Assessment of Forest Sensitivity to Nitrogen and Sulfur Deposition in Maine

Conference of New England Governors and Eastern Canadian Premiers
Forest Mapping Group

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Report Submitted to:

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Executive Summary

Acidic Deposition in Northeastern North America

Although sulfur emissions have decreased as a result of SO_2 control programs, projected emissions of acidifying sulfur and nitrogen compounds are expected to have continuing negative impacts on forests. These emissions present some of the most serious long-term threats to forest health and productivity in northeastern North America. Excess sulfur and nitrogen deposition may reduce the supply of nutrients available for plant growth. Nutrient depletion leads to increases in the susceptibility of forests to climate, pest and pathogen stress which results in reduced forest health, reduced timber yield, and eventual changes in forest species composition.

Forest Sensitivity Mapping Project

Conceived by the Conference of the New England Governors and Eastern Canadian Premiers (NEG/ECP), under the direction of its Committee on the Environment, their 1998 Acid Rain Action Plan called for the formation of a Forest Mapping Working Group to conduct a regional assessment of the sensitivity of northeastern North American forests to current and projected sulfur and nitrogen emissions levels. This group is charged with identifying specific forested areas most sensitive to continued sulfur and nitrogen deposition and estimating deposition rates required to maintain forest health and productivity. This report provides new information on the potential risks posed by sulfur and nitrogen air pollution to Maine's forest resources.

How Was This Assessment Done?

Evaluating forest sensitivity to acidic deposition requires information on: pollution loading to forest landscapes; the interaction of pollutants with forest canopies; plant nutrient requirements; and the ability of soils to buffer acid inputs and replenish nutrients lost due to acidification. Recent scientific advances in estimating each of these factors have made it feasible to produce maps of forest sensitivity to acid inputs from atmospheric nitrogen and sulfur. An integral part of this project was an open dialog with scientists, air resource specialists, foresters, and members of provincial, state and federal governments about data, methodology, and interpretation of results. Appropriate methods, models and mapping techniques and data requirements were developed during a pilot study of Vermont and Newfoundland which was completed in 2003 (NEG/ECP-FMG 2003). This report describes how these methods were applied to the analysis of forest sensitivity to acid deposition in Maine. Summaries of forest sensitivity in Vermont, New Hampshire, Massachusetts, Connecticut, and Rhode Island are also included in this report to provide spatial context for the Maine results.

Forest Sensitivity Assessment Metrics

Critical load of sulfur + nitrogen is the level of deposition below which no harmful ecological effects occur for a forest ecosystem.

Deposition index is the difference between the critical load and current deposition and is used to identify sensitive forest ecosystems.

The approach we have used to determine acceptable levels of deposition is an ecological assessment based on a steady-state[1], ecosystem mass balance (see Appendix) for nutrient cations (calcium, magnesium, and potassium). Two metrics – *critical load* and *deposition index* – express the result of this assessment. The *critical load* of sulfur + nitrogen is the level of deposition below which no harmful ecological effects occur for a forest ecosystem. The *deposition index* is the difference between the critical load and current deposition and is used to identify sensitive forest ecosystems. The magnitude of the deposition index indicates the severity of nutrient depletion caused by sulfur and nitrogen deposition. When exports of nutrient cations are greater than inputs to an ecosystem (a condition known as *cation depletion*), inadequate levels of nutrients may develop in both soils and plants (Fernandez et al. 2003). Inadequate nutrient levels have been linked to a wide range of forest health problems, reduced growth rates, and increased mortality (see Ouimet et al. 2001, Schaberg et al. 2001, Moore and Ouimet 2006, Schaberg et al. 2006).

Forest Sensitivity Findings

Sensitive forest areas were mapped under the current emissions levels of sulfur and nitrogen (Figure 1). In Maine, current levels of S + N deposition create the conditions for cation depletion in 36% of forests (2,430,000 ha).

Sensitive Forest Area Results:
- 36% of Maine forests
- 30% of Region's forests

Factors that increase forest sensitivity include: high levels of nitrogen and sulfur deposition, low mineral weathering rates, and tree species with high nutrient demands. High elevation forests and areas closest to emission sources experience the highest levels of nitrogen and sulfur deposition. Low mineral weathering rates occur in association with particular geologic and climatic factors. Requirements for soil nutrients vary according to the species currently growing in a forest because tree species have different nutrient requirements for health and growth. Sugar maple trees, for example, have a high demand for calcium.

Independent ecological indicators have been used to demonstrate that the assessment results are consistent with tree health observations from the region. Forests classified as sensitive by this approach exhibited crown health problems across Canada and in Vermont and lower tree growth in stands in Québec. In Massachusetts, soil base cation availability and tree height growth were correlated with the assessment deposition index.

Projected Sulfur and Nitrogen Emission Levels and Future Forests

Modifications in pollutant emissions and deposition can affect the area and distribution of sensitive forests. We estimate that a 50% reduction in combined sulfur and nitrogen deposition would remediate the nutrient depletion problem on 52% (1,254,700 ha) of the currently sensitive forest area in Maine. Conversely, if deposition increased again to the levels measured during 1984-1998 (a 36% increase in sulfur deposition), an additional (216,700 ha) of Maine's forests would be classified as sensitive.

Solutions Are Possible:
A 50% reduction in N & S deposition can reverse forest effects by 52% in Maine and by 55% in the New England Region.

[1] A steady-state (rather than dynamic) modeling approach was chosen for this assessment because the lower data requirements of a steady-state model made regional mapping of forest sensitivity feasible.

Forest Sensitivity Maps for Northeastern North America

Forest sensitivity mapping has been conducted for New England and eastern Canada. With this report, the analyses for Maine, Vermont, New Hampshire, Massachusetts, Connecticut, Rhode Island, the Eastern Canadian Provinces, and Ontario have been completed. Additional assessment components may be added (contingent on funding) to estimate the time span until forest health and productivity are compromised in sensitive areas.

