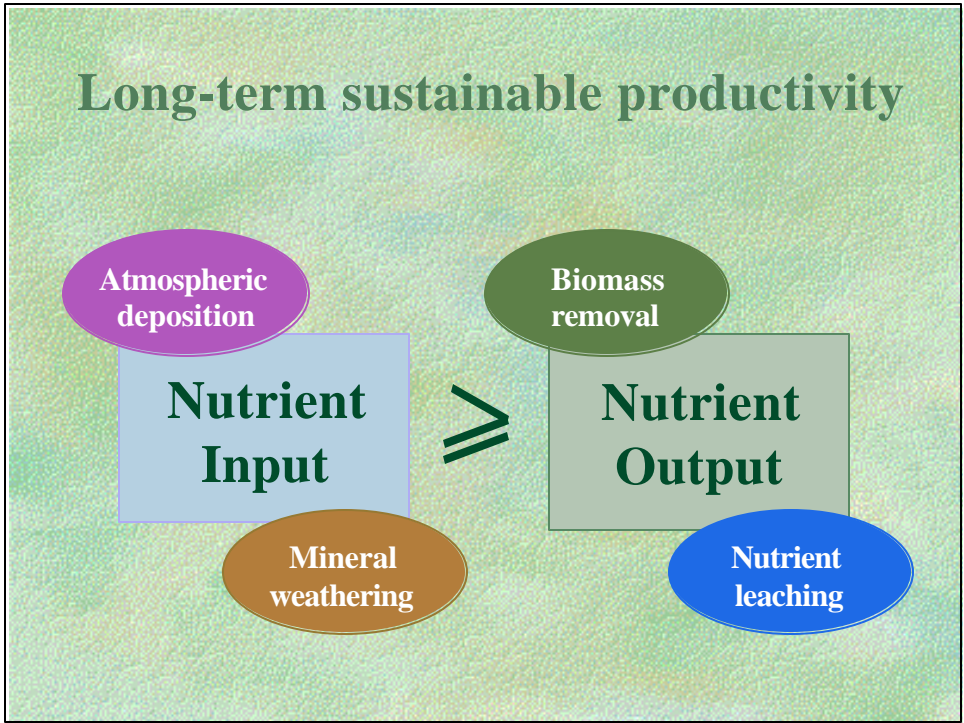
In principle, the net mass balance between external nutrient inputs and outputs determines the overall forest biomass production expectation, i.e., the latter must be numerically consistent with the former (Figure 2.2). One criterion for sustaining current forest biomass production rates can therefore be stated as follows:

*Long-term nutrient inputs to the forest site (atmospheric deposition + mineral weathering) must be at least equal to or higher than long-term nutrient outputs from the forest site (leaching + biomass removal).*

From this concept, sustainable harvest can be defined:

*Biomass harvesting is only sustainable when removal of nutrients via harvesting and corresponded leaching does not exceed the supply of the same nutrients received by chemical weathering of minerals and atmospheric deposition.*

If this condition is met, then, the risk of engendering nutrient imbalances in the forest vegetation will be reduced. Otherwise, forests will suffer serious health and growth reductions on account of insufficient nutrient supplies.



**Figure 2.2** The concept of sustainable forest biomass productivity based on the necessity of balance between external nutrient inputs and outputs.

Excess amounts of acid deposition (N and S) may cause soil acidification, soil nutrient imbalances, and eventually forest biomass growth reduction (Sverdrup and Warfvinge 1994). Therefore, critical acid deposition loads (N and S) can be used as a criterion for sustained forest biomass production (Sverdrup and Rosen 1998). A critical load has been defined as being the highest deposition of a compound that will not cause chemical changes which will eventually or directly lead to harmful effects on ecosystem structure and function (Nilsson 1986). Therefore, this particular sustainability criterion can be formulated as follows:

*The system is not long-term sustainable when the acid deposition (N and S) exceeds critical acid deposition loads in a specific area.*

The amount of excessive N and S loads depends directly on the ability of the plants and soil to absorb N and S, and on site-external factors that influence N and S removals such as harvesting and forest fires (Grennfelt and Thörnelöf, 1992).

## CONCLUSION

Nutrient elements that are available for forest biomass production reside in nutrient pools such as the mineral soil, the organic soil layers on top of the mineral soil, and the vegetation layer. Transfer of elements from one pool to the other occurs by way of geochemical, biogeochemical, and biochemical cycling. Primary nutrients imported to the system through atmospheric deposition, mineral soil weathering, and nutrients exported

from site by leaching and biomass harvesting (fire) are all part of the geochemical cycle. There is an assumption that maintaining long-term forest site productivity is directly dependent on the geochemical balance of primary nutrient inputs and outputs. Based on this assumption, biomass harvesting and/or acid deposition are sustainable only when do not cause nutrient input/output imbalances.

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CHAPTER 3  
INPUT/OUTPUT COMPONENTS OF THE LONG-TERM  
FOREST SUSTAINABILITY: A LITERATURE REVIEW

INTRODUCTION

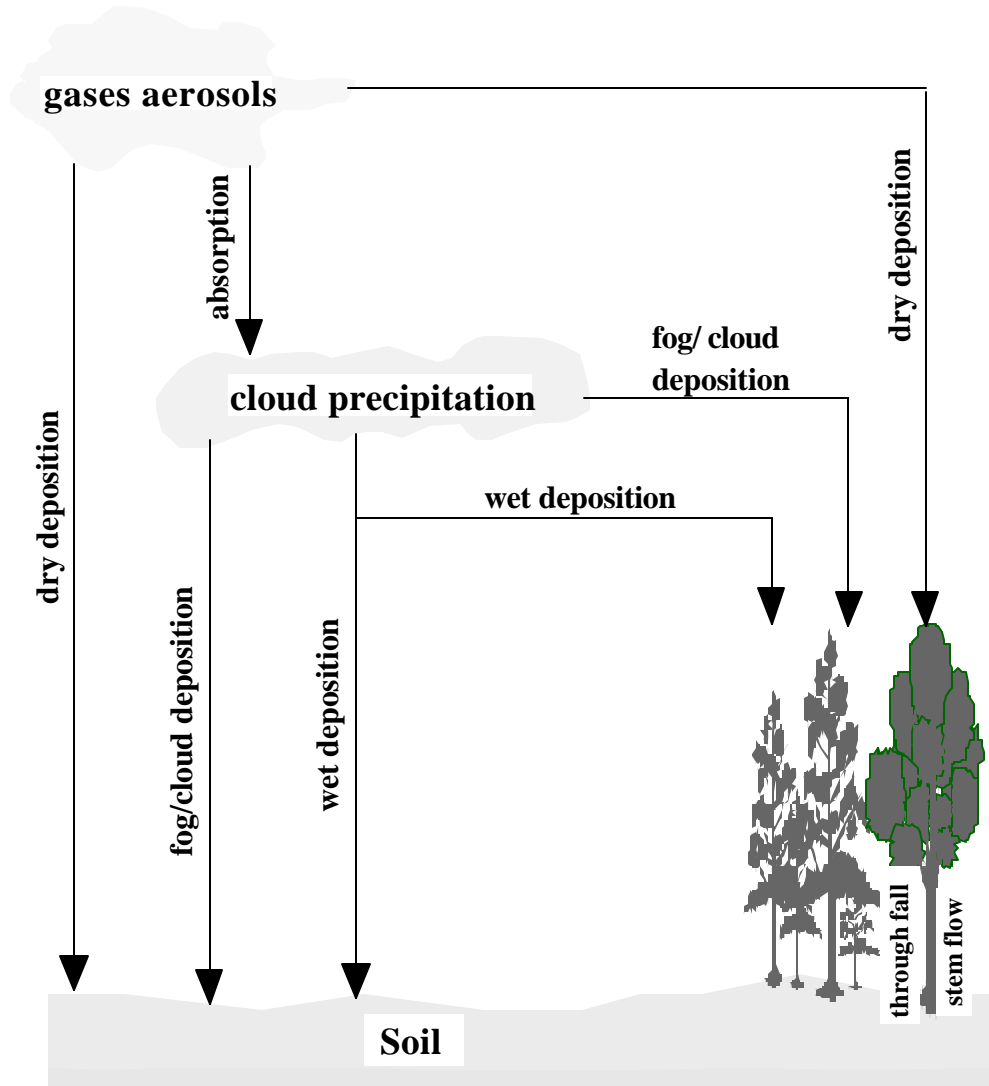
The objectives of this Chapter are:

1. To review the most important factors that control the primary nutrient inputs/outputs that affect long-term forest sustainability.
2. To describe a number of methods used for estimating the primary nutrient input/output rates.
3. To summarize data in annual input/output rates.

ATMOSPHERIC DEPOSITION

Chemical elements from the atmosphere are deposited on soil and/or vegetation through three main processes (Lovblad and Erisman 1992): wet deposition, fog deposition, and dry deposition (Figure 3.1).

Wet deposition is the movement of dissolved gases and large particles (greater than 20 micron) via incident precipitation in the form of rain, snow, sleet or hail (Smith 1990; Lin 1994). The amount of a nutrient deposited into a forest ecosystem is related to nutrient concentration in precipitation and the total amount of precipitation received.



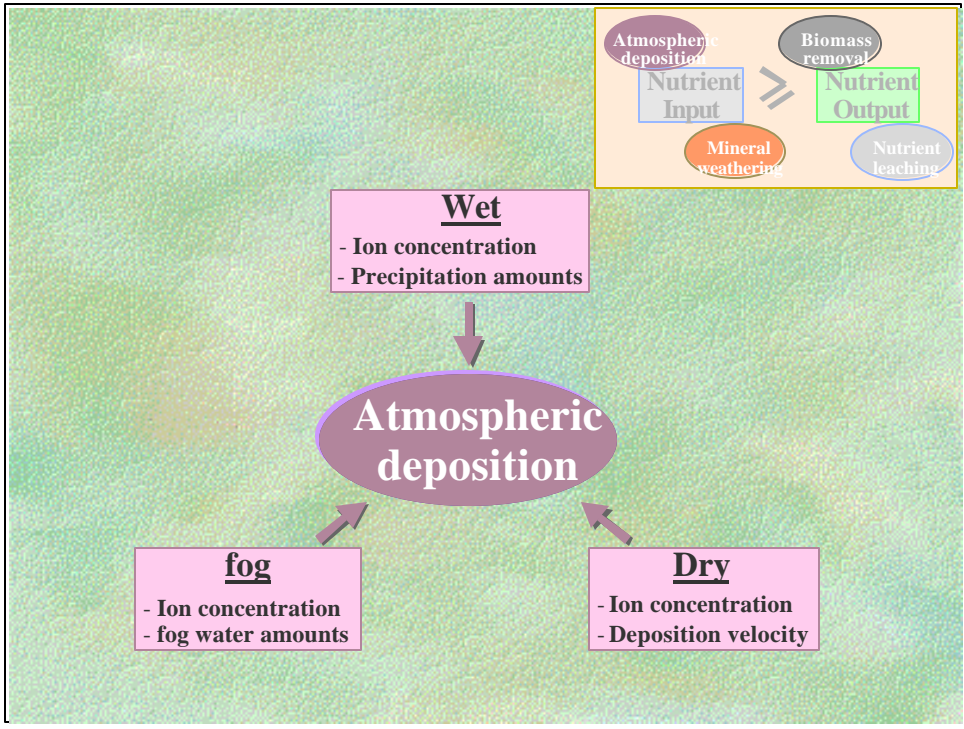
**Figure 3.1** Three processes of atmospheric deposition (re-drawn from Lovblad and Erisman, 1992).

Fog deposition is the movement of dissolved gases and small particles (5-30 micron) via fog (coastal areas) or cloud water (high elevation areas) droplets. Since fog and cloud droplets normally contain much higher concentrations of many elements than those in precipitation (Grennfelt and Thörnelöf 1992; Yanni 1996), significant amounts of elements might be deposited on forest soil and vegetation via fog deposition. In Canada, significant fog occurs frequently in the coastal areas of the Atlantic, Pacific, and Arctic oceans (Hare and Thomas 1974).

Dry deposition includes the deposition of aerosols, including fine particles (1-20 micron) and very fine particles (< 1 micron), on forest and soil surfaces through sedimentation and diffusion as well as absorption of gases (Smith 1990; Lin 1994). Collection and retention of aerosols are enhanced by surface roughness such as leaf hairs (Wedding *et al.* 1977). Trace gases, including acid rain precursors, are absorbed through the leaf stomata (Hicks *et al.* 1985).

#### Controlling Factors of Atmospheric Deposition Rates

Depending on the process of atmospheric deposition (wet, fog, dry), rates of ion deposition are controlled by various factors (Figure 3.2). Wet deposition is controlled by ion concentration in precipitation and amount of precipitation during a period of time. Fog deposition is controlled by ion concentration in fog droplets, intensity of fog water droplets, and fog frequency (fog hours) during a period of time. Dry deposition is controlled by ion concentration in dry deposition and amount of dry deposition.



**Figure 3.2** Controlling factors of three different processes of atmospheric deposition.

## Methods of estimating atmospheric deposition

### *Wet deposition*

Wet deposition can simply be calculated by the concentration of chemical contained in the precipitation times the amount of precipitation recorded (Smith 1990). Wet deposition data is usually the most accessible data.

### *Fog deposition*

Since quantifying fog water and fog deposition is not as simple as estimating precipitation deposition, various methods and mathematical models have been developed. These methods can be categorized as micrometeorological methods (Dollard *et al.* 1983), throughfall (Lovett and Kinsman 1990), and artificial surfaces (Joslin and Wolfe 1992).

Difficulty of calculating chemical elements deposited by fog is mainly related to lack of data (monthly or annual) for amounts of fog water droplets. Yin and Arp (1994) estimated fog deposition in the Maritime Provinces, Canada, by calculating the rate of deposition of fog water droplets and the number of fog hours. They calculated the rates of fog water droplets for closed forest canopies based on the following formula (Unsworth and Crossley 1987):

$$D_f = W \left\{ k_2 * u / \left[ \ln \left( \frac{Z - P}{Z_0} \right) \right] + V_s \right\} \quad (1)$$

where

$D_f$  = the rate of fog water droplets ( $\text{g m}^{-2} \text{s}^{-1}$ ),

$W$  = liquid water content in the fog ( $\text{g m}^{-3}$ ),

$k$  = von Kaman's constant (0.41),

$u$  = wind speed ( $\text{m s}^{-1}$ ) at  $Z$ ,

$Z$  = reference height (m),

$P$  = displacement plane (m),

$Z_o$  = roughness length (m),

$V_s$  = deposition velocity by sedimentation ( $\text{m s}^{-1}$ ).

Yin and Arp (1994) developed an empirical formula for calculating fog and mist frequencies in the Maritime Provinces, Canada as follows:

$$\ln(1 + H_f) = 16.2 - \frac{d}{266} + \frac{M(M^2 - 605)}{497} - \frac{(T + 2)^2}{208} \quad (2)$$

where

$H_f$  = the fog-and-mist frequency ( $\text{hrs month}^{-1}$ ),

$d$  = the distance (km) to the southern coast (the Bay of Fundy for New Brunswick, or the Atlantic for Nova Scotia and Prince Edward Island),

$M$  is the digital seasonal month (August: 8.5, September: 9.5, ..., July: 19.5),

$T$  is the monthly mean air temperature ( $^{\circ}\text{C}$ ).

### *Dry deposition*

Dry deposition is the most difficult to quantify. For field (direct) measurement different procedures can be used (Lovett 1987). *Micrometeorological procedures* estimate deposition as a function of pollutant depletion in the air above the canopy. *Accumulation inventories* measure the accumulation of deposition material on natural or artificial surfaces. Due to difficulties of direct measurement (Sirois and Vet 1988), dry deposition was usually estimated via indirect measurements called *inferential methods*. For example, one technique is estimating dry deposition rates from measured concentration of the ions in the air and representative dry deposition velocities for the ions as follows:

$$\text{Dry deposition rate} = \text{deposition velocity (vd)} * \text{ion concentration}$$

where deposition velocity (vd) is the relative rate at which gases and aerosols are removed from the atmosphere. This method is suitable for estimating dry deposition, but it is hard to find local data (particularly for base cation) on deposition velocity and concentrations.

### *Bulk deposition*

Bulk deposition is operationally defined as that which falls in a continuously open collector (Whitehead and Feth 1964) and thus contains both wet deposition and an unknown fraction of the particulate, aerosol, and gaseous material which eventually reaches the earth's surface. In order to measure ion inputs to an ecosystem from the

atmosphere, bulk deposition is more appropriate than wet-only deposition (Galloway and Likens 1978). However, bulk deposition often provides a poor estimate of the dry deposition, particularly for ions having significant deposition from the gaseous phase (Dillon *et al.* 1982).

### Rate of atmospheric deposition in Canada

Shown in Table 3.1 are values of atmospheric deposition for the Turkey Lake Watershed (TLW, Ontario), Lake Clair watershed (Quebec), and Kejimikujic National Park (Nova Scotia).

According to data provided by three different studies (Foster and Nicolson 1988; Nicolson 1988; Jeffries *et al.* 1988), bulk deposition values at TLW ranged 142-195, 67-73, and 27-37 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for Ca, Mg, and K, respectively. Mean values of bulk deposition ranged 608-769, 337-414, and 221-268 (eq ha<sup>-1</sup> yr<sup>-1</sup>) deposition for SO<sub>4</sub><sup>2-</sup>-S, NO<sub>3</sub><sup>-</sup>-N, and NH<sub>4</sub><sup>+</sup>-N, respectively (Table 3.1).

Sirois and Vet (1988) reported total deposition for SO<sub>4</sub><sup>2-</sup>-S, NO<sub>3</sub><sup>-</sup>-N over 4-year (1981-84) sampling and calculations of the wet-only and dry (excluding NO<sub>2</sub>) deposition in the TLW. Their mean values of total deposition (wet plus dry) were 726 and 432 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for SO<sub>4</sub><sup>2-</sup>-S, NO<sub>3</sub><sup>-</sup>-N, respectively. Sirois and Vet (1988) have shown that 13-20% of the total SO<sub>4</sub><sup>2-</sup> and 24-30% of the total NO<sub>3</sub><sup>-</sup> deposition occur in the dry form in the TLW.

**Table 3.1** Mean annual rates of precipitation and atmospheric deposition for various forest sites in eastern Canada.

Latitude	Longitude	Prov	Site	Years of records	Type of deposition	Precipitation (mm yr <sup>-1</sup> )	Ca <sup>2+</sup>	Mg <sup>2+</sup>	K <sup>+</sup>	NH <sub>4</sub> <sup>+</sup>	NO <sub>3</sub> <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	Source of data
							(eq ha <sup>-1</sup> yr <sup>-1</sup> )						
7.03	84.25	ON	Turkey Lake	1981-85	Bulk dep.	1225	194	68	28	221	337	608	Foster and Nicolson 1988
7.03	84.25	ON	Turkey Lake	1981-84	Bulk dep.	1240	195	73	37	221	398	769	Nicolson 1988
7.03	84.25	ON	Turkey Lake	1981-83	Bulk dep.	1336	142	67	27	268	414	699	Jeffries et al 1988
5.57	71.40	QC	Lake Clair	1988-94	Bulk dep.	1300	150	33	23	333	472	660	Houle et al. 1997
5.57	71.40	QC	Lake Clair	1988-94	Wet dep.	1300	91	21	13	317	442	618	Houle et al. 1997
4.38	65.25	NS	Kejimkujik	1979-91	Wet dep.	1391	63	83	16	67	136	377	Yanni 1996
4.38	65.25	NS	Kejimkujik	1992	Fog dep.	100	27	81	8	57	84	187	Yanni 1996
4.38	65.25	NS	Kejimkujik	1992	Dry dep.	0	12	34	4	3	12	19	Yanni 1996
4.38	65.25	NS	Kejimkujik		Total dep.	1491	102	198	28	128	232	583	Yanni 1996

Values of bulk deposition in the Lake Clair Watershed (estimated from values of ion concentration and precipitation reported by Houle *et al.* (1997) were 150, 33, and 23 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for Ca, Mg, K, and were 660, 472, and 333 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for SO<sub>4</sub><sup>2-</sup>-S, NO<sub>3</sub><sup>-</sup>-N, and NH<sub>4</sub><sup>-</sup>-N, respectively. Values of estimated wet-only deposition were equal to 61, 64, and 57% of the bulk deposition of Ca, Mg, K, and were 94, 94, and 95% for SO<sub>4</sub><sup>2-</sup>-S, NO<sub>3</sub><sup>-</sup>-N, and NH<sub>4</sub><sup>-</sup>-N, respectively.

Yanni (1996) estimated wet, fog, and dry deposition for Kejimikujik National Park (Table 3.1). Total (wet + fog + dry) deposition rates were 102, 198, 28 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for Ca, Mg, K, and were 583, 232, 128 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for SO<sub>4</sub><sup>2-</sup>-S, NO<sub>3</sub><sup>-</sup>-N, and NH<sub>4</sub><sup>-</sup>-N, respectively. The total S and N deposition rates in this area tend to be lower than bulk S and N deposition at TLW and in the Lake Clair Watershed. Of total deposition in Kejimikujik, the contribution of wet deposition was 42-62% for base cations and was 52-65% for SO<sub>4</sub><sup>2-</sup>-S and NO<sub>3</sub><sup>-</sup>-N. Fog deposition was found to contribute 26-41% for base cations and 32-45% for SO<sub>4</sub><sup>2-</sup>-S and NO<sub>3</sub><sup>-</sup>-N. Annual precipitation rate was 1391 mm and annual fog deposition was 100 mm. The amounts of cations and anions deposited by fog were relatively high because of high ion concentrations in fog water. The ratio of dry to total deposition in Kejimikujik was 12-17% for base cations and 2-5% for SO<sub>4</sub><sup>2-</sup>-S and NO<sub>3</sub><sup>-</sup>-N. The rates of dry deposition in the Yanni study were obtained by analyzing the monthly contents of an air sedimentation sampler that operated from May 1992 to October 1992.

Sirois and Barrie (1988) reported a 4 year (1979-1982) total deposition rate (excluding fog deposition) for SO<sub>4</sub><sup>2-</sup> and NO<sub>3</sub><sup>-</sup> for Kejimikujik. The mean total SO<sub>4</sub><sup>2-</sup> and NO<sub>3</sub><sup>-</sup> were 540 and 190 (eq ha<sup>-1</sup> yr<sup>-1</sup>), respectively. Of these total depositions, 21% of

$\text{SO}_4^{2-}$  and 18% of  $\text{NO}_3^-$  contributed by dry deposition. These ratios of dry/total deposition are much higher than those reported by Yanni (1996), mainly due to excluding fog deposition.

Local data for dry deposition is rare in Canada so that several attempts have been made to introduce a regional ratio of dry/total deposition or dry/wet deposition. For example, Galloway and Whelpdale (1980) estimated that in all of eastern Canada 29% of total  $\text{SO}_4^{2-}$  deposition was dry deposition. A budget estimate by Logan (1983) indicated that in eastern Canada dry deposition of  $\text{NO}_3^-$  was in the range 17% to 29% of the total deposition. Shannon and Sisterson (1992) estimated regional dry deposition rates for Canada based upon the provincial average rates of wet deposition (obtained from Environment Canada) and a regional dry/wet deposition ratios. For eastern Canada, they applied the dry/wet ratios of 0.28 for S and 0.26 for total nitrate deposition (obtained from Sirois and Barrie 1988). For the Prairie Provinces, they assumed that the dry/wet ratio is twice that in the east because rainfall is reduced in the Prairie Provinces. For British Columbia, they assumed to have the same relationship to wet deposition as in eastern Canada.

In Canada, local or regional information of dry deposition for cations ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ , and  $\text{NH}_4^+$ ) is rare (if any). According to data provided for Kejimikujik National Park, Nova Scotia (Yanni 1996), the ratios of dry/wet deposition (excluding fog water deposition) for  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ , and  $\text{NH}_4^+$  were 0.19, 0.41, 0.23, and 0.04, respectively. These ratios are estimated based upon a one-year record of dry deposition. More temporal and spatial dry deposition data are needed to suggest an appropriate regional dry/wet ratio for cations.

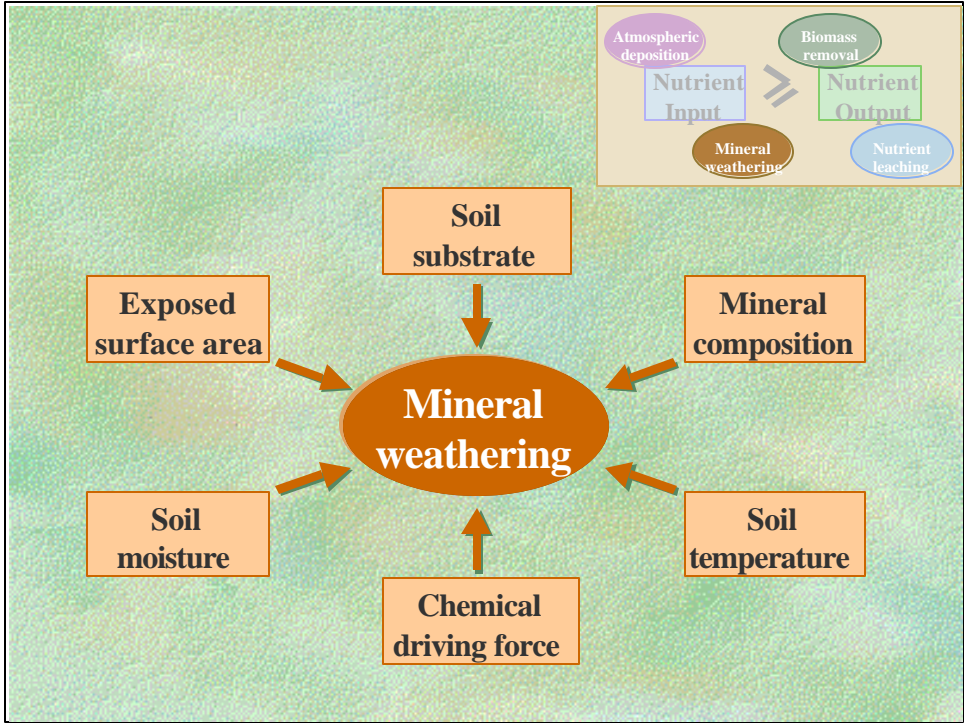
## WEATHERING INPUT

Mineral weathering is an important source of nutrient input for plant growth. Except for nitrogen, which is rarely available in minerals, other essential nutrients such as Ca, Mg, K, and P are largely supplied by the dissolution of primary and secondary minerals. In the following, a general review of weathering processes and rate-controlling factors is provided. A critical review of various laboratory and field methods of estimating weathering rates is conducted to determine a reliable and applicable estimation method.

The main concern of this review is with the chemical weathering of minerals. Physical (mechanical) weathering is not directly considered here. Physical weathering by itself does not release nutrients for plant growth (White *et al.* 1990). However, it can increase potential chemical alteration and nutrient release rates by breaking rock masses into smaller particles, thereby increasing the reacting surface area. This is important where soils are forming over bedrock. Where the soil substrate has been previously broken down (sedimentary deposits such as a glacial or alluvial deposits), mechanical processes are less important (Trudgill 1977).

### Controlling Factors of Weathering Rates

The rate of chemical weathering is governed by various factors that can be categorized into soil substrate factors, climatic factors, and chemical driving forces (Brady 1990; Sverdrup and Warfvinge 1993), see Figure 3.3.



**Figure 3.3** Controlling factors of soil weathering.

Soil substrate factors include chemical composition, crystal size, and structure of minerals. Depending on structure and chemical composition, the resistance of minerals to weathering differs. For example, Goldich (1938) showed the order of weathering resistance of some common minerals: quartz (most resistant) > muscovite > K-feldspars > biotite > amphibole > pyroxine > olivine (least resistant). Typically, soil parent materials contain various minerals. Therefore, the type of substrate mix indicates the ability of the soil to provide nutrients for plant growth. Dissolution of minerals occurs at mineral surface (Hodson *et al.* 1998) and thus particle surface areas are an important factor affecting weathering rates. Finer mineral crystals provide larger surface areas compared to coarser particles.

Soil temperature and soil moisture are two important factors which affect the weathering rates. Soil temperature, for example, controls the weathering rate by affecting the rate of chemical reaction of the soil minerals. Increased soil temperature generally increases chemical weathering rates (White *et al.* 1990). The availability of water is also crucial for driving the gradual dissolution of the minerals, and for removing the soluble products of weathering. The availability of water is governed by local rates of precipitation, evapotranspiration, and percolation.

Kinetic expressions describe the dependency of the weathering rates on the solution concentration of weathering reactants. Several chemical reactions between the mineral and constituents in the liquid solution (e.g., the reaction with the  $H^+$ ,  $Al^{3+}$ ,  $CO_2$ , base cations, and organic acids) contribute to the total rate of weathering of silicate minerals (Sverdrup and Warfvinge 1993).

The pH value of soil solution is a measure of the  $H^+$  ion concentration or the soil solution. High  $H^+$  concentrations (low pH) contribute to a rapid dissolution of minerals by increasing the rate of hydrolysis reactions and by governing species solubility (Kittrick 1969). Under this condition (low pH and base cation concentration), a congruent (complete) dissolution of the available minerals may occur. But, under less acid condition (higher base cation concentration), incongruent weathering of minerals will produce insoluble clay minerals (van der Salm 1999).

Atmospheric deposition can cause substantial changes in soil solution chemistry and thereby influence the extent of weathering (White *et al.* 1990; van der Salm 1999). For example, S and N deposition will generally lead to increased  $H^+$  concentrations and to decreased base cations concentration in the soil solutions. With low pH, the release of base cations from the soil minerals will increase due to both an increased weathering of primary and secondary minerals (van der Salm 1999).

The soil solution pH also affects mineral stability by controlling the solubility of Al oxides/hydroxides. In general, more Al is soluble in an acid medium than in a neutral or alkaline medium. As the dissolution of Al compounds increases, part of the  $H^+$  ions will be neutralized. Sverdrop and Warfvinge (1993) noted that minerals rich in Al show a stronger weathering rate dependence on dissolved Al than minerals poor in Al. Under certain circumstances, large amounts of Al in the soil may overwhelm the effect of pH, and the weathering rate may decrease.

Carbon dioxide is always present in soil pores in a wide range of concentrations (commonly 0.5-5% but it can be as high as 10-20%), and is known as an active

contributor to soil mineral weathering. The dissociation of CO<sub>2</sub> in water is source of carbonic acid (Kern 1960). This acid is unstable and dissociates as follows:



This equation indicates that effect of CO<sub>2</sub> on the weathering rate will increase as the CO<sub>2</sub> concentrations increase in the soil pores.

Organic acids also have a capacity to facilitate the weathering of soil minerals by producing H<sup>+</sup> ions and CO<sub>2</sub> (Trudgill 1977). Weathering in the presence of organic acids may proceed at a faster rate than would be expected in the presence of inorganic acids alone (Huang and Keller 1972). In this context, the role of water percolation is vital to carry the organic substances from the upper layers to the lower layers of the soil.

In general, we can conclude that the weathering rate of minerals will be changed positively by increased concentrations of H<sup>+</sup>, CO<sub>2</sub>, and organic acids, and it will be changed negatively by increased concentrations of Al<sup>3+</sup> ions and base cations in the soil solution. The total rate will be the sum forward rate minus the sum backward rate.

#### Methods of estimating weathering rates

Quantitative estimates of weathering rates are complicated owing to sampling and analytical errors, and high degree of spatial and temporal variations within the biogeochemical system (Clayton 1979). The recognition of this complexity led to a wide variety of methods for estimating weathering rates (e.g., Sverdrup and Warfvinge 1988; White *et al.* 1990; Van Miegroet *et al.* 1994; Bain and Langan 1995; Kolka *et al.* 1996; van der Salm 1999). These methods can be categorized as (1) field studies including

watershed mass-balance and historical mineral (elemental) depletion, (2) laboratory studies such as mineral dissolution experiments, soil batch studies, and soil column studies, and (3) mathematical models based on statistical and mechanistic methods.

Mass-balance studies of watersheds estimate weathering rates over relatively large areas through consideration of nutrient inputs from atmosphere, nutrient losses by leaching or erosion, and net changes in nutrient storage in plants and soil pools (Clayton 1979; Zabowski 1990; White *et al.* 1990). This approach is appropriate in areas where bedrock is impermeable so that deep seepage can be considered zero. Mass-balance studies, however, are time-consuming because such studies require many flux measurements, many of which are difficult to do (van der Salm 1999).

Elemental depletion studies of soil estimate long-term average weathering rates based on the reduction of bases cation contents in the soil profile (April *et al.* 1986). Assumptions include minimal surface erosion, uniform parent material, insignificant storage of base cations in biomass, and loss of all weathered base cations from the system (Kolka *et al.* 1996). This field method focuses on weathering rates of the past, and provides information over long time periods. Therefore, the weathering rates of this approach may deviate from present date weathering rates due to differences in climate and soil chemical conditions (van der Salm 1999).

Laboratory studies estimate release of base cations by percolating chemical solutions through a soil sample (e.g., Sverdrup and Warfvinge 1987; Wesselink *et al.* 1994; Kolka *et al.* 1996). Mineral dissolution experiments emphasize individual mineral species that are important to understand weathering mechanisms while batch studies (similar approach) use native soils with mixed mineral suits (Kolka *et al.* 1996). In

column studies, soil columns (i.e., soils packed into a cylinder) are leached either by gravity flow or by pumping leachate solutions (White *et al.* 1990). Laboratory studies provide a relatively quick and reliable means to determine weathering rates (van der Salm 1999). However, extrapolation of rates to natural systems is difficult. Measured cation released rates by laboratory approaches are commonly up to three order of magnitude higher than rates determined from field studies (Velbel 1990). These differences may be due to differences in climatic and hydrological conditions, and due to solution concentrations used in the laboratory studies (Wesselink *et al.* 1994).

Statistical models are developed based on generalizing the results of many studies of weathering. In this context, weathering rates obtained in the field or laboratory studies are related to common soil characteristics (e.g., soil substrate, soil texture, bulk density, exchangeable cations, etc.) by way of regression equations. These empirical equations are very simple to estimate weathering rates, but can only be applied for a limited range of soils (van der Salm 1999).

An alternative method to estimate local or regional weathering rates is to apply a mechanistic model. One example of doing so is represented by the steady-state soil chemistry model PROFILE (Warfvinge and Sverdrup 1992), which is a multi-soil-layer model for predicting weathering rates and the degree of soil acidification. The rate of weathering in this model is assumed to be proportional to (1) the exposed surface area of soil mineral, (2) the degree of soil moisture saturation, (3) soil temperature, and (4) chemical driving forces ( $H^+$ ,  $Al^{3+}$ ,  $CO_2$ , and dissolved organic acids).

### Rates of mineral weathering in Canada (examples)

Shown in Table 3.2 are weathering rates for tolerant hardwood and jack pine sites in eastern Canada. The weathering rates of base cations range 312-750 ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) for hardwood sites, and 222-484 ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) for jack pine sites.

Wishart Lake and Norberg Creek sites (Morrison 1990) are closely located within the Turkey Lakes Watershed, Ontario, and have identical soil parent materials and cover types. The released rates of Ca, Mg, and K were estimated to be 382, 324 and 44 ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) at Wishart Lake, and 304, 171 and 55 ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) at Norberg Creek. Overall base cation weathering values were higher at Wishart Lake than at Norberg Creek due to a higher base content of the soil substrate. Base cation weathering rates of the Lake Clair site (Houle *et al.* 1997) were relatively low ( $312 \text{ eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) due to a low base content in the soil substrate. The release rates of Ca, Mg, and K were 286, 19, and 7 ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ), respectively.

Among jack pine sites, base cation weathering rates were relatively high ( $484 \text{ eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) at Wells, intermediate ( $329 \text{ eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) at Dupuis, and low ( $222 \text{ eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) at Dryden.

**Table 3.2** Estimated annual rates of soil weathering for various forest sites in eastern Canada.

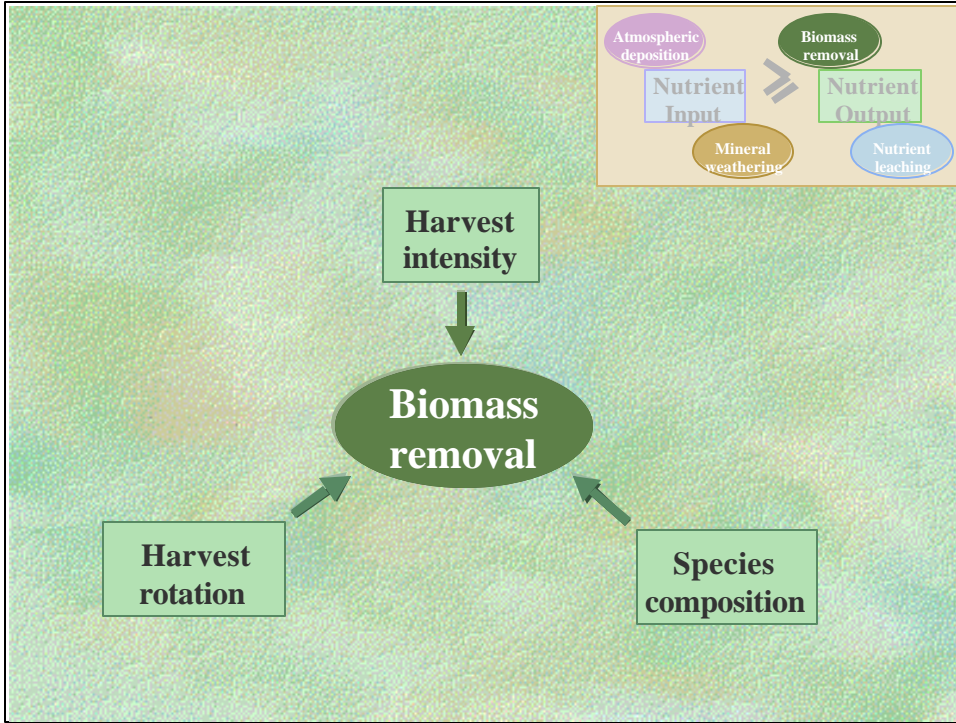
Latitude	Longitude	Prov.	Site	Soil substrate	Cover type	Ca <sup>2+</sup>	Mg <sup>2+</sup>	K <sup>+</sup>	BC	Site Reference
						(eq ha <sup>-1</sup> yr <sup>-1</sup> m <sup>-1</sup> )				
7.03	84.25	ON	Wishart Lake	Abl. & basal Till (higher base)	SM <sub>82</sub> YB <sub>10</sub>	382	324	44	750	Morrison 1990
7.03	84.25	ON	Norberg Creek	Abl. & basal Till (lower base)	SM <sub>93</sub> YB <sub>3</sub>	304	171	55	530	Morrison 1990
6.57	71.40	QC	Lake Clair	Till (gneiss & granite)	SM <sub>77</sub> YB <sub>15</sub>	286	19	7	312	Houle et al. 1997
6.25	83.23	ON	Wells		JP <sub>100</sub>	382	60	42	484	Foster et al., 1995
7.38	83.15	ON	Dupuis		JP <sub>100</sub>	160	105	64	329	Foster et al., 1995
0.02	92.32	ON	Dryden	Outwash	JP <sub>100</sub>	147	39	36	222	Morrison and Foster, 1974

## NUTRIENT EXPORT BY HARVESTING

Harvesting removes essential nutrients from the site through removing biomass, through leaching of nutrients from logging slash, and from the forest floor. Harvesting can therefore cause nutrient depletion and, consequently, growth reductions in the next rotation. Exports of nutrients from the forest site are mainly associated with direct biomass removal. Losses of nutrients through accelerated erosion, leaching, or gaseous loss are relatively small.

Direct removal of nutrients via exporting biomass can be quantified by multiplying nutrient concentration in tree components with biomass weight of removed tree components. Furthermore, the amounts of nutrients exported by biomass removal depend on the intensity of harvesting (Hüttl and Scaaf 1995), on stand age (rotation length), and on species composition (Figure 3.4).

Harvesting might be operated through clear-cutting (highest intensity), partial cutting (moderate intensity), or selection cutting (lowest intensity). The intensity of a harvest may include removing stems only (SOH), or stems plus branches (SBH), or even entire trees (WTH). Nutrient removals are increased greatly if branches and leaves are included (Binkley 1986). Kimmins (1977) revealed that whole-tree harvesting (WTH) removes from 1.5-4.0 times more nutrients than does bole-only harvesting.



**Figure 3.4** Controlling factors of harvest removal.

The removal of nutrients increases with short rotations. As Kimmins (1987) has noted, sapwood generally has higher nutrient concentrations than heartwood, and nutrient concentrations in crowns are higher than in stems. The ratio of sapwood to heartwood in younger stands is higher than in older stands, as is the ratio of crown biomass to stem biomass. Harvesting stands in younger age (short rotation) increases the proportion of sapwood to heartwood and crown to stem materials in harvested biomass. Therefore, more nutrients are removed from site with shortening rotation. For example, Boyle (1975) showed that reducing rotation age from 30 years to 10 years with the same harvesting method (WTH) would increase the nutrient removal of N, P, K, and Ca by 345, 239, 234, and 173%, respectively, for aspen.

Species composition of the forest cover affects the rate of nutrient uptake and therefore, and the rate of nutrient export via biomass removal (Johnson *et al.* 1988a, 1988b). Hardwoods take up greater amounts of nutrients and, therefore, export more nutrients through biomass removal than conifers.

Indirect nutrient losses via increased post-harvest leaching could be due to increased rates of decomposition, nitrification, and reduced nutrient uptake (Binkley, 1986). Increased post-harvest leaching usually occurs for limited years following the time of harvesting. Therefore, post-harvest is much more important for short-term sustainability studies than the long-term studies. A study of clear-cutting effects in British Columbia revealed that losses of dissolved nitrogen in stream-water 2 years after clear-cutting increased about 4 times the first year, and 14 times the second year, compared to the losses before clear-cutting (Kimmins 1987). Jewett *et al.* (1995) showed that from a

harvested forest watershed in New Brunswick, N, K, and Ca were the main leaching losses during the first 6 post-harvest years. Leaching losses of nitrogen and other nutrients following harvesting are normally much less than nutrient exports in biomass.

Sites vary in their response to nutrient removal, depending on their potential for nutrient replacement. Many detailed studies have documented the effects of stem-only and whole-tree harvesting (Kimmins 1977; Mann *et al.* 1988; Foster and Morrison 1989; Compton and Cole 1991; Foster 1996; Olsson *et al.* 1996a, 1996b). Not surprisingly, it was found that sites with small nutrient pools are particularly sensitive to enhanced rates of nutrient removal. On nutrient-rich sites, whole-tree harvesting may or may not affect total ecosystem nutrient content in a significant way (Hornbeck and Kropelin 1982; Hendrickson *et al.* 1989; Compton and Cole 1991). It also reported that calcium is the nutrient most significantly depleted from most sites (e.g., Johnson *et al.* 1988a, 1988b). In some cases, it is expected that even high rates of weathering will not prevent the development of calcium deficiencies in many forest ecosystems (Federer *et al.* 1989).

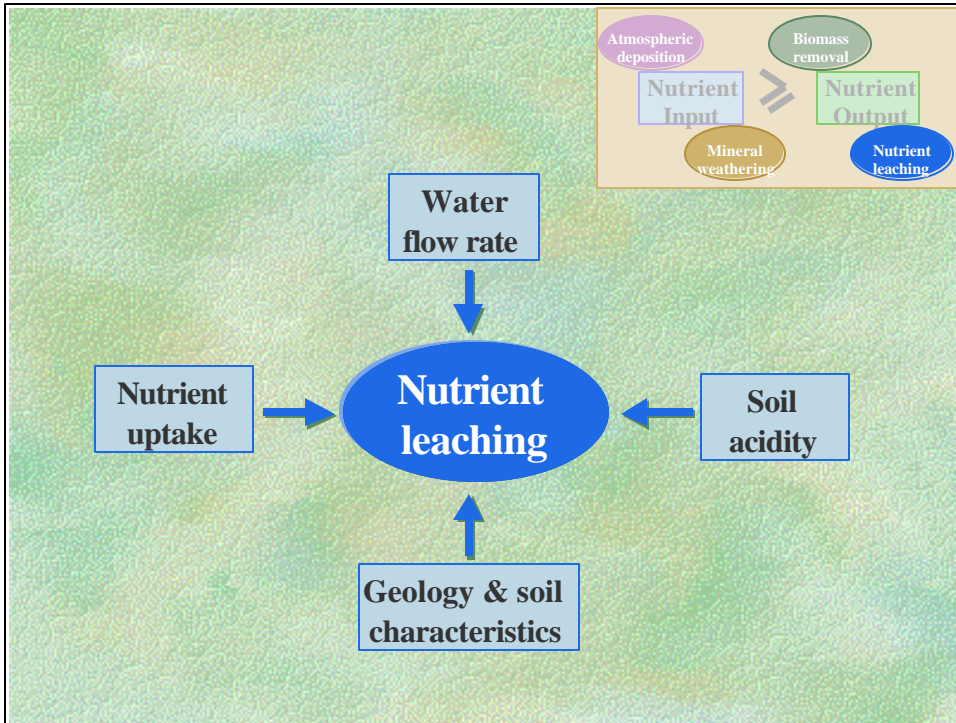
## LEACHING OUTPUT

The removal of nutrients via leaching is generally controlled by water flow rate, vegetative uptake, soil solution availability, soil acidity, soil properties and geological characteristics (Figure 3.5).

Leaching of nutrient is associated to amounts of draining water from forest site. In general, nutrient leaching from areas with wet climate and well-drained soils is higher than that from areas with dry climate or poor soil drainage. In certain areas, leaching of nutrient is much higher during spring than during summer.

Uptake is the opposite process of leaching and there is an interaction between uptake and leaching of nutrients. For example, Johnson and Todd (1987) found that calcium uptake was four times greater with mixed oak than with loblolly pine. However, calcium leaching was considerably greater with loblolly pine than with mixed oak. The overall calcium exports via leaching and uptake (harvesting) were almost identical for the two sites.

The rate of leaching is also correlated to the amounts of available nutrient pools in soil solution. For example, the amount of available N in boreal forests is usually low so that it is known to be the most growth-limiting nutrient. Many studies revealed that nitrate is strongly retained within ecosystems so that losses may decline to near zero as the stand matures (Vitousek and Reiners 1975). Leaching of non-limiting nutrients (e.g., K, Mg, and Ca) will not approach zero.



**Figure 3.5** Controlling factors of soil leaching.

Soil acidification has positive effects on leaching of base cations. Increasing soil acidity changes more insoluble Al into soluble  $Al^{3+}$  and therefore increases the concentration of  $Al^{3+}$  ions in the soil solution. Since there is higher tendency to absorb  $Al^{3+}$  by colloids than base cations, more Ca, Mg, and K are released into the soil solution and these are eventually leached from the site. Atmospheric deposition of acid anions ( $SO_4^{2-}$  and  $NO_3^-$ ) can either be beneficial or harmful for long-term forest biomass production, depending on soil nutrient status and on the amount of input (Johnson and Ball, 1990). Trees need both S and N for biomass growth. However, excess amounts of these elements cause soil acidification, soil nutrient imbalances, and eventually forest biomass growth reductions (Sverdrup and Warfvinge, 1994). On nutrient-poor soils (e.g. northern Ontario), most nitrogen from acid deposition is absorbed by foliage or is retained by the soil. Therefore, in these areas, where nitrogen is the most limiting factor for tree growth, retention of atmospheric nitrogen has a positive effect on tree growth. However, continued deposition may eventually lead to soil acidification and leaching of nutrients from the soil (Hall *et al.* 1997).

#### Rates of nutrient leaching in Canada

Study on  $NO_3^-$  exports from a group of 21 watersheds within the Duffin Creek drainage basin in southern Ontario (Hill 1978) showed that concentrations of  $NO_3^-$  were lower than  $5 \text{ mg L}^{-1}$  in all stations. A minimum concentration happened during summer (May-October) while  $NO_3^-$  concentrations increased and remained at elevated level during November to April. The average annual  $NO_3^-$  export was reported to range from

1.41 to 7.31 kg ha<sup>-1</sup> (50-70% during Feb.-Apr., but < 10% during June-Sep.). High positive correlation with crop area and significant negative correlation with abandoned farm and wetland ponds were found. From another study, it was found that the average (1976-1984) annual export of NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> among 32 study catchments in central Ontario were 0.06-0.67 and 0.36-33.36 kg ha<sup>-1</sup>, respectively (Dillon *et al.* 1991).

