

# Application of the SMART2 model to a forested catchment in Finland: comparison to the SMART model and effects of emission reduction scenarios

Johanna Ahonen, Katri Rankinen, Maria Holmberg, Sanna Syri and Martin Forsius

*Finnish Environment Institute, P.O. Box 140, FIN-00251 Helsinki, Finland*

Ahonen, J., Rankinen, K., Holmberg, M., Syri, S. & Forsius, M. 1998. Application of the SMART2 model to a forested catchment in Finland: comparison to the SMART model and effects of emission reduction scenarios. *Boreal Env. Res.* 3: 221–233. ISSN 1239-6095

SMART2, an extended version of the dynamic soil acidification model SMART, was tested with data from a forested catchment in Finland. The catchment belongs to the network of the Integrated Monitoring Programme and is situated in North Karelia (Eastern Finland). The aim of the study was to predict the long-term acidification of soil and runoff water at the study site under different scenarios of future sulphur and nitrogen deposition. Another objective was to study the model behaviour by comparing the results obtained with this extended version to those of SMART. Compared to previous SMART model calibrations, only the nitrate concentration in surface water was clearly closer to the mean observed value. For the specific purposes of the current study, the results did not indicate any great advantage in using the more complex model version SMART2. The scenario runs with both models showed a decline in soil base saturation and surface water pH in response to the rapid increase in acidifying deposition that started in the 1960s. The studied reduction scenarios will result in the cessation of environmental deterioration and the onset of recovery. However, the timing of the response depends on the stringency of the chosen abatement strategy.

## Introduction

Both steady-state and dynamic models were developed to predict soil, lake, stream, and ground water acidification (e.g., Cosby *et al.* 1985, De Vries

*et al.* 1989, Warfvinge and Sverdrup 1992). For the calculation of critical loads, i.e. quantitative estimate of exposure to one or more pollutants, below which significant harmful effects on specified sensitive elements of the environment do not oc-

cur according to present knowledge (Nilsson and Grennfelt 1988), and their exceedance, steady-state models are presently being extensively employed under the framework of the Convention on Long-range Transboundary Air Pollution (LRTAP) of the United Nation's Economic Commission for Europe (UN/ECE). These models do not take into account time-dependent processes or finite pools, which could buffer excess acidity for a limited time period. They can be applied with a limited amount of information and are suitable for mapping purposes on a regional scale (Posch *et al.* 1997).

Dynamic models are used to predict the gradual chemical response of a receptor to changing deposition and land use practices. Forest soils and lakes are the most common receptors investigated with respect to acidification. Various buffer and adsorption/desorption mechanisms are included in soil acidification models, and the changes in finite element pools over time are calculated. It is important to know the temporal development of acidification in order to determine the timing of necessary measures for emission control, and to assess the dynamic response of a potential critical load exceedance. Dynamic models have also been applied on a regional scale (e.g., Jenkins *et al.* 1990). For a regional application, however, derivation of the large number of required input data presupposes the use of transfer functions or statistical distributions. This kind of data derivation increases the uncertainty. For the sake of developing and assessing emission reduction policies, site-specific applications employing measured data are useful complements to regional or national mapping and modelling efforts.

A dynamic acidification model called SMART (Simulation Model for Acidification's Regional Trends) (Posch *et al.* 1993) has been shown to successfully reproduce, at the catchment scale, the current soil and water chemistry (Posch *et al.* 1989, Kämäri *et al.* 1995). However, it is likely that the nitrogen deposition input to forests, which varies as a result of emission control strategies, affects the dynamics and availability of nitrogen in the soil. This has an influence on soil acidification through the charge balance of the soil solution and thus on the other abiotic site factors, such as pH. SMART, as a simple acidification model, does not take into account all the nitrogen processes in the soil, and the processes included in SMART do not depend on the abiotic site factors. How-

ever, changes in abiotic site factors may pose a threat to the structure and functioning of ecosystems (Kros *et al.* 1995). Therefore, in order to improve the acidification studies with respect to nitrogen and to study the ecosystem feedback to emission strategies, a model with a more detailed description of the nitrogen cycle related to abiotic site factors was needed.