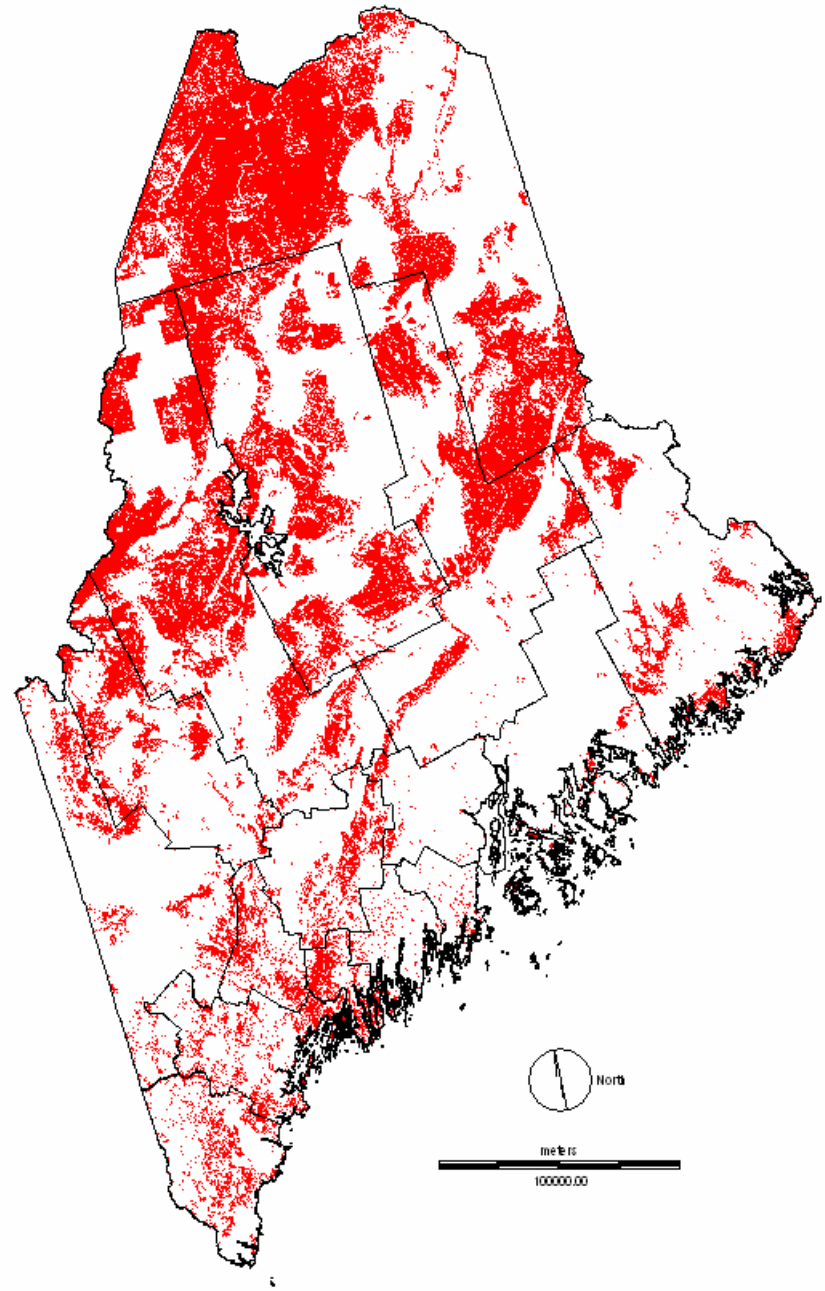

Figure 1. Forested Areas of Maine that are sensitive to the negative effects of combined atmospheric sulfur and nitrogen deposition. Red areas indicate current sulfur and nitrogen atmospheric deposition rates greater than the critical load. The critical load is influenced by land use. Land use information was obtained from Maine Office of GIS in 2003 and may not accurately represent present or future land use.

1. Background

Although sulfur emissions have decreased as a result of SO_2 control programs, projected emissions of both sulfur and nitrogen compounds are expected to have continuing negative impacts on forests, presenting some of the most serious long-term threats to forest health and productivity in northeastern North America. Anthropogenic sulfur and nitrogen deposition can cause excessive nutrient cation (calcium, magnesium, and potassium) leaching, reducing the supply of nutrient cations available for plant growth, a process called *cation depletion*. Inadequate nutrient supplies frequently lead to increased susceptibility to climate, pest and pathogen stress, and result in reduced forest health, reduced timber yield, and eventual changes in forest species composition (see Schaberg et al. 2001). For example, forest growth is significantly lower in Québec forest stands where current deposition exceeds the critical load (Ouimet et al. 2001).

The approach we have used to determine acceptable levels of deposition is an ecological assessment based on a steady-state, ecosystem mass balance for nutrient cations[2] (NEG/ECP-FMG, 2001). Two metrics – *critical load* and *deposition index* – express the results of this assessment. The *critical load* of sulfur + nitrogen is the level of deposition below which no harmful ecological effects occur in a forest ecosystem. The *deposition index*[3] is the difference between the critical load and current deposition and is used to identify sensitive forest ecosystems[4]. The deposition index provides information to policy makers about present and potential future status of forests and provides a useful general target for understanding what deposition levels might be deleterious to forest ecosystems.

This report provides a summary of the results of forest sensitivity mapping in Maine. The report describes how atmospheric deposition, forest nutrient demand associated with growth after harvest, and soil mineral weathering rates were estimated for the region and provides a discussion of the forest sensitivity assessment.

2. Atmospheric Deposition

Atmospheric deposition of sulfur (S), nitrogen (N), chloride (Cl), calcium (Ca), magnesium (Mg), sodium (Na), and potassium (K) representative of recent conditions was estimated for the study regions using the best available data for each state or province. We used a 5-year average (1999-2003) in order to provide some smoothing of year-to-year variations in climate and patterns of atmospheric transport. The period 1999-2003 was selected based on data availability and the timing of funding for atmospheric deposition modeling. Total deposition, including precipitation, cloud droplet interception, and dry deposition, was estimated for Maine (Figure 2). Total deposition to the region was modeled at a 30-m ground resolution using

[2] Base cations were evaluated individually to determine whether there was an insufficient supply of any individual base cation. If there was an inadequate supply of any one base cation, the ecosystem was considered sensitive to the current levels of S+N deposition. The critical load was, therefore, calculated based on the most limiting nutrient cation.
[3] Deposition index = Critical Load – Current Deposition.
[4] A positive value of the deposition index indicates the additional deposition that can be tolerated by the forest ecosystem. A negative value of the deposition index indicates the magnitude of deposition reduction required to eliminate all cation deficits (ensuring an adequate long-term nutrient supply).

atmospheric chemistry data from the US NADP, CASTNet, and NOAA-AirMon deposition monitoring networks and Ecosystems Research Group, Ltd.'s High-Resolution Deposition Model (Miller, 2000; NEG/ECP Forest Mapping Group, 2001, Miller 2001, Miller et al. 2005). Deposition monitoring sites are sparsely distributed in the region and factors affecting deposition rates are highly variable, particularly in mountainous regions (e.g. Miller et al., 1993), thus the deposition estimates carry some unquantifiable uncertainty.