Shown in Table 3.3, are the mean annual export (leaching from soil) of cations, anions, and total available N (NH<sub>4</sub><sup>+</sup> plus NO<sub>3</sub><sup>-</sup>) of several forest sites located in Ontario (ON), in the Maritime Provinces (NB, NS, PEI), and in Newfoundland (NF). The soil leaching rates were 116-1350 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for Ca<sup>2+</sup>, 80-308 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for Mg<sup>2+</sup>, 17-83 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for K<sup>+</sup>, 7-77 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for NH<sub>4</sub><sup>+</sup>, 2-427 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for NO<sub>3</sub><sup>-</sup>, 352-1294 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for SO<sub>4</sub><sup>2-</sup>, and 13-505 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for total available N (NH<sub>4</sub><sup>+</sup> plus NO<sub>3</sub><sup>-</sup>).

## CONCLUSION

The literature was reviewed to collect relevant information about geochemical inputs/outputs of forest ecosystems in eastern Canada. The focus was on summarizing information regarding controlling factors, on estimation methods, and on field-derived data. This information can be used for development, calibration, and validation of a suitable forest sustainability model.

**Table 3.3** Mean annual Leaching rates for various forest sites in eastern Canada.

Latitude	Longitude	Prv	Site (watershed)	Species	Gross terrestrial export (eq ha <sup>-1</sup> yr <sup>-1</sup> )						Source of data
					Ca <sup>2+</sup>	Mg <sup>2+</sup>	K <sup>+</sup>	NH <sub>4</sub> <sup>+</sup>	NO <sub>3</sub> <sup>-</sup>	SO <sub>4</sub> <sup>2-</sup>	
47.03	84.15	ON	Turkey Lake	Hard wood	1188	236	46	8	211	767	Nicolson 1988
47.03	84.15	ON	Turkey Lake	Hard wood	1350	265	49	7	191	860	Jeffries et al 1988
46.21	83.23	ON	Wells (Kirkwood)	Jack pine	352	130	42	7	7	1294	ENFOR
47.38	83.15	ON	Chapleau	Jack pine	222	80	31	11	2	700	ENFOR
46.46	66.56	NB	Deersdale	Mixed wood	116	113	83	39	116		ENFOR
44.38	65.25	NS	Kejimkujik	Mixed wood	269	308	41		6	352	Yanni 1996
45.12	62.38	NS	West Brook Rd	Mixed wood	296	122	55	34	347		ENFOR
45.18	62.50	NS	Dickey Lake Rd	Mixed wood	372	126	36	33	177		ENFOR
45.18	62.55	NS	West River Stn	Mixed wood	403	95	37	77	427		ENFOR
45.23	61.24	NS	McIntyres Mtn	Mixed wood	493	220	46	53	340		ENFOR
46.07	62.45	PEI	Grandview	Mixed wood	468	199	67	38	94		ENFOR
49.01	56.09	NF	Badger West	Birch	372	138	17	20	35		ENFOR
49.07	55.59	NF	Moose Pond	Birch	215	137	22	17	27		ENFOR
49.03	55.59	NF	Middletown Lake	Birch	318	188	20	29	33		ENFOR

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## CHAPTER 4

### SOIL WEATHERING MODEL

#### INTRODUCTION

Weathering of primary minerals is an important process to consider in site productivity assessments because the weathering process provides essential nutrients for plant growth, and also plays a major role in neutralizing naturally and anthropogenically induced soil acidity. Quantitative estimates of weathering rates, however, are difficult to obtain because of sampling and analytical errors, high degrees of complexity and spatial and temporal variations of the biogeochemical components and functions within soils (Clayton 1979).

Mechanistic models such as the PROFILE model (Sverdrup and Warfvinge 1993) provide a means to estimate local or regional weathering rates and acid neutralizing capacities for given soils from knowledge about:

1. the soil mineralogy complex (i.e., a specification of the soil mineral composition (%)),
2. the soil internal surface area that undergoes the weathering reactions,
3. the degree of soil moisture saturation,
4. the soil temperature, and
5. chemical driving forces such as the  $H^+$  and  $Al^{3+}$  concentrations (or activities),  $CO_2$  pressure, and dissolved organic acid.

The PROFILE model is in HyperCard, which is a user-friendly but cumbersome card-by-card programming, data entry, and model evaluation tool. In this chapter, the PROFILE model is re-programmed and evaluated by way of a high performance programming software called ModelMaker (ModelMaker 3, 1997). With ModelMaker:

1. all model equations, parameters and model component connections can be displayed on the computer screen for easy reference and cross-checking,
2. data are read directly from existing spreadsheets according to the specified data entry sequence,
3. the resulting PROFILE reformulation is fully compatible with the proposed Forest Sustainability model (ForSust, see Chapter 5).
4. the performance of the model can be evaluated by way of built-in procedures for iterative sensitivity analysis, parameter optimization, and statistical evaluation.

The objectives of this Chapter are:

1. to establish a forest soil weathering model by the ModelMaker programming software based on the concepts, principles and equations of the PROFILE model.
2. to check the consistency of the reprogrammed soil weathering model (ModelMaker) with the PROFILE (HyperCard) using soil mineralogy data from seven well-studied forest sites in North America.
3. to do sensitivity analysis to determine the most important variables affecting on weathering rates.

## METHODS

The assumptions and theories of the PROFILE model (Sverdrup and Warfvinge 1993) were used to provide a soil weathering model (sub-model) by the ModelMaker programming software. The PROFILE model is based on the theories of chemical kinetics for the context of soil chemical weathering. The prevalent assumption of this model is that the weathering equations as derived from laboratory data for pure minerals are universally valid for any soil matrix. With PROFILE, weathering rates of 14 common minerals, namely K-feldspar, plagioclase, albite, hornblende, pyroxene, epidote, garnet, biotite, muscovite, chlorite, vermiculite, apatite, kaolinite and calcite are calculated (Sverdrup 1990) for field conditions. As such, the model requires input data for: soil substrate mineralogy, soil depth, moisture content, bulk density, soil temperature, CO<sub>2</sub> pressure, dissolved organic carbon in solution, weight fractions of soil particles, weight fractions of soil oxides, pH, BC ion concentrations in solution and the acid neutralizing capacity (ANC) of the soil solution.

### Formulations

The weathering rate, as already stated, is assumed to be a function of soil mineral composition, exposed surface area of each mineral, soil moisture content, soil temperature, and soil depth. The total weathering rate for a soil type is calculated as the sum contribution from each mineral as follows (Sverdrup and Warfvinge 1993):

$$R_w = \sum_{j=1}^{mineral} r_j \cdot A_{exp} \cdot X_i \cdot \theta \cdot Z \quad (1)$$

where  $R_w$  = total weathering rate in soil horizon ( $\text{keq m}^{-2} \text{ s}^{-1}$ ),  $r_j$  = reaction rate of mineral  $j$  ( $\text{keq m}^{-2} \text{ s}^{-1}$ ),  $A_{\text{exp}}$  = exposed surface of soil minerals ( $\text{m}^2 \text{ m}_s^{-3}$ ),  $X_i$  = surface area fraction of mineral  $j$  in the mineral matrix,  $\theta$  = soil moisture saturation ( $\text{m}^3 \text{ m}^{-3}_{\text{soil}}$ ),  $Z$  = soil depth (m).

For most minerals, reactions with  $\text{H}^+$ ,  $\text{H}_2\text{O}$ ,  $\text{OH}^-$ , dissolved  $\text{CO}_2$ , Al and base cations, and strongly complexing organic acids in the liquid phase have been quantified (Sverdrup 1990). A general equation for the base cation release rate (dissolution rate) of a single mineral at constant temperature is given by (Sverdrup 1990, Sverdrup and Warfvinge 1993):

$$r = k_{\text{H}^+} \cdot ([\text{H}^+]^{\eta_{\text{H}}}/f_{\text{H}}) + (k_{\text{H}_2\text{O}}/f_{\text{H}_2\text{O}}) + k_{\text{CO}_2} \cdot P_{\text{CO}_2}^{\eta_{\text{CO}_2}} + k_{\text{R}} \cdot ([\text{R}^-]^{0.5}/f_{\text{H}}) \quad (2)$$

where  $k_{\text{H}^+}$  = rate coefficient for the reaction with  $\text{H}^+$  (m/s),  $k_{\text{H}_2\text{O}}$  = rate coefficient for the reaction with  $\text{H}_2\text{O}$  ( $\text{keq m}^{-2} \text{ s}^{-1}$ ),  $k_{\text{CO}_2}$  = rate coefficient for the reaction with  $\text{CO}_2$  ( $\text{keq atm}^{-1} \text{ m}^{-2} \text{ s}^{-1}$ ),  $k_{\text{R}}$  = rate coefficient for the reaction with DOC ( $\text{m s}^{-1}$ ),  $\eta_{\text{H}}$  and  $\eta_{\text{CO}_2}$  = reaction order of  $\text{H}^+$  and  $\text{CO}_2$  reactions, respectively.  $[\text{H}^+]$  and  $[\text{R}^-]$  = hydrogen ion concentration and free dissociated organic anion concentration in solution, respectively ( $\text{kmol m}^{-3}$ ),  $P_{\text{CO}_2}$  = partial pressure of  $\text{CO}_2$  in the soil solution (atm),  $f_{\text{H}}$ ,  $f_{\text{H}_2\text{O}}$ , and  $f_{\text{H}}$  = rate reduction factors for product inhibition.

The reduction factors reduce the rate of base cation production due to increased concentrations of weathering products on or near the mineral surfaces are defined as (Sverdrup and Warfvinge 1993):

$$f_H = (1 + [Al^{3+}] / k_{Al})^{\chi_{Al}} \cdot (1 + [BC]/k_{BC})^{\chi_{BC}} \quad (3)$$

$$f_{H_2O} = (1 + [Al^{3+}] / k_{Al})^{Z_{Al}} \cdot (1 + [BC]/k_{BC})^{Z_{BC}} \quad (4)$$

$$f_R = (1 + [R^{3+}] / k_R)^{0.5} \quad (5)$$

where  $k_{Al}$ ,  $k_{BC}$ , and  $k_R$  = aluminium, base cation, and organic reaction saturation constants ( $\text{keq m}^{-3}$ ), and  $\chi_{Al}$ ,  $\chi_{BC}$ ,  $Z_{Al}$ ,  $Z_{BC}$  = reaction orders. The  $\text{CO}_2$  reduction factor,  $f_{\text{CO}_2}$ , is not included into the equation (2) because it is equal to one (Sverdrup and Warfvinge 1995).

The release rate of base cations is proportional to the field surface area activity of each base releasing mineral. As shown in equation (10), the field surface area activity of each mineral is calculated by multiplying total exposed surface area, the fractional surface area of each mineral ( $A_{\text{exp}} \cdot X_i \cdot \theta$ ), and fraction of wetted surface which is approximated by the soil moisture content. In particular, the total exposed surface area of soil minerals ( $\text{m}^2 \text{ m}_s^{-3}$ ) can be estimated using the formula (Sverdrup and Warfvinge 1993; Sverdrup and Warfvinge 1995):

$$A_{\text{exp}} = (8.0 f_{\text{clay}} + 2.2 f_{\text{silt}} + 0.3 f_{\text{sand}}) \rho_{\text{soil}} \cdot 1000 \quad (6)$$

with the condition that the particle size fractions add up to one:

$$f_{\text{clay}} + f_{\text{silt}} + f_{\text{sand}} + f_{\text{coarse}} = 1$$

where  $f_{clay}$  is the weight fraction of clay (< 2 micron),  $f_{silt}$  is the weight fraction of silt (2-60 micron),  $f_{sand}$  is the weight fraction of sand (60-250 micron), and  $\rho_{soil}$  is the bulk density ( $\text{kg m}^{-3}$ ).

The surface area fraction of each mineral (percentage weight content) in the mineral matrix,  $X_i$ , can be calculated based on percent weight content of the oxides in the soil horizon by applying the following equations (Posch *et al.* 1995):

$$K\text{-Feldspar} = \text{Max} \{0, 5.88. K_2O - 0.588. Na_2O\} \quad (7)$$

$$Plagioclase = \text{Max} \{0, 11.1. Na_2O - 0.22. K\text{-Feldspar}\} \quad (8)$$

$$Apatite = 2.24. P_2O_5 \quad (9)$$

$$Hornblende = \text{Max} \{0, 6.67. CaO - 3.67. Apatite - 0.2. Plagioclase\} \quad (10)$$

$$Muscovite = \text{Max} \{0, 2.08. K_2O - 0.208. Na_2O\} \quad (11)$$

$$Chlorite = \text{Max} \{0, 3.85. MgO - 0.39. Hornblende - 0.39. Muscovite\} \quad (12)$$

$$Epidote = \text{Max} \{0, 0.1. Hornblende - 0.03. Plagioclase - 0.3\} \quad (13)$$

$$\begin{aligned} Quartz = & SiO_2 - 0.63. Plagioclase - 0.68. K\text{-Feldspar} - 0.38. Muscovite \\ & - 0.33. Chlorite - 0.45. Hornblende - 0.42. Epidote. \end{aligned} \quad (14)$$

In calcareous soils, equations (11-13) are omitted and equation (10) is replaced by:

$$Calcite = \text{Max} \{0, 1.79. CaO - 3.67. Apatite - 0.2. Plagioclase\} \quad (15)$$

The calculation should be checked by summing the fractions of quartz and all the other minerals. The sum of all the mineral fractions should lie within the range 95-105%.

Otherwise, the estimated weathering rate would not be not correct (would be biased towards one mineral or another).

Soil moisture is a controlling factor for base cation release rates because the reactions are assumed to take place on wetted surfaces only. Therefore, the degree of surface wetting is taken to be proportional to the soil moisture content, as follows (Sverdrup and Warfvinge 1993):

$$\theta = (\rho_{mineral} \cdot MC) / (\rho_{mineral} + \rho_{water} \cdot MC - \rho_{soil}) \quad (16)$$

where  $\theta$  is soil moisture content ( $m^3 m^{-3}_{soil}$ ),  $\rho_{mineral}$  = mineral density ( $2700 \text{ kg m}^{-3}$ , Sverdrup and Warfvinge 1995),  $\rho_{water}$  = water density ( $1000 \text{ kg m}^{-3}$ ),  $\rho_{soil}$  = soil bulk density ( $\text{kg m}^{-3}$ ), and MC = the soil moisture content ( $m^3 m^{-3}_{soil}$ ).

The rates of weathering reactions in natural soil are temperature dependent. The rate coefficients used in this model are derived from standard laboratory conditions ( $25^\circ\text{C}$ ), which are then further standardized to a soil temperature of  $8^\circ\text{C}$  ( $281^\circ\text{Kelvin}$ ) using an Arrhenius relation (Sverdrup 1990). The following equation is used to adjust the coefficients to ambient soil temperature (Sverdrup and Warfvinge 1993):

$$K_T = k_S \exp((E_A/R) * ((1/281) - (1/T))) \quad (17)$$

where  $K_T$  = rate coefficient at Temperature T,  $k_S$  = rate coefficient at  $8^\circ\text{C}$ ,  $E_A$  = activation energy ( $\text{KJ Kmol}^{-1}$ ),  $R$  = universal gas constant ( $\text{KJ Kmol}^{-1} \text{ K}^{-1}$ ), and T = absolute temperature (K).

All these relationships were entered into the ModelMaker modeling framework. The main purpose for doing this was to develop a soil weathering module that can be built into the ForSust model. In the following, a comparison study is presented that compares the results obtained with the ModelMaker formulation and with the original PROFILE model: a HyperCard application for MacIntosh computers.

#### Testing the model performance

Seven sites of Integrated Forest Study (IFS sites, Johnson and Lindberg 1992), located in Canada and Northern parts of US, were used to test model performance. As described in Table 4.1, these sites were mainly located on productive glacial and volcanic soil substrates, except for the Maine site. Oxide data at these sites are summarized in Table 4.2. Soil temperature, moisture content, CO<sub>2</sub> pressure, and dissolved organic carbon levels were all standardized to 5 (°C), 0.2 (m<sup>3</sup> m<sup>-3</sup>), 10 (times atmosphere), and 10 (mg L<sup>-1</sup>), respectively. The weathering rates were calculated for the IFS sites by the PROFILE model via its HyperCard formulation and the ModelMaker formulation (this Thesis). Then, ModelMaker weathering outputs plotted versus HyperCard weathering outputs to check consistency.

**Table 4.1** Location and site characteristics of the Integrated Forest Study (IFS) sites.

Site Code	Forest Location	Latitude (°N)	Longitude (°W)	Elevation (m)	Forest Type	Parent Material	Depth (m)	Clay	Silt (wt %)	Sand
FL	Findley Lake	47.04	121.25	1130	Fir, hemlock	Volcanic ash	0.71	0.045	0.360	0.595
HF	Huntington	43.59	74.14	530	Hardwoods	Glacial till	0.58	0.040	0.220	0.740
MS	Maine	45.10	68.40	65	Red spruce	Basal till	0.52	0.035	0.260	0.705
DF	Thompson	47.23	121.56	220	Douglas fir	Glacial till	0.45	0.045	0.395	0.560
RA	Thompson	47.23	121.56	220	Red alder	Glacial till	0.45	0.060	0.390	0.550
TL	Turkey Lake	47.03	84.25	350	Hardwoods	Glacial till	0.57	0.025	0.410	0.565
WF	Whiteface	44.22	73.54	1000	Red spruce	Anorthosite	0.41	0.010	0.200	0.790

**Table 4.2** Average major element chemistry of IFS sites (Johnson and Lindberg 1992).

Site Code	Average (wt %) oxide for all soil horizons									
	Fe <sub>2</sub> O <sub>3</sub>	MnO	TiO <sub>2</sub>	CaC	K <sub>2</sub> O	P <sub>2</sub> O <sub>5</sub>	SiO <sub>2</sub>	Al <sub>2</sub> O <sub>3</sub>	MgO	Na <sub>2</sub> O
FL	6.43	0.09	0.93	3.10	1.11	0.12	63.93	18.97	2.76	3.
HF	7.24	0.08	1.46	3.56	4.02	0.17	64.75	14.69	1.71	2.
MS	5.43	0.03	0.86	0.71	1.81	0.07	77.52	11.22	0.53	1.
DF	6.08	0.16	0.93	2.97	1.12	0.23	68.01	16.44	2.16	2.
RA	6.54	0.20	0.97	3.00	1.12	0.24	66.73	17.42	2.42	2.
TL	5.66	0.07	0.66	3.29	1.83	0.09	69.53	14.75	2.02	3.
WF	5.27	0.07	1.11	4.99	2.26	0.30	67.59	14.78	2.19	1.

### Sensitivity analyses

The model output for the soil weathering rate (total base cation release rate) was investigated by way of a sensitivity analysis with the soil data from the Turkey Lakes site (TL). This analysis was done by varying soil temperature, soil moisture content, particle surface area, CO<sub>2</sub> pressure, and dissolved organic carbon content. The base values and ranges used for the analysis are listed in Table (4.4). Relative changes in model output were evaluated with respect to each base and range value using the following equation (McCuen 1973; McCuen and Snyder 1986):

$$S = [(O_2 - O_1)/O_{12}] / [(I_2 - I_1)/I_{12}]$$

where S is sensitivity of the model output to the specific input parameter, I<sub>1</sub> and I<sub>2</sub> are the least and greatest values of input used, respectively; I<sub>12</sub> is the average of I<sub>1</sub> and I<sub>2</sub>, O<sub>1</sub> and O<sub>2</sub> are the associated outputs for the two input values, and O<sub>12</sub> is the average of the two outputs.

**Table 4.3** Model parameters and base values used for sensitivity analysis of Turkey Lake site.

Parameter	Unit	Base value	Range of test
Soil temperature	°C	5	3 - 7
Moisture content	m <sup>3</sup> m <sup>-3</sup>	0.20	0.18 - 0.22
Specific surface area	m <sup>2</sup> m <sup>-3</sup>	1.37E+06	1.17E+06 - 1.57E+06
CO <sub>2</sub> pressure	atm.	10	6 - 14
Dissolved organic carbon	mg L <sup>-1</sup>	10	6 - 14

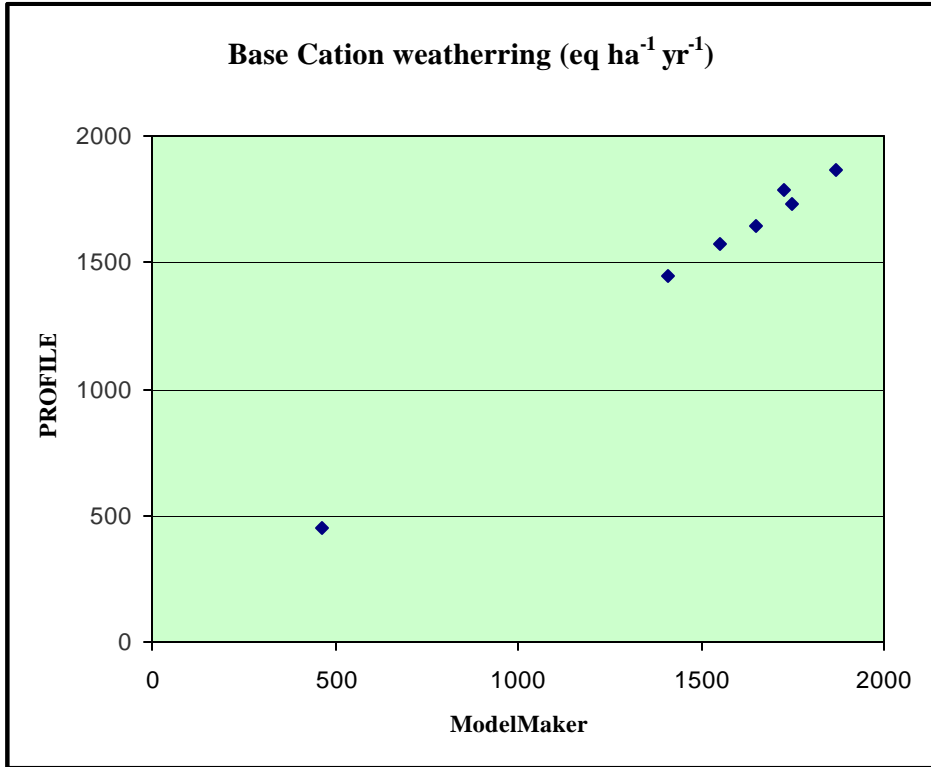
## RESULTS AND DISCUSSION

The soil weathering rates for the 7 sites were each calculated with the HyperCard and ModelMaker formulations of the PROFILE model, each for fixed conditions (soil depth = 1 m; soil temperature = 5°C, moisture content = 0.2 m<sup>3</sup> m<sup>-3</sup>), CO<sub>2</sub> pressure = 10 times atmosphere, and dissolved organic carbon = 10 mg L<sup>-1</sup>). The results are compiled in Table (4.5) and displayed in Figure (4.1). As shown, the outputs of the two formulations are consistent to within +/- 3%.

The PROFILE model has already been verified using 23 different independent determinations of field weathering rate from 15 sites in Scandinavia, Central Europe, and North America (Sverdrup and Warfvinge 1993). Comparing the predicted weathering rates by PROFILE with field and laboratory methods for five different soils in U.S.A. showed that this model adequately quantifies current weathering rates (Kaolka *et al.* 1996). However, an uncertainty of +/- 20% to 25% has been reported for PROFILE weathering rates in other studies (Sverdrup and Warfvinge 1993; Jonsson *et al.* 1995).

**Table 4.4** Weathering rates at seven IFS sites predicted with ModelMaker weatehring model compared with those calculated with PROFILE model.

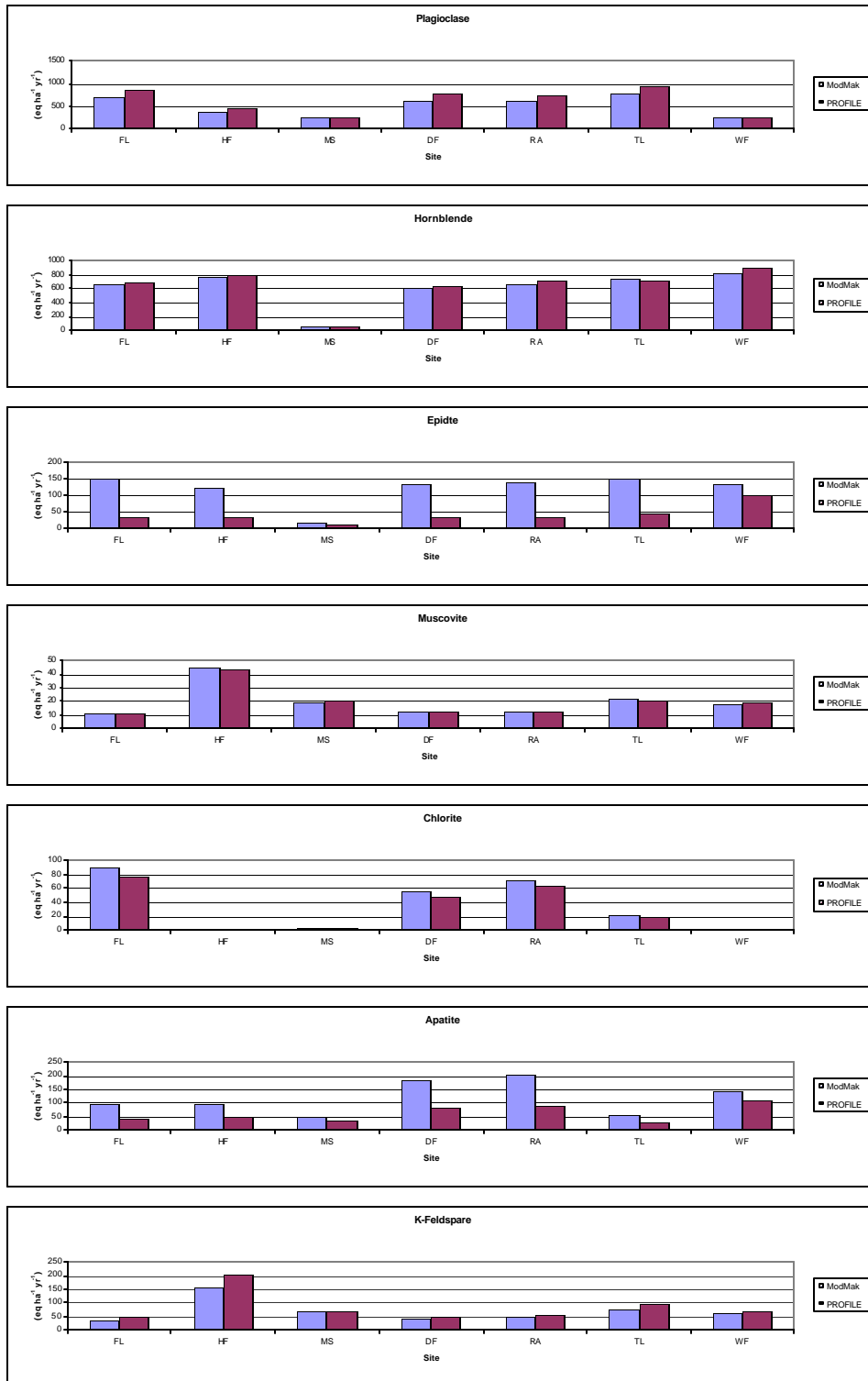
Site Code	BC+Na release rates (eq ha <sup>-1</sup> yr <sup>-1</sup> )	
	ModelMaker	PROFILE
FL	1727	1783
HF	1550	1575
MS	464	451
DF	1648	1643
RA	1747	1729
TL	1868	1867
WF	1408	1445



**Figure 4.1** Comparison between the weathering rate of seven IFS sites calculated with ModelMaker weathering model versus PROFILE model.

The individual contributions to the total weathering rate from K-feldspar, plagioclase, hornblende, epidote, muscovite, chlorite, and apatite, and how contributions vary with model formulation, as shown in Fig. (4.2). From this Figure, it can be seen that the main contributions to the total weathering rates are related to the weathering of plagioclase and hornblende. The second groups of minerals that contribute somewhat to total base cation release are K-feldspar, epidote, and apatite. Contributions to total weathering rates from muscovite and chlorite appear to be minor.

Figure 4.2 also illustrates that the calculated BC+Na release rates are consistent for hornblende, muscovite, and chlorite for both PROFILE formulations. With the ModelMaker formulation, the weathering rates of K-feldspar and plagioclase are somewhat lower than those obtained with the HyperCard formulation (3-25%). In contrast, the weathering rates of epidote and apatite as calculated with ModelMaker are somewhat higher than those calculated with the HyperCard formulation (25-120%). These differences are likely due to method of overall and iterative alkalinity estimation: the HyperCard formulation follows a custom-made iteration scheme to match specified pH with the specified acid-neutralizing capacity (ANC) of the theoretically derived composition of the soil chemical solution. With Modelmaker, a routine internal to the ModelMaker program is used to minimize the differences between a set ANC value and the model generated ANC estimates. Total base cation release rates (Ca + Mg + K + Na) are very close as estimated by way of either formulation (agreement ranges between -3 to +3%). Total base cation release rates were calculated to range from 464 eq ha<sup>-1</sup> yr<sup>-1</sup> (for the MS soil) to 1868 Eq ha<sup>-1</sup> yr<sup>-1</sup> (for the TL soil).



**Figure 4.2** Contribution to the total weatehring rate from different minerals at the IFS sites. The calculated weathering rates with ModelMaker are consistent with those calculated with PROFILE, except in case of Epidote and Apatite that are higher about one-third to twice and one-fourth to three-fourth, respectively.

Based on the weathering potential of the studied IFS, these sites can be ranked in the order of TL, RA, FL, DF, HF, WF, and MS. This sequence is very similar to the ranking derived by April and Newton (1992) for the same sites. These authors estimated the relative weathering rates among the IFS, as derived by way of laboratory analysis with various soil extractants.

The sources of uncertainties with the PROFILE model are due to uncertainties dealing with the kinetic parameters and with the input data. As Sverdrup and Warfvinge (1993) reported: uncertainties about the kinetic parameters for epidote and apatite may cause a slight over-estimation at high weathering rates. The results of the sensitivity analysis for the Turkey Lakes (TL) site (Table 4.6) indicated that model outputs are highly sensitive to specific surface area, soil moisture content, and soil temperature. Sensitivity to CO<sub>2</sub> pressure and dissolved organic carbon was low. There is an agreement between the sensitivity results of this study and those reported for the PROFILE model by Jonsson et al. (1995). These authors concluded that the PROFILE model output was most sensitive to particle surface area, followed by soil bulk density and soil water content, and then by the soil mineralogy specifications and the related kinetic parameters.

**Table 4.5** Measure of sensitivity for model parameters based on Turkey Lake site.

Variable	Value	BC+Na (eq ha <sup>-1</sup> yr <sup>-1</sup> )	Sensitivity of BC+Na
Base	Base <sup>1</sup>	1868	NA
Soil temperature	3	1733	0.19
	5	1868	
	7	2014	
Moisture content	0.18	1700	0.89
	0.20	1868	
	0.22	2033	
Specific surface area	1.17E+06	1595	1.00
	1.37E+06	1868	
	1.57E+06	2142	
CO <sub>2</sub> pressure	6	1796	0.09
	10	1868	
	14	1929	
Dissolved organic carbon	6	1849	0.02
	10	1868	
	14	1880	

1- See Table 5.4 For parameter base values and units.

## CONCLUSION

The soil weathering model was reformulated by the ModelMaker based on the concepts of PROFILE model. The results so obtained were then compared with the original HyperCard formulation, for 7 IFS sites. The resulting comparisons show considerable agreement in general (within +/-3% for total base cation release rates), but also show a few numerical differences. A likely source of these differences could be due to a fundamental difference in the iterative process to anchor the PROFILE calculations around a set ANC value for a given soil. Another source could be due to differences between some of the published kinetic parameters for some of the minerals addressed with PROFILE, and the actual parameters as used within the hidden HyperCard script.

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CHAPTER 5  
MODELING LONG-TERM POTENTIALLY  
SUSTANABLE FOREST BIOMASS PRODUCTIVITY

INTRODUCTION

The forest sustainability model (ForSust) is a process-oriented model that simulates potential forest biomass growth and quantifies long-term base cation balances in terms of:

- (i) stand-level nutrient input/output/storage balances;
- (ii) a balance between nutrient supply and demand;
- (iii) a balance among nutrients and related nutrient availability limitations.
- (iv) anticipated extra leaching losses from harvest slash (or from any other stand-level disturbance effects).

With this model, we focus on geochemical nutrient inputs/outputs and related storage differences by steady-state mass balance (SMB) approach on the basis of averaged annual data. As such, the model does not address the dynamics that deal with temporary changes (seasonal, monthly) of input/output storage rates. With this approach (SMB), the model can imply and estimate how much of the forest biomass production is potentially sustainable with existing or projected site-specific nutrient supplies and related availability restrictions, over the long term, i.e., one to several stand rotations.

The ForSust model links forest biomass productivity with nutrient supply and nutrient availability and offers several advantages in terms of site-specific model application and model initialization:

- (i) model input requirements are restricted in terms of
  - (a) addressing primary nutrient inputs;
  - (b) establishing nutrient demand based on species-specific nutrient concentrations as pooled for foliage, bark, stem-wood, and twigs;
  - (c) basic soil descriptors such as depth, clay content, soil substrate type as classified in terms of weathering potential;
  - (d) basic climate parameters (mean annual air temperature, precipitation, and actual evapotranspiration).
- (ii) The model itself is realized by way of high-level software programming tools i.e., STELLA II (1994) and ModelMaker 3 (1994). These tools have built-in graphs and tabular functions for viewing all model calculations, components, inputs, outputs, parameter values, and intermediate results. With ModelMaker, the model can be further refined by way of parameter optimization, input/output/parameter sensitivity analyses, etc.
- (iii) The model can evaluate a large number of sites, by accessing information about each site by way of simple spreadsheet formats (text files).

The objective of this chapter is to present a mathematical model that predicts stand-level potential biomass growth (productivity) and quantifies long-term base cation balances in response to soil acidification and harvesting (fire) scenarios.

The original aspects of this work relate to the integration and the quantitative formulation of several key components within ForSust. Key aspects are:

1. the extension of the ForSust model by adding a mechanistic forest soil weathering sub-model (ForSW, see Chapter 4);
2. the modification and extension of forest potential productivity model (ForPoP) that simulates potential biomass growth based on the nutrient limitation approach as modified by the anticipated effects of soil acidification on forest biomass growth;
3. the extension of model to allow the simulation of various forest disturbance scenarios, e.g., whole-tree harvesting, stem-only harvesting, forest fire effects on forest biomass production.

Model performance tests were carried out by comparing model-simulated and field-estimated values (annual) for biomass growth, Ca, Mg, K, and N net uptake, and Ca, Mg, K, and N leaching.

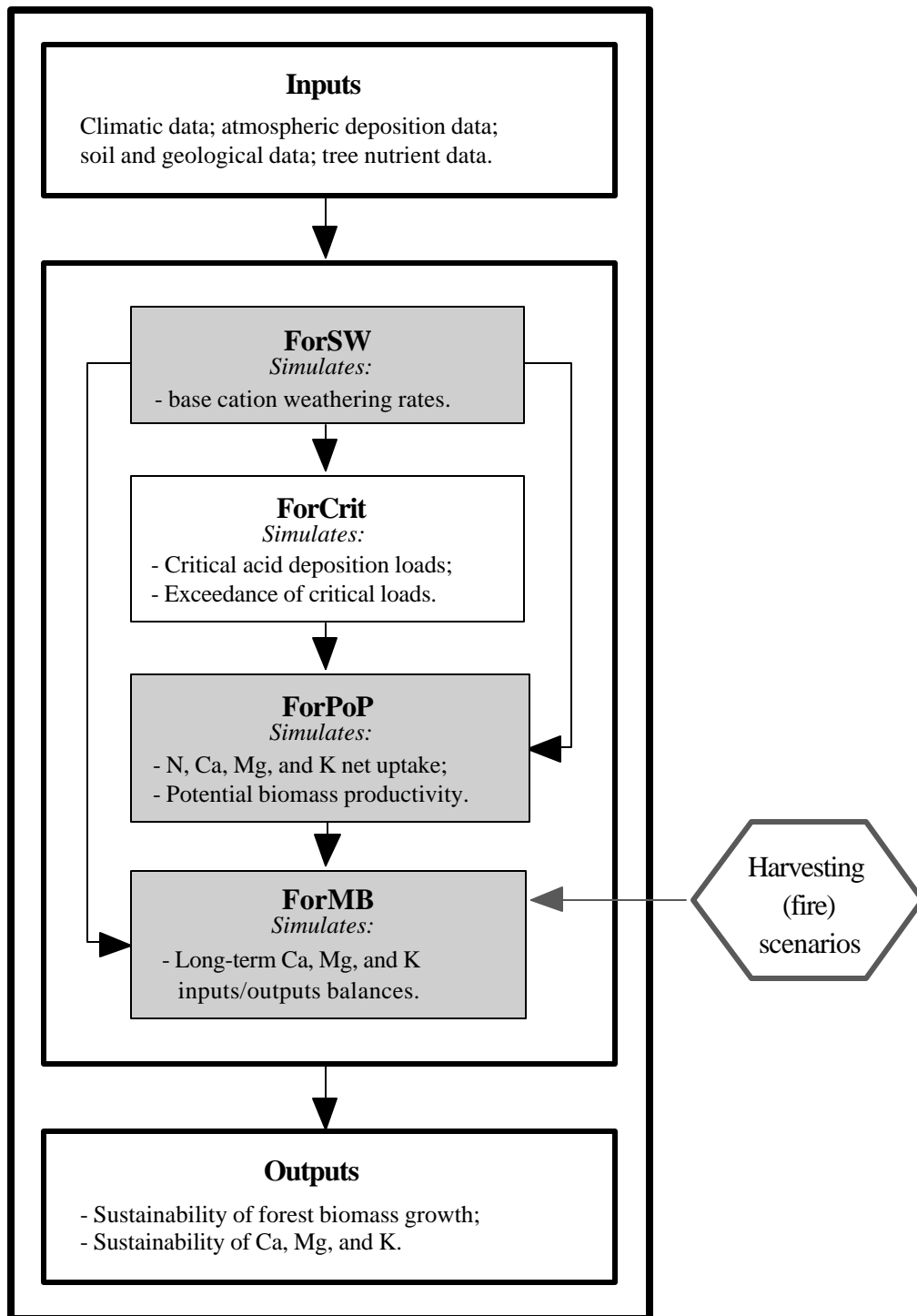
## MODEL CONSTRUCTION

### Structure

Structure and components of the ForSust model are shown in Figure 5.1. There are four sub-models as follows:

1. The forest soil weathering sub-model (ForSW) which simulates BC weathering rates.
2. The forest critical loads sub-model (ForCrit) which calculates critical soil acidification load and related exceedances.

# ForSust



**Figure 5.1** Structure of ForSust model and the relationships among its sub-models.

3. The forest potential productivity sub-model (ForPoP) which simulates potential annual biomass growth.
4. The forest mass balance sub-model (ForMB) which calculates long-term Ca, Mg, and K input/output balances.

## ForSust MODEL SPECIFICATIONS

### Model input

#### *Climate and atmosphere:*

- annual summaries for precipitation (mm);
- annual summaries for air temperature (°C);
- local estimates for annual actual evapotranspiration (AET, mm);
- atmospheric (wet and dry) deposition for S, N, Ca, Mg, K (eq ha<sup>-1</sup> yr<sup>-1</sup>);

#### *Soil:*

- depth of rooting space (m);
- soil clay, silt, and sand content (% wt);
- soil bulk density (kg m<sup>-3</sup>);
- soil Ca, Mg, K exchangeable (eq ha<sup>-1</sup>);
- soil parent material type (1- acidic, 2- intermediate, 3- basic);
- percentage weight content of oxides in soil mineral (%), e.g., SiO<sub>2</sub>, CaO;
- soil mean annual temperature (°C);

- soil moisture content ( $\text{m}^3 \text{ m}^{-3}$ );
- soil  $\text{CO}_2$  pressure (relative number in relation to atmospheric  $\text{CO}_2$  content);
- dissolved organic carbon, DOC ( $\text{mg L}^{-1}$ ).

*Vegetation:*

- stand composition (major tree species);
- mean age of dominant/co-dominant trees (years);
- mean annual increment (MAI) for above-ground biomass ( $\text{Mg ha}^{-1} \text{ yr}^{-1}$ );
- species specific nutrient concentrations in foliage, branches, stem wood, and bark (ppm);
- species specific biomass fractions of the aboveground tree biomass components (foliage, bark, branches, wood).

Model Output

- Base cation weathering ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );
- Critical soil acidification load [various definitions, ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ )];
- Exceedance of critical soil acidification ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );
- Base cation uptake ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );
- N uptake ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );
- Base cation leaching ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );
- N leaching ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );
- Potential annual biomass growth ( $\text{Mg ha}^{-1} \text{ yr}^{-1}$ ).

### Model Validation

- on-site assessment of mean annual biomass increment ( $\text{MAI}_{(f)}$ ,  $\text{Mg ha}^{-1} \text{ yr}^{-1}$ );
- normal assessment of mean annual biomass increment ( $\text{MAI}_{(n)}$ ,  $\text{Mg ha}^{-1} \text{ yr}^{-1}$ );
- on-site assessment of net nutrient uptake [N, K, Ca, Mg, ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ )];
- normal assessment of net nutrient uptake [N, K, Ca, Mg, ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ )];
- soil leachate fluxes for Ca, Mg, K, N, ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ );

### Model scenarios (potential)

- Various atmospheric deposition scenarios;
- Climate change scenarios;
- Forest disturbance scenarios;
- Stand interventions;
- Stand rotation.

### Scope (Scales)

- Aspatial (assumes uniformity within stand);
- Decades to centuries;
- Mean Annual Mass-balances (input = output + storage).

### Formulations

- Steady state (assumes inputs and outputs and storage increments remain the same, until the next change or disturbance);
- Dynamic response to one-time loss of nutrients;

- Dynamic response to one-time input of nutrients;
- Dynamic response to gradual loss of nutrient;
- Various combinations of the above, as needed.

#### Analysis capability

- Optimizes parameters (by least square method), using field data to check model output.
- Performs sensitivity analysis on any model component, to evaluate range of model outcome based on parametric uncertainties, scenario evaluations, variation of site parameters, etc.
- Estimates confidence intervals for model outputs.

#### Application philosophy

- Uses the successive approximation principle: start simple from what is known, fill in blanks through substitution, replace substitutions when possible, critically evaluate outcome at each step, and revise model, as needed.
- Through successive approximation, customize model to particular application.

#### Modeling Platforms

- Stella (simple version);
- ModelMaker (advanced version).

## BASIC CONCEPTS

The following concepts are important parts of the overall ForSust model formulation.

### The sustainability concept

The concept of sustainability implies that the long-term geochemical nutrient removals (leaching and harvest removals) from the forest ecosystem should not exceed the long-term nutrient supplies (atmospheric deposition and weathering). At this condition, which is a repeatable balance between long-term nutrient input and long-term nutrient output, biomass productivity (growth) of the forested ecosystem can be sustainable. Otherwise, decline of nutrient availability and productivity (growth) will occur.

### The limiting nutrient concept

This concept is based on the Liebig's law of the minimum that states there will always be one resource least in supply relative to the demands, and this will be the determinant of growth and production. In this regard, N is well recognized as a limiting element for terrestrial plant growth (Ågren 1985a; Vitousek and Howarth 1991). Quantitatively, the limiting effect of N is expressed as the term *nitrogen productivity* concept.

### The Nitrogen Productivity Concept

This concept implies that the amount of biomass produced per amount of nitrogen in the biomass and per unit of time is constant for a given species under fixed environmental conditions (Ågren 1985b).

### The Biomass Growth Concept

This concept is about control of plant growth by nitrogen, based on a theory that was mathematically formulated (Ågren 1985b). These are two basic assumptions: (i) the biomass growth rate is proportional to nitrogen productivity (the amount of biomass produced per amount of nitrogen in the biomass and per unit of time); (ii) nitrogen productivity (or N efficiency) is a linear function of  $1 / (\text{internal nitrogen concentration})$ .

### The Soil Acidification Concept

This concept state that excess N and S loading can give rise to accelerated rates of soil acidification and can cause nutrient imbalances. In this regard, excess N and S loading depend in part on the ability of the plants to absorb N and S, and on site-external factor that influence N and S removals such as harvesting and forest fires (Grennfelt and Thörnelöf, 1992).

## THEORIES AND MODEL ASSUMPTIONS

### Ca, Mg, and K input/output balance

*Assumption 1: The long-term balance (difference) between primary nutrient supply (nutrient sources) and expected nutrient removal (nutrient sinks) determines site sustainable productivity.*

According to this assumption, the productivity of forest site is sustainable as far as the rates of Ca, Mg, and K removal by biomass (net uptake) and by leaching are not exceeded from the rates of primary nutrient supply (deposition + weathering), i.e.,

$$X_{dep} + X_{we} \underset{X=Ca,Mg,K}{\geq} X_{up} + X_{le} \quad (5.1)$$

where

$X_{dep}$  = supply of nutrient X by deposition (eq ha<sup>-1</sup> yr<sup>-1</sup>),

$X_{we}$  = supply of nutrient X by weathering (eq ha<sup>-1</sup> yr<sup>-1</sup>),

$X_{up}$  = removal of nutrient X by plant uptake (eq ha<sup>-1</sup> yr<sup>-1</sup>),

$X_{le}$  = removal of nutrient X by soil leaching (eq ha<sup>-1</sup> yr<sup>-1</sup>).

Increased nutrient leaching and nutrient exporting by biomass means increased nutrient loss, to the point that site impoverishment occurs when the outgoing rates exceed the incoming rates. In contrast, increased nutrient additions and/or increased nutrient retention can lead to gradual site improvements, by way of nutrient storage and related soil quality enhancement. For convenience, it is assumed that most pristine forests in

mature conditions are at a balance point, i.e., mature production rates match mature nutrient supply rates. Considering the equation (5.1) and assuming constant rates of nutrient supply, deficit or surplus of long-term input/output is calculated from:

$$\Delta X_{X=\text{Ca,Mg,K}} = X_{dep} + X_{we} - X_{up} - X_{le} \quad (5.2)$$

where

$\Delta X$  = deficit (-) or surplus (+) rates of nutrient X (eq ha<sup>-1</sup> yr<sup>-1</sup>).

### Base cation weathering

The rate of base cation weathering (BC<sub>we</sub>) can be calculated with the geochemical soil weathering sub-model (ForSW) which is formulated base on the assumptions and concepts of weathering calculation by PROFILE model (see Chapter 4). Since calculation of weathering rates by the weathering model is limited due to lack of required geological data (model input), an alternative method for estimating BC<sub>we</sub> rates is formulated by setting the following assumption:

*Assumption 2: weathering rates assumed to be correlated to the type of soil parent material, to soil texture, and to site condition.*

Depending on type of parent material (mineral composition of soil substrate), each site has a certain potential to produce base cations. The assumption that texture class has a strong influence on the weathering rate is based on a linear relationship between

weathering rates and internal soil surface areas. The latter is directly related to clay content, as realized by Sverdrup (1990). In addition to these two basic factors, site conditions such as soil temperature and soil depth affect weathering rates as well.

A generalized weathering rate (De Vries 1991) can be assigned by soil parent material type (acidic, moderate, basic), and by soil texture class (percent clay content), and this rate can be modified by soil temperature, and soil rooting depth. Switching to this weathering method, the model calculates  $BC_{we}$  from:

$$BC_{we} = BC_{we0} \times f_{(T)} \times Z \quad (5.3)$$

where

$BC_{we0}$  = a generalized BC weathering rate ( $\text{eq ha}^{-1} \text{ yr}^{-1} \text{ m}^{-1}$ ) corresponding to the parent material class and soil texture, obtained from De Vries (1991).

$f_{(T)}$  = the correction factor for soil temperature, and

$Z$  = depth of mineral soil layers at rooting zone (m).