The first aim of the present study was to test the SMART2 model (Kros *et al.* 1995), an extended version of SMART that includes a description of the nutrient cycle, and to calibrate it to a data set derived from a monitored forested catchment in Finland. The calibration results were then compared to the results obtained from a previously conducted SMART calibration in the same catchment (Forsius *et al.* 1998). The site belongs to the Integrated Monitoring Programme (ICP IM), which is one of the monitoring activities established under the UN/ECE LRTAP Convention for the Assessment and Monitoring of Air Pollution Effects (Kleemola and Forsius 1996). In the comparison, emphasis was placed on studying the effects of more detailed descriptions of nitrogen processes. The second aim of the study was to use the calibrated model to predict the long-term acidification of the soil and the surface water in the Hietajärvi catchment (Eastern Finland), given different scenarios of future sulphur and nitrogen deposition. The site-specific deposition scenarios used in the study were based on agreed and suggested measures for emission reductions of the UN/ECE and the EU (UN/ECE 1994, EC 1997).

## Materials and methods

### Site description

The area of the Hietajärvi (63°10'N, 30°43'E) forested catchment is about 540 hectares. Two lakes, Iso Hietajärvi (83 ha) and Pieni Hietajärvi (2.4 ha) are located in separate subcatchments. In the present study, only the Iso Hietajärvi subcatchment (464 ha) was included. The soils have developed on the 2 500-million-year-old Archean acidic granitoid base. In the area, some eskers and esker-like remnants of glaciofluvial deposits occur. More than one third of the area is covered by fibric histosols (*Sphagnum* peat), the remaining

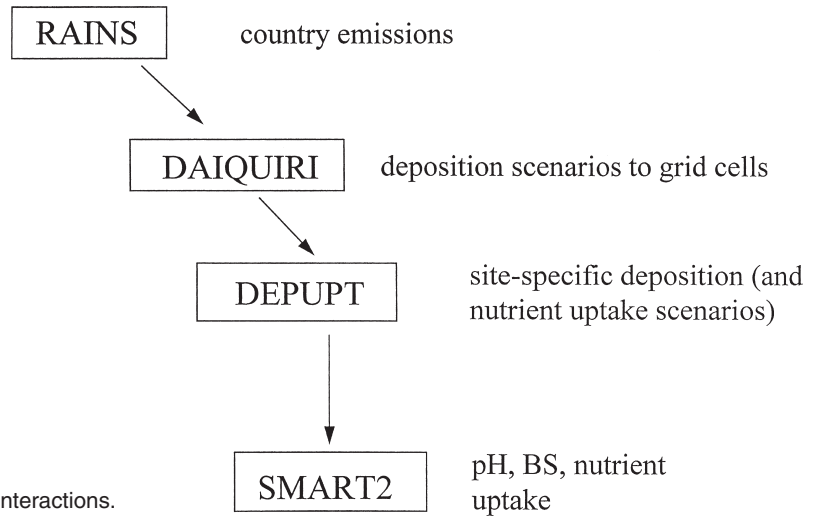


Fig. 1. Flow chart of model interactions.

area being mainly haplic and ferric podzols (till). The forests are mainly mature or old, but some young planted stands are also present. The dominant tree species is Scots pine (*Pinus sylvestris*) (64% of the forest). Norway spruce (*Picea abies*) (26%), downy birch (*Betula pubescens*), silver birch (*Betula pendula*), and aspen (*Populus tremulus*) (10% in all) also occur. In Hietajärvi, the last major forest fire occurred 130–200 years ago. Precipitation chemistry is measured at the site, and the runoff water quality is measured at the outlet of Iso Hietajärvi (Bergström *et al.* 1995). A palaeolimnological study carried out at the site (Simola *et al.* 1991) indicated that pH has been slowly decreasing, but has remained in the range of 6.4–6.8 during this century. The soil parameters used in the SMART2 model calibration are presented in Table 1.