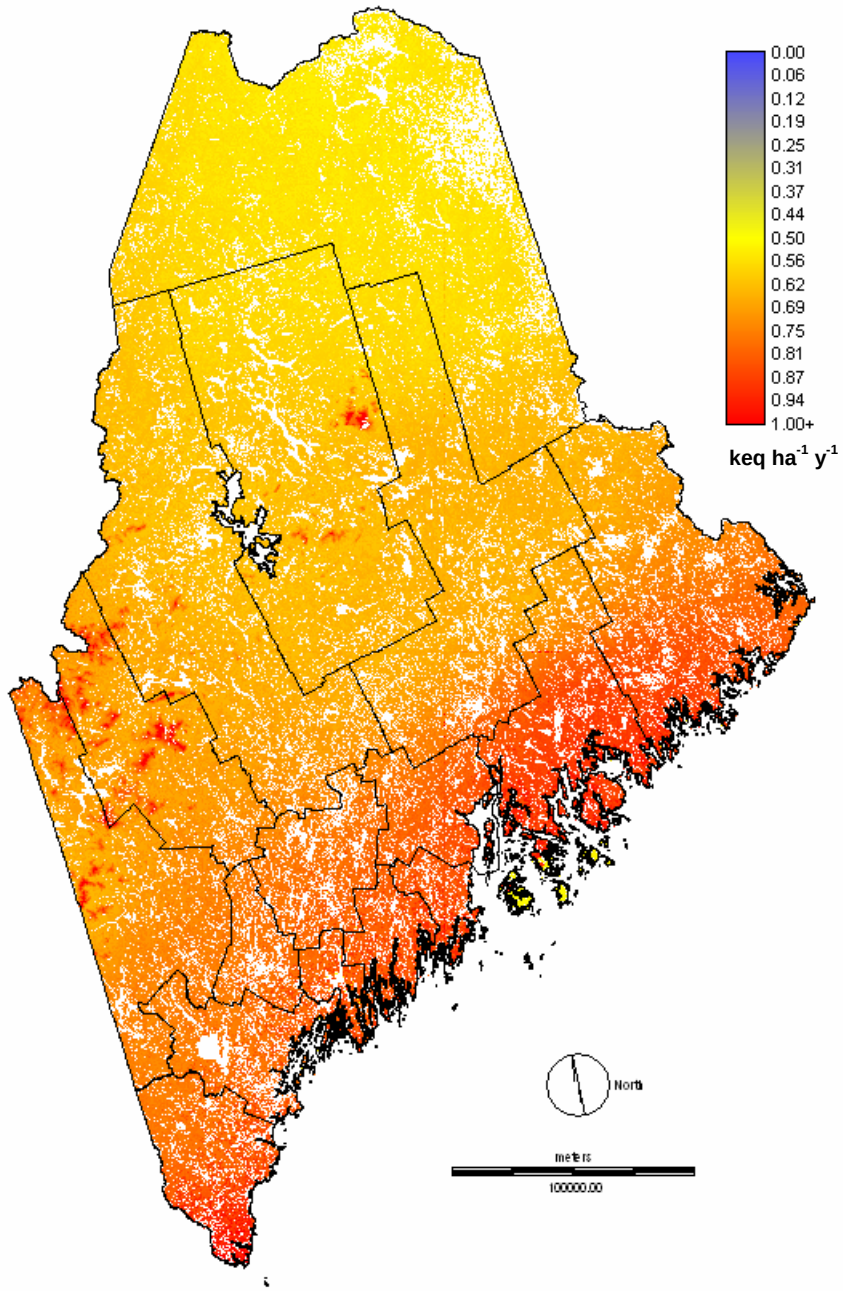

Figure 2. Average annual atmospheric deposition of sulfur and nitrogen to Maine (1999-2003). Total deposition (particle + SO_2 + precipitation + cloud water) expressed in terms of kilo-equivalents of charge per hectare per year; nitrogen deposition includes both ammonium + nitrate forms. This represents the total base-neutralizing and cation-leaching power of S and N atmospheric deposition.

3. Forest Nutrient Requirements

Forest nutrient demand was quantified as part of the mass balance (NEG/ECP-FMG, 2001). In undisturbed forests that have reached their climatic potential biomass there is no ***net*** annual requirement for nutrients because nutrients in dead trees are recycled into new forest growth. When forests are burned or harvested, part of the nutrient capital of the stand is removed with the ash or timber. The amount removed depends on the intensity of the fire or harvest and the parts of the tree that are removed. In order to calculate a critical load that will adequately protect the working forest, it is necessary to quantify the demand for nutrients required for growth after fire or harvest. Forest tree species and communities vary substantially in their inherent growth rates, demand for specific nutrients, fire recurrence interval, and level of forest management activity in different parts of the study regions. There are also variations in harvesting rates and practices on privately and publicly owned lands and in different jurisdictions. For these reasons, we characterized the rate of nutrient extraction associated with fire or harvest by forest type, land-ownership category[5], and location.

Distribution of Forest Types

A 30-m resolution spatial data layer describing the distribution of 9 major forest types was produced by determining the probability of forest type occurrence as a function of climate, and using the USEPA/USGS National Land Cover Data to discriminate between evergreen, deciduous or mixed forest types that could potentially occupy the same climatic conditions. A ground truth survey determined that the forest type map was 75% accurate overall, with higher accuracies for the dominant forest types.

Nutrient Exports Associated with Fire and Harvesting

The annual demand for nutrients required to regrow the biomass exported via harvesting was estimated from timber extraction rates and wood nutrient content. This information was generally available for the New England States from combinations of state and federal sources. In Maine, annual biomass extraction (average of 2.1% of live biomass per year during 1982-1995) was estimated by county, land-ownership category (public, private) and forest type (softwood, hardwood, mixed) from forest inventory data provided by the USDA Forest Service Forest Inventory and Analysis Program (FIA)[6]. FIA does not directly determine timber extraction rates, but rather how much of the inventory is removed between surveys. Therefore, when land is converted from forestry to residential and industrial uses, the trees harvested are counted as a removal from inventory. However, this type of removal does not reflect a long-term recurring harvest, because the harvested land is no longer in forest production. Due to these limits on information, rates of average recurring biomass extractions are probably over estimated in rapidly developing counties.

[5] The public and private land ownership categories refer to all lands where harvesting would be permissible. There was no harvesting modeled from private, state and federal reserves or wilderness areas. Because many private reserves allow harvesting, the critical load may be underestimated in these areas. Data source Maine GIS 2003.
[6] FIA data by county was obtained from the FIA web interface (http://fia.fs.fed.us/) in 2003.