The weathering rate can be corrected for the effect of temperature as follows (Sverdrup, 1990):

$$f_{(T)} = \text{Exp} \left( \frac{A}{T_0} - \frac{A}{T} \right) \quad (5.4)$$

where

$A$  = a pre-exponential factor (3600 °K),

$T_0$  = reference temperature (°K), and

$T$  = the local temperature (°K).

Soil weathering rates for Ca, Mg, and K ( $X_{we}$ ) are calculated from:

$$X_{we} \underset{(X=Ca,Mg,K)}{=} BC_{we} \times \left( \frac{X_{avail}}{\sum X_{avail}} \right) \quad (5.5)$$

where

$BC_{we}$  = supply of base cation (Ca + Mg + K) by weathering at rooting depth (eq ha<sup>-1</sup> yr<sup>-1</sup>),

$X_{avail}$  = available rates of nutrient  $X$  at mineral layers of rooting depth (eq ha<sup>-1</sup> yr<sup>-1</sup>).

### Nutrient uptake

*Assumption 3: annual net N uptake is limited by the nutrient in least supply.*

Here, net uptake refers to the parts of nutrient uptake that accumulates into biomass at the end of year. In the model, Ca, Mg, and K supply is coupled with N supply by the requirement that the ratios between N and Ca, Mg, K in each tree component (foliage, twigs, branches, bark, wood) remains constant for rest of life of a tree (Kimmins *et al.* 1985). Then, either N or base cation (Ca, Mg, K) availability can limit N uptake (N biomass) as follows:

$$N_{up} = \min_{X=Ca,Mg,K} \left( \frac{X_{dep} + X_{we} - X_{le,crit}}{[X]/[N]}, N_{avail} \right) \quad (5.6)$$

where

$N_{up}$  = net N uptake ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),

$X_{dep}$  = supply of nutrient X by deposition ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),

$X_{we}$  = supply of nutrient X by weathering ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),

$X_{le,crit}$  = unavailable part of nutrient X supply ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),

$N_{avail}$  = available N for plant uptake ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),

$[X]/[N]$  = the weighted concentration ratio of X element to N in the plant (ppm/ppm).

*Assumption 4: Nutrient concentrations of tree components (foliage, branches, stem wood, stem bark) remain the same over age (after crown closure).*

The principle of successive approximations starts with a simplified assessment of the situation, i.e., we expect that nutrient concentrations within the tree biomass (foliage, branches, stem wood, and stem bark) do not change very much, especially once the trees are 15 years old or older and under pristine conditions. This is not to say that nutrient concentrations within trees do not change. For example, foliar nutrient concentrations do increase with increasing nutrient loads due to, e.g., fertilizer applications. Low foliar nutrient concentrations are often used for diagnostic assessment regarding inadequate tree nutrition with respect to particular nutrient elements. Changes in wood chemistry have been observed in relation to tree age, and possibly environmental factors such as acid rain. The point is that valuable insight and understanding can be gained about nutrient loadings, nutrient requirements, and related biomass growth expectations when nutrient concentrations are assumed to remain the same. As part of the next approximation, this condition can be relaxed by, e.g., forcing an age-dependent nutrient concentration pattern

for foliage, bark and stem wood, or forcing such concentrations to respond to overall nutrient input or nutrient availability.

*Assumption 5: With Equation (5.6), it was assumed that all the nutrient (N, Ca, Mg, and K) received and retained by the forest is used for aboveground tree biomass production.*

This assumption is purposefully aimed at calculating **potential harvestable production**, not **actual total production**. Actual total production would include the production of aboveground biomass other than trees, and underground biomass production as well. Actual and potential production rates, however, would not necessarily differ from one another in terms of long-term steady-state averages, as long as the non-harvested biomass nutrient stocks are already at steady state, remain part of the ecosystem, or are subject to efficient nutrient cycling. For example, and in principle, nutrient needed for root production can be re-supplied over the long-term through indefinite recycling of the existing root biomass of a mature stand. Also, annual nutrient required for tree maintenance (total uptake – net uptake) would be re-supplied by either litterfall or retranslocation, and of which is part of nutrient cycling.

*Assumption 6: Base cation (Ca, Mg, and K) uptake assumed to be proportional to N uptake.*

Once net N uptake ( $N_{up}$ ) is determined by the growth limiting element, net Ca, Mg, and K uptake rates ( $X_{up}$ ) are calculated from:

$$X_{up} \underset{(X=Ca,Mg,K)}{=} N_{up} \times \left( \frac{[X]}{[N]} \right) \quad (5.7)$$

### Biomass growth

*Assumption 7: biomass growth (potential) is assumed to be proportional to nitrogen limiting uptake and nitrogen productivity.*

The amount of biomass produced per amount of nitrogen in the biomass and per unit of time (nitrogen productivity) is constant for a given species under fixed environmental conditions. Considering this assumption, the nutrient limiting growth rate (potential) is calculated by the following equation:

$$G_{Pot,Nutr} = \frac{14 (N_{up})}{[N]} \quad (5.8)$$

where

$G_{Pot,Nutr}$  = nutrient-limited (potential) biomass growth rate ( $Mg \text{ ha}^{-1} \text{ yr}^{-1}$ ),

$N_{up}$  = N (net) uptake defined in Eq. (5.6), ( $eq \text{ ha}^{-1} \text{ yr}^{-1}$ ),

$[N]$  = weighted tree N concentration for given species ( $g \text{ Mg}^{-1}$ ).

*Assumption 8: soil acidification due to exceedances of critical acid deposition load is assumed to cause a decline in potential biomass growth rate.*

This assumption expresses the action of soil acidity, base cation (BC) availability and Al solubility on biomass growth. Soil acidification causes changes of the BC/Al ratio in the soil solution and reduces available base cations for uptake. Therefore, the potentially sustainable biomass growth is calculated from:

$$G_{Pot,Sust} = G_{Pot,Nutr} - G_{Pot,Exc} \quad (5.9)$$

where

$G_{Pot,Sust}$  = sustainable (potential) biomass growth ( $\text{Mg ha}^{-1} \text{ yr}^{-1}$ ),

$G_{Pot,Nutr}$  = nutrient-limited biomass growth rate, defined in Eq. (5.8), ( $\text{Mg ha}^{-1} \text{ yr}^{-1}$ ),

$G_{Pot,Exc}$  = biomass growth decline (potential) rate, caused by exceedances of critical load ( $\text{Mg ha}^{-1} \text{ yr}^{-1}$ ) which are calculated from

$$G_{Pot,Exc} = \max \left( 0, \frac{Exc_{(CL)}}{f_{(exc)}} \right) \quad (5.10)$$

where

$Exc_{(CL)}$  = the rate of critical load exceedance ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),

$f_{(exc)}$  = a biomass growth decline factor ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ).

The effect of other environmental factors (e.g., effect of available water or effect of climate change) is ignored, by assuming those factors are constant and they are not a limiting factor for biomass growth.

### Critical soil acidification loads

The assumptions that are built into the critical loads (CL) sub-model follows the Simple Mass Balance (SMB) approach by De Vries (1991), Hettelingh *et al.* (1992), Posch *et al.* (1993), De Vries *et al.* (1995), and Arp *et al.* (1996). This formulation is based on a generalized input/output balance for soil acidity, as follows:

$$CL(S + N) = BC_{dep} + BC_{we} - BC_{up} + N_{up} + N_{dnit} + N_{imm} + AC_{le, crit} \quad (5.11)$$

where CL (N + S) is critical loads for acid (S+N) deposition ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ), BC stands for base cations, and subscripts *dep*, *we*, *up*, *dnit*, *imm*, and *le* refer to total deposition, weathering, net uptake, denitrification, immobilization and leaching, respectively.

As shown by this equation, the acceptable level of incoming acidity (due to S and N deposition) is determined by:

1. the net input of base cations ( $BC_{dep} + BC_{we} - BC_{up}$ ),
2. the net removal of N from soil solution ( $N_{up} + N_{dnit} + N_{imm}$ ), and
3. an acceptable acid leaching rate from the soil ( $AC_{le, crit}$ ).

From the N balance, a critical load for N, namely CL(N), can be obtained as follows:

$$CL(N) = N_{up} + N_{dnit} + N_{imm} + N_{le, crit} \quad (5.12)$$

where  $N_{le, crit}$  is the allowed leaching of N at critical load.

From Eq. 1 and Eq. 2, a critical load for S, namely CL(S), can be derived as:

$$CL(S) = BC_{dep} + BC_{we} - BC_{up} + AC_{le, crit} - N_{le, crit} \quad (5.13)$$

Exceedances (Ex) of S, N, and acid (Ac) deposition loads are therefore defined as follows:

$$Ex(S) = CL(S) - S_{dep} \quad (5.14)$$

$$Ex(N) = CL(N) - N_{dep} \quad (5.15)$$

$$Ex(Ac) = Ex(S) + Ex(N). \quad (5.16)$$

N loss by denitrification ( $N_{dnit}$ ) is calculated with the following kinetic equation (Warfvinge *et al.* 1992):

$$N_{dnit} = \frac{k \times N_{dep}}{K + N_{dep}} \times 10^{(4770 \times (1/280 - 1/T))} \times f_{(m)} \quad (5.17)$$

where

k = kinetic constant (1000),

K = saturation coefficient (3200 eq ha<sup>-1</sup> yr<sup>-1</sup>),

T = local temperature (°K), and

$f_{(m)}$  = moisture factor (dry soil: 0.2; moist soil: 0.6; wet soil: 1.0).

An alternative way for estimating the denitrification rate ( $N_{dnit}$ , eq ha<sup>-1</sup> yr<sup>-1</sup>) is (De Vries *et al.* 1992):

$$N_{dnit} = f_{(dnit)} \times \min(0, N_{dep} - N_{up} - N_{imm}) \quad (5.18)$$

Specifically,  $f_{(dnit)} = 0.8, 0.7, 0.5, 0.1,$  and 0 for peat soils, clay soils, sandy soil with gleyic features, loess and sandy soils without gleyic features, and deeply well-drained sandy soils, respectively (Hettelingh *et al.* 1992; Posch *et al.* 1993).

N immobilization ( $N_{imm}$ , eq ha<sup>-1</sup> yr<sup>-1</sup>) refers to the net effect of two N cycle components on N critical loads and exceedances for forest soils, i.e.,

$$N_{imm} = N_{hum} - N_{fix} \quad (5.19)$$

with  $N_{hum}$  accounting for the gradual net incorporation of N into the essentially undecomposing part of soil humus and  $N_{fix}$  accounting for the gain of the forested ecosystem by way of nitrogen fixation.

N humus ( $N_{hum}$ , eq ha<sup>-1</sup> yr<sup>-1</sup>) is assumed to be proportional to the C/N ratio as follows:

$$N_{hum} = 3.2 \times \left( 1 - \text{Exp} \left( -0.0028 \times \left( \frac{C}{N} \right)^{1.87} \right) \right) \times \frac{1000}{14} \quad (5.20)$$

N fixation ( $N_{fix}$ , eq ha<sup>-1</sup> yr<sup>-1</sup>) is assumed to be proportional to the soil weathering rate for base cations. The rationale for this assumption is that productivity of N-fixing organisms is likely limited by supply of Ca, Mg, and K:

$$N_{fix} = BC_{we} \times f_{fix} \quad (5.21)$$

where  $f_{fix}$  is the N-fixation coefficient.

The rate of critical acid leaching ( $Ac_{le,crit}$ ) is set as follows:

$$Ac_{le,crit} = Al_{le,crit} + H_{le,crit} \quad (5.22)$$

with

$$Al_{le,crit} = [Al]/[BC]_{le,crit} \times BC_{le,crit} \quad (5.23)$$

$$BC_{le,crit} = BC_{dep} + BC_{we} - BC_{up} \quad (5.24)$$

$$H_{le,crit} = \left( \frac{([Al]/[BC]_{le,crit}) * BC_{le,crit}}{K_{gibb}} \right)^{1/3} \times Q^{2/3} \quad (5.25)$$

where

$[Al]/[BC]_{le,crit}$  = soil acidification criterion (e.g., 1.5 eq eq<sup>-1</sup>, Arp *et al.* 1996),

$K_{gibb}$  = equilibrium constant for gibbsite (300 m<sup>6</sup> eq<sup>-2</sup>), and

$Q$  = flux of water through the rooting space (m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>).

The critical N leaching rate ( $N_{le,crit}$ ) is set as follows:

$$N_{le,crit} = [NO_3]_{le,crit} \times Q \quad (5.26)$$

where

$[NO_3]_{le,crit}$  = minimum concentration of N in the percolating soil solution (eq m<sup>-3</sup>).

### Soil nutrient leach

*Assumption 9: the part of available nutrient (N, Ca, Mg, K) that is not taken up is leached.*

Net mineralization of X from soil organic matter, as well as net immobilization, is considered to be part of the steady state condition, and is therefore set to zero. N leaching rate ( $N_{le}$ , eq ha<sup>-1</sup> yr<sup>-1</sup>) is calculated by setting:

$$N_{le} = N_{dep} + N_{fix} - N_{up} - N_{dnit} + N_{le,crit} \quad (5.27)$$

The Ca, Mg, and K leaching rates ( $X_{le}$ , eq ha<sup>-1</sup> yr<sup>-1</sup>) are calculated from:

$$X_{le} = X_{dep} + X_{we} - X_{up} + X_{le,crit} + X_{le,exc} \quad (5.28)$$

by setting:

$$X_{le,crit} = [X]_{le,crit} \times Q \quad (5.29)$$

$$X_{le,exc} = ([X] \times G_{dec}) / X_{Eq,wt} \quad (5.30)$$

where

$X_{le,crit}$  = unavailable part of nutrient X supply (eq ha<sup>-1</sup> yr<sup>-1</sup>),

$X_{le,exc}$  = excess of leaching nutrient X due to critical load exceedances (eq ha<sup>-1</sup> yr<sup>-1</sup>),

$[X]_{le,crit}$  = minimum concentration of nutrient X in the percolating soil solution (eq m<sup>-3</sup>),

$[X]$  = weighted average concentration of nutrient X in tree (ppm),

$Q$  = flux of water through the rooting space (m<sup>3</sup> ha<sup>-1</sup> yr<sup>-1</sup>),

$G_{dec}$  = biomass growth decline due to critical load exceedances (Mg ha<sup>-1</sup> yr<sup>-1</sup>), and

$X_{Eq,wt}$  = equivalent weight of nutrient X (eq g<sup>-1</sup>).

### Harvest (fire) removal

*Assumption 10: Sustainable forest harvesting (or fire) means base cation (Ca, Mg, and K) removal by harvesting (or fire) matches limiting nutrient uptake.*

Sustainable nutrient removals via biomass exporting are those that do not exceed the established nutrient uptake rate of the forest system. Therefore, over long-term sustainable production (several rotations), we have that

$$X_{exp} \leq_{X=Ca, Mg, K} X_{up} \quad (5.31)$$

where

$X_{up}$  = X (net) uptake defined in Eq. (5.7), (eq ha<sup>-1</sup> yr<sup>-1</sup>), and

$X_{exp}$  = X (net) export due to harvesting or fire (eq ha<sup>-1</sup> yr<sup>-1</sup>).

Assumption 11: *Nutrients can be exported from the site once in a while, or continuously.*

For long-term and steady state calculations, it was assumed that it does not matter whether nutrients are being exported from the site once in a while (i.e., once every few decades), or continuously (i.e., once every year). In principle, the calculated outcome is independent of the forest renewal rate, or forest rotation age, as long as the sustainability criterion can be matched for same overall rate of biomass production. Harvesting or fire, for example, may be a one-time event in the life (rotation) of a forest stand. In this case, nutrient exports caused by harvest or fire disturbance are the amounts of nutrients in the harvested portion of the aboveground biomass plus any additional nutrient leaching losses from residues. Average losses are then calculated by dividing total losses to the age of the stand (or mean stand age for uneven-aged stand) at the time of disturbance. If so, then  $X_{exp}$  ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ) can be calculated from the following equations:

(a) for various harvesting scenarios,

$$X_{\text{exp}}_{X=\text{Ca, Mg, K}} = (X_{\text{harv}} + X_{\text{le,slash}}) / \text{age} \quad (5.32)$$

(b) for particular fire scenarios,

$$X_{\text{exp}}_{X=\text{Ca, Mg, K}} = (X_{\text{fire}} + X_{\text{le,resid}}) / \text{age} \quad (5.33)$$

where

$X_{harv}$  = portions of nutrient X exports (eq ha<sup>-1</sup>) that are associated with the harvested portion of the aboveground tree biomass,

$X_{le,slash}$  = extra post-harvest X leaching losses (eq ha<sup>-1</sup>), i.e., that portion of nutrient X in harvest residues (slash) that does not remain on site due to post-harvest leaching,

$X_{fire}$  = the amount of nutrient X exports (eq ha<sup>-1</sup>) that are returned to the atmosphere by combustion,

$X_{le,resid}$  = extra post-fire X leaching losses (eq ha<sup>-1</sup>), i.e., portion of nutrient X in fire residues that does not remain on site due to post-fire leaching.

*Assumption 12: Extra post-disturbance N leaching is proportional to the amount of slash left on site.*

This assumption quantifies as follows:

$$X_{le,slash} = X_{slash} \times f_{(le,slash)} \quad (5.34)$$

and

$$X_{le,resid} = X_{resid} \times f_{(le,resid)} \quad (5.35)$$

with

$X_{slash}$  = nutrient X (eq ha<sup>-1</sup>) that is associated with the post-harvest residues (slash),

$f_{(le,slash)}$  = post-harvest leach factor (dimensionless),

$X_{resid}$  = nutrient X (eq ha<sup>-1</sup>) that is associated with the post-fire residues,

$f_{(le,resid)}$  = post-fire leach factor (dimensionless).

In general, post-harvest leach factor ( $f_{(le,slash)}$ ) would depend on post-harvest vegetation recovery rates, which, in turn, would depend on the extent of site and soil disturbance, harvesting intensity, management practices, etc. (Gholz *et al.* 1985; Vitousek and Matson 1985; Knight *et al.* 1991). Any combination of factors that would discourage vegetation recovery would increase nutrient leaching from residuals. Evidence emerging from sites covered with and without slash suggests that slash residues do lead to increased N leaching in the first post-harvest years on account of increased availability of N for nutrient leaching, in spite of increased substrate availability for microbial N immobilization (Vitousek and Matson 1985).

For the case of forest fire, most fire residues have a high C/N ratio (mostly wood, and unburned pieces of wood), so that N losses from fire residues are unlikely. However, the post-fire leach factor ( $f_{(le,resid)}$ ) would be  $> 0$  for Ca, Mg, K, and possibly P.

## METHODS

### Program design and development

The ForSust model was developed with Stella and ModelMaker, which are both high-level programming languages for PC computers. ModelMaker has many new and improved features that provide high efficiency and flexibility for systems modeling. For example, some important features of this software are as follows:

1. allows for a visual display of model components, variables, flows, and of their relationships either by diagram or by mathematical definition;

2. provides considerable flexibility for the user to seek and generate alternative formulations and solutions;
3. has look-up functions that facilitate entering input data via external spreadsheet files and shifting to various already compiled data sets;
4. facilitates the processes of model calibration, verification, hypothesis testing, and scenario evaluation through its built-in analysis tools such as minimization, optimization, repeated run, goodness of fit, confidence interval estimation, and sensitivity analysis;
5. provides a flexible view of output data (user define) through its built-in table and graph features;
6. has workbook feature that facilitates organizing model structure, parameters, and results;
7. enables joining of separately developed sub-models into one model with little effort.

#### Model parametrization

Model parameters (Table 5.1) were identified by (1) searching the literature for relevant information regarding the model parameters, and (2) generating data to determine biomass growth decline factor ( $f_{(exc)}$ ). Data for item (2) belong to Acid Rain National Early Warning System (ARNEWS) which obtained from Canadian Forest Service (CFS). The growth decline factor ( $f_{(exc)}$ ) was identified by way of generating regression equations and searching for potential correlation between critical load exceedances and field-estimated biomass growth rates.

**Table 5.1** Fixed model parameter

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A	a pre-exponential factor (=3600 °K)
T <sub>0</sub>	reference temperature (= 275.6 °K)
k	kinetic constant (=1000, dimensionless)
K	saturation coefficient (=3200, mol <sub>c</sub> ha <sup>-1</sup> yr <sup>-1</sup> )
K <sub>gibb</sub>	equilibrium constant for gibbsite (=300 m <sup>6</sup> mol <sub>c</sub> <sup>-2</sup> )
f <sub>fix</sub>	fixation coefficient (= 0.1, dimensionless)
[Al]/[BC] <sub>le,crit</sub>	soil acidification criterion (=1.5 mol <sub>c</sub> mol <sub>c</sub> <sup>-1</sup> )
[Al] <sub>crit</sub>	critical Al concentration (= 0.2 mmol <sub>c</sub> L <sup>-1</sup> )
f <sub>(exc)</sub>	a biomass growth decline factor (= 1000 mol <sub>c</sub> ha <sup>-1</sup> yr <sup>-1</sup> )

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### Model verification

Model verification was done by (1) comparing field-estimated mean annual biomass growth (actual productivity) and simulated mean annual biomass growth (potential productivity), (2) comparing actual net uptake and simulated net uptake for N, Ca, Mg, and K, (3) comparing actual leaching rates and simulated leaching rates for N, Ca, Mg, and K.

Six well-studied forest stands including three tolerant hardwood sites (Morrison 1990, Houle *et al.* 1997) and three jack pine sites (Foster *et al.* 1995, Morrison and Foster 1974) were chosen to verify the ForSust model (Table 5.2).

**Table 5.2** Inventory data and estimated mean annual biomass increment for three tolerant hardwoods and three jack pine sites used for model validation.

Site	Latitude	Longitude	Site Location	Prov	Cover Type	Stand Density	Mean DBH (cm)	Basal Area (m <sup>2</sup> ha <sup>-1</sup> )	Mean Height (m)	Mean Age (year)	Mean Annual biomass Increment <sup>b</sup>			Site Reference
											MAI <sub>(f)</sub>	MAI <sub>(n)</sub>	MAI <sub>(n,r)</sub>	
											(Mg ha <sup>-1</sup> yr <sup>-1</sup> )			
1	47.03	84.25	Norberg Creek	ON	SM <sub>82</sub> YB <sub>10</sub>	682	19.5	28.6	22.1	135	1.56	1.73	2.06	Morrison 1990
2	47.03	84.25	Wishart Lake	ON	SM <sub>93</sub> YB <sub>3</sub>	787	16.8	24.7	22.0	120	1.50	1.85	2.06	Morrison 1990
3	46.57	71.40	Lake Clair	QC	SM <sub>77</sub> YB <sub>15</sub>	394	27.6	28.0	17.2	130	1.36	1.46	1.66	Houle et al. 1997
4	46.25	83.23	Wells	ON	JP <sub>100</sub>	1175	19.0	35.0	19.3	56	2.76	2.81	3.06	Foster et al., 1995
5	47.38	83.15	Dupuis	ON	JP <sub>100</sub>	1478	17.0	34.0	18.2	68	1.75	1.83	2.33	Foster et al., 1995
5	50.02	92.32	Dryden	ON	JP <sub>100</sub>	1893	12.4	21.4	13.9	55	1.19	1.42	1.41	Morrison and Foster, 1995

<sup>a</sup> JP = jack pine, SM = sugar maple, YB = yellow birch.

MAI<sub>(f)</sub> = field-stocked Mean annual biomass increment.

MAI<sub>(n)</sub> = normal-stocked Mean annual biomass increment.

MAI<sub>(n,r)</sub> = normal-stocked Mean annual biomass increment at rotation age (JP=40, SM=80).

Data for model input (climate and deposition data, soil data, tree nutrient data) and data required for checking model outputs (biomass growth, net uptake, leaching) were collected from the above-cited literature and other literature articles (e.g., Jeffries *et al.* 1988 and ENFOR data).

Climatic inputs include mean annual temperature ( $^{\circ}\text{C}$ ), mean July temperature ( $^{\circ}\text{C}$ ), mean annual precipitation (mm) and mean annual actual evapotranspiration (AET, mm). Atmospheric deposition (bulk deposition) for  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^{+}$ ,  $\text{NH}_4^{+}$ ,  $\text{NO}_3^{-}$ , and  $\text{SO}_4^{2-}$  were calculated by multiplying volume-weighted ion concentrations with the local precipitation volumes.

Soil data include soil rooting depth (m), clay content (%), soil substrate class, exchangeable Ca, Mg, and K, and weathering rates for Ca, Mg, and K. Weathering rates were calculated with the PROFILE model (Sverdrup and Warfvinge 1993). There was insufficient information about soil P. Hence, all calculations were done without P.

Tree nutrient data include Ca, Mg, K, and N concentrations for aboveground tree components (foliage, branch, stem bark, and wood). These data were obtained from the Canadian Forest Services (foliage data) and a site- and species-specific wood and bark sampling effort followed by laboratory analyses. Concentrations of N, Ca, Mg, and K in branches were estimated through regression equations that were generated from foliage, bark, branch, and stem wood concentrations in Maliondo *et al.* (1990).

Mean annual biomass increment (MAI) was calculated by setting:

$$\text{MAI}_{(t)} = (\text{BA} \times \text{H} \times \text{FF} \times \text{WD}) / \text{AGE}_{(t)}$$

where  $MAI_{(f)}$  is the field-stocked mean annual biomass increment ( $Mg\ ha^{-1}\ yr^{-1}$ ), BA is basal area ( $m^2\ ha^{-1}$ ), H is the mean tree height (m), FF is a tree form factor that was set 0.44 and 0.38 for tolerant hardwood and jack pine, respectively. WD ( $Mg\ m^{-3}$ ) is wood density that was set at 0.84 and 0.58 for tolerant hardwood and jack pine, respectively. The form factor and the wood density were calculated from data presented by Maliondo *et al.* (1990).  $AGE_{(f)}$  is the mean age (yr) of dominant and co-dominant trees.

To determine fully stocked biomass growth rates, mean annual biomass increments were adjusted to normal stocking levels as follows:

$$MAI_{(n)} = (VOL_{(n)} \times WD) / AGE_{(f)}$$

where  $MAI_{(n)}$  is normal-stocked mean annual biomass increment ( $Mg\ ha^{-1}\ yr^{-1}$ ) and  $VOL_{(n)}$  is normal gross total volume ( $m^3\ ha^{-1}$ ), which was obtained from a normal yield table (Plonski 1981), using mean stand age and mean tree heights look-up variables.

To reduce errors caused by maturity (age status) variations,  $MAI_n$  was further adjusted as follows:

$$MAI_{(n,r)} = (VOL_{(n,r)} \times WD) / AGE_{(r)}$$

where  $MAI_{(n,r)}$  is the normal-stocked mean annual biomass increment at rotation age ( $Mg\ ha^{-1}\ yr^{-1}$ ),  $VOL_{(n,r)}$  is the normal gross total volume at rotation age ( $m^3\ ha^{-1}$ ), which was obtained from the normal yield table (Plonski 1981) according to the selected rotation age

and mean tree height (on site),  $AGE_{(r)}$  is the rotation age (yr) which was set 40 and 80 for jack pine and tolerant hardwood sites, respectively.

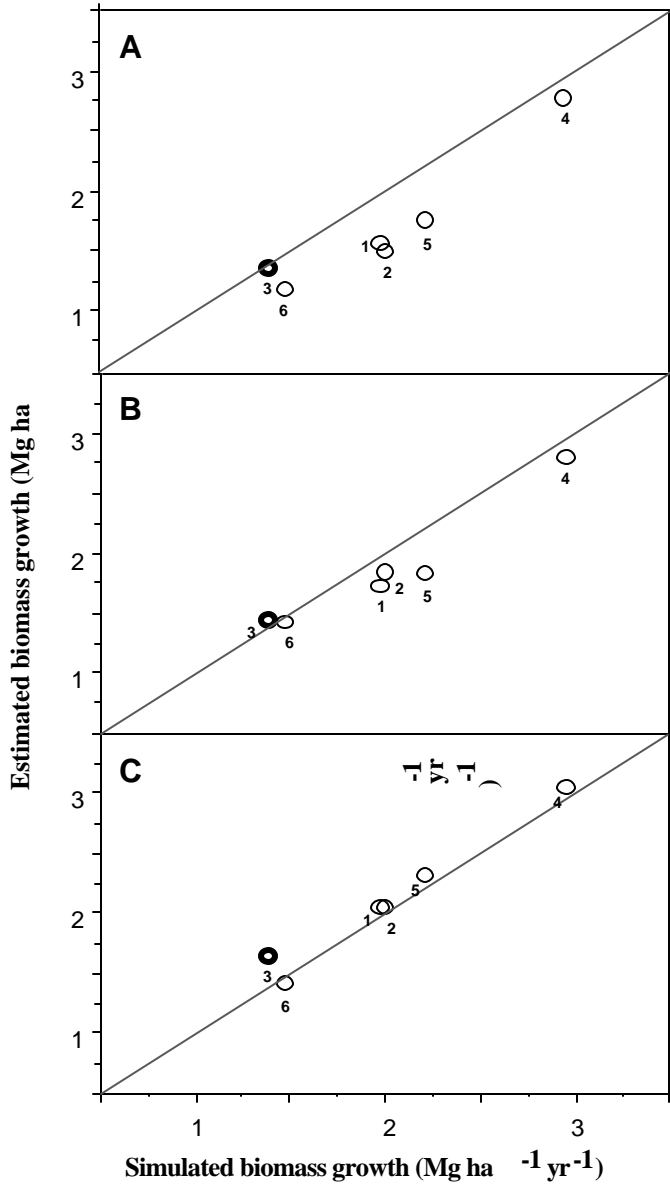
N, Ca, Mg, and K net uptake ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ) was calculated by multiplying actual biomass growth and with the corresponding weighted tree nutrient concentration. The calculations were repeated for three different growth values (field-stocked, normal-stocked at stand age, and normal-stocked at rotation age).

## RESULTS AND DISCUSSION

### Potential biomass growth (productivity)

A preliminary analysis of the modeling results showed that the potential biomass productivity was simulated to be controlled by base cation supply (weathering + deposition) for the tolerant hardwood stands (Sites 1-3). In contrast, potential biomass production was while it was controlled by N supply for the jack pine stands (Sites 4-6). Potential biomass growth was calculated to be adversely affected by exceedances of the critical soil acidification loads at Site 3 (Lake Clair, Quebec).

Shown in Figure 5.2 are values of simulated potential biomass growth versus values of estimated actual biomass growth. These calculations are in general agreement with respect to overall magnitude, but differ substantially from each other in detail for some sites (Fig. 5.2A).



**Figure 5.2** Simulated (potential) biomass growth plotted versus estimated (actual) biomass growth: (A) estimated from field-stocked biomass, (B) estimated from normal yield table at stand age, (C) estimated from normal yield table at fixed rotation age (40 years jack pine and 80 years for tolerant hardwood stands). Bolded symbol is indication of exceedances of critical acid deposition loads.

The disagreements can be related to errors caused by either estimation of actual biomass growth or prediction of potential biomass growth. For example, stands may have been under-stocked; hence, actual biomass growth is less than the potential biomass growth. To solve this problem, field-estimated biomass growth was adjusted to normal biomass growth according to the existing tree inventory data and the normal yield expectations (Plonski 1981). Adjusting the actual biomass growth values to the normal biomass growth values generated considerable agreement with simulated biomass growth values (Fig. 5.2B).

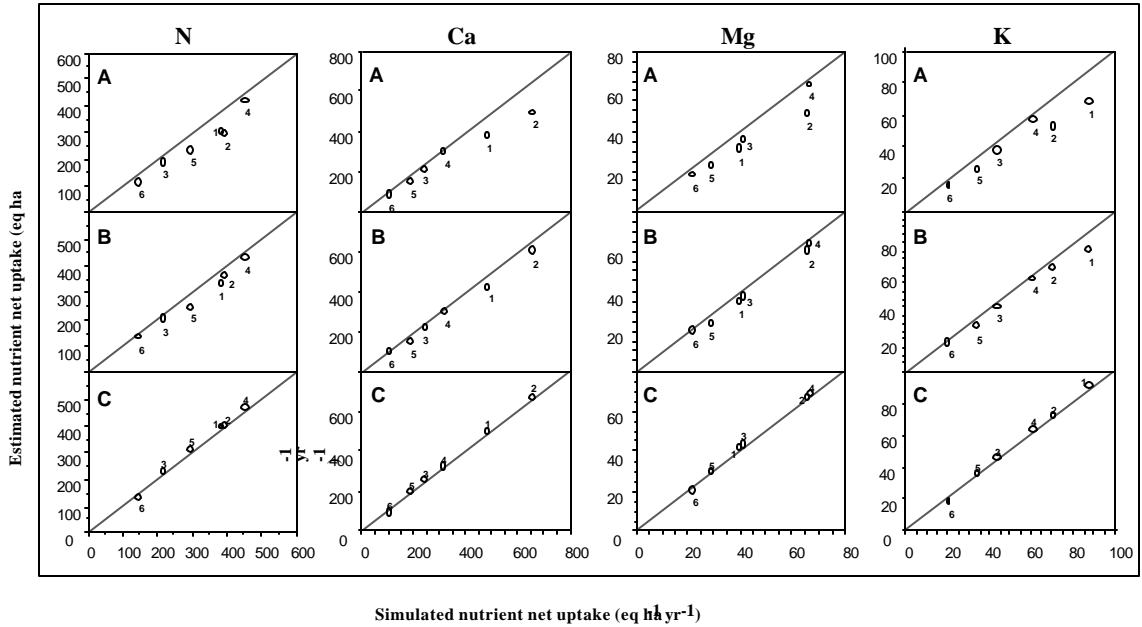
Another possibility for disagreement would be due to stand age variations. Estimated actual biomass growth is age dependent and varies at each maturity stages. For example, mean annual biomass growth usually increases at the early development stage, relaxes at maturity stage, and decreases at over maturity. However, the simulated potential biomass growth is dependent on nutrient supply and demand, and is essentially age independent. The temporal scale, here, is long-term (at least one rotation). To remove the effect of stand age variations, actual biomass growth was further adjusted by modifying stand age to an equal species-specific rotation age (80 year for tolerant hardwoods and 40 years for jack pine stands). The adjusted normal biomass growth values show the highest agreement with the simulated potential biomass growth values (Fig. 5.2C). Site 3 (Lake Clair, Quebec) remained an exception.

### Nutrient net uptake

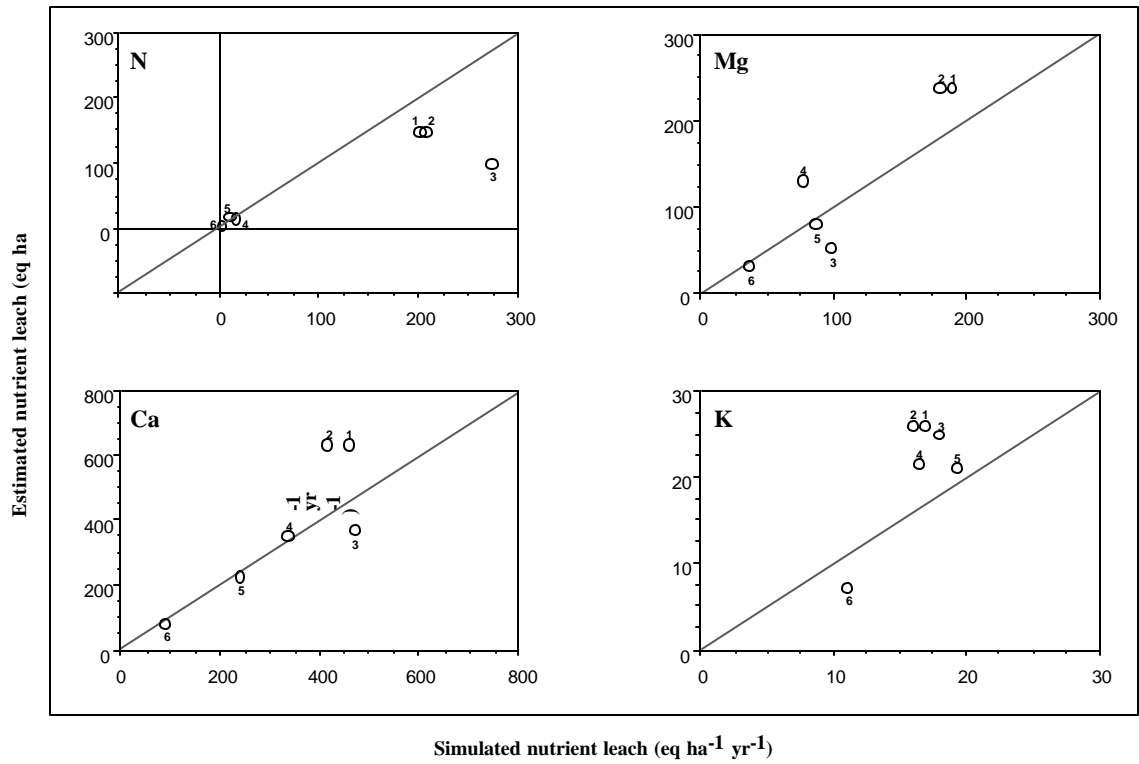
In Figure 5.3, simulated net uptake rates are plotted against estimated net uptake rates for N, Ca, Mg, and K. Model predicted net uptake values are in general agreement with the field-estimated net uptake rates (Fig. 5.3A), except for Sites 1 and 2 (tolerant hardwoods in Ontario). An improved agreement between simulated and field-adjusted net uptake (Fig. 5.3B and 5.3C) imply that the disagreement between values for sites 1 and 2 was correlated to stocking and/or age variations.

### Soil nutrient leach

A comparison of model-calculated and field-estimated leaching values for N, Ca, Mg, and K is shown in Figure (5.4). The model generates very low N leaching rate estimates (owing to limiting nutrient uptake) for jack pine stands (Sites 4-6). This is in agreement with the field-estimated N leaching rates. Agreements was also obtained between simulated and estimated leaching values at jack pine sites for Mg and K. For tolerant hardwoods (Sites 1-3), the calculations are in general agreements with respect to overall magnitude, but differ substantially from each other in detail. For example, a considerable disagreement was found for N leaching at Site 3 (Lake Clair, Quebec) and for base cations (Ca, Mg, and K) leaching at Sites 1 and 2 (Norberg Creek and Wishart Lake, Ontario).



**Figure 5.3** Simulated (potential) nutrient net uptake plotted versus estimated (actual) nutrient net uptake: (A) estimated from field-stocked biomass, (B) estimated from normal yield table at stand age, (C) estimated from normal yield table at fixed rotation age (40 years for jack pine and 80 years for tolerant hardwood stands).



**Figure 5.4** Simulated (potential) N, Ca, Mg and K leach plotted versus estimated (actual) N, Ca, Mg and K leach.

The model verification regarding the simulated nutrient leaching rates is tentative because long-term leaching data are not available. Instead, the leaching data used for this test were collected from short-term studies (2-3 years). For example, N, Ca, Mg, and K leaching values for Sites 1 and 2 were estimated from a two-year stream data record (Jeffries *et al.* 1988) at corresponding watershed. Also, N leaching data were not available for Site 3. Instead, N leaching was estimated from the lake outlet data (Houle *et al.* 1997).

## CONCLUSION

The established forest sustainability model (ForSust) combines existing empirical, mechanistic, and steady state mass balance (SMB) modeling techniques. For example, weathering rates are simulated with a mechanistic sub-model (ForSW). However, an alternative empirical method is also formulated for cases for which the required ForSW input data are not available. Critical soil acidification load and exceedances of critical loads are modeled with SMB sub-model.

The proposed ForSust model appears to be a useful tool for assessing long-term sustainability of forest nutrient balances and forest biomass production. The consistent agreement between model simulations and available field observations suggest that potential forest biomass productivity can be calculated on the basis of assessing mean annual primary nutrient inputs with the context of nutrient supply and demand, and by limiting N uptake accordingly.

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## CHAPTER 6

### SUMMARY FOR ARNEWS CLIMATIC AND ATMOSPHERIC DATA

#### INTRODUCTION

The Acid Rain National Early Warning System (ARNEWS) plot network was initiated in 1984 by Canadian Forest Service (CFS) to establish a national program to detect early signs of air pollution damage to Canada's forest. Since that time, more than 150 ARNEWS plots have been established across Canada to monitor changes in forest vegetation and soils caused by air pollution (D'Eon *et al.* 1994).

To apply the ForSust model, 125 ARNEWS plots are selected to be the primary data source for this study. Each ARNEWS plot is a 10 m by 40 m rectangle (0.04 ha). To reduce measurement errors, the longer side of each plot is normally positioned parallel to a contour line in the stand. The ARNEWS plots are generally located in large stands (minimum stand area of 2 ha), away from stand edges and local disturbances (e.g., roads), and contain enough off-plot trees for destructive sampling.

To run the ForSust model, several climatic and atmospheric variables need to be compiled for each site. These variables include mean annual precipitation, annual fog water deposition, mean annual temperature, mean July temperature, actual evapotranspiration (AET), wet and dry deposition of anions ( $\text{NO}_3^-$ ,  $\text{SO}_4^{2-}$ ,  $\text{Cl}^-$ ) and cations ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ ,  $\text{Na}^+$ ,  $\text{H}^+$ ,  $\text{NH}_4^+$ ). Temperature is an important factor affecting evapotranspiration, and therefore controls, in part, soil percolation rate, and available soil water. Precipitation affects AET and flow rate. Atmospheric wet deposition is a function of precipitation amount and ions concentration in precipitation.

This chapter summarizes climatic and atmospheric data for the ARNEWS plots.

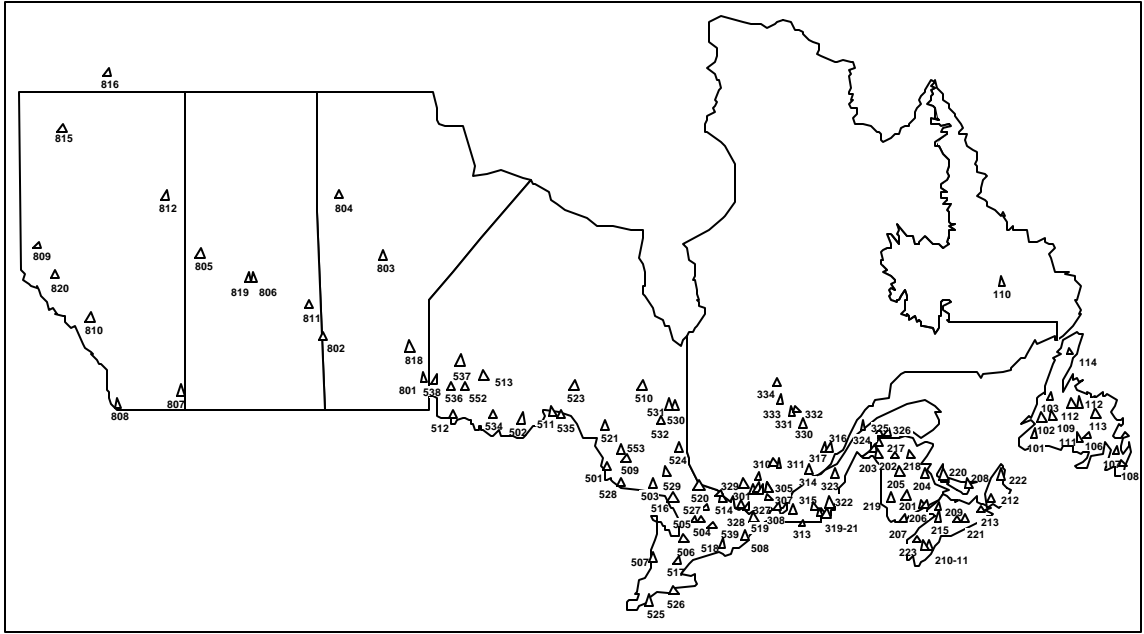
The objectives of this chapter, therefore, are:

1. To describe the methods applied for compiling climatic and atmospheric data for ARNEWS sites.
2. To provide a complete data set as required for the model inputs for sustainability assessments of long-term biomass productivity and nutrient input/output balances at the ARNEWS sites.
3. To analyze and discuss the distributions, magnitudes, and variations of data in the selected ARNEWS sites.

## MEHTODS

### Study sites

Distributed from Newfoundland to Alberta, 125 monitoring ARNEWS plots (D'Eon and Power, 1989) were selected to be the primary data source for this study (Figure 6.1). The main reason for excluding the BC and Yukon region's plots was related to inaccessibility to atmospheric deposition data for that region. The selected plots vary in terms of location (latitude, longitude, and altitude), geology and soil types, cover type (species composition), age status (e.g., even- or uneven-aged stand), maturity (e.g., immature or mature stand), stocking and growth characteristics (e.g., tree density, average tree diameter, basal area, and tree height), see Table A.1 and Table A.2 (Appendix A). Inventory data for dominant tree species per plot are available in ARNEWS CD-ROM, 2000 (Inv-data folder, Inv-186 file).



**Figure 6.1** The Acid Rain National Early Warning System (ARNEWS) plot network.

## Compiling climatic and atmospheric data

### *Precipitation and temperature*

The data from Canada climate CD-ROM (Environment Canada, 1994) was used to compile mean annual precipitation ( $\text{mm yr}^{-1}$ ), mean annual temperature ( $^{\circ}\text{C}$ ), and mean July temperature ( $^{\circ}\text{C}$ ) for each ARNEWS site. Mean annual precipitation and temperature are averages of the normal data (1963-1993). Mean July temperature is a monthly average. MapInfo software (MapInfo Professional, 1996) was used to choose the closest weather station to each ARNEWS site. Spatial averages were calculated (weighted by distance from the site) for plots with more than one station nearby.

Fog water deposition is very difficult to record so that a lack of information at local scale is common. Because of its difficulty of direct measurement, several mathematical formulas have been developed. Normally, fog water deposition is estimated by predicting the number of fog hours and the rate of deposition of fog water droplets. In this study, fog water deposition for the ARNEWS sites located in coastal areas of the Atlantic Provinces was estimated based on the following equation:

$$\text{Fog} = (0.018 * \text{MAP}) - (83.874 * (1 / (\text{MAT} - 6.5))) \quad r^2 = 0.97$$

where Fog is fog water deposition ( $\text{mm yr}^{-1}$ ), MAP is mean annual precipitation ( $\text{mm yr}^{-1}$ ), and MAT is mean annual temperature ( $^{\circ}\text{C}$ ). This equation was generated from data belong to 25 sites in New Brunswick (Simpson and Arp, 1996).

### *Evapotranspiration*

A set of data including stream discharge (Q) and precipitation (P) at 765 hydraulic stations across Canada were used to calculate actual evapotranspiration based on the water balance approach ( $AET = P - Q$ ). Calculated AET (dependent variable) was related to climatic and geometric variables (independent variables) by way of multiple regression (Figure 6.2). The following equation was obtained:

$$AET = [(16.298 * T_{July} ) + (0.015* T_{July} * Ppt) - (10.474 * Latitude) + (4.943*Longitude)] \quad r^2 = 0.97$$

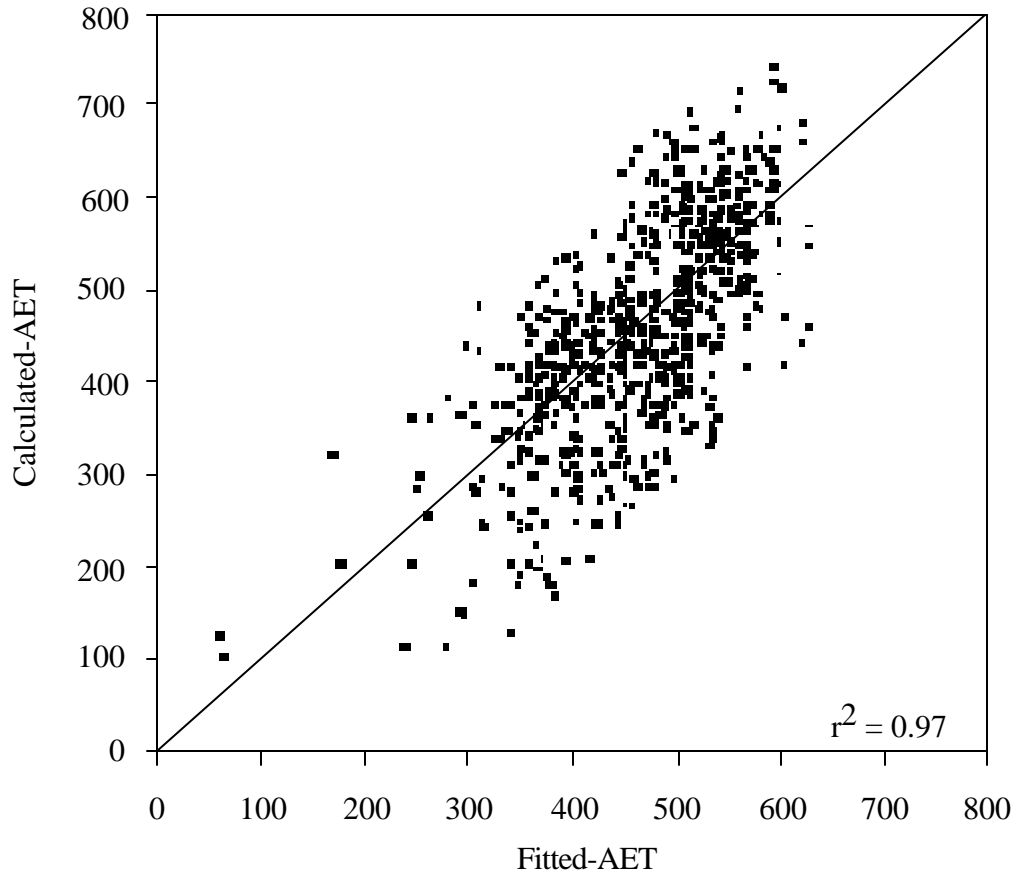
where AET = actual evapotranspiration ( $\text{mm yr}^{-1}$ ),  $T_{July}$  = mean July temperature ( $^{\circ}\text{C}$ ), and Ppt = mean annual precipitation ( $\text{mm yr}^{-1}$ ).