### Modelling concept

A linked model system was used to estimate the effects of the emission scenarios (Fig. 1). Historical sulphur depositions were available from the study by Mylona (1993), average nitrogen histories were estimated on the basis of the work by Sverdrup *et al.* (1995) and Asman and Drukker (1988), and the non-marine, dry deposition component of base cations was assumed to follow the historical sulphur deposition curve. Four future emission scenarios (*see* Table 2) of the EU Acidification Strategy (EC 1997) were formulated us-

ing the RAINS optimisation model (Amann *et al.* 1996) of the IIASA institute. Current and future depositions at the site were derived from coun-

Table 1. Main characteristics of the Hietajärvi catchment (Bergström *et al.* 1995) used in the SMART and SMART2 model applications

Variable	Value
Latitude	63°09'N
Longitude	30°40'E
Total catchment area	4 640 000 m <sup>2</sup>
Lake area	830 000 m <sup>2</sup>
Mean lake depth	3.6 m
Peatland	35%
Standing tree volume	7.8 kg m <sup>-2</sup>
Soil thickness	0.80 m
Bulk density of organic layer*	0.10 g cm <sup>-3</sup>
Bulk density of mineral soil layer	1.32 g cm <sup>-3</sup>
Cation exchange capacity (CEC):	
organic layer*	228.6 meq kg <sup>-1</sup>
mineral soil layer	3.76 meq kg <sup>-1</sup>
C/N ratio in the organic layer	38.0
Organic matter content in the mineral topsoil	0.014 kg kg <sup>-1</sup>
Precipitation surplus (av. 1988–1993)	420 mm a <sup>-1</sup>
Bulk deposition in 1991	
SO <sub>4</sub> <sup>2--</sup> -S	24.4 meq m <sup>-2</sup> a <sup>-1</sup>
NO <sub>3</sub> <sup>-</sup> -N	12.1 meq m <sup>-2</sup> a <sup>-1</sup>
NH <sub>4</sub> <sup>+</sup> -N	10.2 meq m <sup>-2</sup> a <sup>-1</sup>
Throughfall deposition in 1991	
SO <sub>4</sub> <sup>2--</sup> -S	29.8 meq m <sup>-2</sup> a <sup>-1</sup>
NO <sub>3</sub> <sup>-</sup> -N	11.9 meq m <sup>-2</sup> a <sup>-1</sup>
NH <sub>4</sub> <sup>+</sup> -N	9.7 meq m <sup>-2</sup> a <sup>-1</sup>

\* not needed in SMART

try-level emissions using the DAIQUIRI model (Syri *et al.* 1998), on the basis of ten-year averaged transport matrices and emissions for 1985–1995 provided by the EMEP/MSC-W in Norway (Barrett *et al.* 1995). DAIQUIRI produces site-specific deposition values as a function of time by quadratic interpolation from the  $150 \times 150$  km gridded results of Mylona (1993) and Barrett *et al.* (1995), thus giving the historical, current and future deposition (for a particular year) as output.

The outputs of DAIQUIRI were used to derive the site-specific historical and future scenarios for deposition (from 1800 to 2010) by the methodology, later called the DEPUPT model, described by Johansson *et al.* (1996) and Alveteg *et al.* (1998). The DEPUPT model divides the deposition of each compound into different components (wet and dry deposition, and anthropogenic and sea salt components) using measured bulk and throughfall deposition. The forest filtering of dry deposition, linearly correlated to canopy size, is taken into account. In order to estimate the temporal development of the canopy, DEPUPT calculates the potential forest growth from a logistic curve, but slows down the biomass accumulation if nitrogen via atmospheric deposition and mineralization are not sufficient, thus maintaining the nitrogen content of the tree stand constant. Forest growth was scaled to the estimates of volume and increment of standing biomass for Hietajärvi.

The estimated future depositions are shown in Fig. 2 according to four emission scenarios: (1) Current Reduction Plans (CRP) incorporates of-

ficially adopted or internationally announced ceilings on national emissions, (2) Reference scenario (REF) selects the more stringent outcome of CRP and national or international legislation for each country, (3) B1 aims to reduce the areas not protected from acidification by at least 50%, and (4) the Maximum Technically Feasible Reduction (MFR) illustrates the potential of full application of current control technology. The reduction targets by different scenarios are presented in Table 2. The site-specific deposition scenarios prepared with DEPUPT and average soil and vegetation data for the catchment were used as inputs to the dynamic acidification model SMART2. The model was calibrated to measured current runoff water chemistry data and to the estimates of present standing biomass volume and increment and the assumed maximal amount of biomass. The calibrated model was used for the assessment of future ecosystem responses.