The Maine state-wide public and conservation lands GIS data layers used to map harvesting rates by land-ownership categories may contain inconsistencies or omissions that influence the estimated spatial distribution of nutrient extraction rates. It was beyond the scope and budget of this project to address these problems with state-wide coverages. Due to limited information on management plans for specific land parcels, all forest type-ownership categories were assigned the county-wide average rate of timber extraction for that class. The effect of these assumptions is that nutrient losses due to recurring timber extraction may be overestimated on some lands and underestimated on others. Fire recurrence intervals average tens of thousands of years in New England; therefore, nutrient losses due to fire were ignored. Nutrient concentrations for each tree species for healthy foliage, branch, bark, and stem wood were compiled from the literature (e.g. Pardo et al. 2005).

4. Soil Mineral Weathering

The chemical breakdown of rock-forming minerals and their conversion to soil minerals, termed *soil mineral weathering*, is the primary means of replenishing the nutrients Ca, Mg and K that are lost from soils via acidic deposition-induced leaching and/or biomass removal. The landscape and geologic factors that control the rate of weathering are: 1) mineral assemblage, 2) climate, and 3) physical properties of the soil. Common minerals that may co-occur in the same rock or soil may have widely varying Ca, Mg, and K contents and inherent rates of chemical breakdown that could vary by up to 8 orders of magnitude. Thus, the proportion of easily weathered minerals (which are often the highest in Ca and Mg) exerts the dominant control on the overall soil weathering rate. The mineral assemblage is governed by the geologic history of a site including the bedrock mineralogy, transport of minerals to the site by water or glaciers, and the length of time the assemblage has been subject to weathering. Weathering rates increase with increasing temperature and water flux through a soil. The more mineral surface area that is exposed to water, the higher the weathering rate and this factor is governed by soil texture and climate. The depth to which roots can penetrate the soil (a function of both plant and soil characteristics) and the presence or absence of a fluctuating water table at this depth influence the volume of soil over which weathering is relevant to plant nutrition. Not surprisingly, the weathering rate is a highly localized parameter and very difficult to evaluate on a regional basis given the complexity of factors involved and data required. The estimation approach we have employed provides values of the average weathering rate for upland soils (NEG/ECP-FMG, 2001). Local weathering rates may depart substantially from the averages derived, but the estimates provide a rational basis for differentiating the ability of different areas within the region to replenish lost nutrients.

The mountainous landscape, range of climate, diverse bedrock geology, glacial history, and lack of any data that were scale-appropriate for the entire region presented a series of challenges to estimating mineral weathering rates. Through a combination of field studies, modeling, and literature review we: 1) identified the directions of glacial transport, 2) developed an empirical model describing the glacial transport of minerals in the <2mm size fraction, 3) developed a comprehensive state-wide database of bedrock mineralogy to be used with the glacial transport model, 4) developed landscape context sensitive empirical models of key climatic factors and soil characteristics, and 5) modified the PROFILE (Sverdrup and Warfvinge, 1993; see also NEG/ECP-FMG, 2001) weathering rate model to process this information stored in a geographic information system.

5. Forest Sensitivity Assessment

Forest areas sensitive to current levels of S + N deposition were identified by computing the steady-state mass balance for sources and sinks of acidity in a forest ecosystem (NEG/ECP-FMG, 2001). Briefly, we evaluated whether the base cations (Ca^{2+}, Mg^{2+}, K^{+}, Na^{+}) lost in conjunction with leaching of SO_4^{2-} and NO_3^{-} from the ecosystem and via harvesting or fire can be replaced on an ongoing basis by base cations released into the soil by mineral weathering reactions and those deposited from the atmosphere. Mass balances for Ca, Mg, K, and Na were evaluated individually. If the steady-state mass balance for any one individual cation is negative, the ecosystem is considered to be sensitive to the effects of current levels of S and N deposition over the long term. A negative mass-balance for Ca, Mg or K indicates a long-term nutrient limitation. If all cation mass balances are positive, this indicates the system has the capacity to tolerate additional S or N deposition.

Critical loads of sulfur plus nitrogen ranged widely in Maine (0 – 35 keq ha^{-1} y^{-1}) as a result of the diverse geology and climate of the state (Figure 3), encompassing the range observed for New England. Areas of Ca-rich rocks and soil materials scattered throughout the state support the highest critical loads in excess of 3 keq ha^{-1} y^{-1}. The lowest critical loads were found primarily in northern Maine where soils are developed in tills derived from base-poor rocks. Northern Maine has some of the highest timber extraction rates of the region which contributed to the lower estimated critical loads there.

Atmospheric deposition ranged widely across Maine during 1999-2003 (Figure 2). Sulfur deposition ranged between 3.2 and 12.2 (average 6.6) kg ha^{-1} y^{-1} and nitrogen deposition (ammonium + nitrate) ranged between 3.3 and 14.1 (average 6.1) kg ha^{-1} y^{-1}, producing an aggregate acidifying and nutrient leaching potential of 0.43 to 1.76 (average 0.68) keq ha^{-1} y^{-1}. The highest elevation areas received the highest S + N deposition due to orographically-enhanced precipitation and cloud water inputs. Deposition was also high in the southern and western parts of the state due to proximity to emission sources. Maine experienced the lowest sulfur and nitrogen deposition of the New England states (Table 1), receiving 50% less acidic deposition than Connecticut. As a result of low sulfur and nitrogen deposition rates, Maine also had the lowest leaching losses of nutrient base cations. However, nutrient demand for forest growth was highest in Maine due to high harvesting rates and forest species with moderate to high base cation requirements. The statewide average rate of nutrient cation resupply via mineral weathering occurring in Maine was similar to the other states in New England (Table 1).

Atmospheric Deposition of sulfur and nitrogen during 1999-2003 exceeded the critical load in 36% of Maine's forested area as compared to 30% of the forested area in the region. Critical loads were generally exceeded more in northern Maine where geologic conditions and high timber utilization resulted in the lowest critical loads (Figure 3).

Forest tree species occupy different portions of the landscape as a function of climate, soil conditions and land-use history. This distribution results in some types of forests being more severely impacted than others by the nutrient cation depletion caused by S + N deposition (Table 2). For example, critical loads are exceeded in 65% of Maine's central hardwoods forests, but in

just 20% of the state's northern hardwood forests. This discrepancy exists because the central hardwood forests (1.9% of total forest area) tend to occur more frequently on poor sandy soils. The northern hardwood forest (12% of forest area) species prefer somewhat richer sites. The critical load is exceeded in 53% of sugar-maple dominated northern hardwood forests (14% of forest area) because sugar-maple has a higher demand for calcium than other northern hardwoods.