### *Atmospheric deposition*

#### *Wet deposition*

Rates of wet (precipitation) deposition for cations ( $\text{K}^+$ ,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{Na}^+$ ,  $\text{H}^+$ ,  $\text{NH}_4^+$ ) and anions ( $\text{NO}_3^-$ ,  $\text{SO}_4^{2-}$ ,  $\text{Cl}^-$ ) were taken from atmospheric stations with deposition data (Environment Canada, special data transfer) closest to each ARNEWS site, and by converting observed ion concentrations into deposition fluxes, based on local precipitation rates.

	Coefficient	Std. Error	Std. Coeff.	t-Value	P-Value
Longitude	4.943	.420	.606	11.756	<.0001
Latitude	-10.474	.742	-.281	-14.116	<.0001
T_July	16.298	1.871	.255	8.712	<.0001
Ppt*T_July	.015	.001	.619	12.577	<.0001



**Figure 6.2** Fitted-AET vs. estimated-AET in 765 stations across Canada, relating AET to the variables such as temperature, precipitation, latitude, and longitude.

### *Fog deposition*

The concentration of chemical element in fog deposition was obtained from atmospheric stations (Environment Canada, special data transfer) with deposition data closest to each ARNEWS plot. The observed ion concentrations were converted into the deposition fluxes by multiplying each ion concentration to estimated fog water deposition rate.

### *Dry deposition*

Because of insufficient data to estimate dry deposition based on available dry deposition models, dry deposition rates sites were estimated from wet deposition rates, and using a ratio of dry to wet deposition as follows:

$$DD_i = WD_i * (DD/WD)_E * f$$

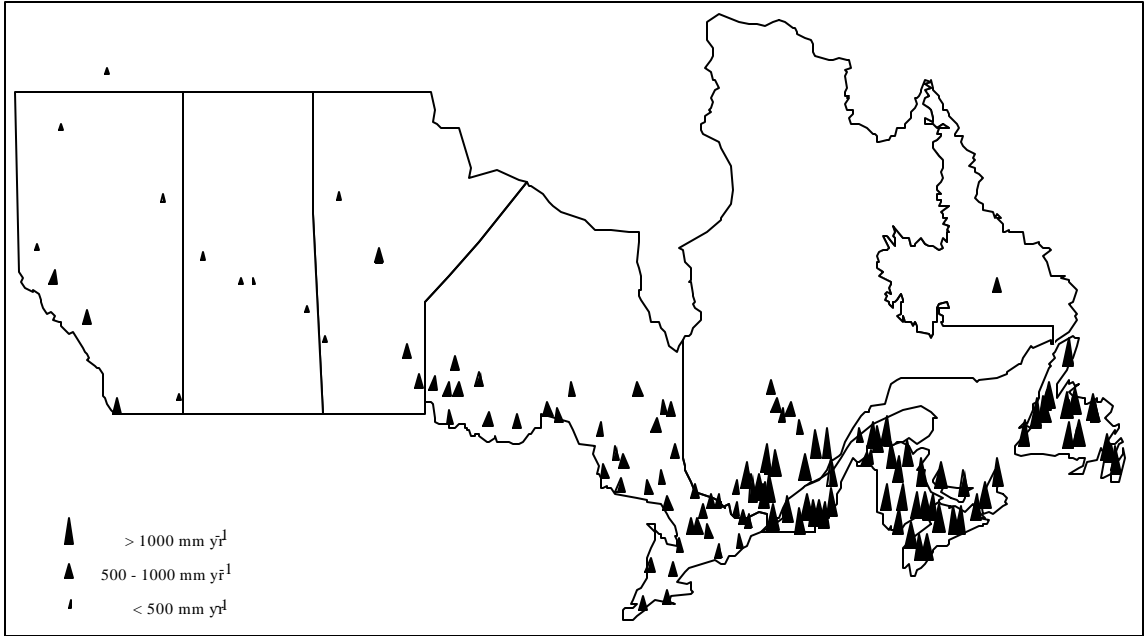
where  $DD_i$  is dry deposition rate ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),  $WD_i$  is wet deposition rate ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ),  $(DD/WD)_E$  is the ratio of dry to wet deposition in eastern Canada, and  $f$  is an adjustment factor for differences in weather condition. According to an intensive reviewing of the available documents on wet and dry deposition in Canada and the U.S (see chapter 3), dry/wet deposition ratio values of 29 % and 28 % were chosen for  $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$ , respectively. For cations ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ , and  $\text{NH}_4^+$ ), a ratio of 15% was used.  $f$  was assumed to be 1 for eastern provinces (Newfoundland to Ontario), and 2 for the typically drier surface conditions in the Prairie Provinces (Shannon and Sisterson, 1992).

## RESULTS AND DISCUSSION

The climatic and atmospheric data were compiled for 125 studied ARNEWS sites as shown Table A.3 (Appendix A). These data were converted to appropriate units, and were coordinated with the ARNEWS vegetation data for the purpose of model input (ARNEWS CD-ROM 2000, Atm-data folder, Atm-186 file).

Figure 6.3 illustrates the pattern of mean annual precipitation for ARNEWS plots across Canada. In general, precipitation rates decrease from east to west (excluding British Columbia province). Atlantic Provinces, southern Quebec, and southwestern Ontario have high precipitation rates ( $> 1000 \text{ mm yr}^{-1}$ ) which is likely due to the frequent passage of frontal systems over southeastern Canada (Bourque, personal communication). Precipitation in central Quebec, most parts of southern and northern Ontario, east of Manitoba is ample (500-1000 mm). Areas located in Manitoba, Saskatchewan, and most of Alberta receive low annual precipitation. Higher annual precipitation rates in southwestern Alberta near B.C. are due to local factors, especially altitude.

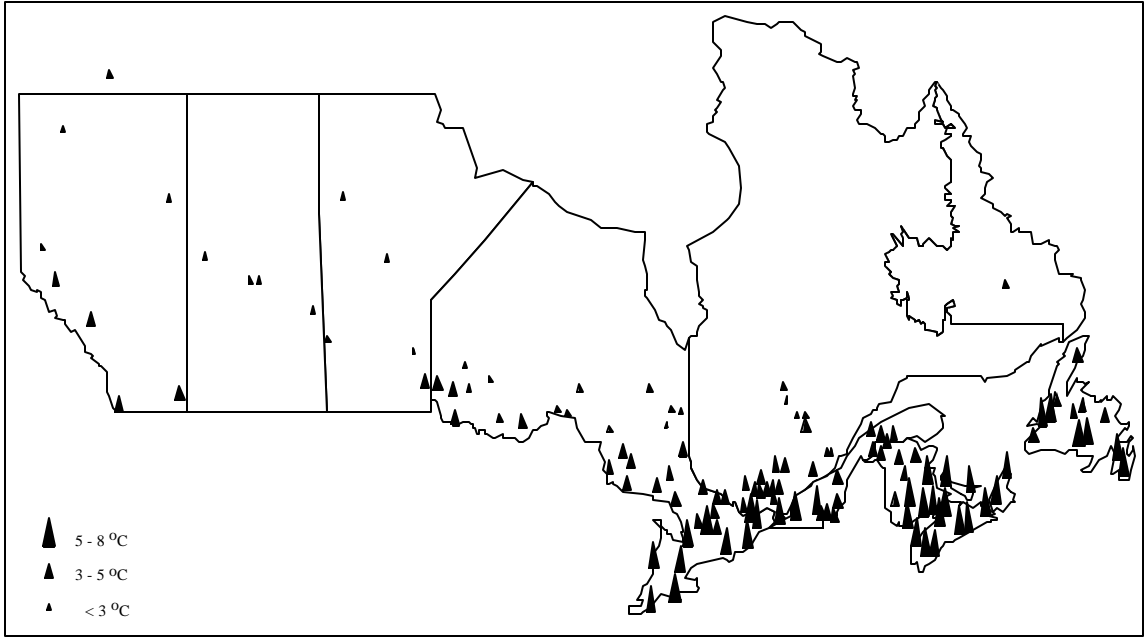
The estimated annual rates of fog water deposition for the 37 ARNEWS sites in the Atlantic Provinces show that a significant amount ( $30\text{-}196 \text{ mm yr}^{-1}$ ) of moisture is added to these areas by fog. Less than  $100 \text{ (mm yr}^{-1}\text{)}$  of fog occurs in the provinces of NF, NB, and eastern PEI. Along the coasts of NS and PEI, fog deposition rates are estimated to exceed  $100 \text{ (mm yr}^{-1}\text{)}$ . In general, fog water deposition ranged 3-17% of the annual total precipitation. Yin and Arp (1994) reported that the simulated annual fog water deposition for 6 sites in Maritime Provinces amounted to 60-110 mm, or 5-8% of the recorded annual precipitation.



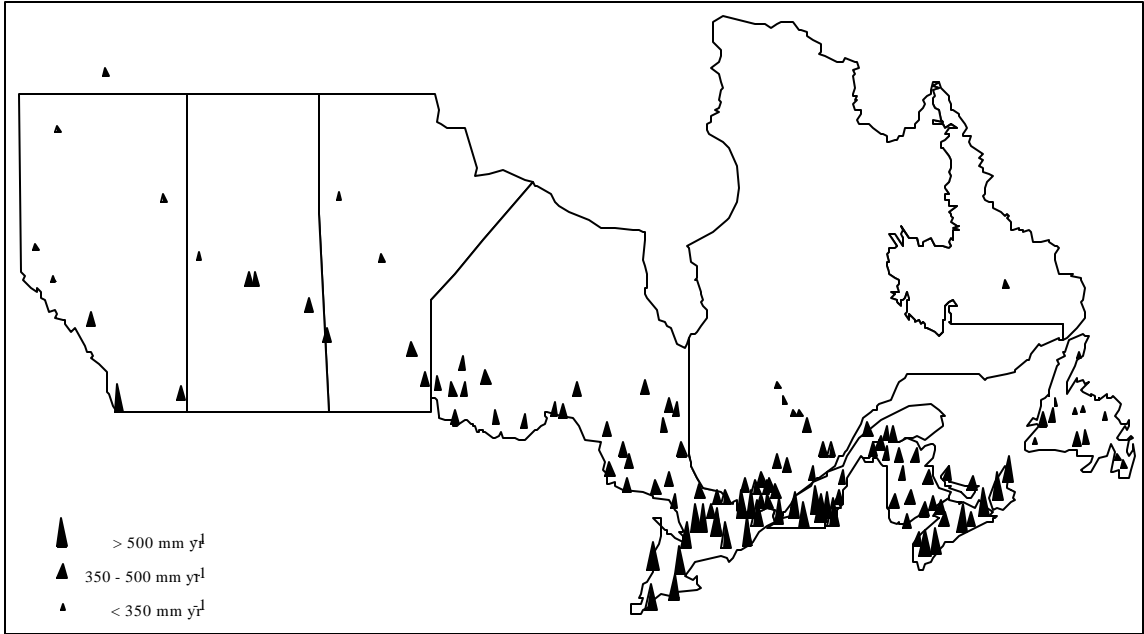
**Figure 6.3** Mean annual precipitation (ARNEWS plots).

Figure 6.4 shows variation of mean annual temperature from Newfoundland to Alberta. As can be seen, the mean annual temperatures in the Atlantic Provinces, in southern Quebec and in Ontario (3-8 °C) are higher than in northern provinces (Alberta, Saskatchewan, and Manitoba), northern Ontario and Quebec (< 3 °C). The areas with highest temperature are located in the Southeastern Climatic Region and have warm summers with at least 4 months above 10 °C. The areas with lowest temperature are mostly located in the Northern Climatic Region where no more than 4 months have a mean temperature of 10 °C or above (Bourque, personal communication). In general, annual temperatures decrease with increasing latitude but are also affected by local microclimate. For example, areas near the Great Lakes and along the Atlantic coast have highest mean annual temperatures of 5-8 °C. The southern parts of the Prairies are quite warm (hot) in summer.

The highest annual rates (> 500 mm) of actual evapotranspiration (Figure 6.5) take place in the most southern parts of Ontario, Quebec, and in the coastal areas of Nova Scotia. AET rates are low (< 350 mm) in the northern parts of the Prairies, in central north of Quebec, and in most areas of Newfoundland. Most areas across the country lose 350-500 mm water by annual evapotranspiration. Considering the distributions of actual evapotranspiration, mean annual precipitation, and mean July temperature, we can conclude that the reason for low AET in Newfoundland sites is due to low July temperatures (precipitation is very high). In the North, AET is low because of both low precipitation rates and low July temperatures.



**Figure 6.4** Mean annual air temperature (ARNEWS plots).



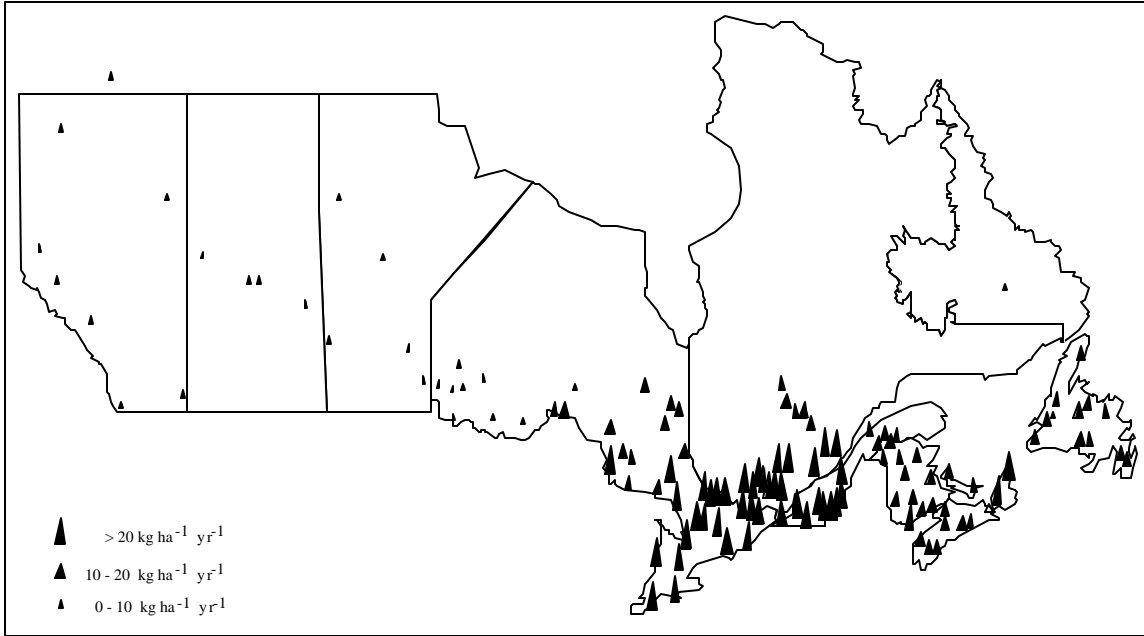
**Figure 6.5** Actual Evapotranspiration (ARNEWS plots).

Table 6.1 shows the annual rates of precipitation deposition ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) for cations ( $\text{K}^+$ ,  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{Na}^+$ ,  $\text{H}^+$ ,  $\text{NH}_4^+$ ) and anions ( $\text{NO}_3^-$ ,  $\text{SO}_4^{2-}$ ,  $\text{Cl}^-$ ) for 125 ARNEWS plots distributed from Newfoundland to Alberta. As can be seen, average rates of sulfur ( $\text{SO}_4^{2-}\text{-S}$ ) and nitrogen ( $\text{NO}_3^-\text{-N}$ ) deposition are relatively high (16.3 and 11.8  $\text{eq ha}^{-1} \text{yr}^{-1}$ , respectively). Among base cations, calcium deposition is higher (1.4  $\text{eq ha}^{-1} \text{yr}^{-1}$ ) than magnesium and potassium deposition (0.4  $\text{eq ha}^{-1} \text{yr}^{-1}$ ).

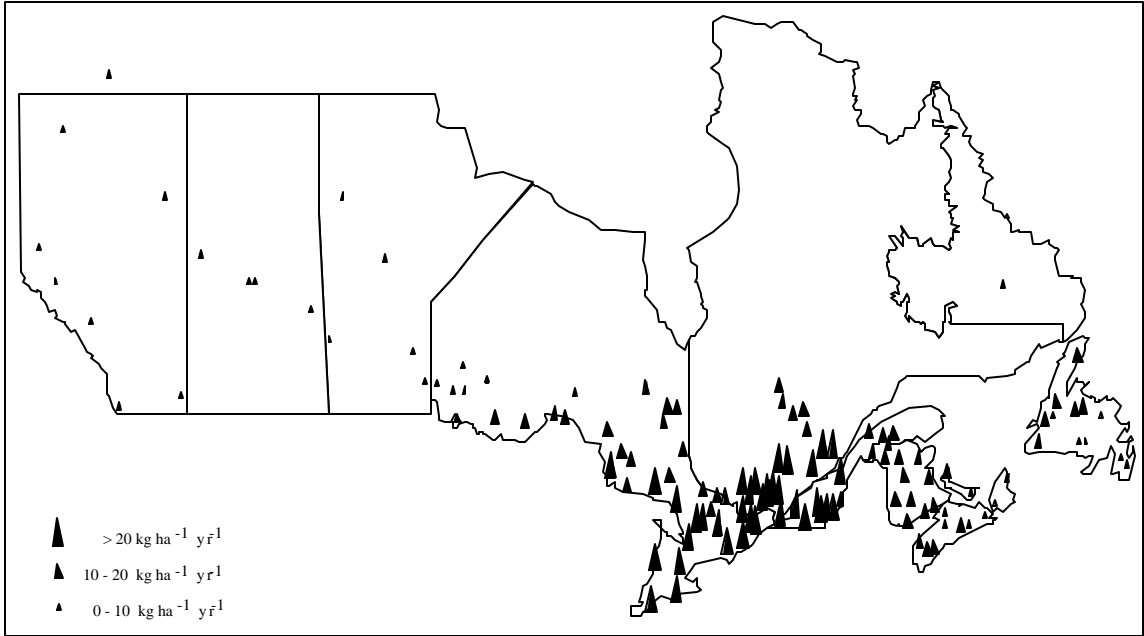
**Table 6.1** Annual rates of wet deposition (average, minimum, maximum, and standard deviation) of cations and anions in the 125 ARNEWS plots.

Ion	Average	Minimum	Maximum	SD
	$\text{Kg ha}^{-1} \text{yr}^{-1}$			
$\text{K}^+$	0.40	0.09	1.31	0.24
$\text{Ca}^{2+}$	1.39	0.30	5.19	0.77
$\text{Mg}^{2+}$	0.44	0.10	2.44	0.45
$\text{Na}^+$	2.17	0.10	18.42	3.33
$\text{H}^+$	0.30	0.01	0.67	0.18
$\text{NH}_4^+$	2.49	0.50	6.61	1.37
$\text{NO}_3^-$	11.78	2.00	26.96	6.65
$\text{SO}_4^{2-}$	16.30	1.74	37.43	8.18
$\text{Cl}^-$	4.11	0.23	33.97	6.15

Although significant amounts of chemical elements are added to forest ecosystems via atmospheric deposition, the addition of sulfate and nitrate ions are most important. Depending on nutrient status of soil and vegetation, deposition of  $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$  could be beneficial or harmful for plant growth. Figures 6.6 and 6.7 illustrate the spatial distribution  $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$  deposition across Canada (except BC), respectively.



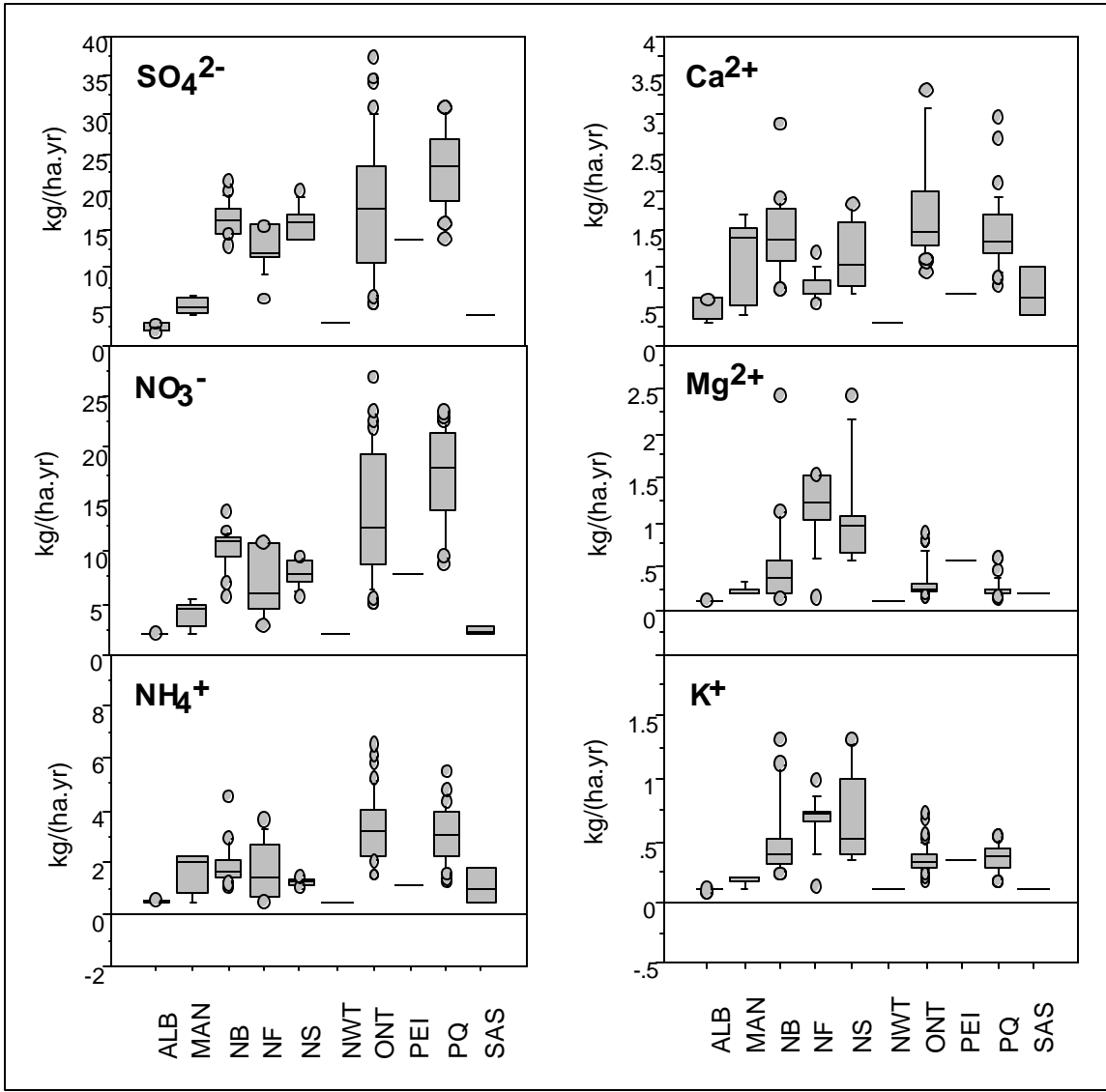
**Figure 6.6** Sulfur ( $\text{SO}_4^{2-}$  -S) deposition in precipitation (ARNEWS plots).



**Figure 6.7** Nitrogen ( $\text{NO}_3^-$ -N and  $\text{NH}_4^+$ -N) deposition in precipitation (ARNEWS plots).

Southern Quebec and Ontario, and part of Nova Scotia (Cape Breton Island) obtain the highest rates of  $\text{SO}_4^{2-}$  deposition ( $> 20 \text{ eq ha}^{-1} \text{ yr}^{-1}$ ). Central areas of Quebec and Ontario and most parts of the Atlantic Provinces receive moderate rates of  $\text{SO}_4^{2-}$  deposition ( $10\text{-}20 \text{ eq ha}^{-1} \text{ yr}^{-1}$ ). Other areas including northern Ontario and Northern Provinces obtain low rates ( $< 10 \text{ eq ha}^{-1} \text{ yr}^{-1}$ ) of  $\text{SO}_4^{2-}$  deposition. The highest rates of the  $\text{NO}_3^-$  precipitation deposition happen in the southern parts of Ontario and Quebec. New Brunswick, central to southern parts of Ontario and Quebec receive 10 to 20 ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ )  $\text{NO}_3^-$  from precipitation. Wet deposition of  $\text{NO}_3^-$  is relatively small in Northern Provinces, northern Ontario, NS, PEI, and NF ( $< 10 \text{ eq ha}^{-1} \text{ yr}^{-1}$ ).

Provincial/regional rates of atmospheric deposition are shown with box plots in Figure 6.8. The highest deposition rates of sulfate and nitrate happen in Ontario and Quebec. Atlantic Provinces receive the second highest, and the Northern Provinces (Alberta, Manitoba, Saskatchewan, and North West Territories) receive the lowest deposition rates of sulfate and nitrate. The pattern of ammonium deposition differs slightly with the sulfate and nitrate deposition pattern: the rates of ammonium deposition in Manitoba, and Saskatuan are almost the same as for the Atlantic Provinces. Highest rates of calcium deposition occur in Ontario, Quebec, and New Brunswick. Magnesium and potassium deposition rates are relatively higher in Newfoundland and Nova Scotia than the other provinces. The lowest deposition rates of these elements happen in the Northern Provinces.



**Figure 6.8** Provincial precipitation deposition rates of sulfate, nitrate, ammonium, calcium, magnesium, and potassium for ARNEWS plot network.

The estimated rates of fog deposition for the ARNEWS sites in Atlantic Provinces are shown in Table (6.2). Like precipitation deposition, fog deposition rates of  $\text{NO}_3^-$  and  $\text{SO}_4^{2-}$  were relatively high, ranging 1.6-10.2 ( $\text{kg ha}^{-1} \text{ yr}^{-1}$ ) and 2.7-17.7 ( $\text{kg ha}^{-1} \text{ yr}^{-1}$ ), respectively. The amounts of  $\text{NO}_3^-$  and  $\text{SO}_4^{2-}$  deposited by fog are 18-174% and 21-126% of those deposited by precipitation, respectively. These high ratios are caused by high chemical concentrations in (Yanni 1994).

**Table 6.2** Estimated annual rates of fog deposition of cations and anions in the 37 ARNEWS plots (Atlantic Provinces).

Ion	Average	Minimum	Maximum	Sd. Dev.
$\text{kg ha}^{-1} \text{ yr}^{-1}$				
$\text{K}^+$	0.3	0.1	0.6	0.2
$\text{Ca}^{2+}$	0.5	0.2	1.1	0.3
$\text{Mg}^{2+}$	0.9	0.3	1.9	0.5
$\text{Na}^+$	7.6	2.4	15.7	4.4
$\text{H}^+$	0.1	0.01	0.3	0.1
$\text{NH}_4^+$	1.0	0.3	2.0	0.6
$\text{NO}_3^-$	5.0	1.6	10.2	2.9
$\text{SO}_4^{2-}$	8.6	2.7	17.7	4.9
$\text{Cl}^-$	12.8	4.1	26.4	7.4

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## CHAPTER 7

### SUMMARY FOR ARNEWS SOIL DATA

#### INTRODUCTION

This chapter summarizes ARNEWS soil data for the purpose of assessing extent of soil acidification and long-term forest sustainability by way of the ForSust model. The data so summarized refer to soil depth (depth of rooting space), amounts of exchangeable base cations ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^{+}$ ), available P, total N, organic matter content (OM), C/N ratio, pH, soil bulk density (Db), weight percent of clay, and type of soil substrate with regard to soil acidification buffering.

A preliminary analysis revealed that soil data vary by region and by forest type. Subsequently, these data were summarized by **5 regions**:

1. Newfoundland and Labrador Region;
2. Maritime Region (New Brunswick, Nova Scotia, and PEI);
3. Quebec Region;
4. Ontario Region;
5. Northwest (Prairie) Region (Alberta, Saskatchewan, Manitoba, and NWT);

and by **6 forest cover types**:

1. intolerant hardwood (plots dominated with white birch [*Betula papyrifera* Marsh.], largetooth aspen [*Populus grandidentata* Michx.], trembling aspen [*Populus tremuloides* Michx.]);
2. tolerant hardwood (plots dominated with sugar maple [*Acer saccharum* Marsh.], red maple [*Acer rubrum* L.], yellow birch [*Betula alleghaniensis* Britton], beech [*Fagus*

- grandifolia* Michx.), red oak [*Quercus rubra* L.], black oak [*Quercus velotina* Lam.], and white oak [*Quercus alba* L.]);
3. mixed wood (plots dominated with hardwood-softwood or softwood-hardwood);
  4. pine (plots dominated with jack pine [*Pinus banksiana* Lamb.], red pine [*Pinus resinosa* Ait.], white pine [*Pinus strobus* L.], lodgepole pine [*Pinus contorta* Lamb.]);
  5. spruce (plots dominated with red spruce [*Picea rubens* Sarg.], Norway spruce [*Picea abies* (L.) Karst.], white spruce [*Picea glauca* (Moench) Voss]);
  6. Softwood (plots dominated with mixture of softwood species).

Preliminary examination of the data revealed instances for which plot data for specific soil variables were missing. For this reason, it was decided to analyze the correlation pattern among the various variables, and to obtain from this a means to replace the missing data by way of regression analysis.

Since estimating soil weathering rates are important for assessing the potential of forest soils to buffer against soil acidification, it was further decided to estimate the rate of soil weathering for each ARNEWS plots based on the compiled soil information. Variables critical for doing so are clay content, type of soil substrate with regard to soil acidification buffering, and soil depth.

The objectives of the Chapter are:

1. to examine to what extent the summarized soil data for each ARNEWS plot from Newfoundland to Alberta vary with forest cover type and with region;
2. to document the procedures used to compile these data;

3. to document the relationships used for replacing missing data;
4. to include in all of this the soil base cation weathering rates as estimated for each ARNEWS plot.

## METHODS

The original ARNEWS soil data were transferred from the Canadian Forest Service (ARNEWS CD-ROM 2000, Sol-data folder, Soil-CFS file). The data set included three files (forest floor, mineral A soil, mineral B soil) listed by ARNEWS plot number, year collected (1985, 1990, and 1993), and the following variables (bold indicates the data used for the purpose of this study.):

### Organic layer

**Thickness** (cm), **pH**, **OM** (%), **CEC** (cmol kg<sup>-1</sup>), and base saturation (%).

Total (ppm) **N**, **P**, **Ca**, **Mg**, **K**, **Na**, **S**, **Fe**, **Mn**, **Zn**, **Cu**, **Al**, **B**, **Pb**, **Ni**.

Exchangeable (cmol kg<sup>-1</sup>) **P**, **Ca**, **Mg**, **K**, **Na**, **S**, **Fe**, **Mn**, **Zn**, **Cu**, **Al**, **B**, **Pb**, **Mo**.

### Mineral A and B layers

**Thickness** (cm), **pH**, **OM** (%), **CEC** (cmol kg<sup>-1</sup>), and base saturation (%).

Total (ppm) **N**, **P**, **Ca**, **Mg**, **K**, **Na**, **S**, **Fe**, **Mn**, **Zn**, **Cu**, **Al**, **B**.

Exchangeable (cmol kg<sup>-1</sup>) **P**, **Ca**, **Mg**, **K**, **Na**, **S**, **Fe**, **Mn**, **Zn**, **Cu**, **Al**, **B**, **Mo**.

The above data were determined by the following soil analysis procedure (D'Eon *et al.* 1994):

PH in 0.01 M CaCl<sub>2</sub> (field-fresh sample);

Organic matter (loss-on-ignition);

Total N (semi-micro Kjeldahl procedure or equivalent);

Total P, Ca, Mg, K, S, Fe, Mn, Zn, Cu, Al, B, Pb, Ni (HClO<sub>4</sub> digest);

Cation exchange capacity (1.0 M NH<sub>4</sub>Cl extractant, unbuffered);

Exchangeable Ca, Mg, K, Na, S, Fe, Mn, Zn, Cu, Al, B, Pb, Mo (1.0 M NH<sub>4</sub>Cl extractant, unbuffered);

Percent base saturation (by summation).

In some cases, the ARNEWS soil data were not quite complete. Therefore, the available soil data were used to determine to what extent these variables were correlated with one another. Highly correlated variables provide a means to estimated missing values by way of regression analysis. Multiple regression equations were established to estimate missing data (see Tables 7.1-7.3). For conversion purposes, organic matter was assumed to contain 58% organic carbon (Walkley 1946). To apply as the model input, exchangeable Ca, Mg, K, and P were calculated as follows:

$$\text{Ca, Mg, K, P (eq ha}^{-1}\text{)} = \text{Ca, Mg, K, P (cmol kg}^{-1}\text{)} * \text{Db (g cm}^{-3}\text{)} * \text{depth (cm)} * 1000$$

The following procedures were used to estimate the other required variables including soil bulk density (Db), percent clay content, soil substrate, and base cation weathering rates.

1. Data from unrelated studies in New Brunswick were used to determine Db relationships. Three regression equations were established to estimate Db ( $\text{g cm}^{-3}$ ) for organic layer, mineral A and B layers (see Tables 7.1 – 7.3).
2. Clay contents (%) in A and B horizons were calculated by applying the following equations (Meyer *et al.* 1994):

$$\text{Clay (\%)} = [7.0 + \text{CEC (cmol kg}^{-1}\text{)} - 0.82 * \text{org. C (\%)} - 1.4 * \text{pH}] / 0.29 \quad (1)$$

$$\text{Clay (\%)} = [0.2 + \text{CEC (cmol kg}^{-1}\text{)} - 0.67 * \text{org. C (\%)}] / 0.33 \quad (2)$$

3. Two additional clay content estimates were obtained by applying species-specific clay – CEC equations (Meyer *et al.* 1994, see Table 7.3). The resulting four estimates were then averaged to arrive at a final value. Where available, given texture class data were compared with clay estimates to check overall validity. Clay estimates were generally in agreement with values expected for given texture classes (some estimates that seemed "unreasonable" were omitted before averaging). Where estimates were outside given class ranges, class endpoint values were used instead. For the plots in Maritimes, regression-estimated clay content values were much higher than expected and were assumed to be invalid (over-estimates resulted from using the available CEC data, which were unusually high). For these soils, clay contents were estimated by assuming representative texture classes associated with parent materials for each plot.

4. Soil substrate was determined by matching ARNEWS plot locations with general soil types, based on the national soil maps (Clayton *et al.* 1977). Soil substrates were classified based on the classification used by De Vries (1991) as follows:
- (i) *Acidic*: Sand; sandstone; siliceous sediments (schists, shales); gravel; granite (low biotite variety); quartzite; gneiss; glacial till derived from these materials.
  - (ii) *Intermediate*: Granodiorite; siliceous sediments with small amounts of carbonates; glacial till derived from these materials; fluvial, lacustrine, and marine sediments.
  - (iii) *Basic*: Gabbro; basalt; dolomite; volcanic deposits; calcite.
5. Soil weathering rates were calculated with a simple scheme of assuming that weathering rates are related to soil parent material and to the texture class for each site (De Vries *et al.* 1993). The assumption that texture class has a strong influence on the weathering rate is based on a linear relationship between weathering rate and soil internal surface area. The latter is directly related to clay content, as realized by Sverdrup *et al.* (1990). The following equations (Arp, unpublished data) were used to estimate base cation weathering rates ( $\text{eq ha}^{-1} \text{ yr}^{-1} \text{ m}^{-1}$ ) for each parent material (PM) class:
- $$\text{Acidic PM} = [56.7 \times \% \text{clay}] - [0.32 \times \% \text{clay}^2] \quad (3)$$
- $$\text{Intermediate PM} = [53.6 \times \% \text{clay}] - [0.18 \times \% \text{clay}^2] + 500 \quad (4)$$
- $$\text{Basic PM} = [59.2 \times \% \text{clay}] + 500 \quad (5)$$
6. A data set was compiled to include the following information (see soil-dat2 file, Appendix C, CD ROM attachment):

**FF layer:** depth (cm), exchangeable Ca, Mg, K, P (eq ha<sup>-1</sup>), total N (ppm), OM (%), C/N, pH, Db (g cm<sup>-3</sup>).

**A and B layers:** same as for FF layer, clay (%), PM class (1=acidic, 2=moderate, 3=basic), and an estimate for base cation weathering rate (eq ha<sup>-1</sup> yr<sup>-1</sup> m<sup>-1</sup>).

**Rooting zone:** the FF, A and B layer were further summarized for the entire rooting zone (FF, A, and B layers combined) by summing depth, exchangeable Ca, Mg, K, P (eq ha<sup>-1</sup>), and/or weight-averaging (on the basis of soil depth) total N (ppm), OM (%), C/N, and pH.

The data so summarized were then analyzed by way of box plots, by region and by tree species. Each box plot is composed of five horizontal lines that display the 10<sup>th</sup>, 25<sup>th</sup>, 50<sup>th</sup> (median), 75<sup>th</sup>, 90<sup>th</sup> percentiles of a variable. Values above the 90<sup>th</sup> percentile and below the 10<sup>th</sup> percentile are plotted as points (outliers).

In addition, the data so summarized were mapped by ARNEWS plot location to show the spatial distributions of the soil variables.

## RESULTS AND DISCUSSION

### Correlation among soil variables and estimation of the missing data

Relationships between various soil variables used for estimating missing data are compiled in Tables 7.1, 7.2, and 7.3, for the forest floor, the A layer, and the B layer, respectively. These equations are interesting by themselves for their qualitative and quantitative implications, and are discussed below, layer by layer.

### *Forest floor relationships*

Shown in Table 7.1, are equations used to estimate missing data for forest floor layer in ARNEWS plots.

**Table 7.1** Equations used to estimate missing data of forest floor layer in ARNEWS plots.

Forest Floor	Equation	n	r <sup>2</sup>	Data Filled
% Carbon	$= \exp[[0.517 \times \ln (\% \text{ tot. N})] - [0.108 \times \text{pH}] + 4.061]$	104	0.44	2
Avail. P (ppm)	$= 0.215 \times \text{tot. P in FF (ppm)} + 3.266$	57	0.65	54
pH	$= [0.000195 \times \text{exch. Ca (ppm)}] + 3.521$	115	0.54	3
Bulk Density (g/cm <sup>3</sup> )	$= [\exp (-0.0903 \times \% \text{OM}) + 0.0739] \times 1.377$	65	0.92	102
Thickness (cm)	$= [0.0489 \times \% \text{OM}] - [1.751 \times \text{pH}] + 9.634$	60	0.57	14

Percent organic carbon is, as to be expected, positively (+) related to percent total N, but organic carbon is also negatively (–) related to soil pH, such that % carbon decreases with increasing pH. This decrease is likely related to increased soil microbiological activity with increasing pH. Note that forest floor pH is also related to cover type, with tolerant hardwoods often having a high forest floor pH compared to softwood forests.

The positive relationship between available P to total P of the forest floor is to be expected: the higher the total P content, the more P is likely available on account of organic matter mineralization.

Due to the general relationship between base cation content and soil pH, it is not surprising to obtain a strong positive relationship between pH and exchangeable Ca in the forest floor.

Since organic matter tends to bind soil particles, it is not surprising to see a strong negative relationship between soil bulk density and soil organic matter content for the forest floor. There is the complication that forest floor layers may also contain considerable amounts of mineral soil particles, due to biomixing, or by some other processes (including sampling). High soil mineral contents cause high soil bulk densities for the forest floor.

The relationship between forest floor thickness and organic matter (+) and pH (–) refers to a corresponding decrease of soil mineral particles in the forest floor, and to the fact that forest floor layers with high pH values are generally thin.

#### *Mineral soil relationship, A layer*

Shown in Table 7.2 are equations used to estimate missing data for the A soil layer in the ARNEWS plots. In mineral A layers, % carbon content is strongly related to % total N, and vice versa. This relationship is expected because each soil type/layer has a specific C/N ratio such that:

$$C/N (\text{forest floor}) > C/N (\text{A layer}) > C/N (\text{B layer}).$$

Where N data is not available, % carbon is estimated based on exchangeable Ca and exchangeable K.

**Table 7.2** Equations used to estimate missing data of A layer in ARNEWS plots.

A Horizon	Equation	n	r <sup>2</sup>	Data Filled
% Carbon	= exp [[0.946 x ln (% tot. N)] + 3.096]	51	0.86	17
% Carbon (no N data)	= [0.304 x exch. Ca (cmol/kg)] [9.601 x exch. K (cmol/kg)] + 2.030	94	0.51	1
% Nitrogen	= exp [[0.907 x ln (%C)] – 3.067]	51	0.86	32
Avail. P (ppm)	= [0.0040 x tot. N (ppm)] – [0.0084 x exch. Ca (ppm)] + [0.3299 x exch. K (ppm)] – 0.2824	44	0.81	54
pH	= [0.079 x exch. Ca (cmol/kg)] – [0.044 x exch. Al (cmol/kg)] + 3.818	85	0.68	1
pH (no Al data)	= [0.100 x exch. Ca (cmol/kg)] – 0.688 x exch. K (cmol/kg)] + [0.324 x exch. Na (cmol/kg)] + 3.915	110	0.42	1
Bulk Density (g/cm <sup>3</sup> )	= [exp (–0.0903 x %OM) + 0.0739] x 1.377	65	0.92	97
Thickness (cm)	= [5.414 x pH] – [7.0251 x exch. K (cmol/kg)] – 13.15	49	0.63	19

Available P in the mineral soil was related to amount of total N (+), exchangeable Ca (–), and exchangeable K (+). Due to the buffering potential of Ca and the acidifying potential of Al, pH was closely related to exchangeable Ca (+) and exchangeable Al (–), as to be expected. Where Al data was not available, best pH correlation was found with Ca (+), K (–), and Na (+).

Again, as to be expected, soil bulk densities in A layers have a strong negative relationship to organic matter content. Thickness of A layers is related to pH (+), and to exchangeable K (–).

*Mineral soil relationship, B layer*

Shown in Table 7.3 are equations used to estimate missing data for soil B layer in ARNEWS plots.

**Table 7.3** Equations used to estimate missing data for B layer in ARNEWS plots.

B Horizon	Equation	n	r <sup>2</sup>	Data Filled
% Carbon	= exp [[0.892 x ln (% tot. N)] + 3.206]	85	0.82	17
% Carbon (no N data)	= [0.823 x exch. Al (cmol/kg)] + 2.139	110	0.52	1
% Nitrogen	= exp [[0.917 x ln (%C)] - 3.389]	85	0.82	31
Avail. P (ppm)	= [0.7047 x avail. P in A horizon (ppm)] + 2.316	50	0.89	54
pH	= [0.102 x exch. Ca (cmol/kg)] - [0.093 x exch. Mg (cmol/kg)] - [0.086 x exch. Al (cmol/kg)] + 4.684	111	0.64	1
Bulk Density (g/cm <sup>3</sup> )	= [exp (-0.0903 x %OM) + 0.0739] x 1.377	65	0.92	100
Thickness 1 (cm) *	= [5.330 x pH] - [110.0 x exch. K (cmol/kg)] + 16.52	26	0.66	20
Thickness 2 (cm) *	= [1.190 x tot. N (cmol/kg)] - [34.37 x exch. K (cmol/kg)] + 26.36	7	0.69	8
Thickness 3 (cm) *	= [1.922 x tot. N (cmol/kg)] + [61.20 x Db (g/cm <sup>3</sup> )] - 40.28	11	0.73	10

\* Equation 1 used for sites designated as having slightly to non-calcareous PM.

Equation 2 used for sites designated as having moderately to highly calcareous PM.

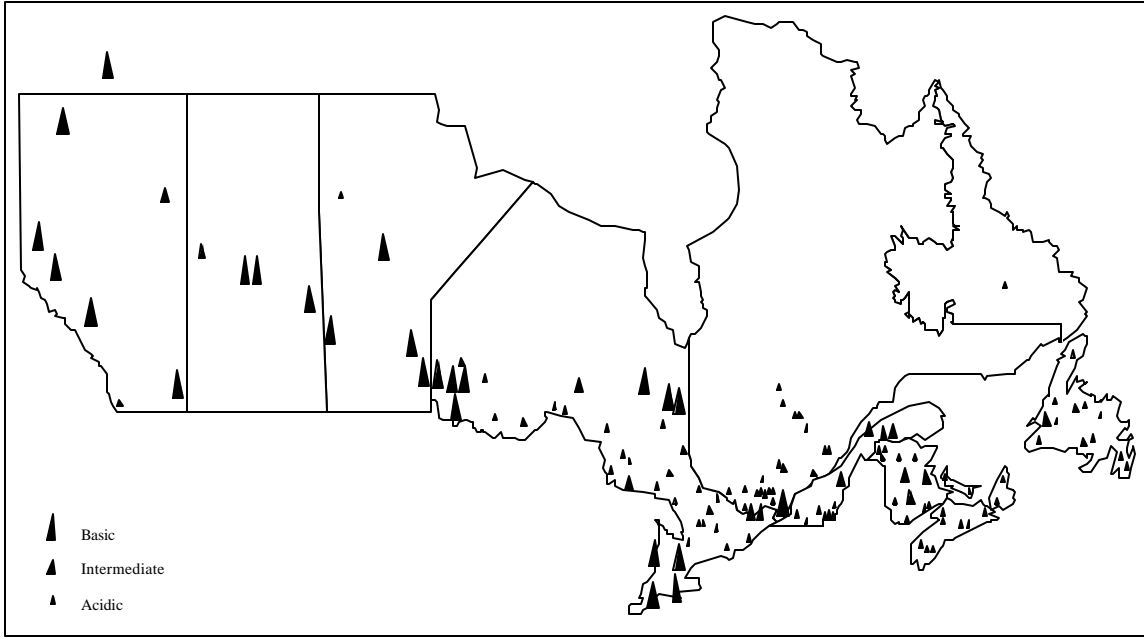
Mean of estimates from equations 1 and 3 used for plots in Newfoundland.

Similar to the A layers, % carbon content in B layers is strongly related to % total N and vice versa. Where N data is not available, exchangeable Al can be used to estimate % carbon levels in the B layers. A positive relationship between available P in the B layer to available P in A layer is to be expected, on account of P leaching from A to B.