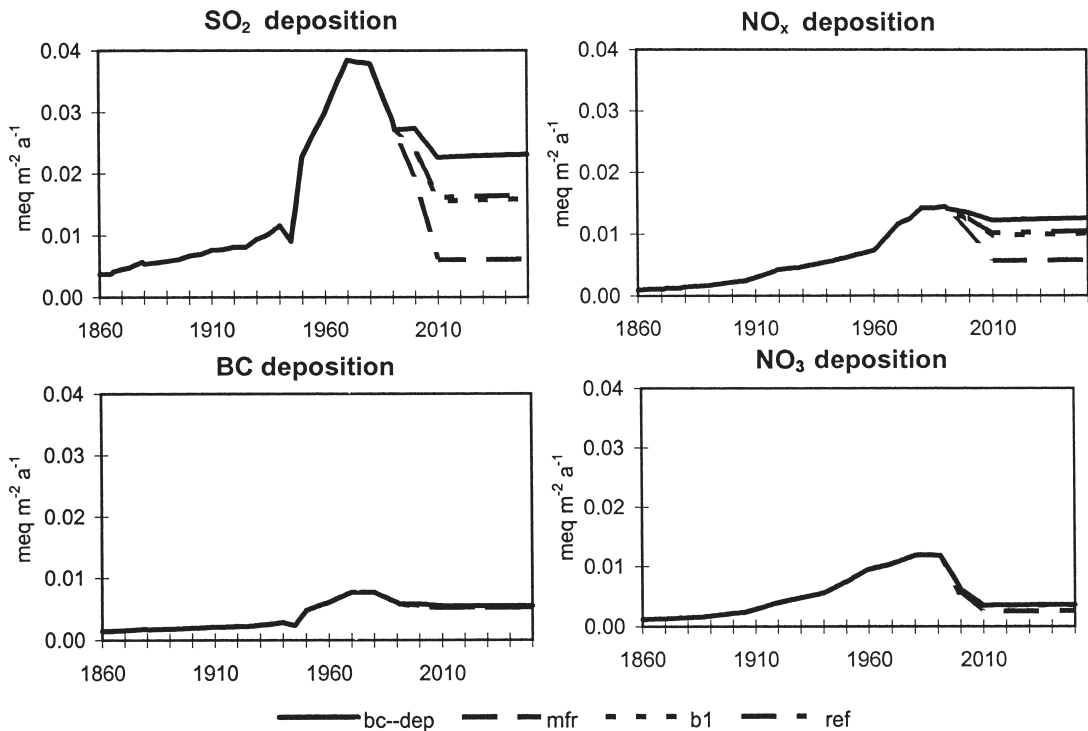
Soil base saturation (BS) and surface water pH were chosen to demonstrate the effects of reduction scenarios on the ecosystem response, since both are considered to be key variables regarding acidifying processes. BS is a measure of the relative amount of base cations (Na, K, Ca, Mg) that are adsorbed on the cation exchange sites at the surfaces of the solid soil particles. A decline in BS may imply that the conditions for forest growth are deteriorating. Runoff water pH reflects the balance of the acid vs alkalinity generating processes in water. A decline in pH usually leads to a deterioration of the conditions for fish spawning and growth.

**Table 2.** Percentage average changes in emissions by the year 2010 in relation to 1990 in Europe (Amann *et al.* 1996) (the numbers may vary from one country to another) and changes in deposition in Hietajärvi derived by the DEPUPT model under four different reduction strategies.

	CRP	REF	B1	MFR
Europe (%):				
SO <sub>2</sub>	-40	-58	-67	-91
NO <sub>x</sub>	-21	-36	-43	-72
NH <sub>3</sub>	-16	-16	-22	-46
Hietajärvi catchment (%):				
SO <sub>2</sub>	-21	-44	-46	-79
NO <sub>x</sub>	-15	-29	-32	-61
NH <sub>3</sub>	-70	-70	-71	-79

## SMART2 model description

The SMART2 model (Kros *et al.* 1995) is an extended version of the SMART model (De Vries *et al.* 1989), developed to estimate long-term chemical changes in the soil and soil water in response to changes in atmospheric deposition. SMART is a simple one-compartment model that only includes geochemical buffer processes. The output of the model is the soil base saturation and the concentrations of the major anions and cations in soil solution and runoff water. A simple lake module describing the retention of sulphate, nitrate and ammonia, as well as inorganic carbon equilibria, is included (Kämäri *et al.* 1994).