Table 1. Comparison of statewide average metrics for elements of the forest sensitivity assessment.

State	S+N Deposition keq ha^{-1} y^{-1}	S+N Range[7]	Base Cation Leaching keq ha^{-1} y^{-1}	Base Cation Harvest keq ha^{-1} y^{-1}	Base Cation Weathering keq ha^{-1} y^{-1}	WX SD[8]	Critical Load keq ha^{-1} y^{-1}
Maine	0.68	1.32	0.53	0.41	2.44	1.87	1.28
New Hampshire	0.90	2.05	0.81	0.26	1.80	0.43	1.35
Vermont	1.01	2.10	0.89	0.23	2.50	2.09	1.60
Rhode Island	1.29	0.87	1.36	0.33	2.20	0.33	1.13
Massachusetts	1.31	1.51	1.40	0.21	2.27	1.24	1.77
Connecticut	1.35	1.18	1.45	0.17	2.68	1.15	2.29

Table 2. Forest sensitivity to acid deposition by forest type in Maine.

Forest Type	Percent of Forest Area	Impaired 1984-1988	Impaired 1999-2003
Red Spruce - Balsam Fir - Hemlock (Low Elevation)	24.4%	33.0%	30.2%
Mixed Evergreen and Northern Hardwoods	19.0%	31.2%	27.3%
Mixed Evergreen and Sugar Maple -Northern Hardwoods	14.8%	50.9%	48.3%
Sugar Maple - Northern Hardwoods	13.7%	55.4%	52.6%
Northern Hardwoods	11.8%	23.8%	20.1%
Balsam Fir - Red Spruce (High Elevation)	4.6%	44.8%	42.5%
Mixed Evergreen and Central Hardwoods	3.0%	38.9%	33.3%
White Pine - Hemlock - Red Spruce	2.7%	18.1%	13.3%
Central Hardwoods	1.9%	68.3%	65.4%
Birch - Northern Hardwoods	1.7%	65.1%	62.4%
Balsam Fir - Red Spruce - Birch	1.6%	47.1%	44.2%
White Pine - Central Hardwoods	0.4%	48.3%	43.2%
White Pine	0.4%	12.9%	8.0%
All Forest	100.0%	39.0%	35.8%

[7] The range between the spatial minimum and maximum estimated atmospheric deposition of sulfur and nitrogen for different locations in the state expressed in keq ha^{-1} y^{-1}.
[8] One standard deviation of the weathering rate estimated for different locations in the state expressed as keq ha^{-1} y^{-1} of base cations released.

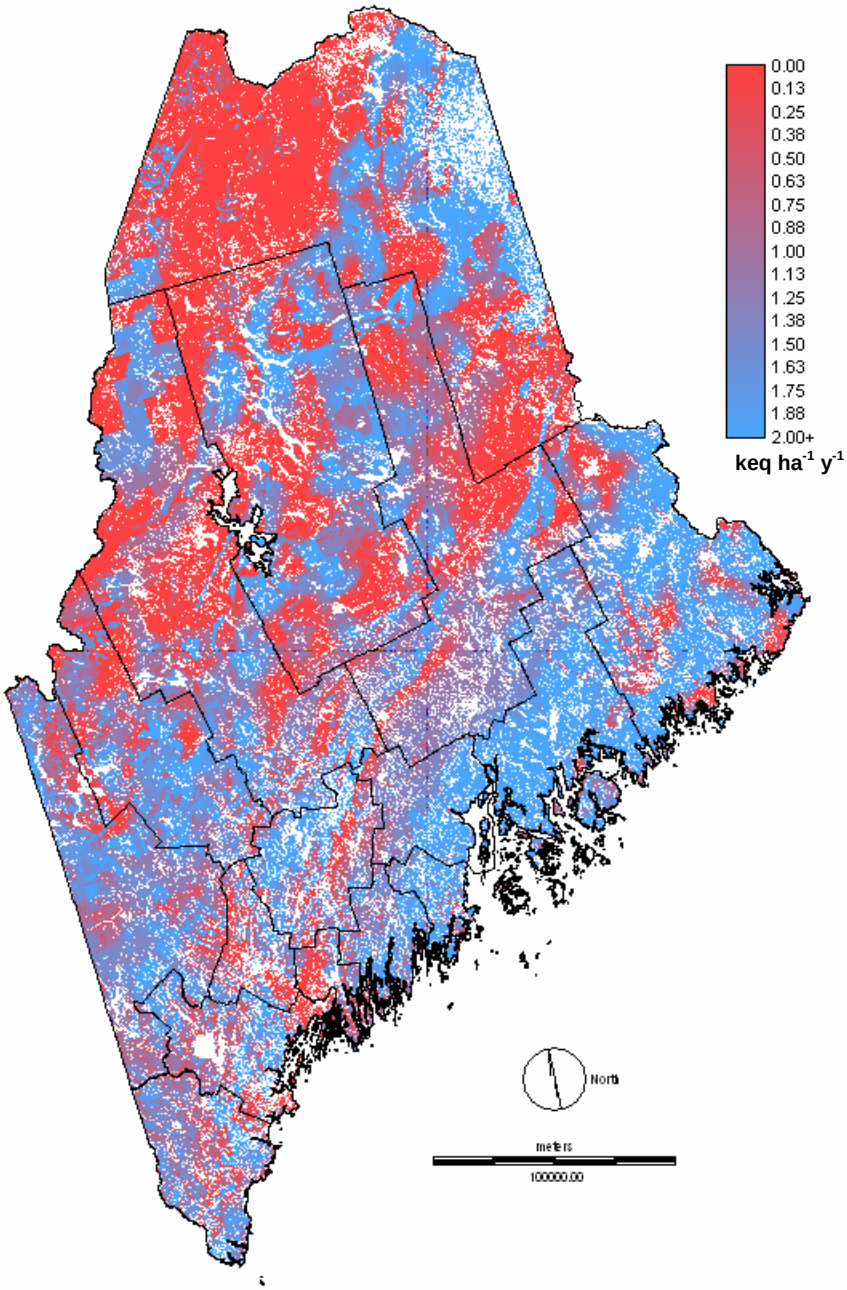

Figure 3. Critical load of sulfur + nitrogen deposition to upland forested areas of Maine. Sulfur + nitrogen atmospheric deposition rates higher than the critical load result in greater exports of nutrient cations (Ca^{2+}, Mg^{2+}, K^{+}) than inputs and eventual deterioration of soil fertility, forest health, and forest productivity. Critical loads are expressed in kilo-equivalents per hectare per year; nitrogen deposition includes both ammonium + nitrate forms. White areas are non-forested land or water. The critical load may be underestimated in some forest reserve areas (see page 8).