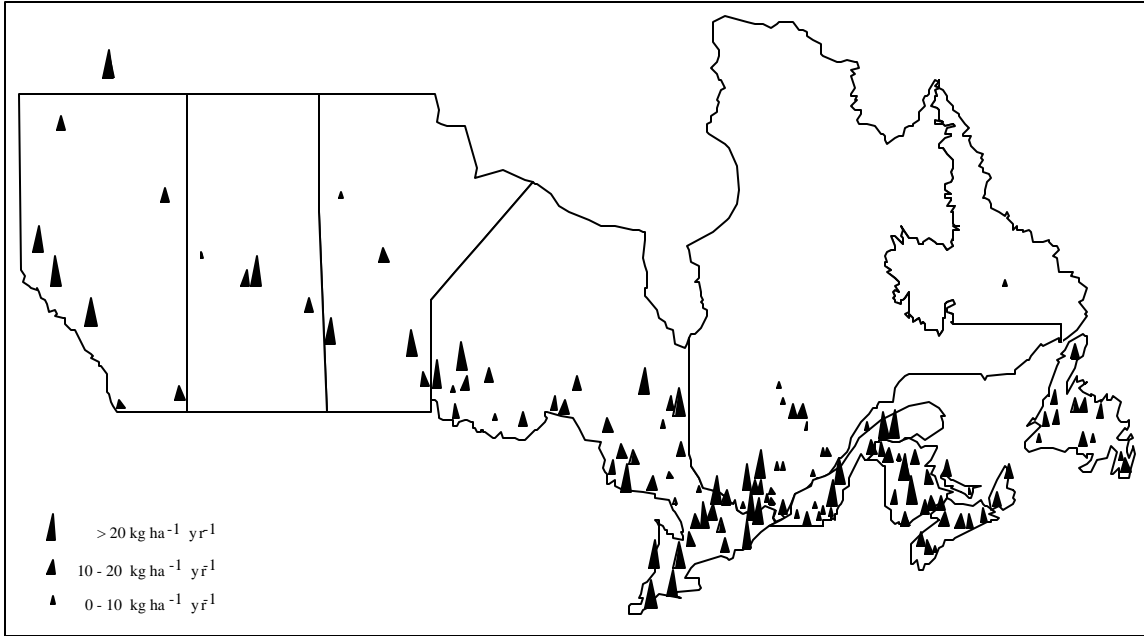
Similar to the A layers, pH in B layers is related to amounts of exchangeable Ca (+) and exchangeable Al (-). However, pH in B layers also appears to be related to exchangeable Mg (-). As usual, soil bulk density in B layers has a strong negative relationship with organic matter content. Depending on type of parent material, the thickness of B layer can be estimated by way of substrate-dependent equations (Table 7.3). Here, the negative relationship between exchangeable K and B layer depth is interesting. This relationship suggests that soil B layers with high K contents tend not to develop strongly. This is either a reflection of low K uptake due to poor rooting, or the reason that soils with high amounts of exchangeable K tend to weather less readily. In particular, high exchangeable K may lead to high levels of K fixation that, in turn, may reduce the rate of clay particle weathering.

#### ARNEWS soil data, by plot

The compiled soil data for 125 ARNEWS plots are shown in Table A.4 (Appendix A). Figures 7.1 and 7.2 illustrate soil substrate classes and magnitude of the estimated weathering rates for ARNEWS plots, respectively.



**Figure 7.1** Soil substrate classes for ARNEWS plots.



**Figure 7.2** Estimated base cation (Ca + Mg + K) weathering rates for ARNEWS plots.

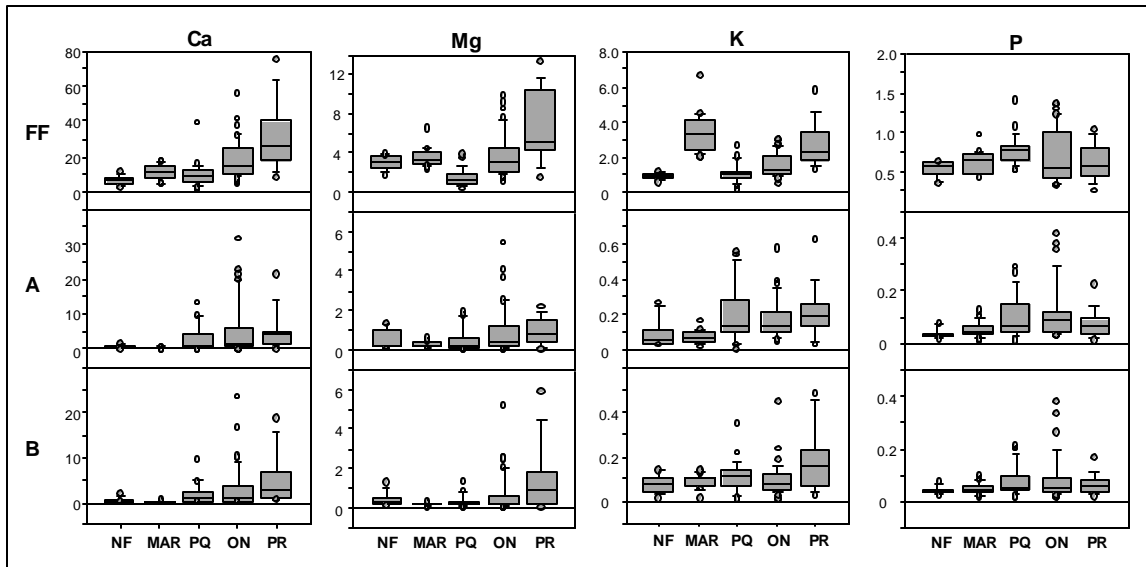
As shown in Figure (7.1), most of plots in Prairie Provinces were located on basic soil substrates. In Ontario, most sites have basic soil substrates except some sites in the northwestern, northeastern, and southwestern parts of province which are generally located on acidic soil substrates. Quebec's sites have mostly acidic and some intermediate soil substrates. Sites with basic soil substrates are located in northern parts of New Brunswick while those with intermediate and acidic soil substrates are in central and the southern parts of the province, respectively. Other sites in the Maritime Provinces and Newfoundland were generally located on acidic glacial tills.

As to be expected and as calculated, soil weathering rates (Figure 7.2) correspond to soil substrate class (acidic, intermediate, basic) and soil texture (clay content). Site with highest weathering values ( $> 20 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ) are mainly located in the Prairies, in Ontario, and in the southern parts of Quebec where parent materials are normally basic and have high clay contents. Sites with the lowest weathering rates ( $< 10 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ) are mainly located in Quebec, central Ontario, and are also scattered in other regions. Intermediate weathering rates ( $10\text{-}20 \text{ kg ha}^{-1} \text{ yr}^{-1}$ ) occur mainly in the Maritimes, in Ontario and also in some plots in Quebec and the Prairies.

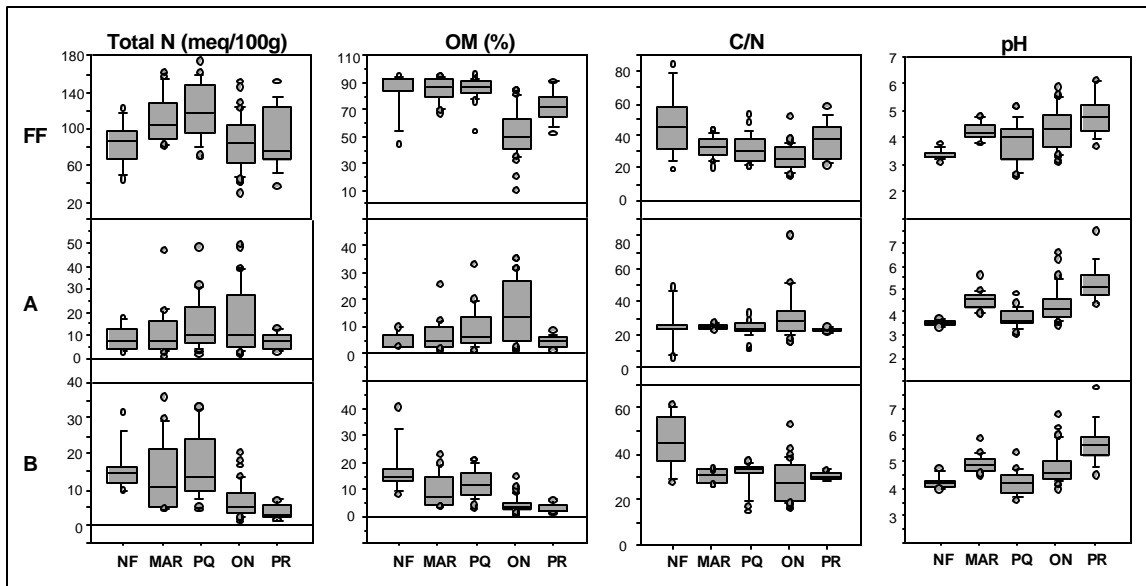
#### Box plots by region

Figures 7.3 – 7.5 illustrate box plots of various soil data for forest floor (FF), A, and B layers by region. From these figures the following results are found:

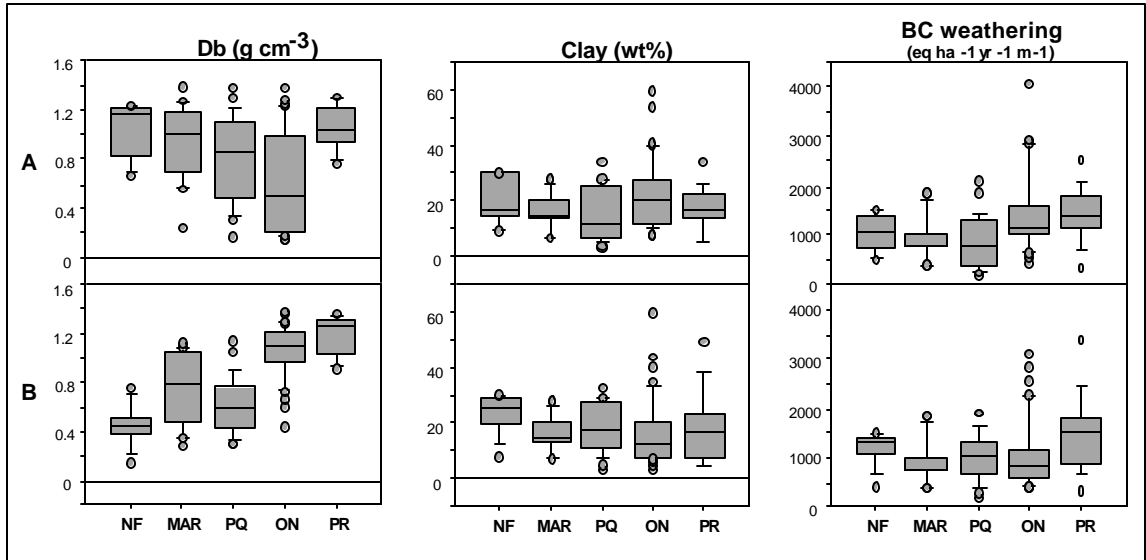
Exchangeable (meq/100g)



**Figure 7.3** Box plots of exchangeable Calcium, Magnesium, Potassium, and Phosphorous by region including Newfoundland (NF), Maritime Provinces (MAR), Quebec (PQ), Ontario (ON), and Prairie Provinces (PR) for ARNEWS plot network.



**Figure 7.4** Box plots of total N, percent of organic matter (OM%), C/N ratio, and pH by region including Newfoundland (NF), Maritime Provinces (MAR), Quebec (PQ), Ontario (ON), and Prairie Provinces (PR) for ARNEWS plot network.



**Figure 7.5** Box plots of bulk density (Db), clay content, and base cation weathering rates by region including Newfoundland (NF), Maritime Provinces (MAR), Quebec (PQ), Ontario (ON), and Prairie Provinces (PR) for ARNEWS plot network.

1. In general, the average and range of exchangeable Ca, Mg, K, and available P decrease from FF to B layer (Figure 7.3).
2. The patterns of Ca, Mg, K, and P concentrations in the FF layers differ from those in the A and B layers. However, Ca, Mg, K, and P concentrations in A and B layers are quite similar (Figure 7.3).
3. Average Ca, Mg, K, and P concentrations vary by region. On average, plots located in Prairies have consistently high values for Ca, Mg in all soil layers, and high values for K in the A and B layers. This is to be expected on account of calcareous soil substrates. Except Mg in FF (lowest values appeared in Quebec), Newfoundland and the Maritimes tend to have the lowest Ca, Mg, and K values, likely due to the prevalence of glacial soil substrates with low base content. Ca, Mg, and K levels in Ontario and Quebec are intermediate between values in the Atlantic region and in the Prairies. Compared to Ca, Mg, K, P concentrations vary less by region (Figure 7.3).
4. In general, total N concentrations decrease from FF to B layer soil. In contrast, N concentrations do not vary by region. However, N values are in the Prairies (Figure 7.4).
5. Organic matter contents generally decrease from FF to B. As with N, organic matter contents do not seem to vary by region (Figure 7.4).
6. Averages C/N ratios are higher in FF than in A and B layers. Least regional C/N variability is found for A layer soils (Figure 7.4).
7. Average pH varies by region as follows (Figure 7.4):

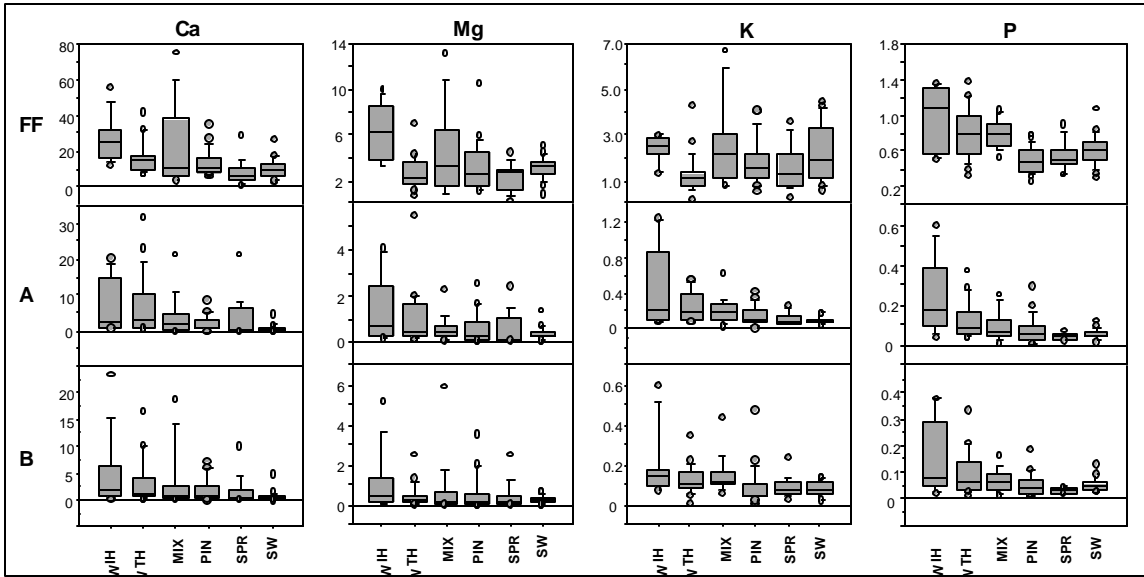
$$\text{PH (NF)} < \text{PH (PQ)} < \text{PH (MAR)} < \text{PH (ON)} < \text{PH (PR)}.$$

8. In A layers, high bulk densities values combine with low bulk densities in Newfoundland, in the Maritimes, and in the Prairies. In B layers, the pattern differs so that average bulk densities are highest in Ontario and in the Prairie Provinces; intermediate values are noted for the Maritimes and Quebec; lowest values are observed for Newfoundland (Figure 7.5).
9. Average clay contents vary from 10 to 20% in A layers and from 10 to 25% in B layers. Ontario and Quebec sites have a wide range of clay contents, due to their soil variability (Figure 7.5).
10. Average weathering rates are estimated to be highest for the Prairie Provinces due to the calcareous nature of the soil substrate. Wide ranges of weathering rates are estimated for Ontario and Quebec, and relatively low weathering rates are estimated for the Maritimes and for Newfoundland (Figure 7.5).

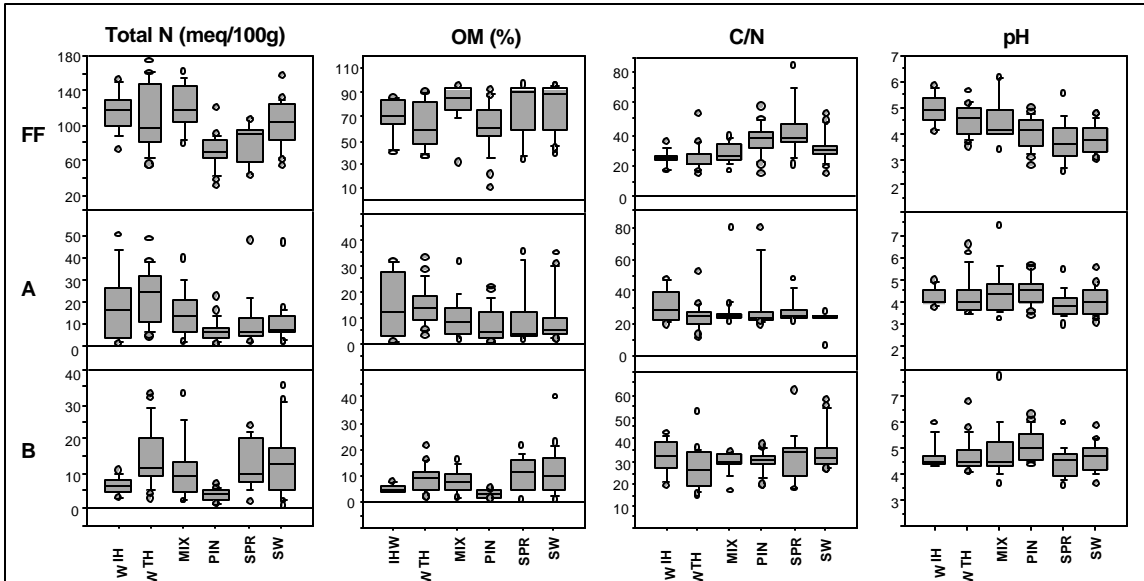
#### Box plots by cover type

Figures 7.6 – 7.8 illustrate box plots of various soil data in forest floor (FF), and in the A and B layers of the ARNEWS plots, by cover type (tree species). From these figures, the following results are found:

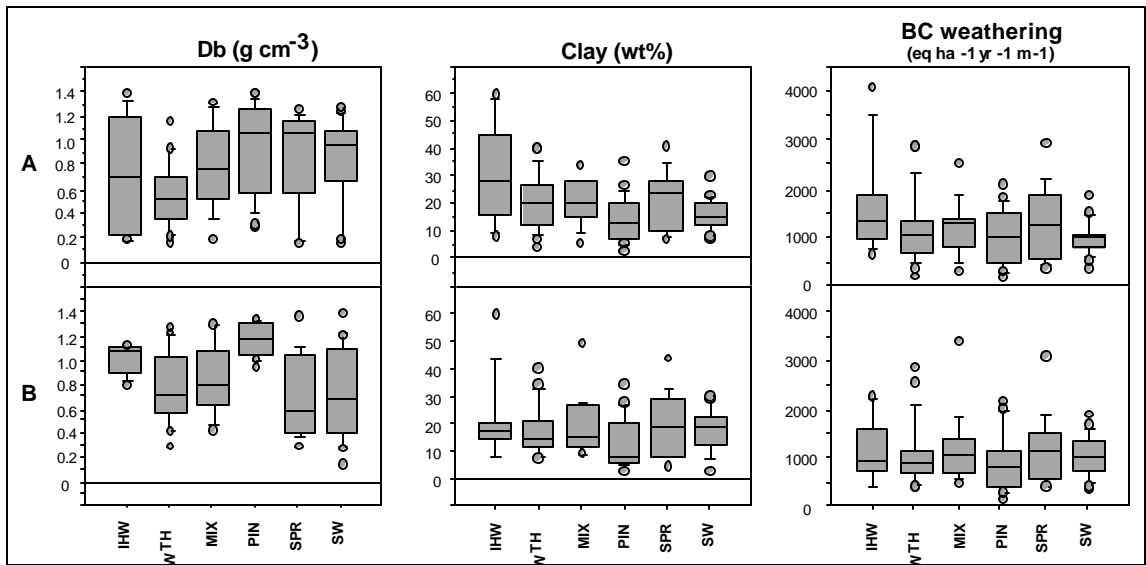
Exchangeable (meq/100g)



**Figure 7.6** Box plots of exchangeable Calcium, Magnesium, Potassium, and Phosphorous by species groups including intolerant hardwood (IHW), tolerant hardwood (THW) mixed wood (MIX), pine (PIN), spruce (SPR), and soft wood (SW), for ARNEWS plot network.



**Figure 7.7** Box plots of total N concentration, percent of organic matter (OM%), C/N ratio, and pH by species groups including intolerant hardwood (IHW), tolerant hardwood (THW), mixed wood (MIX), pine (PIN), spruce (SPR), and soft wood (SW), for ARNEWS plot network.



**Figure 7.8** Box plots of bulk density (Db), clay content, and base cation weathering rates by species groups including intolerant hardwood (IHW), tolerant hardwood (THW), mixed wood (MIX), pine (PIN), spruce (SPR), and soft wood (SW) for ARNEWS plot network.

1. In general, averages and ranges of exchangeable Ca, Mg, K, and available P concentrations decrease from FF to B layer (Figure 7.6).
2. The patterns of Ca, Mg, K, and P concentrations in FF differ with the patterns found for A and B layers. However, the patterns of A and B layers are close to each other (Figure 7.6).
3. There is a correlation between site nutrient availability and tree species composition. For example, intolerant hardwoods are growing on sites with high levels of exchangeable Ca, Mg, K, and available P. Available nutrient concentrations are higher on sites with tolerant hardwoods and mixed woods than on sites with softwood species. An exception is occurs with K in FF, i.e., K (tolerant hardwoods) < K (softwoods). This could be due to the fact that K is a mobile nutrient that can be washed out relatively fast from the forest floor. Among softwood species, soil nutrient concentrations appear in order of spruce < pine < balsam fir (Figure 7.6).
4. In general, total N decreases from FF to B layers. There is no apparent pattern for total N in FF, A and B layers, by species types. Lowest total N values occur for pines (Figure 7.7).
5. Organic matter contents generally decrease from upper to lower layers. As with total N, there is no apparent pattern for total N in FF, A and B layers, by species type. (Figure 7.7).
6. Average of C/N ratios are higher in FF layers than that in A and B layers. Least variability for C/N ratios is found in the A layers (Figure 7.7).

7. Average pH values are less variable for A and B layers than for FF layers, by cover type. For pines, pH values tend to be higher than what is seen for the other species types (Figure 7.7).
8. There is no apparent pattern for soil bulk density by species type (Figure 7.8).
9. Figure 7.8 shows that pines usually grow on soils with low clay content. Intolerant hardwood and spruce species, in contrast, tend to grow on soils with elevated clay content.

## CONCLUSION

Through graphic analysis, variations of physical and chemical soil properties have been examined for the ARNEWS plots from Newfoundland to Alberta, by soil layer, by species type, and by region. Not surprisingly, soil nutrient concentrations (Ca, Mg, K, P, and N) tend to decrease from forest floor to the subsoil. Soil substrates tend to be more acidic in the east (low pH), and less acidic in the west (high pH). Similarly, base cations (Ca, Mg, and K) and P availabilities generally increase from east to west. Hardwoods tend to grow on sites with elevated Ca, Mg, K, P concentrations, i.e., on sites where soil-weathering rates are elevated as well. Organic matter and total N contents are lowest in the Prairies.

Inherently, data that are been compiled from various locations, and by various agencies, tend to have missing data. Fortunately, many of the soil variables are correlated to one another, to some extent. This being so enables one to estimate missing values.

Obtaining missing values is important for completing the existing ARNEWS database as much as possible, to enable further analysis, to enable the evaluation of critical soil acidification loads and related exceedances for each site.

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## CHAPTER 8

### SUMMARY FOR ARNEWS VEGETATION DATA

#### INTRODUCTION

This chapter summarizes ARNEWS tree nutrient data that are required for assessments of soil acidification and long-term forest biomass productivity. The data refer to concentrations of Ca, Mg, K, P, and N in aboveground tree components, i.e., in foliage, stem wood (core), stem bark, and branches. The data belong to various tree species including (i) tolerant hardwoods (comprising red maple [*Acer rubrum* L.], sugar maple [*Acer saccharum* Marsh.], yellow birch [*Betula alleghaniensis* Britton], beech [*Fagus grandifolia* Michx.], red oak [*Quercus rubra* L.], black oak [*Quercus velotina* Lam.], and white oak [*Quercus alba* L.], (ii) intolerant hardwoods (comprising white birch [*Betula papyrifera* Marsh.], largetooth aspen [*Populus grandidentata* Michx.], trembling aspen [*Populus tremuloides* Michx.]), (iii) spruce (comprising red spruce [*Picea rubens* Sarg.], white spruce [*Picea glauca* (Moench) Voss]), (iv) pines (comprising jack pine [*Pinus banksiana* Lamb.], red pine [*Pinus resinosa* Ait.], white pine [*Pinus strobus* L.], lodgepole pine [*Pinus contorta* Lamb.]), (v) fir (balsam fir [*Abies balsamea* (L.) Mill.]).

A preliminary analysis of these data revealed that tree nutrient concentrations vary by region (region refers to Newfoundland, the Maritimes, Quebec, Ontario, and the Prairies). Therefore, it was decided to portray the elemental nutrient concentrations of stem wood (core), bark, and foliage by species and by region.

Since it is generally hypothesized that tree nutrient concentrations depend on soil nutrient availability, the tree nutrient database was combined with the corresponding soil nutrient availability database, in order to examine the underlying soil-tree nutrient relationships. In this work, a hierarchical approach was followed: since tree foliage concentrations refer to the highest nutrient concentrations in the tree, one would expect that soil availability would affect these nutrient concentrations the most. Hence, tree foliage data were regressed against nutrient availability data for N, P, Ca, Mg, and K. This was done to establish functionally important relationships between tree nutrient concentrations and soil nutrient availability. Subsequently, due to within tree allocation of nutrients, it was then decided to establish predictive equations for tree nutrient concentrations in branches, bark, and stem wood based on foliar nutrient concentrations. This was done to provide a means that tree nutrient uptake can essentially be related to nutrient availability as this varies on a wide range of forest soils and forest conditions.

The objectives of this Chapter, therefore, are:

1. to examine to what extent the summarized nutrient concentrations in tree components for each ARNEWS plot from Newfoundland to Alberta vary among components of tree, by species, and by region;
2. to address the procedures used to compile these data;
3. to document relationships used for replacing missing data;
4. to examine the relationships between foliar concentrations to soil data, and between foliage data and the other tree components.

## METHODS

### Nutrient concentration in core and bark

Core and bark samples were taken from 125 dominant and co-dominant trees within ARNEWS plots ranging from Newfoundland to Alberta in summer 1995. Four wood and bark samples of the dominant tree species were taken from each ARNEWS plot. Details of procedures used for laboratory analyses are available in Case *et al.* 1996. All elemental concentrations are expressed in terms of weight of element per weight of oven-dry substance (wood or bark), in ppm. These data are delineated by plot number, species, and province (ARNEWS CD-ROM 2000, Veg-data folder, Nut-wdbk file).

### Nutrient concentration in foliage

Data regarding elemental concentrations in foliage were obtained from the Canadian Forest Service (CFS). As stated by D'Eon *et al.* (1994), foliage sampling and analysis for ARNEWS plots were normally based on sampling 10 trees of the principal species or 10 trees of each of the two most abundant species on each ARNEWS plot, by collecting current leaves from the middle to upper parts of the live crowns of dominant or co-dominant trees. This procedure was usually repeated at 5-year intervals. Oven-dried samples were analyzed for total macro and micro nutrient concentrations.

The foliage data for total N, P, Ca, Mg, K, Na, S, Fe, Mn, Zn, Cu, Al, B, Mo, and Pb province (ARNEWS CD-ROM 2000, Veg-data folder, Nut-fol file) were delineated by plot number, tree number, tree species, and year sampled (1985, 1990, and 1993). In this data set, there were 2 to 10 data entries per dominant species for the sampling year 1985. Except for plots in Prairies, foliage data were repeated in 1990 with 10 additional

data entries per species per plot. Plots that were established later than 1990 (2 plots in Newfoundland, and 11 plots in Ontario) had only one set of foliage data per species in 1993. For the purpose of this study, concentrations (ppm) of N, P, Ca, Mg, and K were aggregated (averaged) by species and by plot. The resulting data set includes 158 rows (ARNEWS CD-ROM 2000, Veg-data folder, Nut-fol file).

#### Compiling wood, bark, and foliage data

The compiled foliage database was merged with the core/bark database, by plot number and by species. In this way, it was found that there were 61 core/bark data that did not find a foliage match, and that there were 28 foliage samples that did not find a wood/bark match. The lack of matching was due to differences in sampling efforts. Not all trees that were sampled for foliage were also sampled for bark and wood. The opposite was true as well, because the wood and bark sampling effort in 1995 involved sampling from new ARNEWS plots that were not sampled previously. To substitute the missing information, the available wood, bark, and foliage data were applied to establish predictive regression equations between foliage nutrient concentration and wood/bark nutrient concentrations, and visa versa.

#### Nutrient concentration in branches

As required for model input, concentrations of N, P, Ca, Mg, and K in branches were also estimated through regression equations (Table 8.1). These equations were established on the basis of independent data in Maliondo *et al.* (1990).

### Wood, bark, and foliage data analysis

The combined wood, bark, and foliage data for Ca, Mg, K, P, and N was summarized by way of box plots to display their variations by species, by tree density (percent basal area), and by regions (see below). To determine variations by species, 6 tree species (balsam fir, black spruce, jack pine, sugar maple, trembling aspen, and white birch) were chosen to represent various species types while having a sufficient sample size. To identify if elemental concentrations change by tree density, two species (white spruce and sugar maple) were examined. For this purpose, element concentrations in each species were grouped by tree density classes (BA%). Finally, the nutrient concentrations in tree components were grouped by region (Newfoundland, New Brunswick, Quebec, Ontario, and Prairie Provinces) to examine the variations by region. For this analysis, black spruce and white birch were chosen because these species were sampled fairly consistently across all regions.

### Relationships between foliage elemental data and soil data, by species type

The nutrient data table was merged with the soil data table to establish which regression equations best describe the relationships between soil nutrient availability and tree nutrient content, especially in reference to foliar nutrient concentrations. The resulting database includes 95 ARNEWS plots to represent the already identified species type groupings, namely fir, pine, spruce, intolerant hardwoods, and tolerant hardwoods.

## RESULTS AND DISCUSSION

### Regression equations for estimating missing data

Regression equations established to estimate missing data for core, bark, foliage, and branch are compiled in Table (8.1).

**Table 8.1** Equations used to estimate missing tree nutrient data for ARNEWS plots (all dependent and independent data are in ppm).

Element	Equation	n	r <sup>2</sup>	Data Filled
<b>Core</b>				
N	= 0.036 x Fol N	90	0.57	61
P	= (0.002 x Fol N)	97	0.53	61
Ca	= (0.441 x Fol P) + (0.039 x Fol Ca)	95	0.29	61
Mg	= (0.045 x Fol P) + (0.004 x Fol N)	96	0.12	61
K	= 0.015 x Fol N	92	0.42	61
<b>Bark</b>				
N	= 0.202 x Fol N	97	0.43	61
P	= 0.219 x Fol P	92	0.18	61
Ca	= 1.674 x Fol Ca	97	0.49	61
Mg	= 0.032 x Fol N	93	0.19	61
K	= (0.557 x Fol P) + (0.083 x Fol Ca)	88	0.42	61
<b>Foliage</b>				
N	= 26.621 x Core N	91	0.77	28
P	= (0.59 x Fol N) + (1.53 x Bark P)	95	0.42	28
Ca	= 0.499 x Bark Ca	97	0.49	28
Mg	= (13.71 x Core P) + (6.346 x Core Mg)	90	0.66	28
K	= 11.276 x Core N	96	0.38	28
<b>Branch</b>				
N	= (0.149 x Fol N) + (1.276 x Bark N)	24	0.97	186
P	= (0.013 x Fol N) + (0.072 x Bark N)	24	0.98	186
Ca	= (0.641 x Fol K) + (0.383 x Bark Ca) + (1.445 x Bark N) - (0.479 x Wood N)	24	0.98	186
Mg	= (0.273 x Fol Mg) + (1.27 x Bark Mg)	24	0.96	186
K	= (0.178 x Fol K) + (0.715 x Bark K) + (0.362 x Wood K)	24	0.98	186

As shown, concentrations (ppm) of N, P, and K in core are all correlated to foliar N concentrations. Core Ca concentrations, in turn, can be estimated from foliar P and Ca; core Mg can be estimated from foliar P and N. This suggests that foliar N concentrations

indicate to what extent nutrients are allocated to stem wood. In contrast, concentrations of N, P, Ca, and Mg in bark are estimated from foliar N, P, Ca, and N, respectively, thereby suggesting a close elemental linkage between foliar concentrations and bark concentrations. Bark K is exceptional: it is positively related to foliar P and Ca.

For the purpose of determining missing data, it was found that foliar concentrations of N and K can be estimated from core N, while foliar Ca can be estimated from bark Ca. Foliar P can be estimated from foliar N, bark N and bark P. Foliar Mg is correlated to Mg and P in wood.

Both N and P concentrations in branches are correlated to N concentrations in foliage and bark. Branch Ca is positively correlated to foliar K, bark Ca and N, and negatively to wood N. Mg concentrations in branch can be estimated from Mg concentrations in foliage and bark. K concentration in branch is correlated to K concentration in foliage, bark, and wood. The compile data set for nutrient concentrations in foliage, bark, wood, and branches contains 186 rows, see Table A.5 (Appendix A).

### Box plots by species

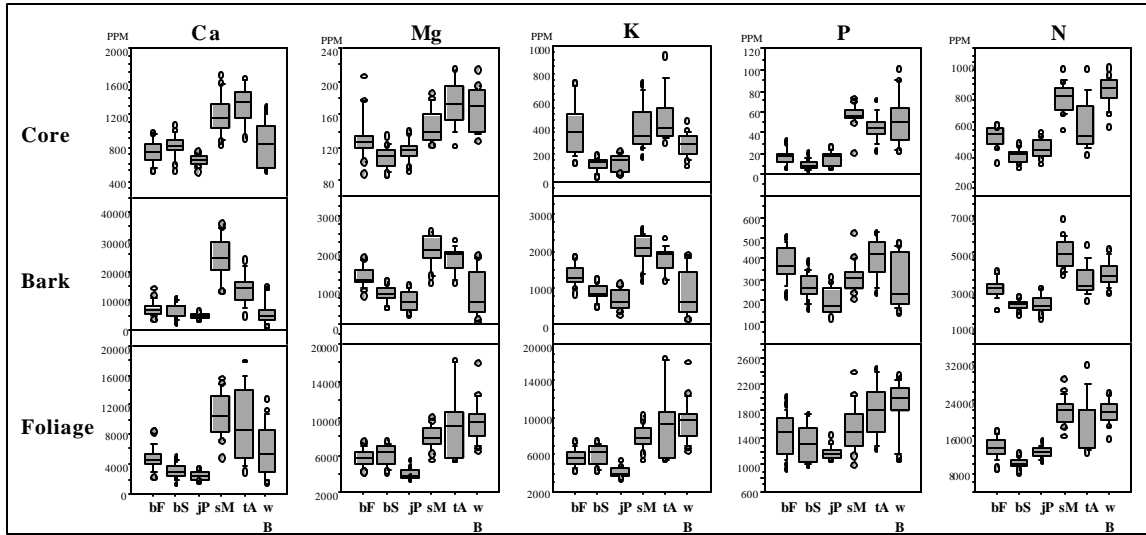
Figure 8.1 illustrates box plots of Ca, Mg, K, P, and N concentrations in core, bark, and foliage by tree species. From this figure, the following results are found:

1. The concentrations of Mg, K, P, N in tree components follow the order

foliage > bark > core

while Ca follows the order

bark > foliage > core.



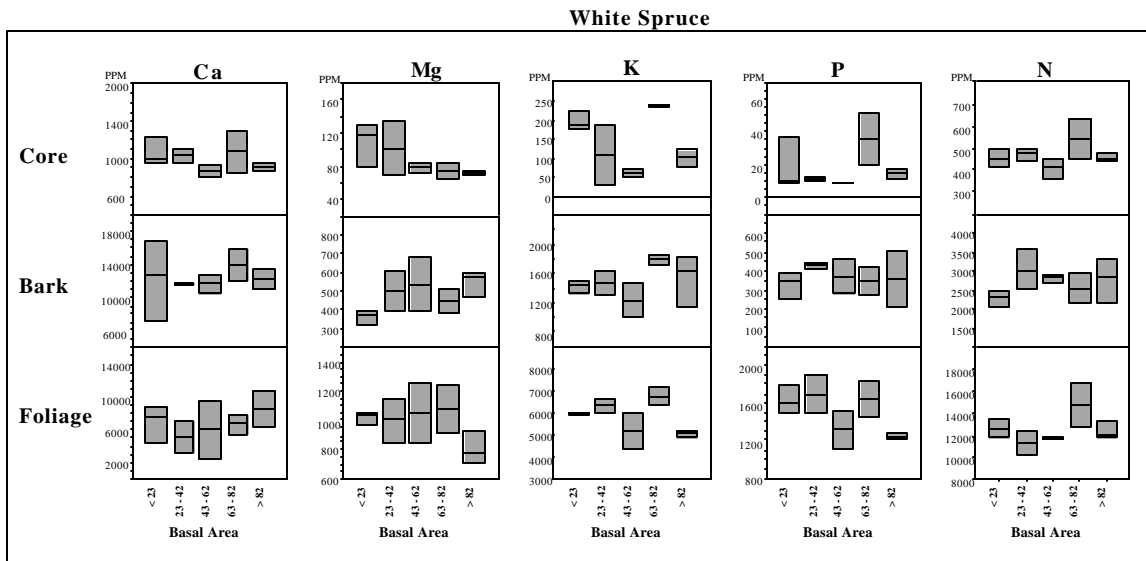
**Figure 8.1** Box plots of nutrient concentration in tree components (core, bark, and foliage) by tree species including Balsam Fir (bF), Black Spruce (bS), Jack Pine (jP), Sugar Maple (sM), Trembling Aspen (tA), and White Birch (wB), in ARNEWS plot network.

2. In general, elemental concentrations in foliage, bark, and core are higher in deciduous species than in coniferous species.
3. Among deciduous trees, sugar maple has particularly high Ca, Mg, and K concentrations in the bark, while white birch has low Ca, Mg, and K concentrations in the bark, i.e. almost in the same range of conifers.
4. Among conifers, jack pine has particularly low Ca, Mg, and K concentrations in bark and foliage. Balsam fir tends to have high P concentrations in bark and foliage.

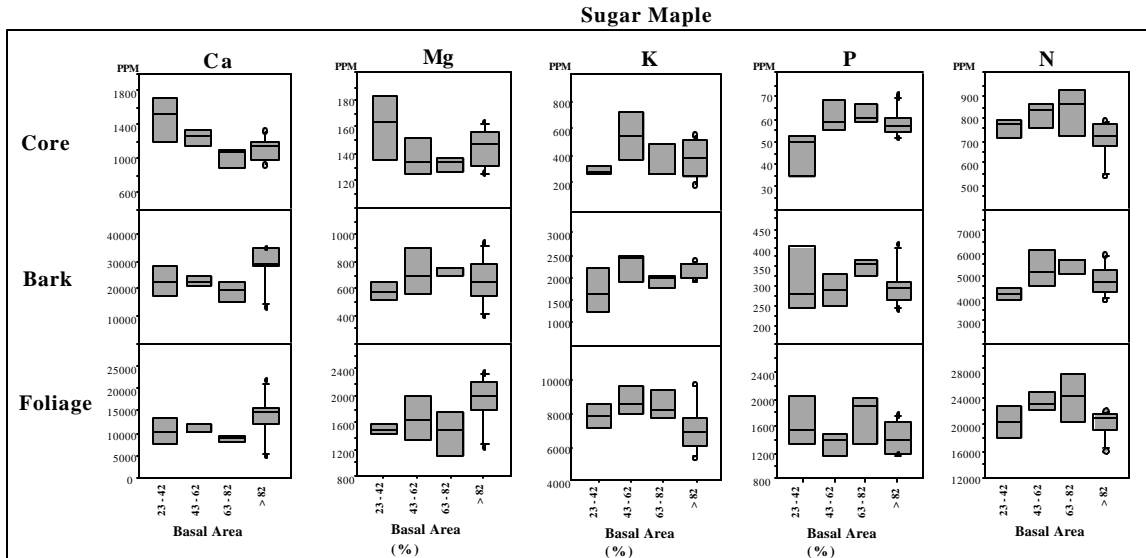
Note: each box plot is composed of five horizontal lines that display the 10<sup>th</sup>, 25<sup>th</sup>, 50<sup>th</sup> (median), 75<sup>th</sup>, 90<sup>th</sup> percentiles of a variable. Values above the 90<sup>th</sup> percentile and below the 10<sup>th</sup> percentile are plotted as points (outliers).

#### Box plots by tree density

The box plots of Ca, Mg, K, P, and N concentrations in core, bark, and foliage of white spruce and sugar maple by tree density are shown by Figures 8.2 and 8.3, respectively. These box plots indicate that there are no trends in concentrations of these tree elements by increasing density. This establishes the trend that elemental concentrations in trees are not measurably affected by tree density, in general. This, however, does not imply that elemental concentrations are not affected by tree age, because elemental concentrations generally decrease with increasing tree age. This change, however, is steep in the first 20 years or so, but levels out afterwards.



**Figure 8.2** Box plots of nutrient concentration in tree components (core, bark, and foliage) of White Spruce by tree density (percent of basal area), in ARNEWS plot network.



**Figure 8.3** Box plots of nutrient concentration in tree components (core, bark, and foliage) of Sugar Maple by tree density (percent of basal area), in ARNEWS plot network.

### Box plots by region

Figures 8.4 and 8.5 illustrate box plots of Ca, Mg, K, P, and N concentrations in core, bark, and foliage of black spruce and white birch by region. From these figures, the following results are found:

#### *1. Black Spruce*

Trees of plots located in the Prairies have relatively high concentrations of K and N in core, bark, and foliage. Foliar P is particularly high in the Quebec plots. Foliar Ca of black spruce is relatively low in New Brunswick. This trend is likely related to type of soil substrate per plot.

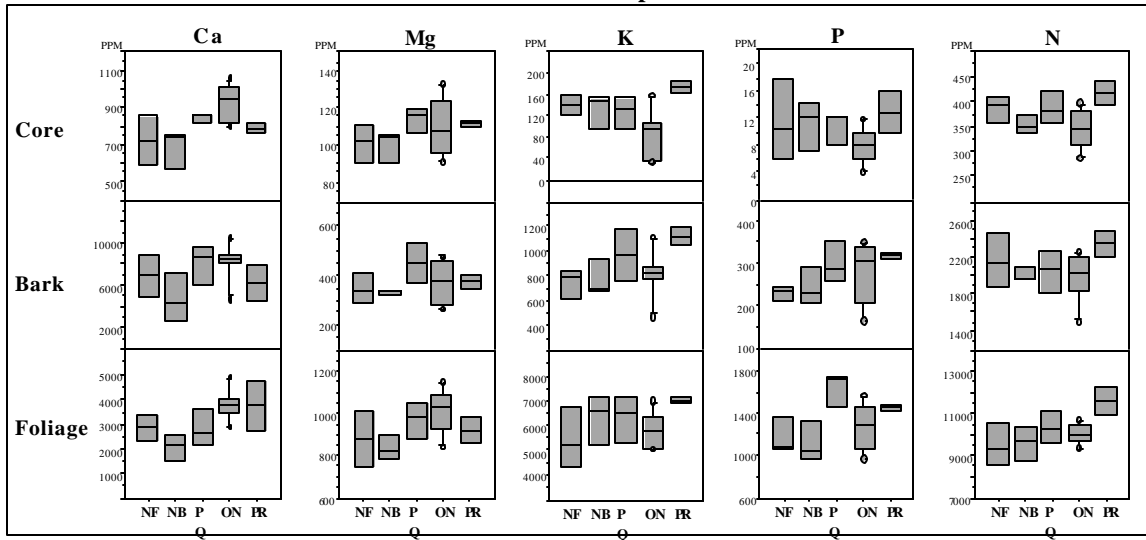
#### *2. White Birch*

In general, the concentrations of Ca, Mg, and K in different tree components of white birch increase from Newfoundland to Ontario (there were no samples from the Prairies). This pattern seems to be due to changes of soil substrate from less base cations in the east to more base cations in the west. P and N follow the same trend, in general. On average, N concentrations are relatively low for white birch in Ontario, but the ranges are wide there as well.

### Relationships between foliage elemental data and soil data, by cover type

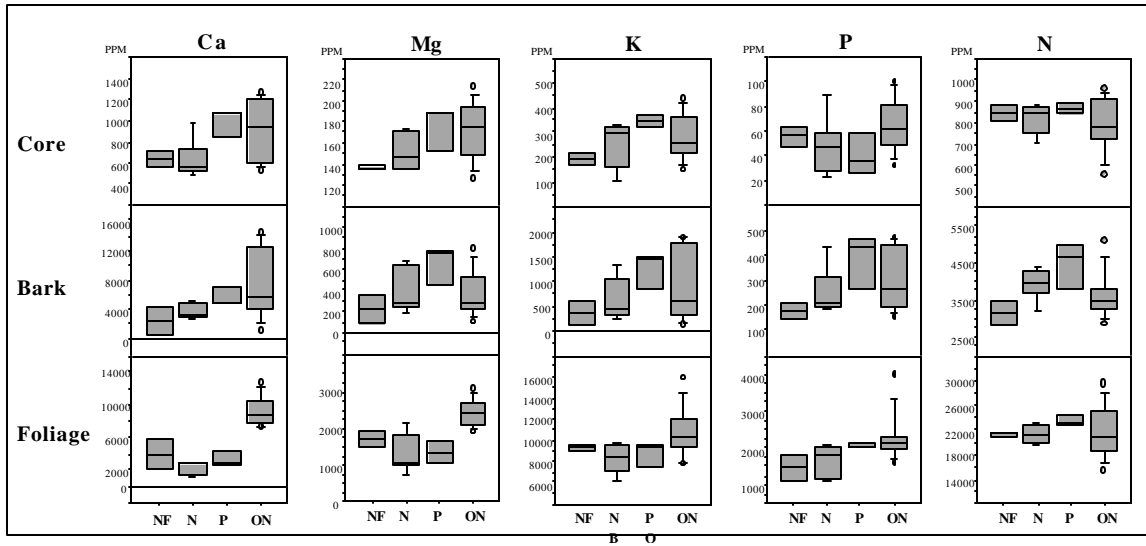
Regression equations for five species types relating foliage elemental concentrations to soil elemental concentrations and other soil variables are shown in Table 8.2. From these regression analyses, the following results were found:

### Black Spruce



**Figure 8.4** Box plots of nutrient concentration in tree components (core, bark, and foliage) of Black Spruce by provinces (regions) including Newfoundland (NF), New Brunswick (NB), Quebec (PQ), Ontario (ON), and Prairie provinces, in ARNEWS plot network.

### White Birch



**Figure 8.5** Box plots of nutrient concentration in tree components (core, bark, and foliage) of White Birch by provinces including Newfoundland (NF), New Brunswick (NB), Quebec (PQ), and Ontario (ON), in ARNEWS plot network.

**Table 8.2** Regression equations for five cover types relating foliage elemental concentrations (in ppm) to soil elemental concentrations (in ppm) and soil properties.