**Fig. 2.** Historical and future deposition scenarios for the Hietajärvi catchment, according to the four scenarios defined in Table 2.

The model structure is based on the anion mobility concept, by incorporating the charge balance principle (Reuss *et al.* 1987). SMART consists of a set of mass balance equations, describing the soil input-output relationships for the cations ( $\text{Al}^{3+}$ ,  $\text{BC}^{2+}$  ( $= \text{Ca}^{2+} + \text{Mg}^{2+}$ ),  $\text{K}^+$ ,  $\text{Na}^+$ ,  $\text{NH}_4^+$ ) and strong acid anions ( $\text{SO}_4^{2-}$ ,  $\text{NO}_3^-$ ,  $\text{Cl}^-$ ), and a set of equilibrium equations that describe the equilibrium soil processes. The soil solution chemistry depends solely on the net element input from the atmosphere and the geochemical interactions (Al hydroxide weathering, cation exchange,  $\text{HCO}_3^-$ -dissociation, sulphate adsorption) in the soil. The concentrations of  $\text{HCO}_3^-$  and  $\text{Al}^{3+}$  are determined by means of an equilibrium, constant with  $\text{H}^+$ , the concentration of which is given by the charge balance equation. The cation exchange reactions ( $\text{H}^+$ ,  $\text{BC}^{2+}$ ,  $\text{Al}^{3+}$ ) are described by the Gaines-Thomas equations. Sulphate adsorption/desorption reactions are described by a Langmuir isotherm. The weathering rate of base cations from silicates is independent of soil pH. Dissociation of organic anions is described as a function of pH.

SMART forms the integral part of SMART2, but additions including a two-layer soil structure, forest growth, biocycling processes and some changes in the description of hydrological processes have been made (Kros *et al.* 1995). The upward seepage of water and lateral water flux have been included. Ion transport by seepage water can be taken into account in the calculation of soil water concentrations. Transpiration and interception fractions of precipitation have also been included. The soil compartment is divided into organic and mineral layers, and SMART is applied to each layer separately.

In SMART applications, nutrient uptake has to be given as an input, while in SMART2 nutrient uptake is described as a demand function consisting of maintenance uptake in the leaves and growth uptake in the stems. The growth uptake is proportional to the forest biomass accumulation, calculated using a logistic growth function. Maintenance uptake is dependent on the direct leaf nutrient uptake, foliar exudation and the leaf nutrient content. The foliage can take up ammonium

and protons and exude potassium, calcium, and magnesium. If some of the nutrients in the soil (and atmospheric deposition) are insufficient, the leaf and stem nutrient content will be reduced (for-growth itself will not be affected).

Nutrients in litterfall and root decay are added to the mineralizing organic pool, where a distinction is made between the rapid decomposition of fresh litter (less than one year old) and slow decomposition of old litter (more than one year). Fresh litter that is not decomposed within one year is moved to the old litter pool. Mineralization is regulated by a mineralization factor or a rate constant. The maximum rate can be reduced by low pH, high C to N ratio or high ground water level. Nitrification of  $\text{NH}_4^+$  and denitrification of  $\text{NO}_3^-$  are, as in SMART, described as rate-limited equations, but they are also regulated by factors that are reduced by low pH and the height of the water table. Immobilisation is calculated with the original SMART and is dependent on the soil C to N ratio. The equations added to SMART2 are presented in the Appendix.

## Parameterization

The time-step in model applications was one year. The value of measured yearly runoff from the catchment was given as the precipitation surplus

**Table 3.** Parameters related to vegetation (Finér 1989, Helmisaari 1990, METLA unpubl., K. Rosén unpubl.) used in the SMART2 model application.