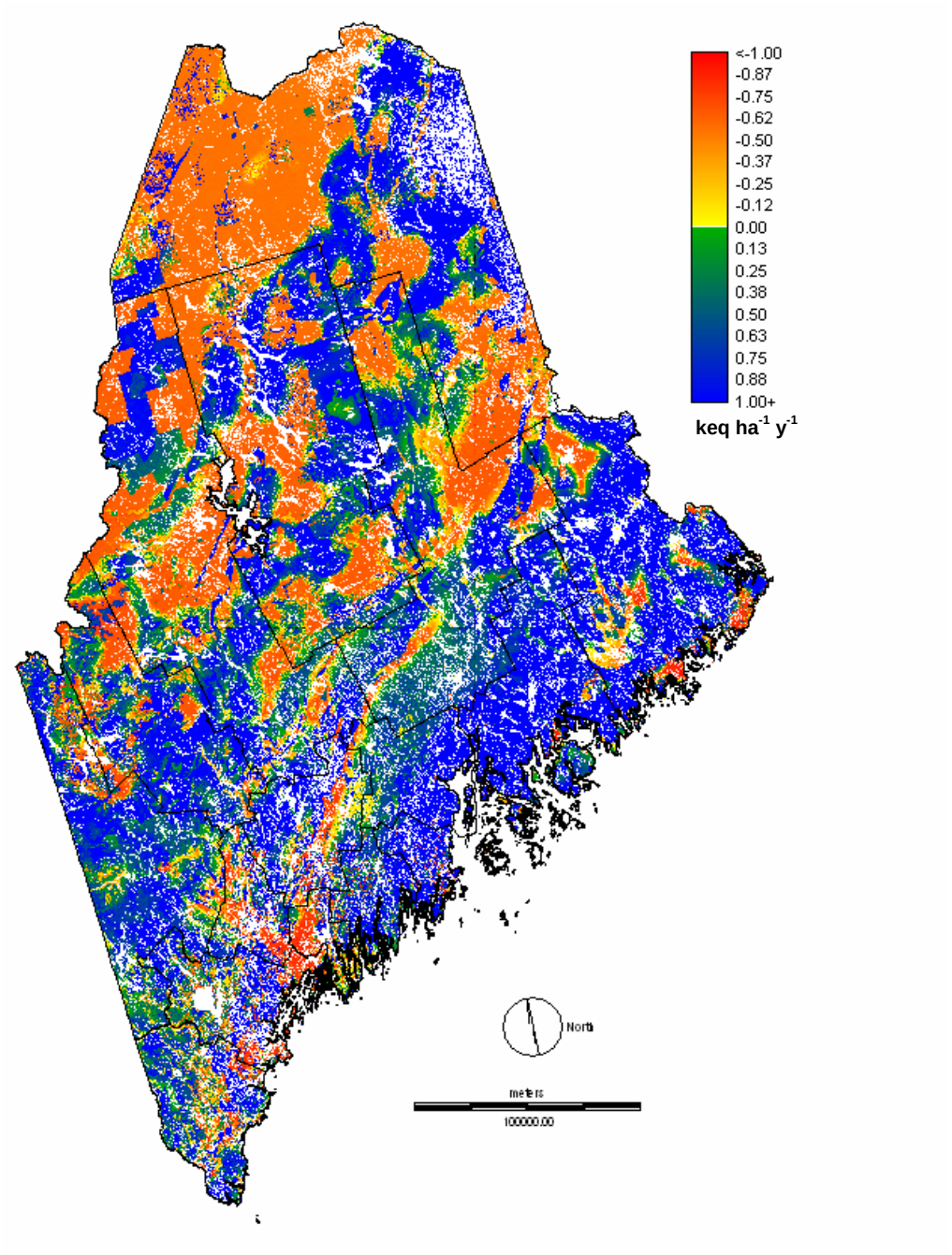

Figure 4. The deposition index for atmospheric sulfur and nitrogen deposition to Maine (1999-2003) with respect to forest ecosystem critical loads. Positive values of the deposition index reflect the capacity of a forest ecosystem to tolerate additional acidic deposition. Negative values of the index correspond to the reduction in S and N deposition required to eliminate present or deter the development of future nutrient limitations. Red-orange-yellow areas indicate current sulfur and nitrogen atmospheric deposition rates greater than the critical load. The deposition index is expressed in terms of kilo-equivalents of charge per hectare per year. Nitrogen deposition includes both ammonium + nitrate forms. White areas are non-forested land or water. The critical load may be underestimated in some forest reserve areas (see page 8).

The difference between the critical load and current atmospheric deposition (Figure 4) indicates the severity of the current nutrient imbalance (negative values) or the capacity to tolerate additional deposition (positive values). At sites where the deposition index is negative, the time required for the manifestation of declines in forest health and growth rate is governed, in part, by the size of the soil-exchangeable pool of nutrient cations. Exchangeable cations are those that are loosely retained in the soil, and can be thought of as the short-term supply of nutrients, while soil mineral weathering provides the long-term supply. If the exchangeable pool is large, the forest may be able to buffer a small nutrient input-output imbalance for tens to hundreds of years, delaying the onset of health and growth limitations. This buffering period allows time for the implementation of air-pollution emissions reductions.

The size of the exchangeable cation pool is governed by a variety of factors including soil depth, texture, organic matter content and the history of nutrient input-output imbalance or surplus. This is an extremely local condition. It is not currently possible to estimate the size of exchangeable nutrient pools on a regional basis. Observations from specific sites throughout the Northeast indicate exchangeable cation pools in the general range of 2 to 80 keq ha^{-1}. Frequently, sites with low weathering rates and a history of nutrient depletion will also have small exchangeable pools of cations. Thus, we can generally assume that where the critical load is exceeded today (negative deposition index), it was also exceeded in the recent past (1960's to present) and the buffering capacity of the exchange pools at such sites has already been somewhat diminished. At sites with a positive value of the deposition index, exchangeable cation reserves are increasing. Therefore, the deposition index also provides an indication of the time to the onset of nutrient shortages. These concepts are graphically illustrated in Figure 5, based on measurements from several plots in Massachusetts. Where the index is strongly negative, health problems and growth declines should be evident now (see section 6 below) or within decades. Where the index is only slightly negative, problems may take 100 to several hundred years to develop.