SpeciesType	Ca (ppm)	Mg (ppm)	K (ppm)	P (ppm)	N (ppm)
<b>Balsam Fir</b>					
n	23	24	23	24	23
Intercept	4420±507	668 ± 87	6690±385	695 ±216	7300 ± 1520
Coefficient	121±31 (FF-Ca) -497 ± 190 (FF-K)	0.063 ± .017 (Ca- <i>fol</i> )	75 ± 20 (FF-Ca) -668 ± 145 (FF-Mg)	1128±322 (FF-P)	57 ± 13 (FF-N)
R <sup>2</sup>	0.48	0.38	0.55	0.60	0.55
R <sup>2</sup> (adj)	0.43	0.36	0.51	0.36	0.51
<b>Pine</b>					
n	23	24	22	24	24
Intercept	4069±589	-156 ± 150	3399 ± 228	-195 ± 119	8846 ± 902
Coefficient	-46±16 (FF-C/N)	828 ± .11 (P- <i>fol</i> ) 35 ± 9 (FF-Mg)	201 ± 47 (B-N)	0.19 ± .02 (K- <i>fol</i> ) 0.048 ± .008 (N- <i>fol</i> )	1.46 ± .36 (Ca- <i>fol</i> )
R <sup>2</sup>	0.29	0.73	0.48	0.88	0.43
R <sup>2</sup> (adj)	0.26	0.71	0.46	0.87	0.40
<b>Spruce</b>					
n	41	41	42	42	42
Intercept	-3922±1737	1094 ± 94	3583 ± 807	16 ± 244	14237 ± 1444
Coefficient	279±56 (A-Ca) 66588 ± 15786 (FF- <i>Db</i> )	0.03 ± .007 (Ca- <i>fol</i> ) -69 ± 23 (A-pH)	-0.16 ± .05 (Ca- <i>fol</i> ) 2.4 ± .6 (P- <i>fol</i> )	0.096 ± .027 (K- <i>fol</i> ) 0.82 ± .21 (Mg- <i>fol</i> )	0.26 ± .26 (Ca- <i>fol</i> ) 86 ± 20 (FF-Ca) -1188 ± 341 (B-pH)
R <sup>2</sup>	0.57	0.35	0.37	0.45	0.60
R <sup>2</sup> (adj)	0.55	0.32	0.34	0.42	0.57

Table 8.2 continues

SpeciesType	Ca (ppm)	Mg (ppm)	K (ppm)	P (ppm)	N (ppm)
<b>l. Hardwood</b>					
	26	26	26	26	24
except	-7932 ± 3742	933 ± 193	1776 ± 2894	727 ± 338	
efficient	5417 ± 694 (FF-pH) -361 ± 86 (FF-C/N)	0.12 ± .02 (Ca -fol)	0.37 ± .13 (N-fol)	0.13 ± .03 (K-fol)	1261 ± 331 (FF-depth) 470 ± 80 (FF-C/N) 198 ± 57 (A-OM%)
	0.76	0.60	0.25	0.40	0.98
(adj)	0.74	0.59	0.22	0.38	0.98
<b>l. Hardwood</b>					
	43	43	43	43	43
except	6019 ± 977	601 ± 382	-923 ± 1207	-62 ± 257	33594 ± 2849
efficient	213 ± 54 (FF-Ca) -1619 ± 400 (FF-K) 321 ± 91 (A-depth)	0.05 ± .02 (N-fol)	0.38 ± .05 (N-fol) 24.3 ± 8.6 (A-C/N)	0.066 ± .011 (N-fol) 6 ± 1.8 (A-C/N)	-466 ± 97 (FF-C/N) 33253 ± 6796 (A-K) -456 ± 116 (A-OM%)
	0.64	0.20	0.61	0.53	0.45
(adj)	0.61	0.18	0.59	0.51	0.41
<b>l species</b>					
	152	152	157	154	152
except	7636 ± 2017	652 ± 351	8720 ± 1338	804 ± 85	32800 ± 6016
efficient	2301 ± 471 (FF-pH) 440 ± 56 (A-Ca) -2670 ± 471 (B-pH)	666 ± 185 (FF-P) 97 ± 25 (B-Ca) -232 ± 63 (B-Mg) 418 ± 97 (FF-pH) -338 ± 92 (A-pH) -16 ± 5 (B-N)	2567 ± 322 (FF-pH) -2623 ± 354 (B-pH)	976 ± 122 (FF-P) 1794 ± 381 (A-P) -10.1 ± 2.5 (A-N)	109 ± 15 (FF-N) -146 ± 25 (FF-OM%) 19333 ± 4386 (A-P) -681 ± 191 (B-OM%) -15608 ± 4081 (B-Db)
	0.49	0.40	0.32	0.42	0.47
(adj)	0.48	0.37	0.32	0.41	0.45

### 1. *Balsam fir*

Ca, K, P, and N concentrations in foliage were highly correlated with nutrient concentrations in forest floor. In particular, Ca in foliage is related to Ca (positive) and K (negative) in forest floor. Foliage K is correlated to forest floor Ca (positive) and Mg (negative); P and N concentrations in foliage are significantly correlated to P and N concentration in the forest floor, respectively. An exception is that foliage Mg is not significantly correlated to soil available Mg. However, foliar Mg can be estimated from foliar Ca. High correlation between foliar nutrients to forest floor nutrients for balsam fir are likely due to (i) similarity of elemental composition between the foliage and the forest floor due to leaf litter fall, (ii) moderately low decomposition rates for fir litter, and (iii) the fact that balsam fir is shallow rooted with more roots growing in the forest floor than in the mineral soil.

### 2. *Pine*

Foliage Ca is negatively correlated to the C/N ratio in the forest floor. Here, increased C content likely reduces the overall availability of Ca for plant uptake. In general, Ca is strongly held by plant organic matter. Examining tree rings with increasing age has found that Ca levels in stem wood increases with increasing stem wood age, while concentrations of other elements such as K and P decrease rapidly with increasing stem wood age (Arp and Manasc, 1988). Foliage K is positively (+) related to N concentrations in the B layer. Foliar Mg is positively correlated to forest floor Mg and foliar P. Foliar N and P concentrations are not correlated to the available soil information. Foliar N, instead, can be estimated from foliar Ca (+), and foliar P can be estimated from foliar K

(+) and Mg (+). Overall, the ranges of foliar nutrient concentrations are narrow, and so are the ranges for the various nutrient concentrations in the soil. This makes it difficult to establish strong regressions between nutrient levels in the tree and in the soil in general. Nevertheless, in relation to the other species, all nutrient concentrations within the pines can be predicted fairly well based on the hierarchical approach taken here.

### 3. *Spruce*

Except for foliage Ca, no significant correlation between foliage elemental concentrations and soil variables could be found. For Ca, foliar Ca is correlated to Ca in A layer (+), and to bulk density of forest floor (+). Here, access to available Ca in the mineral soil would appear to be important for enhancing Ca uptake by spruce. Also important is the Ca relationship to the bulk density of the forest floor. Areas with deep forest floor layers tend to be fluffy and acidic, with low rates of decomposition (e.g. black spruce – kalmia ecosystems). Hence, increasing forest floor bulk density would indicate faster rate of organic matter decomposition, and hence a faster rate of Ca uptake.

Foliar Mg is related to foliar Ca (+) and also to pH (–) of the A layer. This means that increased pH levels of the A layer somewhat restricted Mg uptake. Here, competitive uptake rates between Mg and Ca could play a role: essentially, soils with higher pH have higher Ca levels. Higher Ca levels may decrease Mg uptake.

K and P in foliage can be estimated from the other foliage elements. Foliar N can be estimated from foliar Ca (+) and from Ca in the forest floor (+), and from the pH of the B layer (–). High pH values for the B layer are usually associated with wet, or poorly

drained soils. This, in turn, would confine N uptake to the forest floor and A soil layers, in general.

#### *4. Intolerant hardwoods*

Foliar Ca is correlated to pH (+) and C/N ratio (-) of the forest floor. The latter relationship has already been noted for the pines (see above). The positive relation to pH is likely due to the fact that higher pH forest floors have higher levels of available Ca, or facilitate the uptake of the same. Foliar Mg, K, and P concentrations are not significantly correlated with soil variables, but are correlated with foliar Ca, N, and K concentrations, respectively. Foliar N is correlated to thickness (+) and C/N ratio (+) of the forest floor, and to the organic matter content (+) of the A layer. Overall, this implies that N uptake of intolerant hardwood increases with soils that are enriched with organic matter. The positive effect of the C/N ratio of the forest floor on N uptake likely refers to the fact that intolerant hardwoods tend to grow on soils that are still nutrient rich, but not as rich as soils on which tolerant hardwoods are found. Overall, intolerant hardwood roots grow mainly in the forest floor. Hence it is reasonable to assume that foliar nutrient concentrations of intolerant hardwoods are mainly related to forest floor properties.

#### *5. Tolerant hardwood*

Foliar Ca concentration are related to concentrations of Ca (+) and K (-) in the forest floor and to the thickness of the A layer (+). In general, K concentrations in the forest floor would affect Ca uptake negatively when these concentrations are high. N is correlated to the C/N ratio (-) of the forest floor (suggesting fast rates of decomposition

of the forest floor), and to K and OM% in the A layer (suggesting better Ca uptake on richer sites, and that high levels of K in mineral soils would be less interfering with Ca uptake than high levels of K in the forest floor. Foliar Mg can be estimated from foliar N. Foliar N (+) and C/N ratio (+) in the A layer can be used to estimate foliar K and P.

As shown in the above discussion, nutrient uptake processes vary somewhat with species type, and also with the type of variables that are actually available to use as foliar concentration predictors. Overall, forest floor variables appear to score most highly in many of the relationships found, followed by A and then B horizon characteristics. Forest floor variables are especially important foliar concentration predictors for balsam fir and intolerant hardwoods, because of shallow rooting. With spruce, the relationships are a bit more mineral soil oriented because the spruce type includes black spruce (shallow rooted), and red and white spruce (which have a somewhat deeper rooting habit). The same is true for pines and tolerant hardwoods.

## CONCLUSIONS

Data regarding concentrations (ppm) of Ca, Mg, K, P, and N in foliage, bark, wood, and branch were compiled for 186 dominant and/or co-dominant trees in 125 ARNEWS plots for assessing the extent of soil acidification and long-term forest biomass production.

Through data aggregation, foliar concentrations were obtained for 158 tree-type plot combinations. Elemental concentrations in wood and bark were measured for 125

tree-plot combinations. Missing data (28 for foliage and 61 for wood and bark) were estimated by way of regression. These regression equations indicated that there is significant correlation between foliage and wood/bark nutrient concentrations.

Box plots showed that elemental concentrations vary by tree component, by species, and by region. In general, Mg, K, P, and N decrease from foliage to bark to core. For Ca, the sequence is bark > foliage > core.

Nutrient concentrations in foliage, bark, and core are generally higher in hardwoods than in conifers. Furthermore, elemental concentrations differ among deciduous species and also among coniferous species. Elemental concentrations do not seem to vary much by tree density (percent of basal area per hectare).

Through box plot analysis, it was shown that elemental concentrations for species differ by region. This variability is likely due to variability in soil type and associated variations in nutrient availability and other soil properties.

Regression analysis showed significant relationships between foliage elemental concentrations and soil elemental concentrations and other soil properties, depending on species type. For example, foliar Ca concentrations of most species are positively correlated to Ca concentrations in specific soil layers. As well, foliar K, P, N concentrations of balsam fir are particularly correlated to nutrient concentrations in the forest floor. Foliar N concentrations in tolerant and intolerant hardwoods are partly correlated to the C/N ratio of the forest floor, and to the OM content (%) of the A layer.

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## CHAPTER 9

### EVALUATING CRITICAL ACID DEPOSITION LOADS AND RELATED EXCEEDANCE FOR ARNEWS SITES

#### INTRODUCTION

In this Chapter, the ForSust model (see Chapter 5) is used to calculate critical acid deposition loads (CL) and related exceedances for ARNEWS sites (see Chapter 6). As defined by Nilsson and Grennfelt (1988), critical loads (CL) are “a quantitative estimate of an exposure to one or more pollutants below which significantly harmful effects on specified sensitive elements of the environment do not occur”. In principle, excess amounts of acid deposition [= (N + S) deposition – base cation deposition] above CL enhances soil acidification. This enhancement may also lead to soil nutrient imbalances and to reductions of forest biomass growth (Sverdrup *et al.* 1994). Therefore, evaluating CL and related exceedances based on current or supposed atmospheric deposition rates is important for the context of evaluating long-term forest growth potentials for existing or planned forest growth conditions.

The objectives of this chapter are:

1. to establish critical soil acidification loads (CL) for ARNEWS sites from Newfoundland to Alberta;
2. to determine whether these loads are exceeded or not;
3. to evaluate how CL and related acid exceedances are affected by local atmospheric, soil, and vegetative conditions.

## METHODS

### Modeling approach

The assessment of critical loads is based on the simple mass balance (SMB) approach, which is formulated in the ForSust model (see Chapter 5). In this approach, soil acidification is calculated as the difference between soil-based inputs and outputs of acid generating and acid neutralizing cations and anions. Inputs of base cations (Ca, Mg, K) by mineral weathering and atmospheric deposition are considered acid neutralizing; plant uptake of base cations is considered to be soil acidifying; atmospheric S and N deposition is considered acidifying; but plant uptake S and N is acid neutralizing. Hence, each forest site has a specific potential (or critical acid deposition load) to buffer acid S and N deposition, which further depends on local climate, soil, and vegetation conditions.

### Study area and model inputs

One hundred and twenty five ARNEWS plots (see Chapter 6) have been chosen for assessing critical loads and related exceedances. Site-specific model inputs have been compiled in the preceding Chapters as follows: atmospheric deposition and climatic data (Chapter 6), soil data (Chapter 7), and tree elemental data (Chapter 8).

In order to advance the evaluation of critical loads per site, it was decided to evaluate each plot based on each dominant species separately. For example, a site that contained both sugar maple and yellow birch as dominant trees would be evaluated twice, once for sugar maple, and once for yellow birch. In so doing, site-specific data for climate, atmospheric deposition, climate, and soil parameters were kept the same. In this

way, 186 site-species variations were generated from the originating 125 ARNEWS plots (Appendix A, Tables A.5).

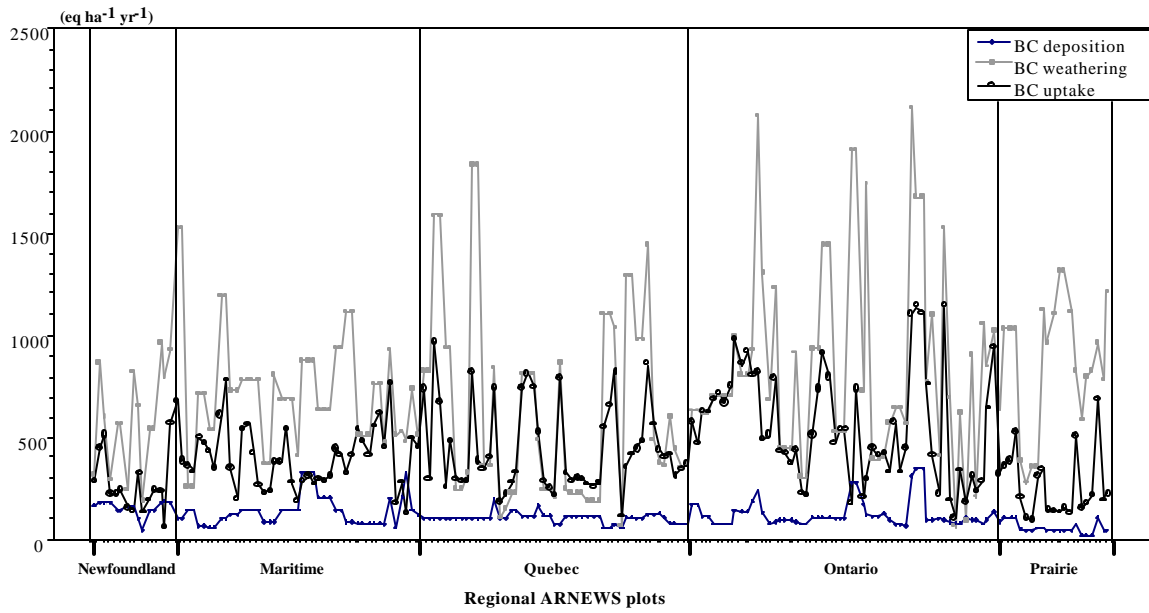
For the purpose of mapping and per plot analysis of the results, model-calculated estimates for critical soil acidification loads, acid exceedances, and N leaching rates were averaged by tree species ratio per plot.

## RESULTS AND DISCUSSION

### Base cation (BC) deposition, weathering, and uptake

Figure 9.1 shows the distributions of BC wet deposition, model calculated BC weathering and BC uptake used. Current BC deposition rates are seen to be less than 300 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) for all ARNEWS plots. Values appear to be similar for all species (Figure 9.4).

Estimated rates of BC weathering range from 66 to  $>2000$  ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ). This range is mainly determined by assigning a basic weathering rate to a given soil parent material, and this value is then prorated according to depth of rooting zone, and to clay content. Values are somewhat higher below deciduous species (especially red oak and sugar maple) which usually grow on base-rich sites. Among conifers, weathering rates tend to be highest for white pine (Figure 9.4).



**Figure 9.1** Rates of base cation wet deposition, base cation weathering, and base cation uptake for ARNEWS plots, from Newfoundland to Alberta. Data are independent for each plot and connection lines used only for comparison.

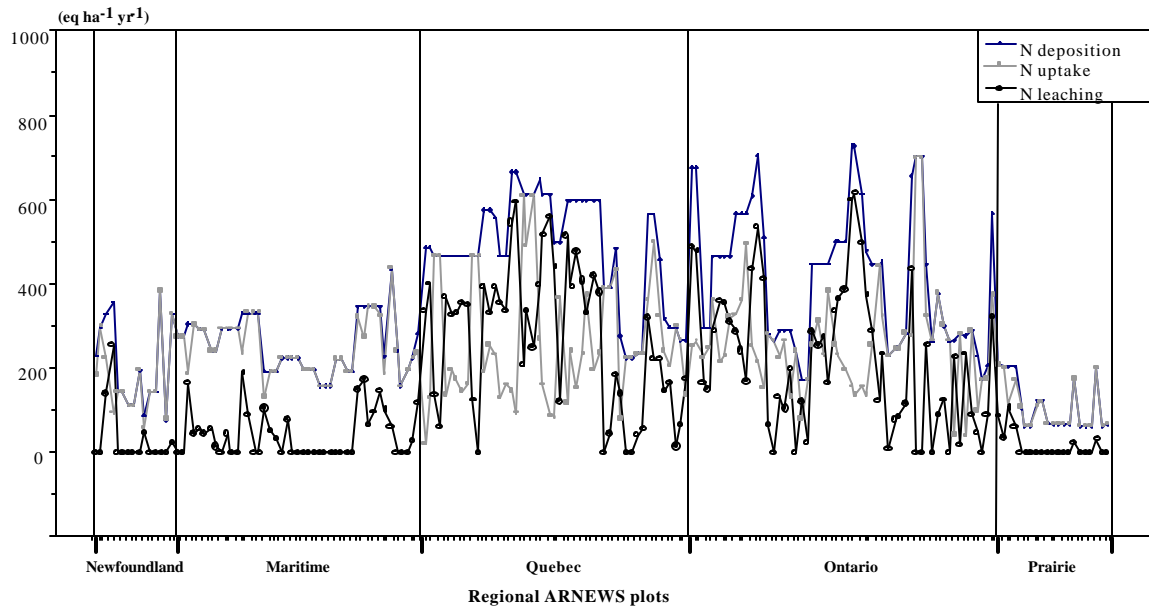
Estimated values for BC uptake are below 1150 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ). Here, estimated values are determined by soil availability, and by species demand. Estimated uptake rates are highest for red oak and sugar maple (Figure 9.4). For European forests, it was reported that N and BC uptake rates are nearly twice as high for deciduous forests than for coniferous forest due to both higher growth rates and higher N and BC contents (De Vries *et al.* 1994).

#### N deposition, uptake, and leaching

Figure 9.2 shows the data distribution for atmospheric N deposition, estimated N uptake, and estimated N leaching. Values for current N deposition range from 60 to 725 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ). Lowest N deposition rates occur in the Prairies. Highest rates occur in Quebec and Ontario (Figure 9.5).

In forestry, N additions to a site are generally beneficial to biomass growth. This, however, is only the case when N availability is actually growth limiting. Forests with this condition are said to be N under-saturated. Forests for which growth is not limited by N availability are said to be N saturated. As indicated in Figure 9.2, there are many ARNEWS plots that are calculated to be highly N under-saturated, and N uptake rates are therefore high. High N uptake, in turn, may mean decreased soil acidification if:

- (i) the dominant form of N uptake occurs with  $\text{NO}_3\text{-N}$  uptake,
- (ii)  $\text{NO}_3\text{-N}$  uptake exceeds the combined charge-equivalent uptake of cations such as  $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$  and  $\text{NH}_4\text{-N}$ .



**Figure 9.2** Rates of N wet deposition, N uptake, and N leaching for ARNEWS plots, from Newfoundland to Alberta. Data are independent for each plot and connection line used only for comparison.

Where N saturated, surplus of available N is likely lost from the soil on account of  $\text{NO}_3\text{-N}$  leaching. Increased N saturation may also be accompanied with increased soil nitrification rates, which, in turn, adds to N-leaching. In general, N in the form of  $\text{NO}_3\text{-N}$  is much more easily leached from soils than N in the form of  $\text{NH}_4\text{-N}$ . The leaching of the latter is constrained on account of ion retention on cation exchange sites. In forest soils, cation exchange sites dominate anion exchange sites by at least an order of magnitude, in general. In addition,  $\text{NH}_4\text{-N}$  may be retained in fine textured soils on account of  $\text{NH}_4^+$ -fixation, whereby  $\text{NH}_4$  becomes part of the surface structure of clay minerals. In some cases, early high rates of N retention and uptake might be replaced later by increased nitrification rates, as is the case for newly established red alder sites.

As shown in Figure 9.2, N uptake rates were calculated to be below 700 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) for all ARNEWS sites. In general, availability and uptake rates of N appear to be most restricted in the east and in the west, thus leading to low to very low (near zero) estimates for N leaching in these areas. In Quebec and Ontario, and select areas in Newfoundland and the Maritimes, forest growth and related N uptake appears to be limited by the availability of other elements, especially Ca and K. At these locations, N not taken up by vegetation is calculated to be leached up to  $\sim 600$  ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ).

In terms of forest vegetation, N uptake is calculated to be higher for deciduous species such as white birch, red oak, and sugar maple than for conifers (Figure 9.5). Similarly, N leaching rates are generally higher for deciduous vegetation such as red oak and sugar maple than for other species (Figure 9.5). This may in part be due to the high

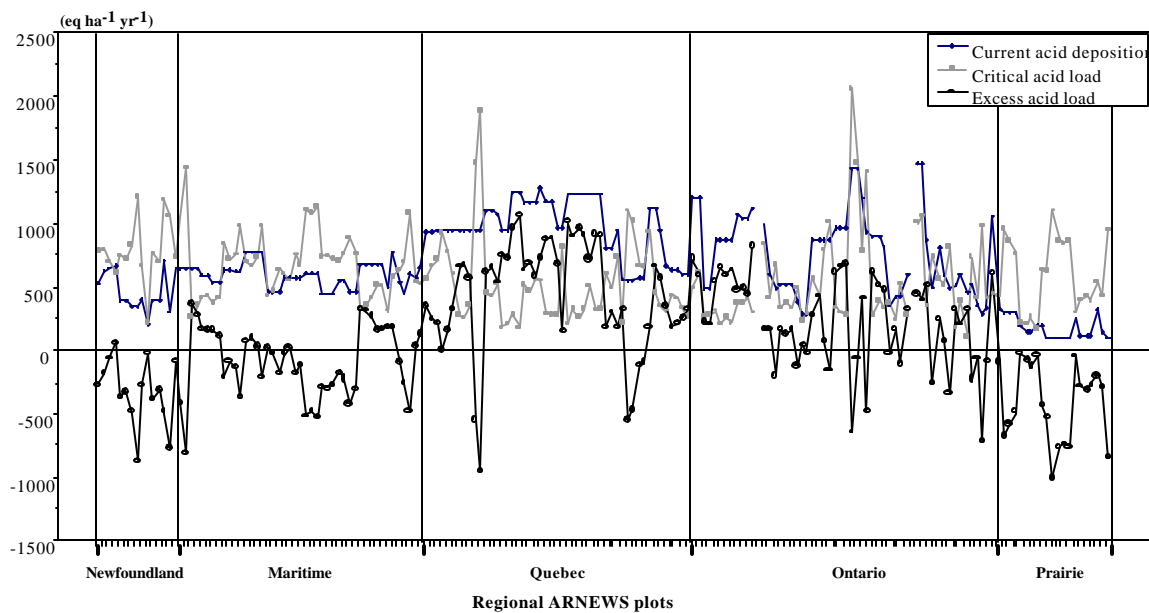
rates of local N deposition, but may also be due to a limited capacity of deciduous forest soils to retain N.

### Critical acid loads and related exceedances

As shown in Figure 9.3, current acid (S and N) deposition rates are calculated to range from 100 to about 1500 (eq ha<sup>-1</sup> yr<sup>-1</sup>). Sites located in the Prairies generally receive lowest acid deposition rates, while some sites in southern Quebec and Ontario receive the highest rates. In Atlantic regions, S and N deposition are estimated to be elevated in coastal areas due to fog deposition and sea-spray; inland areas receive moderate rates of acid deposition. The pattern of acid deposition by species follows the pattern for N deposition, i.e., values are highest for red oak, sugar maple, white pine, and slightly elevated for red spruce comparing with the other species (Figure 9.6).

Critical soil acidification loads (CL) vary from 100 to over 2000 (eq ha<sup>-1</sup> yr<sup>-1</sup>), see Figure 9.3. Lowest and highest values are calculated to occur in Quebec and Ontario where soil substrates vary from low to high base contents. Comparing rates of CL with rates of BC weathering, BC uptake, N uptake, it can be found that CL values are generally related to weathering rates of soil substrate, and are selectively (species-specific) related to uptake of BC (-) and N (+).

As shown in Figure 9.6, CL values were generally below 500 (eq ha<sup>-1</sup> yr<sup>-1</sup>) for jack pine sites, and are relatively high for balsam fir, mainly due to low and high soil weathering rates, respectively. Sugar maple sites tend to have slightly lower CL values than fir. This is due to high base cation uptake in relation to N uptake.

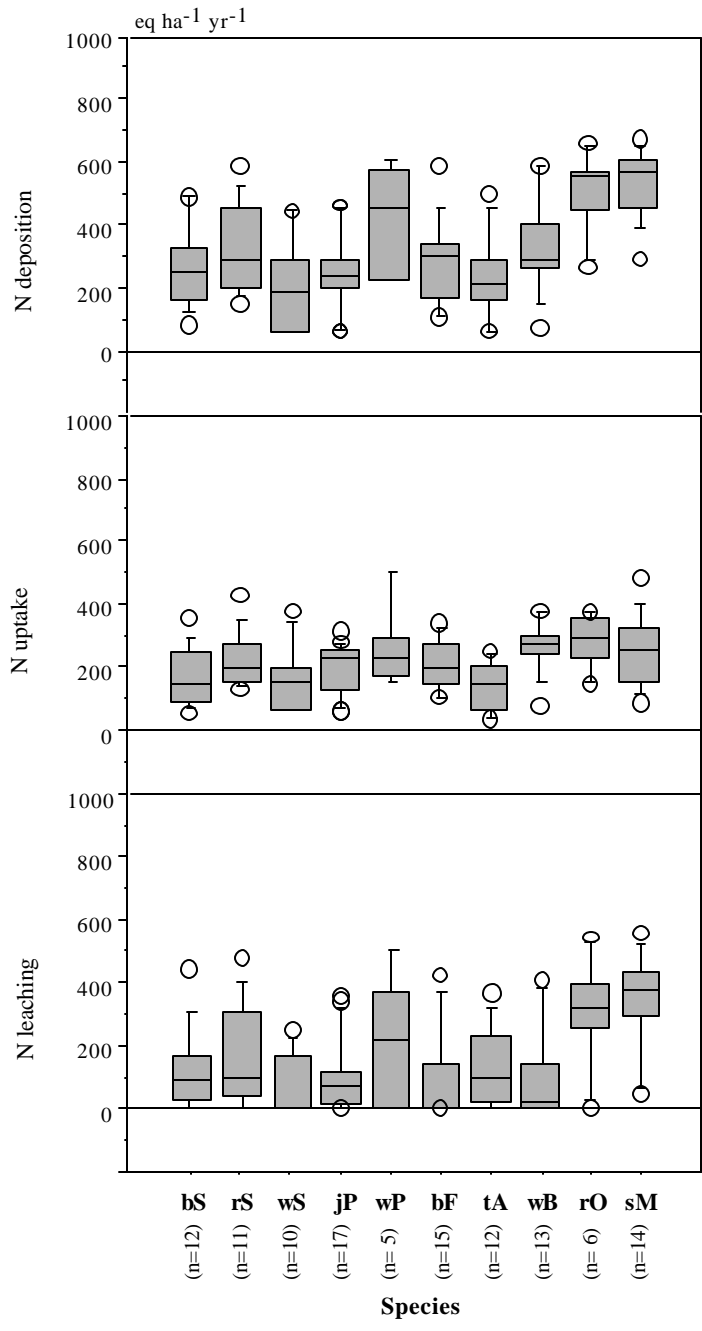


**Figure 9.3** Rates of current acid (S and N) deposition, critical acid load, and excess acid load for ARNEWS plots, from Newfoundland to Alberta. Data are independent for each plot and connection line used only for comparison.

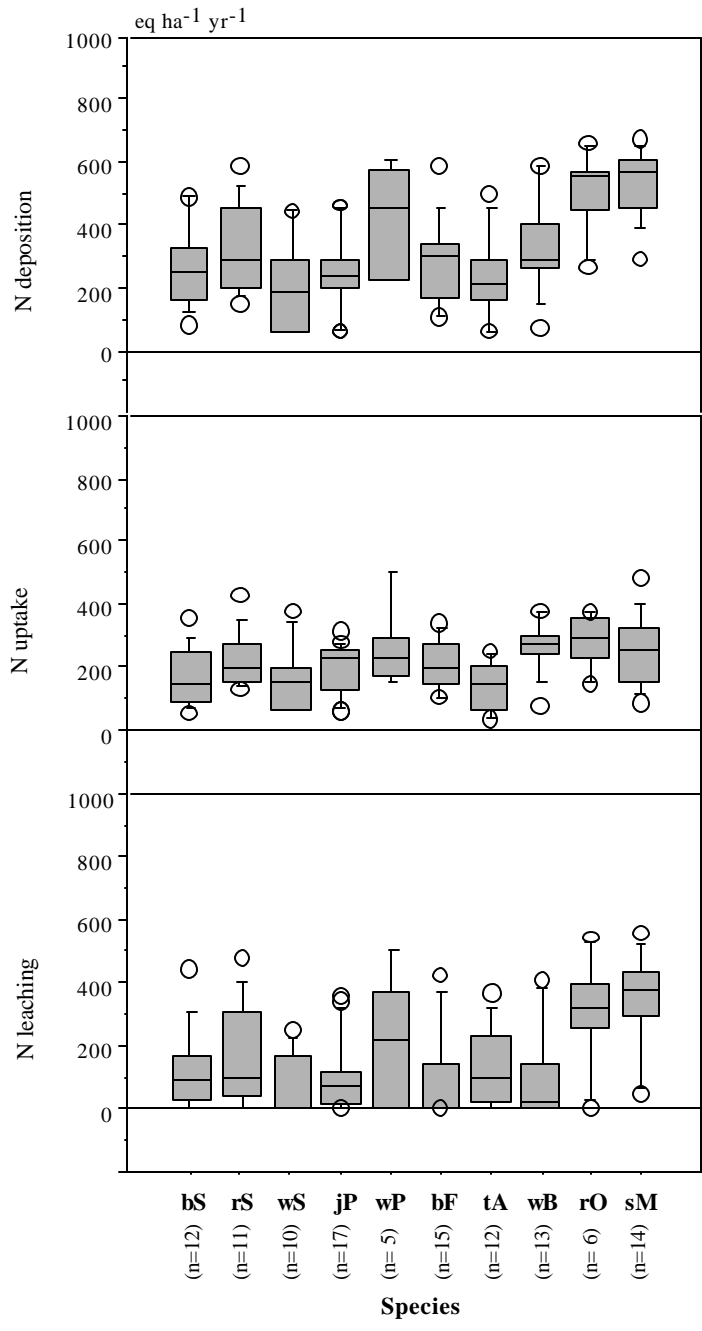
Values for other softwood and hardwood species are intermediate. In Europe, critical acid loads were reported to be high for deciduous forests, possibly due to high soil weathering rates on hardwood sites (De Vries *et al.* 1994). For ARNEWS plots, CL values do not vary much between deciduous and coniferous species.

In terms of region (Figure 9.7), areas with  $< 500$  ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ) CL are prevalent in the North West Territories (NWT), in the northern part of Prairie Region (Alberta, Saskatchewan, and Manitoba), northern and central Ontario (except areas in the northeast), northern and central southern Quebec, northeastern New Brunswick and Labrador. This is again mainly due to low weathering rates (soil substrates with low weathering potentials), and low N and/or S uptake rates. Sites with  $> 500$  ( $\text{eq ha}^{-1} \text{ yr}^{-1}$ ) CL are found in Newfoundland, in Nova Scotia, in western and southern parts of New Brunswick, in southeastern and southwestern parts of Quebec and Ontario (including a patch in the northeast), in southern Manitoba, Saskatchewan, and Alberta, on soils with high BC weathering rates.

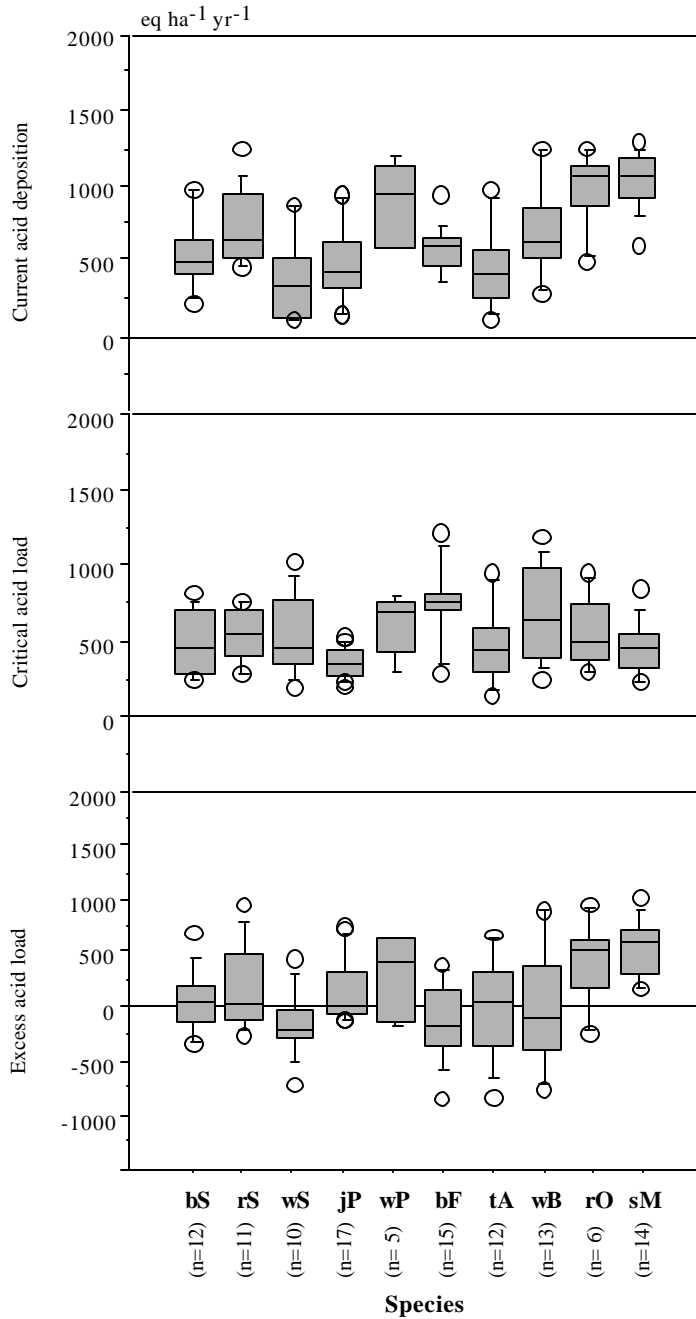
In all of this, the choice of an ARNEWS plot within each region is important. For example, while parts of southern Quebec and Ontario have soils that developed on calcareous soil parent materials, the majority of the ARNEWS plots fall into the area of the Canadian Shield, where soils have essentially developed on igneous substrates with low acid buffering potential. In the Prairies, most plots were established on soil substrates with a calcareous base. In Atlantic Canada, soil plot location varied greatly, ranging with substrates with low to high soil buffering potential.



**Figure 9.4** Atmospheric wet deposition of base cation (BC), model calculated BC weathering, and BC uptake grouped by tree species including black spruce (bS), red spruce (rS), white spruce (wS), jack pine (jP), white pine (wP), balsam fir (bF), trembling aspen (tA), white birch (wB), red oak (rO), and sugar maple (sM), in ARNEWS plots.



**Figure 9.5** Atmospheric wet deposition of N, model calculated N uptake, and N leaching grouped by tree species including black spruce (bS), red spruce (rS), white spruce (wS), jack pine (jP), white pine (wP), balsam fir (bF), trembling aspen (tA), white birch (wB), red oak (rO), and sugar maple (sM), in ARNEWS plots.



**Figure 9.6** Current acid (S+N) deposition, model calculated critical acid load (CL), and Exceedances of acid load grouped by tree species including black spruce (bS), red spruce (rS), white spruce (wS), jack pine (jP), white pine (wP), balsam fir (bF), trembling aspen (tA), white birch (wB), red oak (rO), and sugar maple (sM), in ARNEWS plots.

Values for critical acid deposition exceedances are shown in Figure (9.3). As indicated, ARNEWS plots located in Newfoundland and Prairie regions are mainly calculated to not exceed CL due to relative high critical loads and low acid deposition rates. Critical acid deposition exceedances are calculated to be positive on some plots in the Maritimes, but the exceedances are mostly less than 500 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ). Exceedances are mostly positive for the ARNEWS plots in Quebec and Ontario. Here, calculated exceedances are also highest (up to  $1063 \text{ eq ha}^{-1} \text{yr}^{-1}$ ), due to high acid deposition rates and low critical load values.

As shown in Figure 9.6, exceedances of CL are calculated to be mostly negative for ARNEWS balsam fir plots due to choice of location, where acid deposition values are moderate, and CL values are relatively high. In contrast, sites covered with red oak and sugar maple are calculated to have positive exceedances that is related to high base cation uptake rates, high acid deposition rates, and moderate estimates for rate of soil weathering. In general, calculated exceedance values tend to be mainly positive for deciduous sites and mainly negative for coniferous sites.

Shown in Figure 9.8 are the calculated acid exceedances for each ARNEWS plot from Newfoundland to Alberta. As calculated, critical soil acidification loads are not exceeded in NWT, Alberta, Saskatchewan, and Manitoba due to low acid deposition and high acid buffering rates of the soil substrates. However, there may be areas where local S and N emissions are high, e.g., localities where S is released from sour gas production plants, or where N is released in major amounts from high-powered diesel engines that are used for the surface mining of local oil sands (Dale Doram, personal information).





Positive exceedances are estimated to occur in Ontario (except sites located in northwest and southwest of the province) and Quebec due in part to low CL (acid sensitive parent materials), and in part due to high acid deposition rates. Exceedances in excess of 500 (eq ha<sup>-1</sup> yr<sup>-1</sup>) are calculated to occur at a number of ARNEWS plots in Ontario and Quebec where atmospheric depositions of S and N are highest.

Calculated acid exceedances are also positive in east parts of New Brunswick, west parts of Nova Scotia and Newfoundland where S and N deposition is enhanced by fog deposition. Exceedances of acid deposition in other parts of Maritime Provinces and Newfoundland are negative. The reason for the general non-exceedance in these areas is mainly due to the selection of the ARNEWS plots. Relocating ARNEWS sites on soils with low acid buffering potential would produce exceedances in this region as well.

## UNCERTAINTIES

There are a number of uncertainties regarding the calculated critical load and their exceedance values. These are due to assumptions made about

1. the numbers adopted for critical Al and acid leaching,
2. representativeness of atmospheric deposition,
3. approaches taken to calculate rates for soil weathering,
4. the limitations of the SMB approach (model structure) itself.

### Uncertainty in critical chemical values

Values for critical Al concentration, Al/BC concentration ratio, and H concentration need to be acceptable for calculating CL. In principle, levels should be chosen so that the forest ecosystem as a whole can be protected. Several suggestions have been made:

1. Set the value at that level where plants or animals show immediate negative response.
2. Set the value at that level where selected plants (trees) or animals are safe, and are therefore protected from immediate damage.
3. Set the value at that level where the forest ecosystem as a whole is safe, this includes vegetation, soil organisms, and aquatic life downstream.

In general, an Al/BC ratio of 1.5 have been determined to present threshold values beyond which there is immediate physiological damage to tree roots and other parts of the plants based on laboratory/greenhouse experiments (De Vries *et al.* 1994). There is, however, lack of knowledge about the effects of Al in the field. As well, there is a natural range in sensitivity of various tree species and other species for Al toxicity (De Vries 1991). For the calculation of CL in this chapter,  $[Al]/[BC]_{le, crit}$  was set at  $0.15 \text{ eq eq}^{-1}$ , and  $[NO_3]_{le, crit}$  was set at  $0.02 \text{ meq L}^{-1}$  (Arp *et al.* 1996). These numbers are conservative in comparison with the physiological thresholds. They do, however, offer a considerable safety margin for most species that need to be protected against high Al influx and uptake rates. Such a margin needs to be available, because field conditions are highly variable. For example, if the critical Al leaching value would be on average at the

physiological threshold level, than a considerable part of the vegetation would be affected because of:

1. high soil variability,
2. high variability in terms of Al tolerance from species to species and also within species
3. high sensitivity of aquatic and semi-aquatic species to Al.

#### Uncertainties in atmospheric deposition

Atmospheric deposition data used for this calculation only referenced to wet deposition. Adding rates of dry deposition for BC, N and S should increase the calculated CL exceedances. The review of literature revealed that depending on site location, dry deposition of BC, N, and S can be 15% to 25% of the corresponding wet deposition rates (see Chapter 3). Study for a tolerant hardwood site at Turkey Lake, Ontario (Moayeri *et al.* 1999) showed that including dry deposition rates for BC, N, and S increases critical load values by 4-5%, which can double the values of CL exceedance.

#### Uncertainties in soil weathering rates

Determining the rate of soil base weathering is central to the determination of the critical soil acidification loads (Hodson and Langan 1999). Based on current knowledge, the direct measurement of BC weathering is not possible. Instead, several indirect methods have been developed (see Chapter 3), with each method generating differing results (Kolka *et al.* 1996). The weathering rates in this chapter are estimated based on an empirical model (due to lack of input data for mechanistic model represented in Chapter

4) that relates soil weathering to three basic soil parent material classes (low, medium, and high acid buffering classes), to soil texture (clay content), and to mean annual air temperatures. The uncertainties for this weathering calculation might be large.

#### Uncertainties in SMB model structure

Uncertainties in model structure refer to the assumptions in the model. An important assumption in the ForSust model is the assumed homogeneity of the rooting-zone. In reality, Al/BC ratio, mineral weathering rates and nutrient uptake rates vary with soil depth, in a soil-specific manner. However, overall nutrient taken up by trees and nutrient leached from site are not so much related to local variations of nutrient concentrations, and local acidity levels. Overall nutrient availability and/or acidity levels across the accessible rooting space are much more important. Hence, the one layer assumption is operationally appropriate.

## CONCLUSIONS

Using an Al/BC ratio of 0.15 (eq eq<sup>-1</sup>) as the soil acidification criterion, critical loads (CL) and related exceedances (or none exceedances) have been identified for ARNEWS sites from Newfoundland to Alberta. Based on the current acid (S and N) deposition rates and calculated critical loads, exceedances of critical loads were positive in Ontario, Quebec, coastal areas of New Brunswick, Nova Scotia and Newfoundland due to low CL (acid sensitive parent materials) and/or high acid deposition rates. Exceedances of acid loads were large (> 500 eq ha<sup>-1</sup> yr<sup>-1</sup>) at a number of ARNEWS plots

in central and south of Ontario and Quebec where atmospheric S and N depositions were extremely high. High rates of N deposition in these areas may also cause N saturation.

ARNEWS sites were not acid exceeded in west (NWT, Alberta, Saskatchewan, and Manitoba) and some parts of east (Maritime and Newfoundland provinces except along the coast) due to location of these sites on high acid buffering soil parent materials and/or relatively low acid depositions. However, these results do not necessarily mean that all forest sites in these areas were not exceeded. Exceeded areas may still be present due to local S and N emissions and also by counting dry depositions into the assessments particularly on soils with low acid buffering potential.

Acid exceedances were mostly negative for balsam fir sites due to their moderate acid deposition rates and relatively high CL values. In contrast, exceedances were mainly positive for red oak and sugar maple sites owing to high rates of acid deposition, BC uptake, and low CL values. High rates of acid deposition on these sites relate to high rates of N deposition.

It is important to note that the results of these assessments are not the ultimate values of CL and exceedances of acid deposition for these ARNEWS plots. The values might differ significantly by changes in rates of atmospheric deposition (e.g., increase or decrease of S and N emissions, including dry deposition), rates of BC weathering (e.g., improved weathering estimations by providing input data for a mechanistic model), or changing the critical soil acidification criterion values.

It is also important to emphasize that the interactions between soil acidification and forest growth are complex. However, excess of acid deposition over CL involves a certain risk to the health of forests, which should increase with the magnitude and the

duration of the exceedance (De Vries *et al.* 1994). Therefore, positive exceedances as calculated for several sites in Ontario and Quebec mean that the health of these forest sites may be endangered, and that these sites require attention.

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CHAPTER 10  
ASSESSING IMPACTS OF ACID DEPOSITION ON  
CANOPY CONDITION FOR ARNEWS SITES

INTRODUCTION

Learning how to assess potential impacts of acid deposition on forest damage has been a matter of considerable interest in Europe and North America. Observed impacts have been related to soil acidification and related nutrient imbalances, changes in tree physiology and tree nutrients, twig and branch dieback, increased canopy transparency, tree mortality, and tree growth (height, diameter, and biomass) reduction.

In Canada, several attempts have been made to determine impacts of acid deposition on forest damage (Hall *et al.* 1997). For example, birch decline along the Bay of Fundy coast in Eastern Canada is attributed to acid fog (Cox *et al.* 1996). Observed red spruce decline at high elevation areas, where acid rain impacts are high due to within canopy cloud drift, are related to increased Al/Ca ratios of woody tissue and respiration (McLaughlin *et al.* 1991). High acid rain also leads to increase foliar N and S levels due to increased N and S uptake (Hogan 1998) which in turn may cause reduced frost hardiness (Sheppard 1994). Growth and nutrient status of sugar maple seedlings have been linked to increased soil acidity and decreased base cations (Ouimet *et al.* 1996). In addition, dry and wet acid deposition rates alter the chemical and physical characteristics of leaf cuticles (Percy *et al.* 1994, Huttunen 1994). Ambient levels of acid rain accelerate base cation losses from sugar maple sites (Foster *et al.* 1992). According to my knowledge,

impacts of acid deposition on the degree of canopy transparency have not yet been addressed.

The Acid Rain National Early Warning System (ARNEWS) was designed by Canadian Forest Service (CFS) to detect possible damage to forest trees and soils by acid deposition (D'Eon and Power, 1989). In this study, potential relationships between acid deposition and canopy conditions are investigated using data from 125 ARNEWS plots.

The specific objectives are:

1. To assess the magnitude of % canopy transparency (aggregated crown damages) and combined foliage damage caused by insects, diseases, and abiotic factors.
2. To determine the degree of correlation between canopy damage (% canopy transparency, insect damage, disease damage, and abiotic damage) and acid deposition (S and N wet deposition, critical soil acidification loads, and exceedances of critical loads).
3. To verify the hypothesis that there is a potential relationship between exceedances of critical acid deposition load and canopy transparency.
4. To quantify this relationship by way of multiple regression for various tree species (balsam fir, black spruce, red spruce, white spruce, jack pine, sugar maple, trembling aspen, white birch).

## METHODS

### Data set

Field data of crown condition, insect foliar damage, disease foliar damage, and abiotic foliar damage were obtained from the Canadian Forest Service (CFS). Data were recorded each year for all trees in each plot for the period of 1984-1995. Foliar damage levels were coded as explained in the ARNEWS manual (Magasi 1988; D'Eon *et al.* 1994). All damage data were converted into appropriate percent values, as explained in Tables (10.1 – 10.3). These data were then averaged by tree over the entire recorded period. The data were further averaged by plot to generate 125 data row (Appendix A, Table A.6).