Parameter	Value
Thickness of the root zone	0.60 m
Fraction of fine roots in organic layer	0.35
Maximum amount of litterfall	0.15 kg m <sup>-2</sup> a <sup>-1</sup>
Ammonium foliar uptake fraction	0.30
Proton foliar uptake fraction	0.0
K foliar exudation fraction	0.45
N reallocation fraction for leaves	0.40
Ca + Mg content in leaves	0.55%
K content in leaves	0.50%
N content in leaves	0.80%
N content in stems and branches	0.16%
Ca+Mg content in stems and branches	0.21%
K content in stems and branches	0.06%
Logistic growth rate constant	0.044 a <sup>-1</sup>
Growth half-time	128 a

for the years 1987 to 1993, and thus (evapo)-transpiration and interception were set to zero. Before 1987 and after 1993, the mean runoff was used as constant precipitation surplus. Upward seepage was not included.

The measurements of bulk and through fall deposition were taken from the IM database (Bergström *et al.* 1995). These measurements were used to calculate the foliar uptake fractions of ammonium and H<sup>+</sup> and the foliar exudation fraction of potassium according to De Vries *et al.* (1994). The maximum amount of litterfall was derived from litterfall measurements and the maximum amount of biomass from the biomass estimates carried out by the Finnish Forest Research Institute (METLA). The logistic growth rate constant and the growth half time were set to give the present annual biomass growth and the total amount of biomass, which were estimated by the National Board of Parks and Forestry during the last forest inventory at the site. The nutrient contents in the biomass compartments of different tree species were taken from K. Rosén (unpubl.) and L. Finér (1989). The average stem nutrient contents for the whole catchment were calculated as the weighted sum of the contents in different species. An average nitrogen content in the leaves (K. Rosén unpubl.) was used instead of minimum and maximum leaf N contents. The reason for this was that no clear correlation between leaf and deposition N concentration was found in the measurements. The fraction of reallocated nitrogen from leaves to stems prior to litterfall was taken from the literature (Finér 1989, Helmisaari 1990).

The mineralization fraction was taken from

**Table 4.** Variables taken from the literature (Kurka *et al.* 1995, Finér 1989, Oliver *et al.* 1983) used in the SMART and SMART2 model applications.

Variable	Value
Nitrification factor	1
Denitrification factor	0.5
Mineralization rate constant of litter*	0.35 a <sup>-1</sup>
Nutrient cycling factor*	0.40
Total concentration of organic acids	0.05 eq m <sup>-3</sup>
3 parameters for modeling pK <sub>a</sub> :	
a	0.96
b	0.90
c	0.039

\* not needed in SMART

the measurements made by METLA (Kurka *et al.* 1995). Under optimal conditions, all the ammonium in the soil was assumed to be nitrified and ten percent of the nitrate was assumed to be denitrified. The soil pH reduced dynamically the nitrification and denitrification processes in accordance with the included reduction functions. The reducing impact of pH was effective between the pH values of seven and four (Schmidt 1985). Below pH four, no denitrification was assumed to take place. The influence of the mean height of the spring water table on the mineralization and nitrification processes was not taken into account, because the moisture regime was assumed to remain stable with a yearly time step. The nutrient cycling factor was derived from the literature (Finer 1989).

The parameters derived by Oliver *et al.* (1983) were used to calculate organic acid dissociation. The calibration parameters were the partial pressure of CO<sub>2</sub> in the soil, the exchange coefficients, the cation weathering rates, and the mass transfer coefficients in the lake. Other soil chemical parameters were taken from the IM database (Bergström