Information on the spatial distribution of atmospheric deposition rates and critical loads can be used to estimate the impact of changes in atmospheric deposition rates. We estimate that during the period 1984-1988[9] an additional 216,716 ha in Maine and 597,961 ha in the region beyond the currently sensitive areas experienced sulfur and nitrogen deposition greater than the critical load and conditions of cation depletion. It is clear that reductions in sulfur deposition achieved in recent years (NSTC, 1998) have prevented vast areas of forest from developing nutrient limitations to growth. If combined sulfur and nitrogen deposition were reduced further, to half of the 1999-2003 levels, the nutrient depletion problem could be remediated in 52% of Maine's and 55% of the region's forested area where critical loads are currently exceeded. The remaining sensitive areas would be forests growing on very poor sandy soils, areas with high timber extraction rates, and some balsam-fir, red-spruce, birch, and sugar maple forests growing at high elevations.

[9] This is as far back as we can reliably hindcast atmospheric deposition rates using the NADP observational network data. Nitrogen deposition has not changed significantly, and sulfur deposition has decreased by about 36% in New England from the 1984-1998 averaging period to the 1999-2003 averaging period.

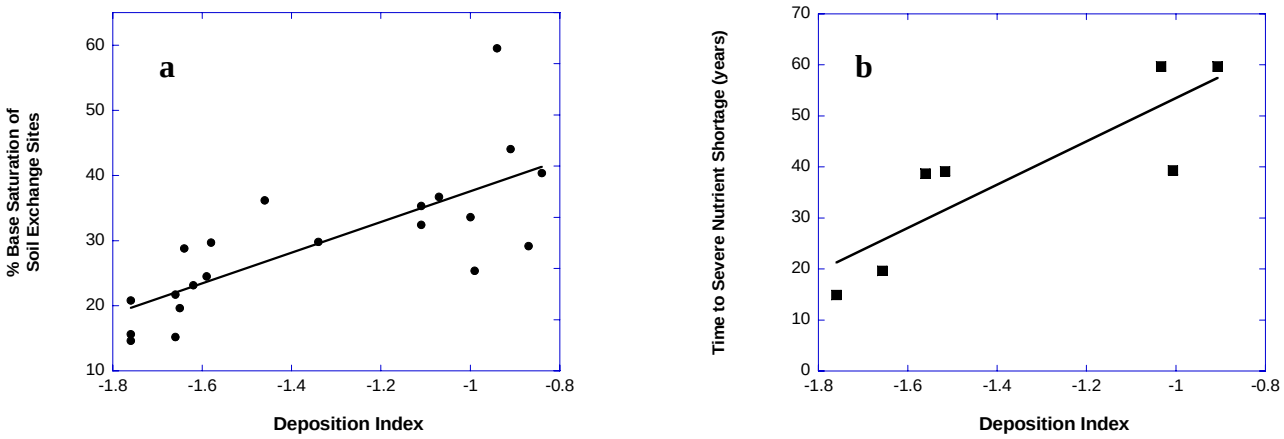

Figure 5. Comparison of soil measurements to modeled deposition index. a) Root-zone base cation availability expressed as base saturation (the percent of exchange sites populated by base cations) as a function of the modeled deposition index for soil pits in Massachusetts. Soil base saturation was correlated with the deposition index ($r^2 = 0.56$, $p < 0.0001$, $n = 21$ pits). **b) Index of time to severe nutrient shortage** as a function of modeled deposition index for plots in Massachusetts. A simple index of time to severe nutrient shortage (base cation pool / base cation depletion rate) was correlated with the modeled deposition index ($r^2 = 0.74$, $p = 0.013$, $n = 7$ plots).

6. Impacts on Forest Ecosystems when the Deposition Exceeds the Critical Load

Excess acidic deposition to forest ecosystems can adversely affect forest growth and productivity. Forest health consequences of elevated nitrogen and sulfur deposition have been documented in the literature and are variable depending on many site-related characteristics (e.g. Ouimet et al. 2001, Schaberg et al. 2001, Schaberg et al. 2006). In general, acidic deposition can cause soil and surface water acidification, increase soluble soil aluminum to toxic levels, and lead to a depletion of soil base cations, especially those required for plant growth and health (Ca, Mg and K). The symptoms of plant nutrient deficits manifest themselves at the cellular level, but also become visible as primary indicators of tree health. Notable tree health problems include increased susceptibility to winter injury, increased crown dieback, and increased proliferation of insect or disease activities. All of these may reduce forest growth and increase mortality. Over time, stand productivity may decrease, and the accumulation of health problems may lead to shifts in species composition and diversity (Schaberg et al. in press).

State and federal agencies as well as private researchers measure established indicators of ecosystem health throughout the study area. We have compared the modeled deposition index from this assessment with many of these independent indicators and demonstrated that our assessment is consistent with region-wide tree health data. For example, Figure 6 shows the

association between tree growth of hardwood and softwood stands in Québec, and the plot-specific deposition index. Similar associations were obtained between the regional assessment derived deposition index and crown dieback and canopy transparency (in Vermont) and soil chemistry (pH, Ca/Al, percent base saturation, Ca amount), canopy dieback, and tree growth (in Massachusetts). Work continues to compare the magnitude of the deposition index with ecological indicators in order to refine estimates of the time to development of forest health problems associated with different values of the deposition index. If forest health is already compromised in large areas with a negative deposition index, then there may be an urgent need to accelerate on-going regional, national and international air quality policy discussions.

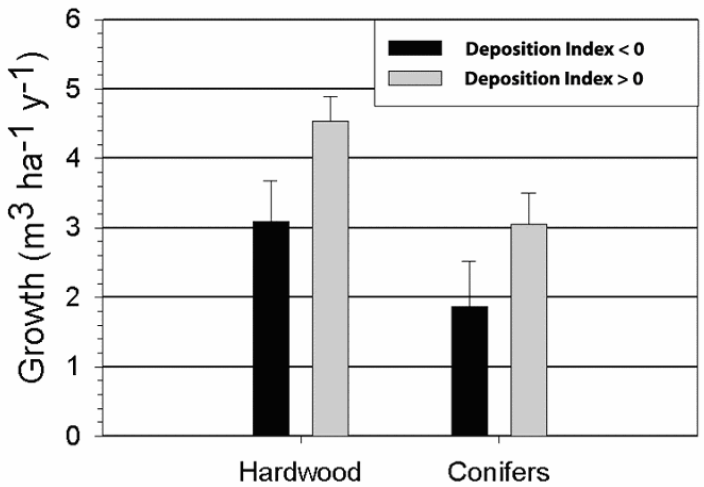

Figure 6. Growth rates of northern hardwood and boreal coniferous forest stands relative to the deposition index. Forest growth was significantly lower over a 19-year period at hardwood and softwood stands in Québec where current deposition levels exceed the critical load (deposition index < 0) than at sites where deposition is less than the critical load (deposition index > 0). Data presented are means adjusted for plot initial volume and stand age. Error bars represent standard errors of the adjusted means (Ouimet et al., 2001).