Another data set was established by averaging the crown condition data (per tree) by tree type and by plot. To be consistent with other ARNEWS data, only dominant and co-dominant trees were used. In this way, an identical data set (186 rows) was compiled (ARNEWS CD-ROM 2000, Dmg-data folder, Dmg-186 file). Because the new data sets are results of aggregating crown condition data (recorder by tree), crown damage (defoliation for conifers) is called “% canopy transparency”.

For the purpose of statistical analysis, canopy data were cross-referenced with the other ARNEWS data including atmospheric deposition (see Chapter 6), critical acid deposition loads, exceedance of critical loads, base cation leaching, and N leaching (Chapter 9), and site information such as latitude, longitude, altitude, and tree age.

**Table 10.1** Hardwood crown classification system for ARNEWS data (D'Eon et al. 1994) and corresponded average % damage.

Code	Crown condition	Ave. % damage
10	Full complement of foliage. Tree exhibits no visible crown damage.	0
20	Foliage thin, off-color. No dead branches present or bare twigs visible.	0
30	No dead branches present. Bare twigs present in up to 5% of the crown.	3
35	No dead branches present. Bare twigs present in more than 6% of the crown.	7
40	Dead branches and bare twigs present in up to 15% of the crown.	12
45	Dead branches and bare twigs present in 16 to 25% of the crown.	20
50	Dead branches and bare twigs present in 26 to 37% of the crown.	31
55	Dead branches and bare twigs present in 38 to 50% of the crown.	44
60	Dead branches and bare twigs present in 51 to 75% of the crown.	63
65	Dead branches and bare twigs present in 76% or more of the crown.	80
70	More than 50% of the crown dead.	96
08	Tree died since last assessment.	100
09	Dead tree.	100

**Table 10.2** Conifer crown classification system for ARNEWS data (D'Eon et al. 1994) and corresponded average % damage.

Code	Crown condition	Ave. % damage
01	No defoliation.	0
02	Only current foliage defoliated. Defoliation less than 25%.	15
03	Current and / or some older foliage defoliated. Defoliation less than 25%.	20
04	25-50% defoliation.	37
05	51-75% defoliation.	63
06	76-90% defoliation.	82
07	More than 90% defoliation.	90
08	Tree died since last assessment.	100
09	Dead tree	100

**Table 10.3** Classification of foliage damages caused by insects, diseases, and abiotic factors for ARNEWS data (D'Eon et al. 1994) and corresponded average % damage.

Code	Observed damage level	Ave. % damage
0	None affected	0
T	1-5% (trace)	3
1	6-15%	10
2	16-25%	20
3	26-35%	30
4	36-45%	40
5	46-55%	50
6	56-65%	60
7	66-75%	70
8	76-85%	80
9	86-100%	93

## Analyses

Percent values of canopy transparency, insect damage, disease damage, and abiotic damage were regressed against S and N deposition, critical load, and exceedances of critical loads. Since some of these variables were not normally distributed, significance of the correlation was checked by way of the non-parametric Spearman rank correlation procedure (StatView 1998).

Since % canopy transparency is the most common criterion for assessing forest damage (Thomsen, 1990), and is also the most readily recorded, therefore, % canopy transparency was selected as the main criterion of forest damage. To facilitate the analysis of the % canopy transparency data, it was decided to group average % canopy transparency into quarterly % canopy transparency percentiles (25<sup>th</sup>, 50<sup>th</sup>, and 75<sup>th</sup> and 100<sup>th</sup> percentiles). Mean values for atmospheric S and N deposition, critical load, and exceedances critical loads were then compared among the four levels of % canopy transparency.

Since exceedance of critical loads is the main variable that covers the degree of acid deposition and also buffering potential of forest site, % canopy transparency was regressed against exceedances of critical load. This analysis was done by average values of equally clustered plots after sorting the data by % canopy transparency from the lowest to the highest rates.

ANOVA and the Tukey-Kramer post hoc test (StatView 1998) were applied to determine significant differences of % canopy transparency by tree type. The following tree species were included: balsam fir (*Abies balsamea* L. Mill.), black spruce (*Picea mariana* Mill.), red spruce (*Picea rubens* Sarg.), white spruce (*Picea glauca* Moench),

jack pine (*Pinus banksiana* Lamb.), sugar maple (*Acer saccharum* Marsh.), trembling aspen (*Populus tremuloides* Michx.), and white birch (*Betula papyrifera* Marsh.).

Forest damage as represented by % canopy transparency is usually due to a combination of factors. In this case, the following variables were all considered as potential contributors to % canopy transparency.

- atmospheric S and N wet deposition,
- soil acidification (as expressed by critical soil acidification loads, exceedance of critical loads, base cations leaching, and N leaching),
- tree age,
- foliar damages due to insects, diseases, and abiotic factors, and
- site location (latitude, longitude, altitude)

A linear multiple regression procedure was used to assess the % canopy transparency variance contributions of each of the variables, by tree species.

## RESULTS AND DISCUSSION

Shown in Tables 10.4 and 10.5, are correlation coefficients among S and N deposition, critical loads, exceedance of critical loads, % canopy transparency, insect damage, disease damage, and abiotic damage. Defoliation was strongly correlated to S and N deposition, and to exceedance of critical loads as well. Disease damage was weakly correlated to S and N deposition. No clear relation was found between the soil acidifying variables and foliar damage caused by insects, diseases, and abiotic factors, see Table 10.4.

**Table 10.4** Correlation coefficients between explanatory variables and crown-condition variables (%).

	Crown-condition damage (%)			
	Defoliation	Foliar insect	Foliar disease	Foliar abiotic
Sulfate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.579 <sup>a</sup>	0.116	0.208 <sup>b</sup>	0.086
Nitrate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.527 <sup>a</sup>	0.120	0.182 <sup>b</sup>	0.060
Critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	-0.008	0.050	0.060	0.057
Excess of critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.467 <sup>a</sup>	0.034	0.130	0.102

<sup>a</sup> p < 0.0001

<sup>b</sup> p < 0.05

**Table 10.5** Spearman rank correlation coefficients between explanatory variables and crown-condition variables (%).

	Crown-condition damage (%)			
	Defoliation	Foliar insect	Foliar disease	Foliar abiotic
Sulfate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.566 <sup>a</sup>	0.187 <sup>b</sup>	0.332 <sup>a</sup>	0.345 <sup>a</sup>
Nitrate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.532 <sup>a</sup>	0.192 <sup>b</sup>	0.292 <sup>a</sup>	0.336 <sup>a</sup>
Critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	-0.028	0.128	0.124	0.099
Excess of critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.510 <sup>a</sup>	0.052	0.180 <sup>b</sup>	0.319 <sup>a</sup>

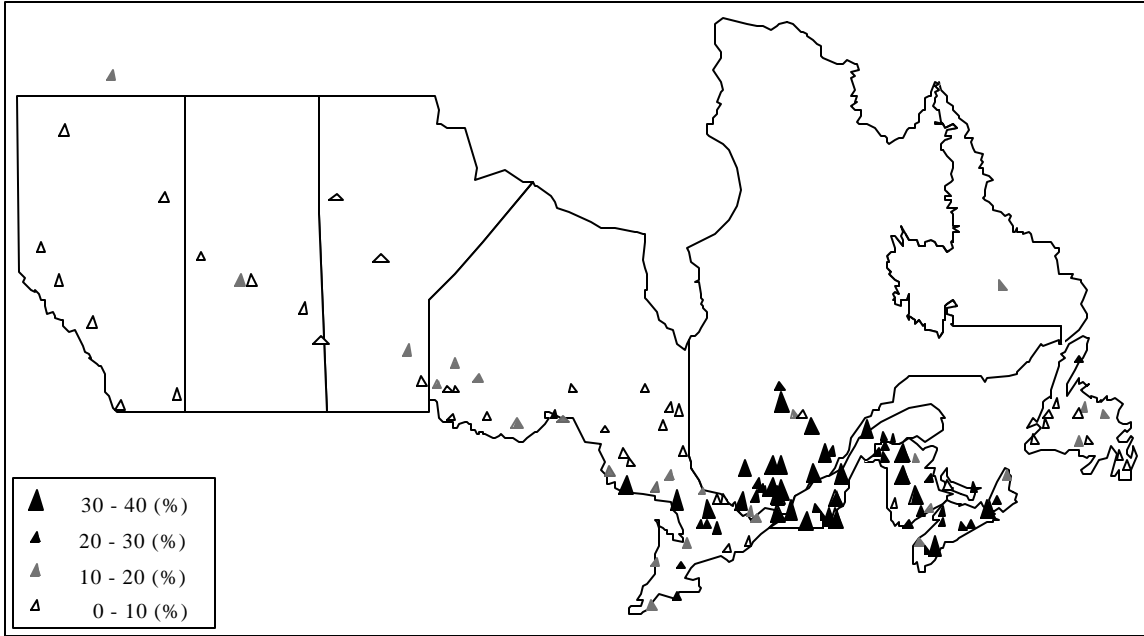
<sup>a</sup> p < 0.0001

<sup>b</sup> p < 0.05

The results of non-parametric correlation (Spearman rank correlation) were identical for canopy transparency (%) but were slightly different for the other crown-condition variables. Based on this analysis, abiotic and disease damages were significantly correlated to S and N deposition. However, insect damage was weakly correlated to S and N deposition.

The magnitude of canopy transparency in the ARNEWS plots is shown in Figure (10.1). Plots with the highest degree of % canopy transparency were aggregated in the southern part of Quebec (> 30 %). The Maritime Region (New Brunswick, Nova Scotia and PEI) had plots with second highest canopy transparency rates. In this region, plots were mainly ranked from relatively high (20-30%) to very high (> 30%). In Ontario, plots appeared to have relatively low (< 10%) to moderate (10-20%) canopy transparencies. However, a few plots in southern Ontario had relatively high (20-30%) to very high (> 30%) canopy transparencies. Percent canopy transparency was also low to moderate for plots in the Northern Region (Manitoba, Saskatchewan, Alberta) and in the Newfoundland Region (Newfoundland and Labrador).

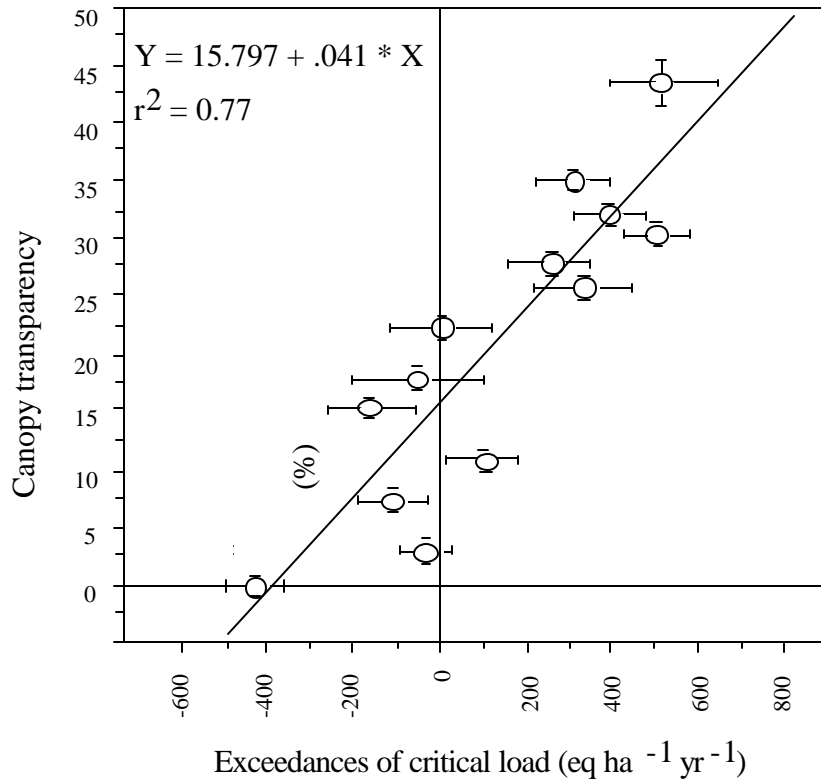
Table (10.6) shows that as the quarter percentiles of canopy transparency increases, mean values of atmospheric S and N deposition, and exceedance of critical loads consistently also increase. Simple regressions (Figure 10.2) confirm that there is a positive linear relationship between exceedance of critical loads and canopy transparency. Specifically, exceedance values > 200 eq ha<sup>-1</sup> yr<sup>-1</sup> are associated with high canopy transparency values.



**Figure 10.1** Defoliation level (percent of total foliage) for ARNEWS plots from Newfoundland to Alberta.

**Table 10.6** Comparison of pollution loads for ARNEWS plots with four quartile-defoliation degrees.

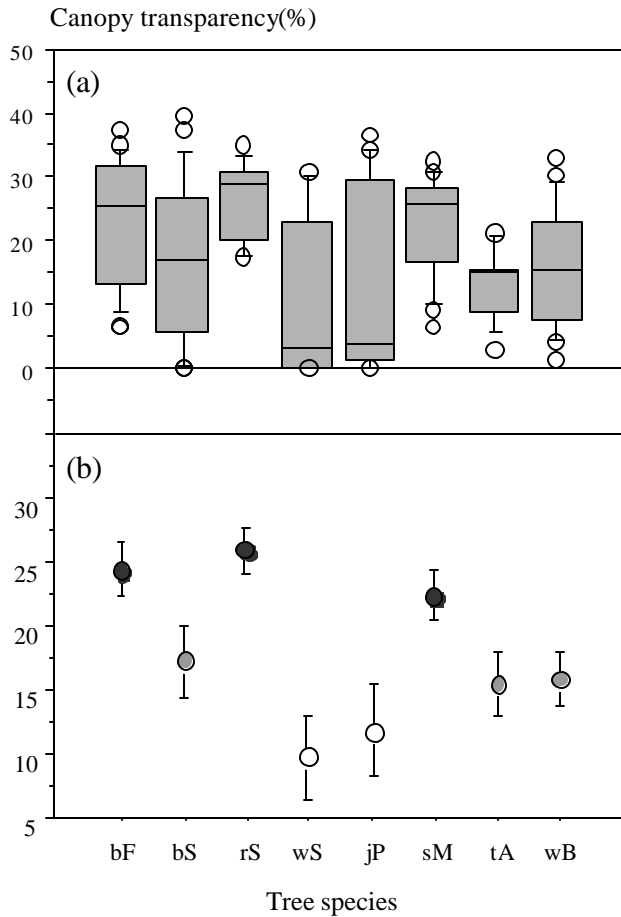
	Defoliation			
	< 10 %	10 - 20 %	20 - 30 %	> 30 %
Sulfate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	217 (19)	340 (29)	428 (16)	449 (20)
Nitrate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	125 (11)	171 (17)	245 (12)	255 (14)
Critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	605 (40)	868 (101)	738 (51)	572 (42)
Excess of critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	-111 (48)	-53 (83)	204 (56)	418 (49)



**Figure 10.2** The relationship between excess of acid deposition loads (ForSust model) and forest damage given as the degree of defoliation in ARNEWS plots. Each point represent mean values and related standard error for equal number sample plots.

According to Figure (10.3a), canopy transparencies ranged from zero to about 40% for most of the trees examined. However, high degree of canopy transparency mainly occurred for red spruce and sugar maple. In contrast, half of the plots (dominated by white spruce or jack pine trees) appeared to have a healthy canopy (near zero % canopy transparency). Shown in Figure (10.3b), mean canopy transparency by tree type was relatively high for red spruce ( $26.0 \pm 1.7\%$ ), balsam fir ( $24.4 \pm 2.1\%$ ), and sugar maple ( $22.4 \pm 1.9\%$ ), and was relatively low for white spruce ( $9.7 \pm 3.3\%$ ) and jack pine ( $11.8 \pm 1.7\%$ ). The difference between these two species groups (red spruce, sugar maple versus white spruce, jack pine) is statistically significant ( $P < 0.05$ ). Mean canopy transparency was moderate for black spruce ( $17.1 \pm 2.8\%$ ), trembling aspen ( $15.4 \pm 2.4\%$ ), and white birch ( $15.8 \pm 2.1\%$ ).

With multiple regression, it was found that canopy transparency was affected by various species-dependent factors, as shown in Table (10.7). In particular, canopy transparency was partly related to critical loads, exceedance of critical loads, and foliar insect and abiotic damages for balsam fir ( $r^2 = 0.73$ ). A weak positive relationship was found between canopy transparency and S deposition for black spruce ( $r^2 = 0.46$ ). Canopy transparency was positively related to exceedance of critical loads, foliar insect and abiotic damages for red spruce ( $r^2 = 0.67$ ). Partial relationships were found between canopy transparency and S deposition (+), N deposition (-), and foliar insect damage (+) for white spruce ( $r^2 = 0.76$ ). Percent canopy transparency was positively correlated to exceedance of critical loads and foliar insect damage for jack pine ( $r^2 = 0.65$ ).



**Figure 10.3** Distribution of the degree of defoliation (a) and mean of defoliation with corresponding error bar (b), grouped by major tree species including balsam fir (bF), black spruce (bS), red spruce (rS), white spruce (wS), jack pine (jP), sugar maple (sM), trembling aspen (tA), and white birch (wB), in ARNEWS plots.

**Table 10.7** Multiple regression equations for eight tree species relating the degree of defoliation (%) to explanatory variables. P values for intercept and independent variable are represented in bracket. The coefficient of determination (R<sup>2</sup>), the number of sample for each species (n) are represented.

Variable	Tree species							
	Balsam fir	Black spruce	Red spruce	White spruce	Jack pine	Sugar maple	Trembling aspen	Whit birch
Intercept	-6.77 (0.377)	-1.179 (0.837)	11.26 (0.021)	-5.007 (0.199)	4.251 (0.151)	8.68 (0.003)	-1.365 (0.625)	13.89 (<0.001)
Sulfate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )		0.069 (0.002)		0.199 (0.0004)				
Nitrate wet deposition, (eq ha <sup>-1</sup> yr <sup>-1</sup> )				-0.279 (0.001)				
Critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.025 (0.006)						0.019 (0.006)	
Excess of critical loads, (eq ha <sup>-1</sup> yr <sup>-1</sup> )	0.021 (0.002)		0.029 (0.005)		0.034 (0.001)		0.011 (0.014)	0.011 (<0.001)
Foliar insect damage, (%)	1.012 (<0.0001)		0.805 (0.044)	0.558 (0.035)	2.354 (0.056)	0.498 (0.061)	0.401 (0.024)	
Foliar disease damage, (%)						1.98 (<0.0001)		
Foliar abiotic symptoms, (%)	4.613 (0.001)		8.134 (0.045)					3.08 (0.0001)
tree age, (yr)							0.114 (0.017)	
r <sup>2</sup>	0.73	0.46	0.67	0.76	0.65	0.83	0.86	0.75
n	25	19	13	15	17	18	13	20

No correlation was found between canopy transparency and in part acid deposition factors for sugar maple. However, canopy transparency was positively correlated to foliar insect and disease damages for sugar maple ( $r^2 = 0.83$ ). Percent canopy transparency was partially related to critical loads, exceedance of critical loads, foliar insect damage, and tree age for trembling aspen ( $r^2 = 0.86$ ). Finally, canopy transparency was positively related to exceedance of critical loads and foliar abiotic damage for white birch ( $r^2 = 0.75$ ).

## CONCLUSION AND DISCUSSION

The results of this study showed that increased canopy transparency is, at least in part related to exceedance of critical soil acidification loads. On average, canopy transparency increased 10% by increasing 200 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) exceedance of critical loads. Since increased canopy transparency translates directly into reduced leaf area, and reduced leaf area translates into reduced photosynthate production, we can expect an overall forest productivity decline with increased exceedance of critical loads.

In Europe, some studies have attempted to investigate potential relationships between crown condition and acid deposition. For example, Nellemann and Fronger (1994) found that a high degree of Norway spruce (*Picea abies*) defoliation (over 30% defoliation or needle loss) in southern Norway was positively correlated to exceedance of S critical loads ( $r^2 = 0.94$ ,  $p < 0.01$ ). In contrast, the study of Solberg and Torseth (1997)

based on data from an extensive network of Norway spruce plots showed no clear relations between S deposition and crown-condition variables.

As shown by multiple regression, canopy transparency in ARNEWS plots was partially related to S and N deposition, to critical loads, to exceedance of critical loads, to insect damage, and to abiotic damage. Tree age, latitude, longitude, and altitude were not dominant factors. Other factors that could be important but are not addressed in this study are specific climate factors (droughts, late frosts, heavy winds, ozone, extended thaw periods in mid-winter). Damage to other parts of trees (stem wood, roots) can also influence canopy transparency. Studies from other areas have shown that canopy conditions are governed by climate, stand and site condition, with emphasis on drought and insect attacks (Brooks 1994, Burton *et al.* 1991).

Foliar discoloration, needle loss, susceptibility to secondary stress, growth decline, and tree mortality have been reported to be symptoms of acidification damage (Schulze and Freer-Smith 1991; Thomsen 1991; Sverdrup *et al.* 1994). Brandt *et al.* (2000) examined relationships between tree mortality rates of various tree species in ARNEWS plots and acid deposition as well as several other factors. They revealed that exceedance of critical loads significantly contributes to mortality, specifically for white spruce and white birch. In addition to exceedance of critical loads, other contributors to mortality were reported to include tree dominance and storm damage. In addition, defoliating insects and woody-tissue disease contributed to white birch mortality.

Concluding the results of this study, exceedance of critical loads appears to be a fairly consistent contributor to the observed canopy transparency at least for some tree species. In principle, this association is an expression of the potential effects of S and N

deposition on tree vitality through soil acidification (Roberts *et al.* 1989) as determined by:

- restrictions in nutrient availability due to increased leaching of base cations,
- possible effects caused by excessive N loading and related nutrient deficiencies,
- potential mobilization and enhanced concentrations of Al, and related impacts on root growth.

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## CHAPTER 11

### LONG-TERM SUSTAINABLE BIOMASS PRODUCTIVITY IN RELATION TO NUTRIENT MASS BALANCE FOR JACK PINE SITES

#### INTRODUCTION

This chapter focuses on the sustainability analysis of long-term nutrient input/output balances and biomass productivity of ARNEWS jack pine stands, by applying the ForSust model (Chapter 5).

Jack pine (*Pinus banksiana* Lamb.) is one of the principal forest species of northern and eastern Canada. this species occurs both in pure stands (normally originated by wildfire) and in mixture with other species, very often on deep acid stands of glacio-fluvial origin (Foster and Morrison 1976). Independent of other factors, jack pine growth increases with increasing moisture availability and, independent of moisture, with increasing fineness of texture and basic mineral content (Chrosciewicz 1963).

Since Jack pine stands are normally grow on nutrient-poor sites, harvesting these site can potentially cause nutrient depletion and can consequently lead to sustainability problems. Nutrient-related sustainability problems are likely to occur when the primary post-harvest replenishment of nutrients is less than what is required to re-establish the pre-harvest biomass production rates. Primary nutrient replenishment rates are due to atmospheric deposition (N, S, Ca, Mg, K, P, in wet and dry form) and soil weathering (Ca, Mg, K, P). Maintaining or achieving pre-harvest biomass production rates depends, therefore, on matching inputs with site-specific outputs due to soil leaching and due to harvesting (Hüttl and Scaaf 1995, Foster *et al.* 1997).

Previous studies have mainly focused on short-term nutrient cycling and related harvesting impacts. For example, Foster and Morrison (1989) reported that whole-tree harvesting of jack pine had about 50% higher removal of Ca, and > 100% higher removal of N, P, K, and Mg than stem-only harvesting.

In this chapter, the ForSust model (see Chapter 5) is applied to assess the long-term nutrient-based sustainability for 12 jack pine sites within ARNEWS plot network. The objectives of this chapter are:

1. to quantify the potential (sustainable) biomass production based on the nutrient limiting approach;
2. to determine the potential effect of acid deposition on potential biomass productivity;
3. to evaluate the sustainability of pre-harvest forest biomass growth rates through comparing simulated potential biomass productivity and estimated actual biomass productivity;
4. to analyze impacts of biomass removal intensities on base cation budget balances.

For the purpose of this study, the potential for sustainable forest biomass productivity is defined as the rate that results from primary nutrient inputs of N, P, K, Ca, and Mg at the stand level is evaluated in reference to the indefinite continuation of the existing forest type, and is done within the context of locally relevant natural and/or anthropogenic disturbance regime (e.g., acid deposition, recurring forest fire, various harvest levels).

## METHODS

The twelve jack pine stands used in this study are part of the Acid Rain National Early Warning System (ARNEWS) plot network (D'Eon and Power 1989; D'Eon *et al.* 1994). A brief description of each site is given in Table (11.1). These sites represent jack pine stands for various soil parent materials (ranging from acidic to calcareous), climatic conditions, and atmospheric deposition rates. Mean stand ages ranged from 23 to 65 years.

The following data was provided for the modeling purpose: mean annual temperature, mean July temperature, mean annual precipitation, and mean annual evapotranspiration. Actual evapotranspiration (AET, mm yr<sup>-1</sup>) was set as follows (see Chapter 6):

$$\text{AET} = [(16.298 * T_{\text{July}}) + (0.015 * T_{\text{July}} * \text{Ppt}) - (10.474 * \text{Latitude}) + (4.943 * \text{Longitude})] \quad r^2 = 0.97$$

Locations for weather, atmospheric deposition, and stream discharge information were chosen to match the jack pine site locations as closely as possible. Data sources included: the climate CD-ROM, Environment Canada (1994); the national stream water discharge rate catalogue (Environment Canada 1992); the CAPMoN data set for Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>, Na<sup>+</sup>, NH<sub>4</sub><sup>+</sup>, NO<sub>3</sub><sup>-</sup>, and SO<sub>4</sub><sup>2-</sup> ion concentrations in precipitation. Volume-weighted ion concentrations were multiplied with location-estimated precipitation volumes to generate the corresponding ion deposition rates.

**Table 11.1** Inventory data and estimated biomass growth for 12 jack pine sites, ARNEWS.

Plot Series	Latitude	Longitude	ARNEWS Plot No.	Prv	Stand Type <sup>a</sup>	Stand Density (trees ha <sup>-1</sup> )	Mean DBH (cm)	Basal Area (m <sup>2</sup> ha <sup>-1</sup> )	Mean Height (m)	Mean Age (year)	Estimated biomass growth		
											Actual	Normal	
												(Mg ha <sup>-1</sup> yr <sup>-1</sup> )	
1	46.74	65.13	204	NB	JP <sub>91</sub> BS <sub>9</sub>	1700	14.8	35.6	14.8	37	3.01	3.15	
2	47.12	74.11	310	QC	JP <sub>95</sub> BS <sub>5</sub>	1975	15.2	38.2	16.6	54	2.76	2.65	
3	47.07	74.06	311	QC	JP <sub>99</sub> BS <sub>1</sub>	1775	13.9	28.1	16.5	53	2.11	2.12	
4	47.28	83.21	509	ON	JP <sub>100</sub>	1550	14.1	25.9	15.6	44	2.19	2.81	
5	48.77	93.67	512	ON	JP <sub>91</sub> TA <sub>9</sub>	1700	16.6	38.9	18.1	48	3.41	3.40	
6	48.43	84.51	521	ON	JP <sub>100</sub>	1025	11.8	11.9	9.5	23	1.32	3.18	
7	49.83	93.14	522	ON	JP <sub>100</sub>	450	22.4	20.0	14.8	36	1.86	3.15	
8	47.66	80.00	524	ON	JP <sub>100</sub>	1875	14.3	31.2	11.3	35	2.66	2.35	
9	50.08	95.52	801	MAN	JP <sub>100</sub>	950	11.7	10.4	11.0	35	0.89	2.35	
10	54.33	97.92	803	MAN	JP <sub>100</sub>	1150	12.6	14.7	13.0	65	0.76	1.35	
11	56.44	100.59	804	MAN	JP <sub>100</sub>	1075	11.5	11.3	11.2	35	0.98	1.34	
12	54.40	109.00	805	SAS	JP <sub>100</sub>	1175	13.5	17.4	10.2	41	1.17	1.41	

P = jack pine; BS = black spruce; TA = trembling aspen.

Estimated actual biomass growth = stand density (10<sup>3</sup> ha<sup>-1</sup>) x 0.0384 x DBH<sup>1.98</sup> x mean height<sup>0.782</sup> / mean age (yr).

[DBH in cm, mean height in m; Ran et al. 1995].

Estimated normal biomass growth = wood density (0.58 Mg m<sup>-3</sup>) x normal volume (m<sup>3</sup> ha<sup>-1</sup>) / mean age (yr).

(wood density calculated from Maliondo et al. 1990).

(Normal volume obtained from Plonski 1981).

Fog deposition (see Chapter 6) was added to plot #1 (a New Brunswick site), where there was potential fog deposition. Dry deposition was set 15 and 25 percent of wet deposition for cations ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^+$ ,  $\text{Na}^+$ , and  $\text{NH}_4^+$ ) and anions ( $\text{SO}_4^{2-}$  and  $\text{NO}_3^-$ ), respectively (see Chapter 3).

Soil data involved specifications for soil depth (m), soil moisture content (%), clay content (%), organic matter content (%), and available N, Ca, Mg, and K. There was insufficient information about P in soils. Hence, all calculations were done without P.

The base cation ( $\text{BC} = \text{Ca} + \text{Mg} + \text{K}$ ) weathering rate was estimated for each site by parent material class (acidic, moderate, basic) and by soil texture class (percent clay content) and then modified for soil temperature, soil moisture, and rooting depth (de Vries 1991) as follows:

$$\text{BC}_{we} = \text{BC}_{we_0} \times f_{(T)} \times f_{(m)} \times \text{depth}$$

where

$\text{BC}_{we_0}$  = a generalized BC weathering rate ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) corresponding to the parent material class and soil texture, obtained from De Vries (1991).

$f_{(T)}$  = the correction factor for soil temperature,

$f_{(m)}$  = the correction factor for soil wetness, and

depth = depth of mineral soil layers at rooting zone (m).

The BC weathering rate was adjusted for soil temperature according to (Sverdrup, 1990):

$$f_{(T)} = \text{Exp} \left( \frac{A}{T_0} - \frac{A}{T} \right)$$

where

A = a pre-exponential factor (3600 °K),

T<sub>0</sub> = reference temperature (°K), and

T = the local temperature (°K).

Soil weathering rates for Ca, Mg, and K ( $X_{we}$ ) was calculated from:

$$X_{we} \text{ (X=Ca,Mg,K)} = BC_{we} \times \left( \frac{X_{avail}}{\sum X_{avail}} \right)$$

where

BC<sub>we</sub> = supply of base cation (Ca + Mg + K) per rooting layer (eq ha<sup>-1</sup> yr<sup>-1</sup>),

X<sub>avail</sub> = available rates of nutrient X at rooting mineral layers (eq ha<sup>-1</sup> yr<sup>-1</sup>).

Tree nutrient data for Ca, Mg, K, N, and P concentrations for above-ground tree components (foliage, branch, bark, and stem wood) were obtained from Canadian Forest Services (foliage data), and from a separate sampling and laboratory analysis of stem wood and bark. Concentrations of N, Ca, Mg, and K in branches were estimated through regression equations based on foliage, bark, branch, and stem wood concentrations in Maliondo *et al.* (1990). Details for vegetative nutrient data are provided in Chapter 8.

Field-estimated biomass productivity was calculated by setting:

$$\begin{aligned} & \textit{Field-estimated biomass productivity (Mg ha}^{-1} \textit{ yr}^{-1}) \\ & = \textit{total aboveground tree biomass (Mg ha}^{-1}) / \textit{age (yr)} \end{aligned}$$

where age is mean age of dominant and co-dominant trees. Total aboveground biomass was estimated in two ways:

1- actual biomass equation, as proposed by Ran *et al.* (1996) for jack pine:

$$\begin{aligned} & \textit{Total aboveground biomass (Mg ha}^{-1}) \\ & = (\textit{stand density} / 1000) \times 0.0384 \times \textit{mean DBH}_{(cm)}^{1.98} \times \textit{mean height}_{(m)}^{0.7819} \end{aligned}$$

2- normal biomass equation, by setting:

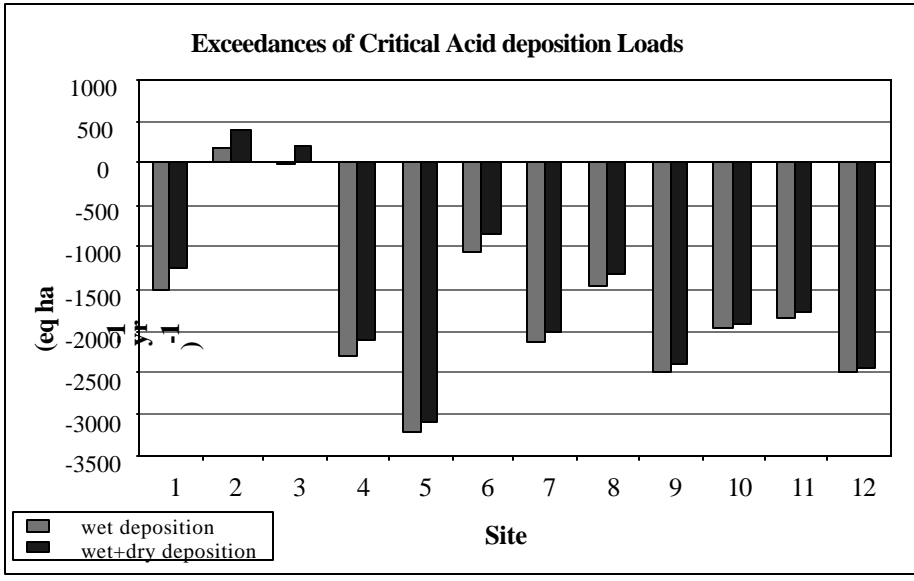
$$\begin{aligned} & \textit{Total aboveground biomass (Mg ha}^{-1}) \\ & = \textit{wood density}_{(dry, Mg m^{-3})} \times \textit{growth total volume}_{(m^3 ha^{-1})} \end{aligned}$$

where wood density set 0.58 (Mg m<sup>-3</sup>), as calculated from jack pine data (biomass and volume ratios) reported by Maliondo *et al.* (1990), and from total volume growth data obtained from a normal yield table (Plonski 1981), according to mean stand age and height.

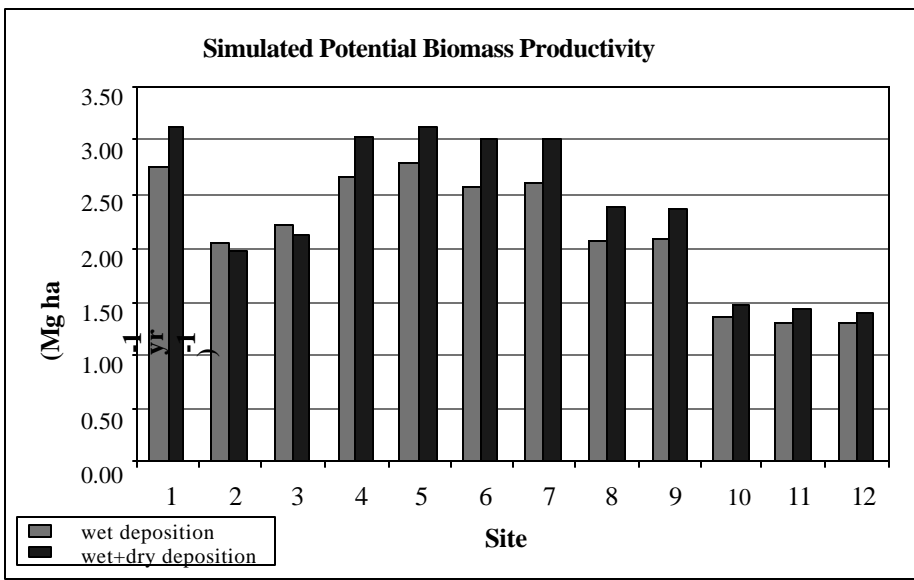
## RESULTS AND DISCUSSION

### Potential biomass productivity

Altogether, jack pine biomass productivity was simulated to be strongly dependent on base cation supply or on N supply, with the latter being the more growth-limiting factor than the former. Figure 11.1 and 11.2 show simulated exceedances of critical acid deposition loads and potential biomass productivity for 12 jack pine sites. Potential biomass productivity was calculated to be adversely affected by exceedances of critical loads in Sites 2 and 3 (Quebec sites). As exceedances of critical loads would further increase by addition of dry deposition, potential biomass productivity was calculated to decrease further. On other sites, where acid deposition rates were below the critical soil acidic loads, potential biomass productivity increased due to added N, Ca, Mg, and K dry deposition.



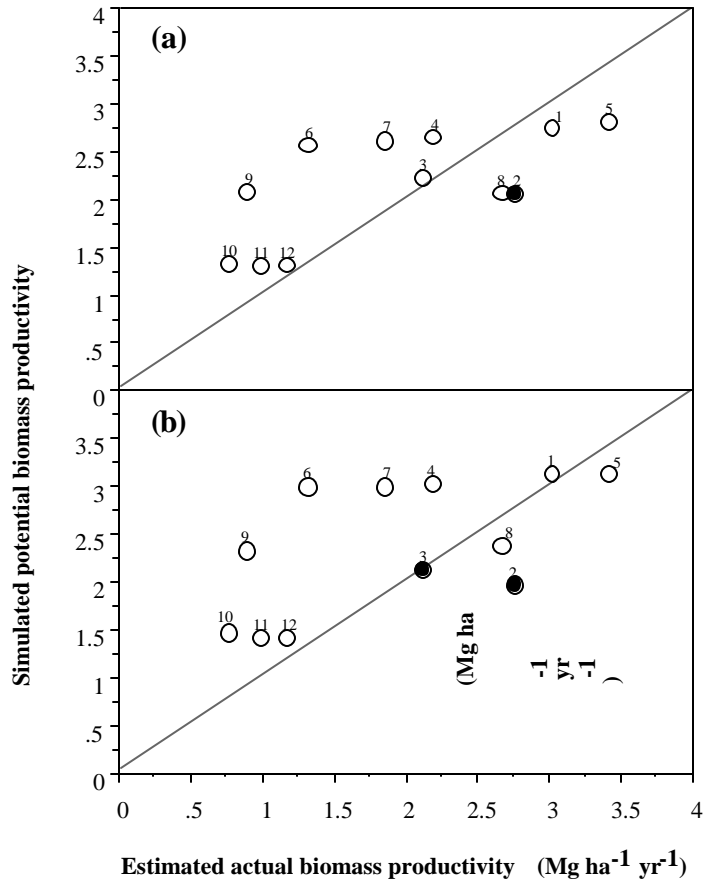
**Figure 11.1** Exceedances of critical acid deposition loads for 12 jack pine sites (ARNEWS).



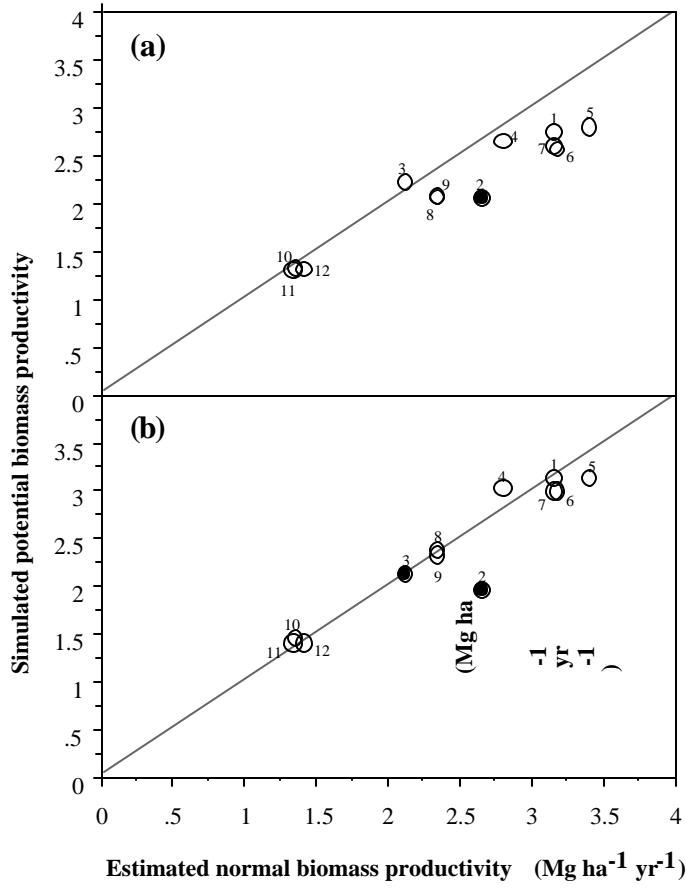
**Figure 11.2** Potential biomass productivity simulated by two atmospheric deposition scenarios for 12 jack pine sites (ARNEWS).

Shown in Figure 11.3 are values of simulated potential biomass productivity and estimated actual biomass productivity. These calculations are in general agreement with respect to overall magnitude, but differ substantially from each other in detail for most of plots. This disagreement is associated to uncertainties related to either predicted or estimated biomass productivity. The related uncertainties can be due to (i) errors in model structure, parameters, and assumptions, (ii) errors in input data. Here, model-related errors were already minimized by calibrating and validating the ForSust model (see Chapter 5). Uncertainties in values of actual biomass productivity can be related to: (1) applying an experimental biomass equation for all jack pine sites that might vary in terms of site condition and stand composition; (2) incorrect field data due to inventory errors; (3) stands may have been under-stocked for various reasons. Of these, (3) was most likely.

Adjusted values of actual biomass productivity to the normal stocking values (Plonski 1981) show a general agreement with simulated potential biomass productivity, especially when dry deposition was part of the model inputs (Fig. 11.4). After this adjustment, the highest disagreement between predicted and adjusted values was for Site 2, for which a critical load exceedance was already reached.



**Figure 11.3** Simulated potential biomass productivity (by two atmospheric deposition scenarios) plotted versus estimated actual biomass productivity for 12 jack pine sites (ARNEWS). Filled symbol indicates an exceedance of the critical acid deposition load.

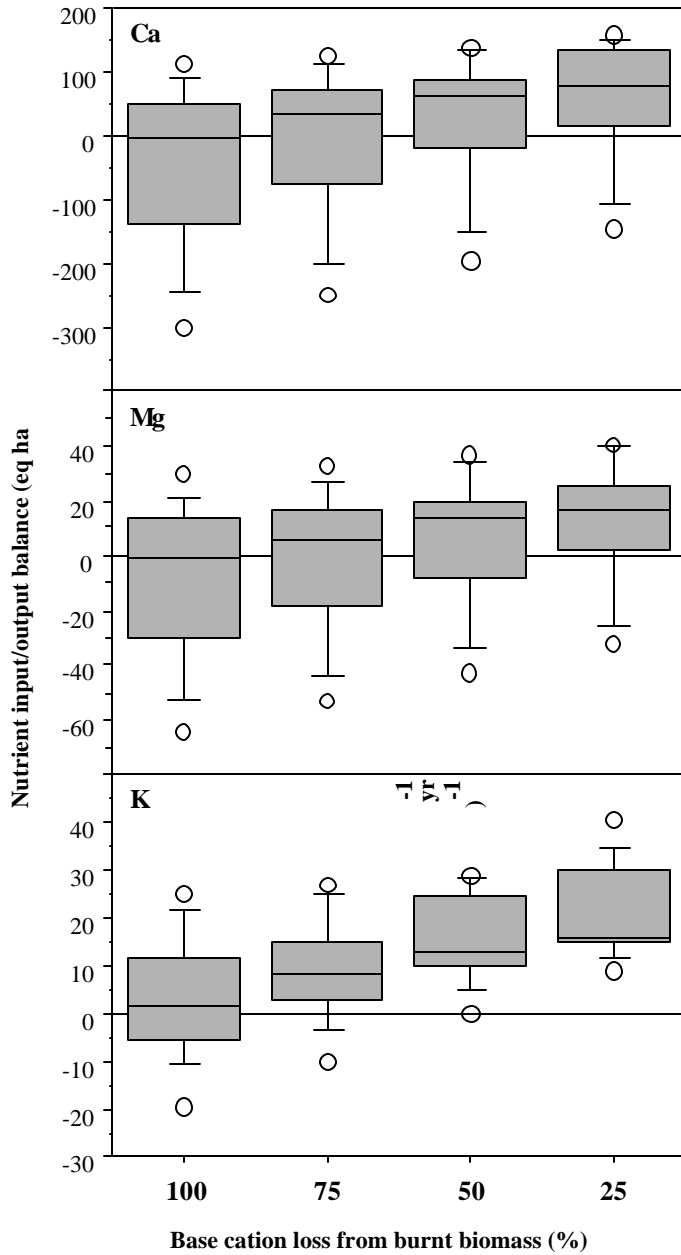


**Figure 11.4** Simulated potential biomass productivity (by two atmospheric deposition scenarios) plotted versus estimated normal biomass productivity for 12 jack pine sites (ARNEWS). Filled symbol indicates an exceedance of the critical acid deposition load.

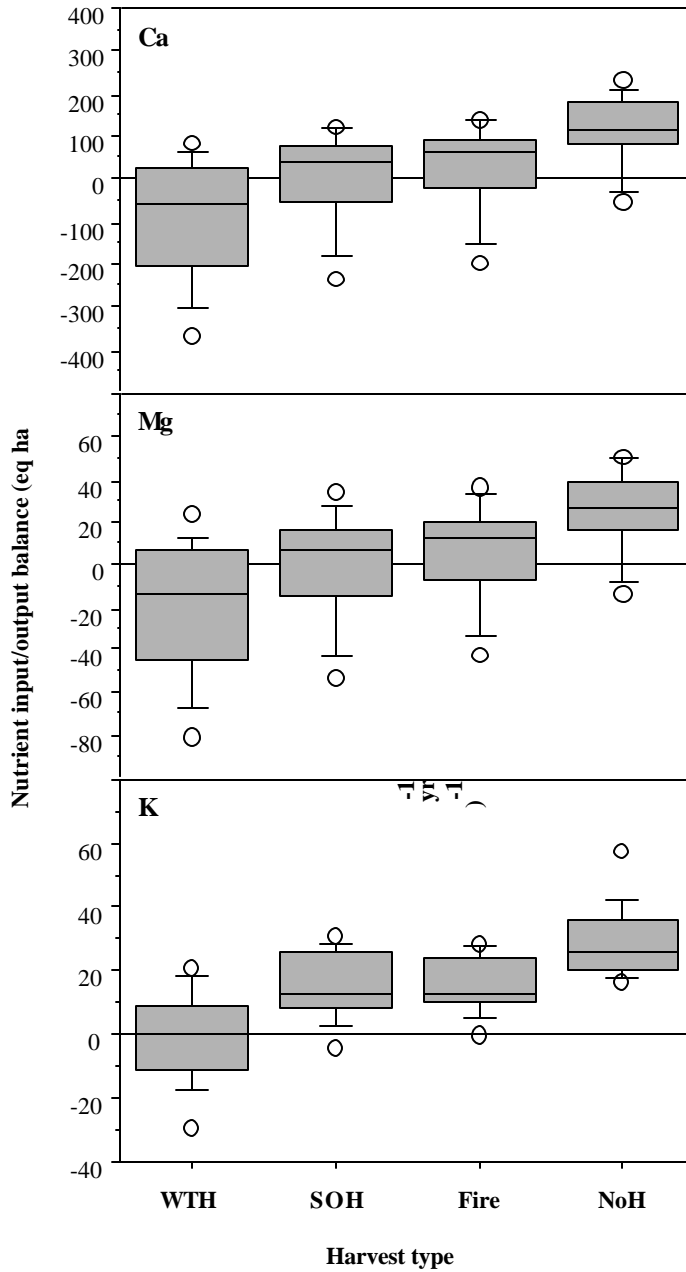
### Input/output imbalances

Figures 11.5 – 11.6 show the results of the Ca, Mg, K input-output balance (differences between supply and removal) under fire and various harvesting scenarios. Negative values imply that the combined outflow due to biomass removal and leaching is larger than inflow due to deposition and weathering. For negative values, long-term biomass productivity would not be sustainable at current rates. Negative balances between supply (inflow) and removal (outflow) were relatively large for both Ca and Mg, but less for K. Sverdrup and Rosen (1998) came to a similar conclusion by analyzing Ca, Mg, and K mass balances for Swedish forests.