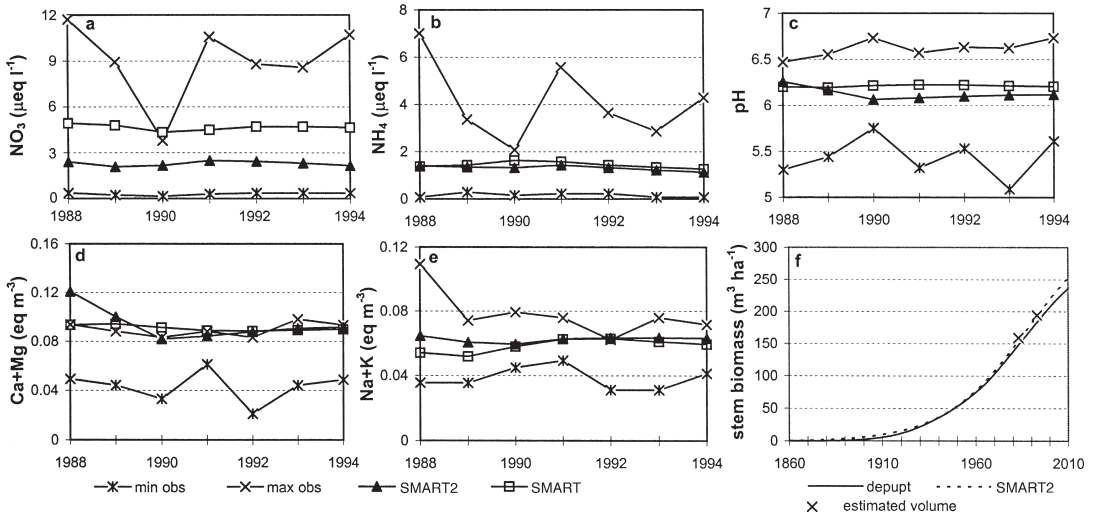
*et al.* 1995). Physical and chemical properties of the soil profile were given separately for the organic layer and the mineral soil and averaged into a single profile, characteristic for the entire catchment. The vertical averaging was realised by calculating the profile-specific weighted sums of the measurements, the weights being the relative thickness and the bulk density of each layer (Posch 1997). The profiles were averaged horizontally over the catchment by weighting the parameters of each profile by the relative proportion of the total area. All the parameters used in the model are presented in Tables 1, 3, 4 and 5.

### Calibration

The starting year of the simulations was 1860, which is also the assumed year of the beginning of vegetation growth after the last forest fire. The SMART2 calibration was started by adjusting the vegetation growth curve (Fig. 3f). The logistic growth rate constant and the growth half-time were set to values giving the correct present amount of

**Table 5.** Parameters obtained as a result of the model calibration

Parameter	Unit	SMART2 Organic/mineral	SMART
<b>Soil parameters:</b>			
Ca + Mg weathering rate	eq m <sup>-3</sup> a <sup>-1</sup>	0.058/0.058	0.040
Na weathering rate	eq m <sup>-3</sup> a <sup>-1</sup>	0.015/0.015	0.023
K weathering rate	eq m <sup>-3</sup> a <sup>-1</sup>	0.030/0.030	0.008
Selectivity constant for:			
Al-BC exchange	–	10 <sup>-1.8</sup> /10 <sup>0.8</sup>	1
H-BC exchange	–	10 <sup>4.084</sup> /10 <sup>6.3</sup>	10 <sup>7.0</sup>
Gibbsite equilibrium constant	–	10 <sup>9.085</sup> /10 <sup>6.7</sup>	10 <sup>8.2</sup>
Maximum SO <sub>4</sub> adsorption capacity	meq kg <sup>-1</sup>	0.0/2.5	3
Half-saturation constant for SO <sub>4</sub> adsorption	eq m <sup>-3</sup>	2 000/0.20	0.1
pCO <sub>2</sub> in soil solution (multiple of pCO <sub>2</sub> in air)	–	60/60	20
<b>Lake parameters:</b>			
Gibbsite equilibrium constant in lake	–	10 <sup>10</sup>	10 <sup>8.2</sup>
Net mass transfer coefficient for retention of			
S	m a <sup>-1</sup>	1.5	0.5
NO <sub>3</sub>	m a <sup>-1</sup>	12.0	5.0
NH <sub>4</sub>	m a <sup>-1</sup>	6.0	5.0
pCO <sub>2</sub> in stream (multiple of pCO <sub>2</sub> in air)	–	8.0	5.0



**Fig. 3.** The fit of the SMART2 and SMART calibration to the six year measurements of (a) nitrate, (b) ammonium, (c) pH, (d) divalent base cations, and (e) monovalent base cations. The calculated forest biomass accumulations by SMART2 and DEPUPT is shown in (f).

biomass ( $78 \text{ t ha}^{-1}$ ) and annual growth ( $3.5 \text{ m}^3 \text{ ha}^{-1} \text{ a}^{-1}$ ). After that, the calibration followed the procedure outlined by Bleeker *et al.* (1994). The model was calibrated to produce present-day (1988–94) runoff water chemistry. The criterion in accepting the calibration was that modelled annual pH, Gran alkalinity, and ion concentrations in runoff water would fall between the measured annual minimum and maximum values. The results of the calibrated SMART2 were compared to the SMART calibration results (Forsius *et al.* 1998), and to the yearly mean of the measurements.