7. Stakeholder Consultations

The maps and products relating forest sensitivity to acidic deposition are new tools for eastern North American resource managers. While their primary purpose is to guide the development of policies focused on air pollution control, they also provide valuable information pertinent to state and provincial air quality standards, forest management, and future land-use assessments. Interactions with stakeholder groups have been an integral part of the development of this project, including data compilation, developing methods and processes of using multi-disciplinary data, and interpreting the results. During this project, regional experts from all areas of forest ecosystem science have been consulted, both individually and in groups. State, provincial, and federal government employees from air resources and forestry have also been involved at numerous stages of the project. Private forest landowners and forestry professionals

had additional opportunities for input. Outreach of results for the general public has been conducted.

8. Conclusions

The maps and analysis presented for Maine quantify the forest area that is potentially sensitive to current levels of acidic deposition inputs. These results will be useful both in the policy context for determining the need for further emissions reductions, and in the forest resource management context for identifying potentially sensitive forest areas. Identification of sensitive forest areas facilitates selection of sites where the consequences of future (elevated or reduced) deposition levels can most effectively be monitored.

Several factors may co-occur that cause the critical load to be exceeded by the current deposition: the mineral weathering rate may be low (and soils may be thin), the vegetation may have a high nutrient cation demand, and deposition levels may be high. Certain forest types (e.g. red spruce-balsam fir) more often have deposition levels which exceed the critical load because they tend to occur at high elevations, where deposition levels are higher, and on low weathering rate soils. Comparisons between ecological indicators of forest health and productivity and critical loads suggest that forest health is poorer and growth rates lower where critical loads are exceeded. A relatively small increase in deposition levels will increase sensitive forest area substantially, while reduction in deposition levels will reduce sensitive forest area.

Regional Context

The Forest Mapping Project has completed assessments for Maine, Connecticut, Rhode Island, Vermont, New Hampshire, and Massachusetts. Assessments have been published for the Eastern Canadian Provinces and Ontario (see Morrison et al. 2005, Ouimet et al. 2006). Summary documents describing the US regional results should be available in 2007.

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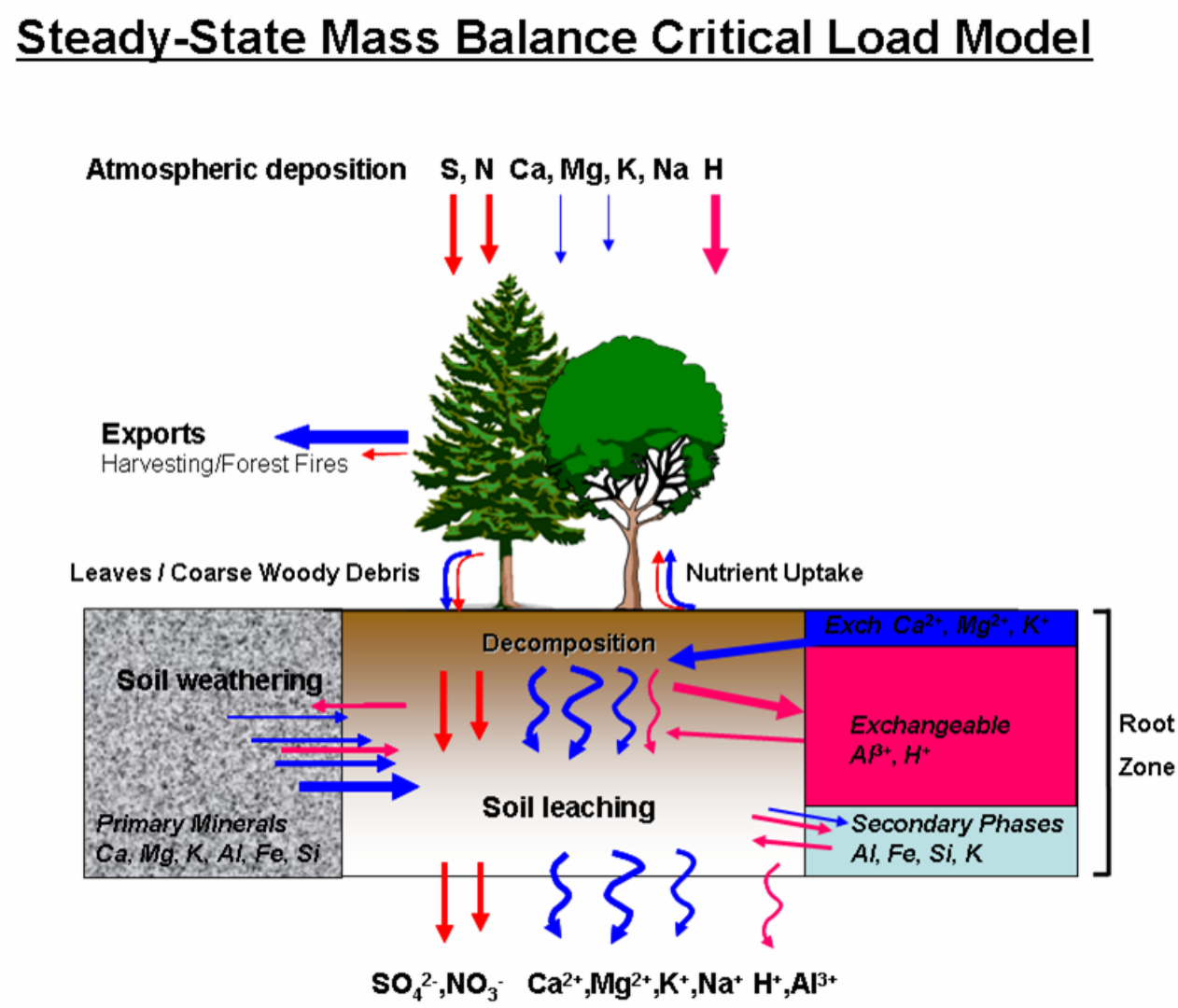
Appendix

Conceptual illustration of the steady-state mass balance model