Figure 11.5 shows that as fire intensity decreases, the degree of deficit and number of sites with Ca, Mg, and K deficit decreases. For example, high-intensity fire (100% nutrient loss) was calculated to cause relatively large deficits for Ca, Mg, and K at ~50% of the jack pine sites. Low-fire intensity (25% loss) was calculated to cause relatively low deficits for Ca and Mg at < 20% of the sites. Intermediate-fire intensities (50% - 75% losses) were calculated to cause Ca and Mg sustainability problems.



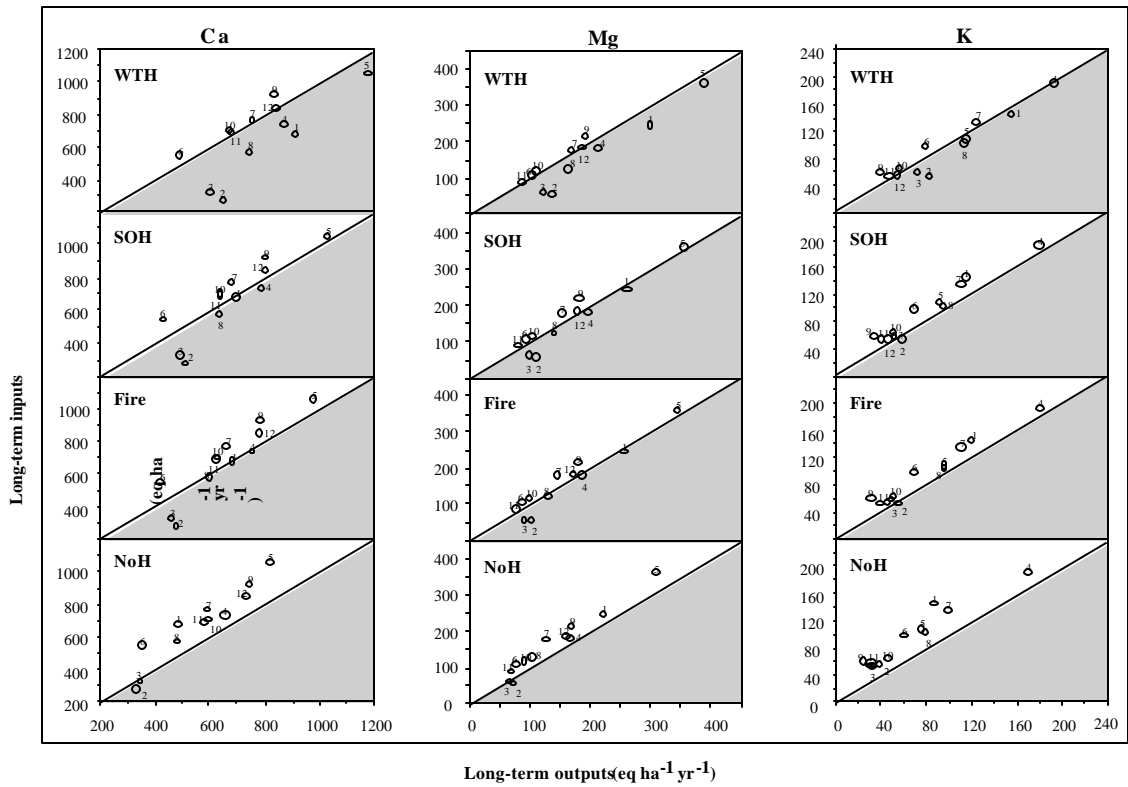
**Figure 11.5** Simulated long-term budget balances (input - output) of base cation values by fire intensity (percent loss of bases from burnt biomass) for 12 jack pine sites (ARNEWS).



**Figure 11.6** Simulated long-term budget balances (input - output) of base cation values by intensity of various biomass removal: whole-tree harvesting (WTH), stem-only harvesting (SOH), natural fire (50%), and no harvesting for 12 jack pine sites (ARNEWS).

With respect to harvesting, base cation imbalances were calculated to increase by increasing harvest intensity (Figure 11.6). Sites are mainly (but not all) sustainable at current nutritional status without any biomass removal (no harvesting). Conventional (stem-only) harvesting with 33% leaching loss from harvest residues would cause an increase in number of sites with base cation deficits (especially Ca and Mg). The impact of stem harvesting on base cation deficits is calculated to be close to that corresponding fire impacts (50% loss). Whole-tree (aboveground) harvesting was calculated to cause high rates of budget deficits for Ca and Mg at ~75% of the sites, and for K at ~50% of the sites. In general, Ca deficiencies after harvesting have been reported in several studies (e.g., Johnson *et al.* 1988a, 1988b; Federer *et al.* 1989).

A plot by plot analysis of base cation inputs / outputs can be seen in Figure 11.7. Here, the distance of plots from the 1:1 line (balance between nutrient supply and removal) implies intensity of deficit (shaded area) or surplus (clear area). These results indicate that whole-tree harvesting will likely cause serious Ca, Mg, and K deficits for most (but not all) jack pine sites. For those sites, estimated nutrient replenishment rates would likely not be able to compensate for whole-tree nutrient exporting at current production. As well, growth decline is expected to occur with the next stand rotation. Plots number 6, 7, 9, 10, and 11 are calculated to be sustainable in the rotation after whole-tree harvesting. This is due to the fact that these plots have a large surplus of base cations prior to harvesting (no harvest scenario). However, continuing whole-tree harvesting will likely cause base cation deficits in second rotation. Olsson *et al.* (1993) came to a similar conclusion by analyzing post whole-tree harvest effects on coniferous forest production and nutrient cycling in southern and northern Sweden.



**Figure 11.7** Long term base cation budget balance (supply versus removal) for 12 jack pine sites (ARNEWS) calculated by whole-tree harvesting (WTH), stem-only harvesting (SOH), fire disturbance (50% loss), and no harvesting (NoH). Plots located on shaded area have involved deficit for the corresponding nutrient.

Sustainability of current base cation supply was generally calculated to be feasible after stem-only harvesting with 33% leaching of bases in the slash, but not on all sites. For plots 1, 2, 3, 4, and 8, Ca and Mg exports were exceeded from their current supply (inputs), i.e., future growth may not be expected to be as high as what is currently observed. Almost the same pattern of base cation sustainability was calculated to occur when jack pine stands would be disturbed by fire with 50% base cation loss from burnt biomass, and 33% leaching loss from residual slash. A small deficit of Ca and Mg was calculated to occur in plots 2 and 3 without biomass removal (NoH). This deficit was related to increased leaching rates due to exceedances of critical acid deposition loads.

#### Management implication

From a management perspective, the results of this study show that serious attention needs to be given to Sites 2 and 3. Reducing acid deposition to the rates of critical loads would significantly reduce base cation leaching at these sites. Doing so may return those sites into the balance nutrient and consequently sustainable productivity.

On some sites, as can be seen in Figure 11.3, actual field-estimated biomass growth rates are lower than potentially simulated biomass growth rates. This suggests that those stands may require a treatment, e.g. a silviculture treatment, to improve biomass production.

Whole-tree harvesting should not be used in general because the harvesting method leads to large nutrient losses. Where acid deposition values do not exceed the critical soil acidification loads, stem harvesting seems to be the most feasible to a balance incoming with outgoing nutrient flows. However, debarked stem harvesting or partial

cutting can minimize the uncertainties for future forest growth, especially for sites with marginal nutrient supplies.

The results suggest that an assessment of long-term nutrient fluxes and reservoirs should be part of sustainable forest management planning and practices. Where the rate of uptake would be increased by high growth plantation and subsequent harvesting, potentially accessing deficits of essential nutrients must be compensated by fertilization.

## CONCLUSION

Potential aboveground biomass productivity on upland jack pine sites has been calculated from knowledge about nutrient deposition, weathering rates, nutrient availability in the soil, and nutrient concentrations in foliage, branches, bark, and wood. The simulated biomass productivity values are in general agreement with field-estimated values of normal biomass productivity. Potential rates of biomass productivity are susceptible to decline when acid deposition significantly exceeds the critical soil acidification loads, and where a deficit of base cation supply may occur even without biomass removal (e.g., Sites #2 and 3).

With whole-tree harvesting, base cation removal is larger than base cation supply for most of sites. In contrast, jack pine sites generally enable to replenish the rates of base cation removal after a recurring forest fire or stem-only harvesting. This comparison, therefore, can be used to identify those sites for which intensive harvesting levels may not be advisable in terms of maintaining current nutrient supply and biomass production levels.

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## CHAPTER 12

### LONG-TERM SUSTAINABLE BIOMASS PRODUCTIVITY IN RELATION TO NUTRIENT MASS BALANCE FOR SUGAR MAPLE FOREST SITES

#### INTRODUCTION

Sugar maple (*Acer saccharum* Marsh.) is a prominent component of the hardwood stands throughout most of the Great Lake – St. Lawrence, and the Acadian Forest Region (Hosie 1979). Recently, major concern has focussed on sugar maple due to apparent dieback growth decline as reported during the last two decades (McLaughlin *et al.* 1992). This dieback and decline has been attributed to various factors such as foliage nutrient deficiencies (K, Mg, Ca, and P), especially on poorly buffered soils in Ontario and southern Quebec (e.g., Bernier and Brazeau 1988a, 1988b, 1988c; Bernier *et al.* 1989; Ouimet *et al.* 1995), air pollution, particularly acid deposition (Foster 1989; Houston 1987; Ouimet *et al.* 1996), and natural disturbances likely drought and insect infestations (Payette *et al.* 1996).

The cause of nutrient deficiency in sugar maple is still unclear (Cote *et al.* 1995), acid deposition and related nutrient imbalances may be involved (Schlegel *et al.* 1992; Shortle and Bondietti 1992; Ulrich 1983). Acid enhances base cation leaching (Johnson *et al.* 1994; Likens *et al.* 1996). Base cation leaching from surface soils has been documented by Foster *et al.* (1989), and Morrison *et al.* (1992).

In this chapter, biomass growth potential and long-term input/output balances of base cations are evaluated in response to acid deposition and various biomass removal intensities. The specific objectives are:

1. To quantify the potential (sustainable) biomass production based on the nutrient limiting approach;
2. To determine the effect of acid deposition on potential biomass productivity;
3. To evaluate sustainability of forest biomass growth rates through comparing potential biomass productivity and estimated actual biomass productivity;
4. To analyze impacts of biomass removal intensities on the base cation input/output balances.

For the purpose of this study, the potential for sustainable forest biomass productivity is defined as the rate that results from primary nutrient inputs of N, P, K, Ca, and Mg at the stand level, in reference to the indefinite continuation of the existing forest type, and within the context of locally relevant natural and/or anthropogenic disturbance regimes (e.g., acid deposition, recurring forest fire, various harvest levels).

## METHODS

Ten sugar maple stands were selected from Acid Rain National Early Warning System (ARNEWS) plot network (D'Eon and Power 1989; D'Eon et al. 1994). A brief description of each site is given in Table (12.1). These sites represent sugar maple stands growing on soil parent materials ranging from acidic to calcareous, within various climatic conditions, and under various atmospheric deposition rates. Mean stand ages range from 54 to 120 years.

**Table 12.1** Inventory data and estimated mean annual biomass increment (MAI) for 10 sugar maple sites, ARNEWS.

Plot Series	Latitude	Longitude	ARNEWS Plot No.	Prv	Stand Type <sup>a</sup>	Stand Density (trees ha <sup>-1</sup> )	Mean DBH (cm)	Basal Area (m <sup>2</sup> ha <sup>-1</sup> )	Mean Height (m)	Mean Age (year)	Estimated biomass growth (Mg ha <sup>-1</sup> yr <sup>-1</sup> )	
											Actual	Normal
1	47.62	68.31	203	NB	SM <sub>86</sub> YB <sub>14</sub>	900	17.2	23.4	14.4	54	2.06	1.95
2	46.92	72.11	314	QC	SM <sub>86</sub> BE <sub>14</sub>	525	26.5	29.3	20.5	74	2.41	2.64
3	45.40	71.30	319	QC	SM <sub>96</sub> YB <sub>4</sub>	600	24.6	30.7	19.7	71	2.41	2.47
4	45.77	70.93	322	QC	SM <sub>83</sub> BE <sub>17</sub>	575	24.8	32.3	18.5	80	2.09	2.06
5	47.02	84.40	501	ON	SM <sub>83</sub> YB <sub>17</sub>	675	19.5	31.6	16.8	120	0.93	1.52
6	45.14	78.84	505	ON	SM <sub>79</sub> YB <sub>21</sub>	700	22.9	32.3	18.9	69	2.45	2.47
7	44.53	79.69	506	ON	SM <sub>100</sub>	825	18.7	25.3	17.0	60	2.07	2.41
8	43.85	81.60	507	ON	SM <sub>100</sub>	375	31.1	32.6	22.4	93	1.99	2.46
9	44.61	76.01	508	ON	SM <sub>100</sub>	375	33.0	35.9	23.0	103	2.06	2.41
10	42.36	81.85	525	ON	SM <sub>100</sub>	225	36.3	24.4	24.2	115	1.39	2.34

<sup>a</sup> Spp abbreviation = SM, sugar maple; YB, yellow birch; BE, American beech.

Estimated actual biomass growth =  $(\exp(-1.8329 + 2.3376 \cdot \ln(\text{DBH})) \cdot \text{stand density}) / \text{mean age} \cdot 1000$

[DBH in cm, mean age in yr; Ker 1980].

Estimated normal biomass growth = wood density (0.84 Mg m<sup>-3</sup>) x normal gross total volume (m<sup>3</sup> ha<sup>-1</sup>) / mean age (yr).

(wood density calculated from Maliondo et al. 1990).

(Normal volume obtained from Plonski 1981).

For the modeling purpose via ForSust model (Chapter 5), site-specific climatic and atmospheric data were manipulated by matching the sugar maple site locations to corresponding data sources as closely as possible. Climatic inputs: mean annual temperature (°C), mean July temperature (°C), and mean annual precipitation (mm yr<sup>-1</sup>) were obtained from the Canada climate CD-ROM (Environment Canada 1994). Actual evapotranspiration (AET, mm yr<sup>-1</sup>) was set as follows (see Chapter 6):

$$\begin{aligned} \text{AET} = & [(16.298 * T_{\text{July}}) + (0.015 * T_{\text{July}} * \text{Ppt}) \\ & - (10.474 * \text{Latitude}) + (4.943 * \text{Longitude})] \quad r^2 = 0.97 \end{aligned}$$

where AET = actual evapotranspiration (mm yr<sup>-1</sup>), T<sub>July</sub> = mean July temperature (°C), and Ppt = mean annual precipitation (mm yr<sup>-1</sup>)

Atmospheric wet deposition rates for Ca, Mg, K, Na, NH<sub>4</sub>, NO<sub>3</sub>, and SO<sub>4</sub> were calculated by multiplying volume-weighted ion concentrations with location-estimated precipitation volumes. Wet deposition ion concentration data were obtained from Environment Canada (National Atmospheric Chemistry, NatChem). Fog deposition was added to plot #1 (a New Brunswick site), where fog deposition would be likely (see Chapter 8). Dry deposition was set 15 and 25 percent of wet deposition for cations (Ca<sup>2+</sup>, Mg<sup>2+</sup>, K<sup>+</sup>, Na<sup>+</sup>, and NH<sub>4</sub><sup>+</sup>) and anions (SO<sub>4</sub><sup>2-</sup> and NO<sub>3</sub><sup>-</sup>), respectively (see Chapter 3).

Soil data: depth (m), soil moisture content (%), clay content (%), organic matter content (%), and available N, Ca, Mg, and K. There was insufficient information about soil P. Hence, all calculations were done without P.

A base cation (BC = Ca + Mg + K) weathering rate was assigned for each site based on its parent material class (acidic, moderate, basic) and its soil texture class (percent clay content) and then modified for soil temperature, soil moisture, and rooting depth (de Vries 1991) as follows:

$$BC_{we} = BC_{we_0} \times f_{(T)} \times f_{(m)} \times \text{depth}$$

where

$BC_{we_0}$  = a generalized BC weathering rate ( $\text{eq ha}^{-1} \text{yr}^{-1} \text{m}^{-1}$ ) corresponding to the parent material class and soil texture, obtained from De Vries (1991).

$f_{(T)}$  = the correction factor for soil temperature,

$f_{(m)}$  = the correction factor for soil wetness, and

depth = depth of mineral soil layers at rooting zone (m).

The base weathering rates were corrected for the effect of temperature according to (Sverdrup, 1990):

$$f_{(T)} = \text{Exp} \left( \frac{A}{T_0} - \frac{A}{T} \right)$$

where

A = a pre-exponential factor (3600 °K),

$T_0$  = reference temperature (°K), and

T = the local temperature (°K).

Then, BC weathering rates were corrected by a soil moisture factor that was assigned for each site. Once the lumped BC weathering rates were determined, soil weathering rates for Ca, Mg, and K ( $X_{we}$ ) were calculated from:

$$X_{we} \text{ (X=Ca,Mg,K)} = BC_{we} \times \left( \frac{X_{avail}}{\sum X_{avail}} \right)$$

where

$BC_{we}$  = supply of base cation (Ca + Mg + K) by weathering at rooting mineral layers

(eq ha<sup>-1</sup> yr<sup>-1</sup>),

$X_{avail}$  = available rates of nutrient X at rooting mineral layers (eq ha<sup>-1</sup> yr<sup>-1</sup>).

Tree nutrient data included Ca, Mg, K, N, and P concentrations for aboveground tree components (foliage, branch, bark, and stem wood). These were obtained from the Canadian Forest Service (foliage data), and a special wood and bark sampling efforts followed by laboratory analyses. Concentrations of N, Ca, Mg, and K in branches were estimated through regression equations that were generated from foliage, bark, branch, and stem wood concentrations in Maliondo *et al.* (1990). Details for vegetative nutrient data are provided in Chapter 8.

Mean annual biomass productivity was calculated by setting:

$$\begin{aligned} & \text{Mean annual biomass productivity (Mg ha}^{-1} \text{ yr}^{-1}) \\ & = \text{total above-ground biomass (Mg ha}^{-1}) / \text{age (yr)} \end{aligned}$$

where age is mean age of dominant and co-dominant trees. Total aboveground tree biomass was estimated for actual (on-site) and normal stocking as follows:

(i) actual above-ground biomass was estimated from inventory data and the biomass equation proposed by Ker (1980).

$$\begin{aligned} & \textit{Actual aboveground biomass} \text{ (Mg ha-1)} \\ & = (\textit{stand density} \text{ (stem ha-1)}) \times (\textit{Exp}(-1.8329 + 2.3376 \times \textit{LN}(\textit{mean DBH}_{(cm)})))/1000 \end{aligned}$$

(ii)- normal aboveground biomass was estimated by setting:

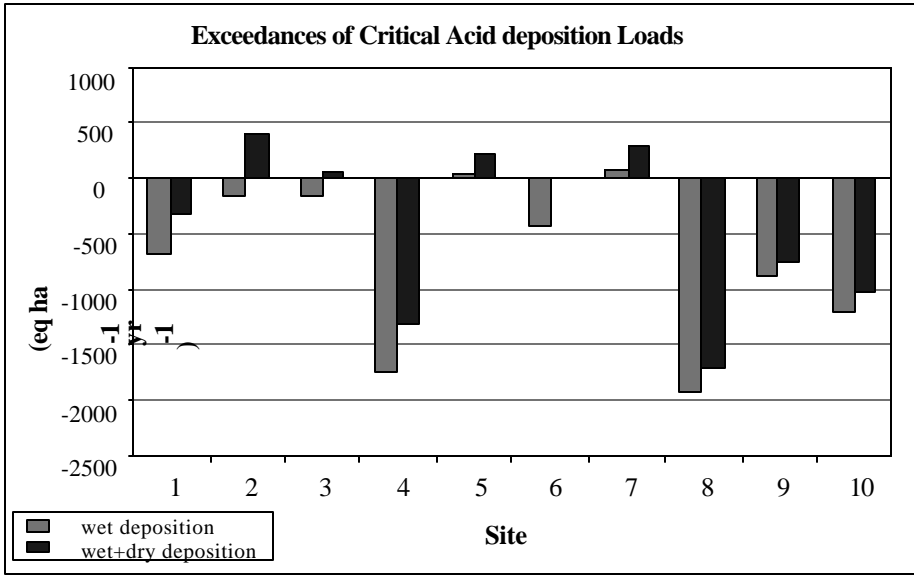
$$\begin{aligned} & \textit{Total aboveground biomass} \text{ (Mg ha-1)} \\ & = \textit{wood density} \text{ (dry, Mg m-3)} \times \textit{normal gross total volume} \text{ (m}^3 \text{ ha-1)} \end{aligned}$$

where wood density is set 0.84 (Mg m<sup>-3</sup>), as calculated from biomass and volume data in Maliondo *et al.* (1990), and normal tree volume is obtained from normal yield table (Plonski 1981) according to field data of mean stand age and mean tree height.

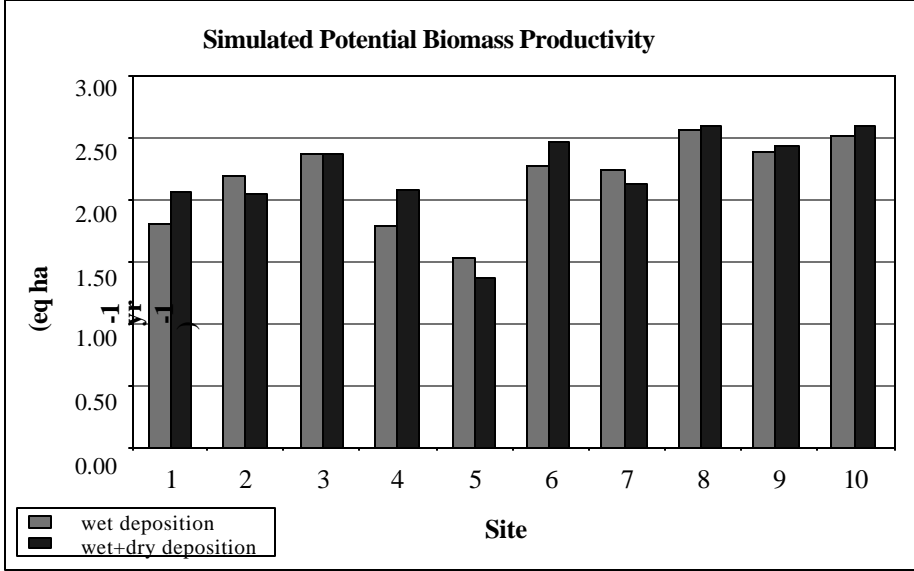
## RESULTS AND DISCUSSION

### Potential biomass productivity

Figures 12.1 and 12.2 show simulated exceedances of critical acid deposition loads and potential biomass productivity for 10 sugar maple sites. Altogether, sugar maple biomass productivity was simulated to be strongly dependent on base cation supply (weathering + deposition). Potential biomass productivity was calculated to be adversely affected by exceedances of critical loads at sites 2 and 3 (Quebec sites) and also at sites 5 and 7 (Ontario sites). As calculated, including of dry deposition would further increase exceedances of critical loads. In other sites, where acid deposition rates were below the critical soil acidic loads, potential biomass productivity was calculated to increase due to added N, Ca, Mg, and K dry deposition. These results imply that the adverse effects of acid deposition on growth of sugar maple may depend on the rates of acid deposition and on the acid buffering potential (critical loads) of the site.



**Figure 12.1** Exceedances of critical acid deposition loads for 10 sugar maple sites (ARNEWS).

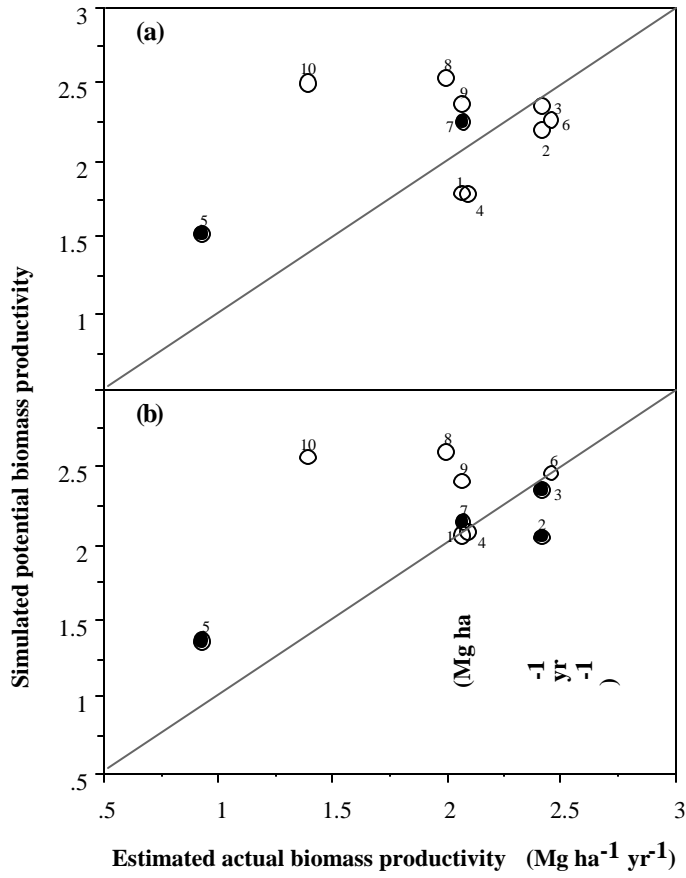


**Figure 12.2** Potential biomass productivity simulated by two atmospheric deposition scenarios for 10 sugar maple (ARNEWS).

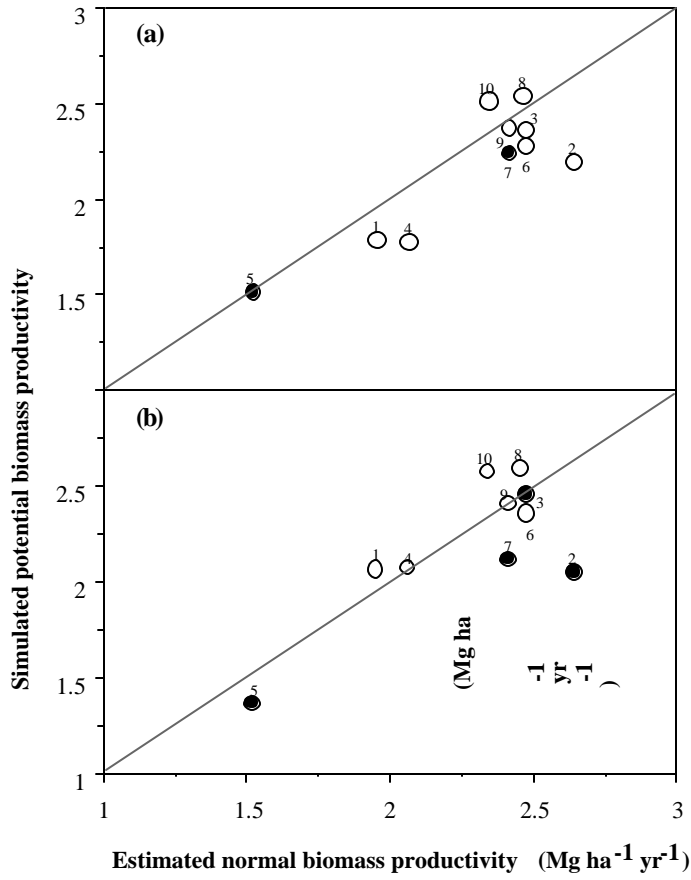
Shown in Figure 12.3 are values of simulated potential biomass productivity and estimated actual biomass productivity. These calculations are in general agreement with respect to overall magnitude, but differ substantially from each other in detail for most plots. For example, stands may have been under-stocked. Hence, actual biomass growth is less than potential biomass growth. To solve this problem, field-estimated biomass production was adjusted to full stocking based on the existing tree inventory data and on normal yield expectations (Plonski 1981). Adjusting actual biomass productivity values to normal stocking values (Fig. 12.4) generated considerable agreement, with dry deposition as part of the model input. Sites 2 and 7 remained exception, possibly because of critical load exceedances.

#### Nutrient input/output balances

Figures 12.5-12.6 show the results of the estimated Ca, Mg, and K input/output balances (differences between supply and removal) for fire and various harvesting scenarios. Negative values imply that the combined outflow due to biomass removal and leaching is larger than inflow due to deposition and weathering. For such negative values, long-term biomass productivity would not be sustainable at current rates. Negative balances between supply (inflow) and removal (outflow) were relatively large for both Ca and Mg, but less so for K. Sverdrup and Rosen (1998) came to a similar conclusion by analyzing Ca, Mg, and K mass balances for Swedish forests.



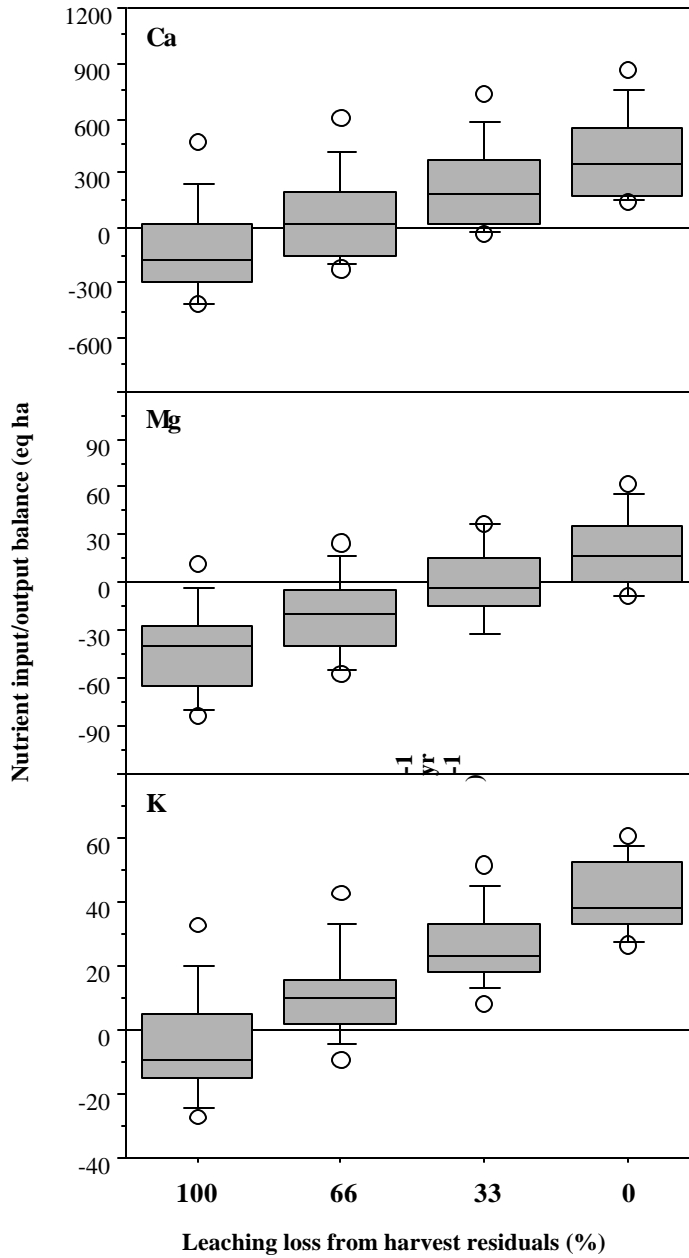
**Figure 12.3** Simulated potential biomass productivity (by two atmospheric deposition scenarios) plotted versus estimated actual biomass productivity for 10 sugar maple sites (ARNEWS). Filled symbol indicates an exceedance of the critical acid deposition load.



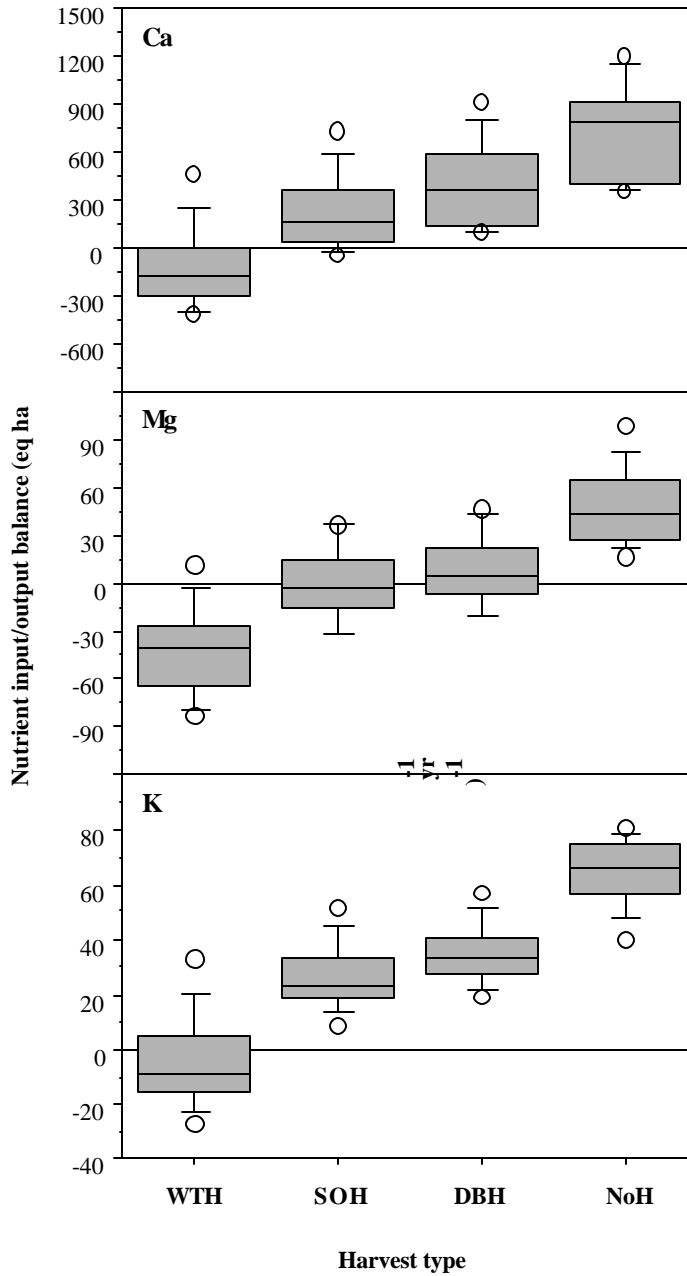
**Figure 12.4** Simulated potential biomass productivity (by two atmospheric deposition scenarios) plotted versus estimated normal biomass productivity for 10 sugar maple sites (ARNEWS). Filled symbol indicates an exceedance of the critical acid deposition load.

Figure 12.5 shows that as the intensity of leaching loss from harvest residues (stem-only harvesting) decreases, the degree of deficit and number of sites with Ca, Mg, and K deficit decreases. For example, high-intensity (100%) leaching losses were calculated to cause nutrient deficits up to 450, 90, and 30 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) for Ca, Mg, and K, respectively, at most sites. Intermediate-intensity (66%) leaching losses were calculated to cause: Ca deficits up to 250 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) at 40% of studied sites; Mg deficits up to 60 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) at 80% of studied sites; and light K deficits up to 10 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) at a few sites. In contrast, low-intensity (33%) leaching losses were calculated to cause Mg deficits up to 30 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) at 50% of the sites. If post-harvest leaching losses from residues were not to occur, then sugar maple biomass growth would essentially be sustainable at current rates.

With respect to harvesting, base cation deficits were calculated to increase with increased harvest intensity (Fig. 12.6). Whole-tree (aboveground) harvesting was calculated to cause nutrients deficits up to 450, 90, and 30 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) for Ca, Mg, and K, respectively, at most sites. In general, Ca deficiencies after harvesting have been reported (e.g., Johnson *et al.* 1988a, b, Federer *et al.* 1989). Conventional (stem-only) harvesting with 33% leaching losses from harvest residues were calculated to cause Mg deficits up to 30 ( $\text{eq ha}^{-1} \text{yr}^{-1}$ ) at 50% of the sites, as noted above. The degree of Mg deficit was lower (up to 20  $\text{eq ha}^{-1} \text{yr}^{-1}$ ) when stems were debarked on site.



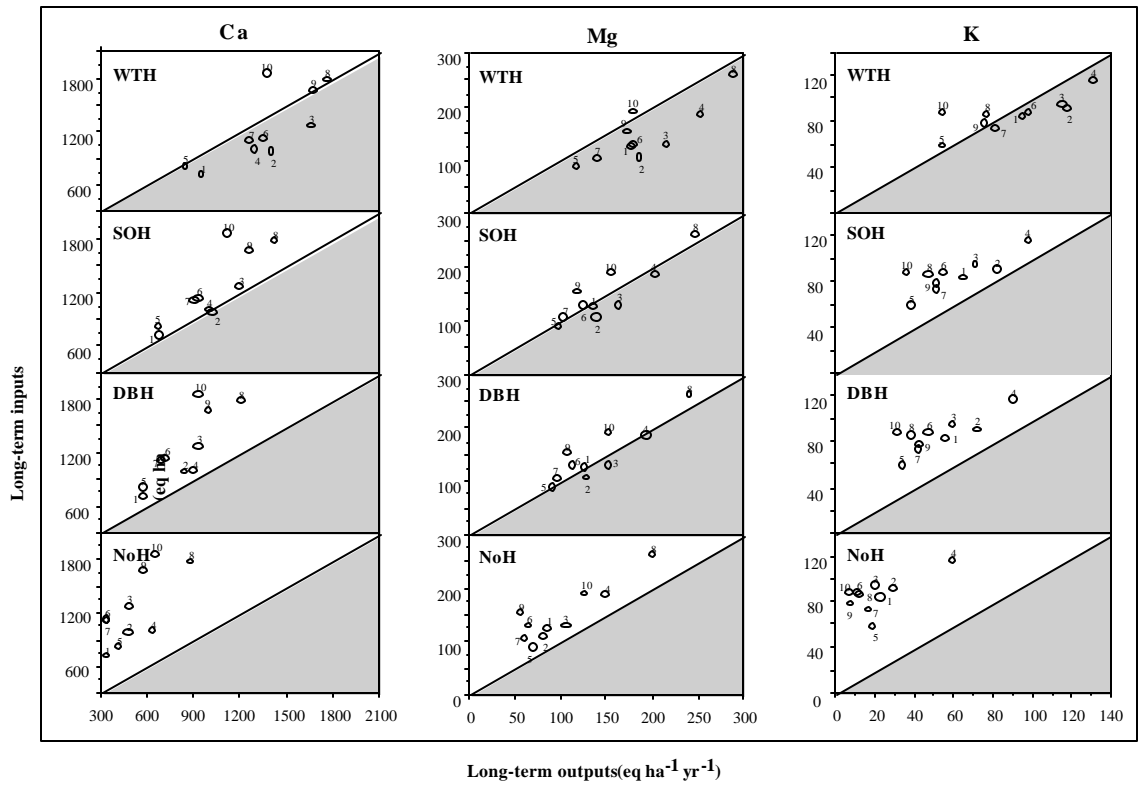
**Figure 12.5** Simulated long-term Ca, Mg, K input-output balance for stem-only harvesting allowing different leaching losses from the harvest residuals at 10 sugar maple sites (ARNEWS).



**Figure 12.6** Simulated long-term Ca, Mg, K input-output balance for different harvest intensities: whole-tree harvesting (WTH), stem-only harvesting (SOH), debarked stem harvesting (DBH), and no harvesting (NoH) at 10 sugar maple sites (ARNEWS).

A plot-by-plot display of the calculated base cation deficits is shown in Fig. 12.7. Here, the distance of individual plots from the 1:1 line indicates intensity of deficit (shaded area) or surplus (clear area). These plots suggest that whole-tree harvesting will likely cause serious Ca, Mg, and K deficits for most (but not all) sugar maple sites. For sites with such deficits, estimated nutrient replenishment rates would not likely be able to compensate for the whole-tree harvesting. As well, growth decline is expected to occur within the next stand rotation. Sustainability of current base cation supply was generally calculated to be feasible after stem-only and debarked harvesting with 33% leaching of bases in the slash, except at plots 2 and 3 (Quebec sites) where Mg deficits would likely occur.

It can be concluded that the ARNEWS sugar maple sites calculated to be vulnerable to on-site base cation depletion. Base cation depletion (particularly Mg and Ca) should increase with increased harvest intensity.



**Figure 12.7** Simulated long term Ca, Mg, K input-output balance (supply versus removal) for different harvest intensities: whole-tree harvesting (WTH), stem-only harvesting (SOH), debarked stem harvesting (DBH), and no harvesting (NoH) at 10 sugar maple sites (ARNEWS). Plots located on shaded area have involved deficit for the corresponding nutrient.

### Management implication

From management perspective, the results of this study show that Sites 2 and 3 need special and individual attention if these sites were to continue to grow at current biomass production rates. Reducing acid deposition may help to reduce base cation leaching.

On some sites (see in Fig. 12.3), actual biomass growth rates are lower than the potential biomass growth rate estimates. This suggests that those stands may require treatments, e.g., silviculture treatments, to realize the full growth potential.

Whole-tree harvesting should not be used in general because this practice leads to large nutrient losses. Where there is no acid deposition exceedance, stem-only harvesting should be sustainable at current biomass production rates. Debarking stems or partial cutting would further minimize nutrient stress for future forest growth, especially on sites where deficits are likely to develop.

### CONCLUSION

Potential aboveground biomass productivity were calculated for selected sugar maple sites from knowledge about nutrient deposition, soil weathering rates, nutrient availability in the soil, and nutrient concentrations in foliage, branches, bark, and wood. The simulated values of potential biomass productivity were in general agreement with the field-estimated values of normal biomass productivity. Potential rates of biomass productivity would likely drop when acid deposition exceeds the critical soil acidification

loads (e.g., sites 2, 5, and 3), due to progressive in base cation depletion (notably Ca and Mg).

With whole-tree harvesting, base cation removal is larger than base cation supply for most of sites. For stem-only harvesting, the overall input/output balance can be maintained. The procedures followed with this thesis can potentially be used to identify those sites where intensive-harvesting levels may not be advisable in terms of maintaining current nutrient supplies and biomass production.

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## CHAPTER 13

### SUMMARY, ORIGINAL CONTRIBUTION, AND RECOMMENDATIONS

#### THESIS SUMMARY

In this thesis, nutrient-related sustainability of the forest biomass productivity was investigated conceptually and practically. This was done by:

1. setting the conceptual definitions and conditions for sustainable forest biomass productivity (Chapter 2);
2. reviewing the published literatures corresponding to input/output requirement components for sustainable forest biomass production (Chapter 3);
3. re-evaluating a geochemical weathering sub-model (Chapter 4);
4. improving, refining, and validating the ForSust model (Chapter 5);
5. compiling and analyzing atmospheric data for ARNEWS plots (Chapter 6);
6. compiling and analyzing soil and geological data for ARNEWS plots (Chapter 7);
7. compiling and analyzing tree nutritional data for ARNEWS plots (Chapter 8);
8. calculating critical soil acidification loads and related exceedances for ARNEWS plots (Chapter 9);
9. analyzing empirical relationships between critical acid deposition exceedances and forest health (canopy transparency), (Chapter 10);
10. projecting the forest sustainability assumptions for select jack pine sites (subset of ARNEWS plots, Chapter 11);
11. projecting the forest sustainability assumptions for select sugar maple sites (subset of ARNEWS plots, Chapter 12);

## THESIS CONTRIBUTIONS

The main contribution of this thesis was to establish a framework that enables to a conceptual and quantitative analysis of sustainable of forest biomass production, on the basis of primary nutrient input/output balances at the stand level. This was done by:

1. defining the relevant concepts, including literature review and review of soil weathering calculations (Chapters 1, 2, 3 and 4)
2. establishing and documenting the Forest Sustainability model (ForSust, Chapter 5),
3. compiling and analyzing all pertinent ARNEWS plot field data (atmospheric deposition, various soil parameters, and nutrient concentration in tree foliage, branches, bark and stem wood (Chapters 6, 7, and 8),
4. calculating critical soil acidification loads for the ARNEWS plots (Chapter 9),
5. examining the implied empirical relationship between forest health (with % canopy transparency as health index) and soil acidification exceedance for ARNEWS plots, by way of multiple regression analysis (Chapter 10),
6. assessing the sustainability of current biomass production rates of select ARNEWS plots, specifically for jack pine and sugar maple plots (Chapters 11 and 12).

Sustainability framework presented by this study incorporates forest ecosystem criteria into forest management practices, which can be used as a guideline for forest managers in the context of “new forestry” (shifting from sustained yield to sustained ecosystem).

## RECOMMENDATION

The model presented in this thesis is essentially an open framework that can either be used as it is, or efforts can be made to further expand this model by examining and replacing some of its assumptions, one-by-one, and/or by adding to the computational framework by tracking changes over time as well. Here are a few suggestions.

1. Model input needs to include dry deposition as well as wet deposition, in order to make the estimates of actual nutrient supplies and related exceedance calculations more realistic. At this stage, nutrient supply and exceedance calculations are conservative, i.e., the overall rate of soil acidification due to calculated exceedance would be conservative.
2. The relationship between soil acidification and forest growth should be examined. For example, increased soil acidification means increased uptake competition from  $Al^{3+}$  ion species and in relation to uptake of nutrient cations such as  $Ca^{2+}$ ,  $Mg^{2+}$ ,  $K^+$  and  $NH_4^+$ . Using an Al / base cation criterion as target for allowable soil acidification is artificial in many respects, and is difficult to evaluate for field conditions. Specifying this ratio is artificial because of large lateral and vertical Al and base cation variability within the rooting space of mature forest trees. Allowing this ratio to potentially become one everywhere will have strong Al impacts on forest health and water quality due to region-wide Al mobilization. Setting this ratio at 0.1 sets an arbitrary safety margin. A less arbitrary criterion is to allow for a no further decrease of the soil base saturation level on account of acid deposition and forest harvesting. Doing so requires model adjustments and further model evaluations that would and should become the topic of another thesis.

3. There needs to be a critical evaluation of the PROFILE soil weathering calculation, and the various parameters used in this model (they are taken for laboratory level experiments of the leaching rates of purified minerals). As well, there needs to be a re-evaluation of the step from determining total oxide contents of soil minerals to the estimating of the soil mineralogy composition.
4. Phosphorus needs to be considered in addition to N, Ca, Mg and K for the uptake calculations: in many cases, P availability may be growth limiting.
5. Effects of air temperature and soil moisture should also be taken into calculation in addition to considering the nutrient limitation factors. For example, a direct relationship can be established by making the amount of forest biomass produced per hectare per year proportional to the amount of actual evapotranspiration, to account for site variations regarding soil moisture availability.
6. Limitations on nutrient uptake because of poor or restricted rooting due to heterogeneity in soil properties should also be considered, in case these limitations exceed the simple specification of soil rooting depth.
7. Comparisons should be made to evaluate forest sustainability in terms of dynamic models and in terms of the steady-state mass balance approach. With dynamic models, details regarding stand dynamics such as post-disturbance recovery and related nutrient losses can be explored. With dynamic extensions of the ForSust model, mass balance implied changes in nutrient pools within the soils can be tracked.
8. Further field examinations are needed to establish species-specific linkages between foliar nutrient concentrations and soil nutrient availability.

9. Model-predicted soil leaching rates should be critically examined with suitable long-term field data obtained by way of soil lysimetry.
10. Nitrogen processes such fixation, retention, immobilization, nitrification are highly variable in nature. It would be interesting to develop a methodology that can either confirm or reject the notion that long-term N status of forest soils is at a steady state, i.e., net long-term N sequestration of soils is zero, assuming that environmental and vegetation changes are effectively zero. Similarly, one could hypothesize that net long-term C sequestration of forest soils is zero, and that the C/N ratio of forest soils remains constant, overall. One difficulty of applying the long-term sustainability calculations to forest stands for which we only have current information is to assume that recent stand histories are not important, and that current levels in terms of, e.g., N storage is equivalent to that of the long-term steady-state expectations. However, N storage in forest stands changes over time, and with intensity of past disturbances, including past forest management practices. Hence, each application of the model needs to discern how to evaluate those steady-state input conditions that are in fact appropriate for any particular forest stand in terms of the desired long-term sustainability assessments.

**APPENDIX (A)**

**ARNEWS DATA**

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### Publications:

- Moayeri, M.H., Meng, F.R., Arp, P.A., and Foster, N.W. 2000. Evaluating critical soil acidification loads and exceedances for a deciduous forest at Turkey Lake, Ontario. *Ecosystems* (accepted).
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Contribution to the Published Reports:

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