## Results and discussion

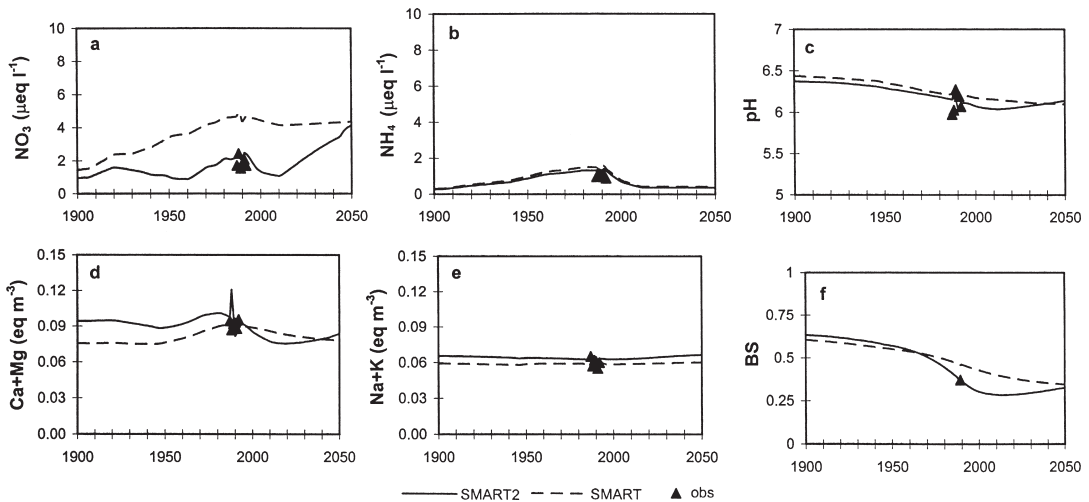
### Calibration and model comparison

The calibration criterion was met by the runoff water pH and alkalinity, and the concentrations of ammonium, nitrate, sulphate, chloride, and monovalent base cations (Fig. 3a–e). Compared to the corresponding SMART model calibration, SMART2 showed a better fit to the yearly mean of the measured nitrate concentrations in runoff water (Fig. 4a). The dynamics of the nitrate concentration also behaved differently in the two models. With SMART, the NO<sub>3</sub><sup>-</sup>-concentration followed the shape of the NO<sub>x</sub> deposition curve,

but with SMART2 the concentration reflected the effect of the simulated uptake of nitrogen by the forest. This can be seen, for instance, as a decrease in growth after the beginning of the period of rapid forest growth in the 1920s. The forest growth curves simulated with DEPUPT and SMART2 were very similar, but there was some discrepancy in the nutrient uptake due to different uptake approaches. This may explain some of the discrepancies in ion concentrations between the two models.

Both models fitted well to the mean of the measured base cations in runoff water, but a clear difference in the dynamical behaviour of divalent base cations was seen (Fig. 4d). The SMART simulation again followed the deposition curve, while SMART2 gave a slight fall during the rapid uptake period followed by a much more pronounced decrease than SMART. The historical runoff water pH was very similar in both simulations; decreasing, as expected (Fig. 4c). SMART2 predicted lower present values, which were also slightly lower than the pH estimated in the palaeolimnological study (Simola *et al.* 1991). The historical soil base saturation was higher in the SMART2 simulations but the decrease between 1950 and 1990 was also much stronger, leading to lower levels by the year 2000 (Fig. 4f). In general, SMART2 had the tendency to show more response to changes than SMART, but it





**Fig. 4.** Comparison of the calibrated SMART2 and SMART models to the observed mean of: (a) nitrate, (b) ammonium, (c) pH, (d) divalent base cations, (e) monovalent base cations and (f) base saturation. Model results are calculated with the CRP scenario.

was difficult to give a preference to either calibration.

## Nitrogen dynamics

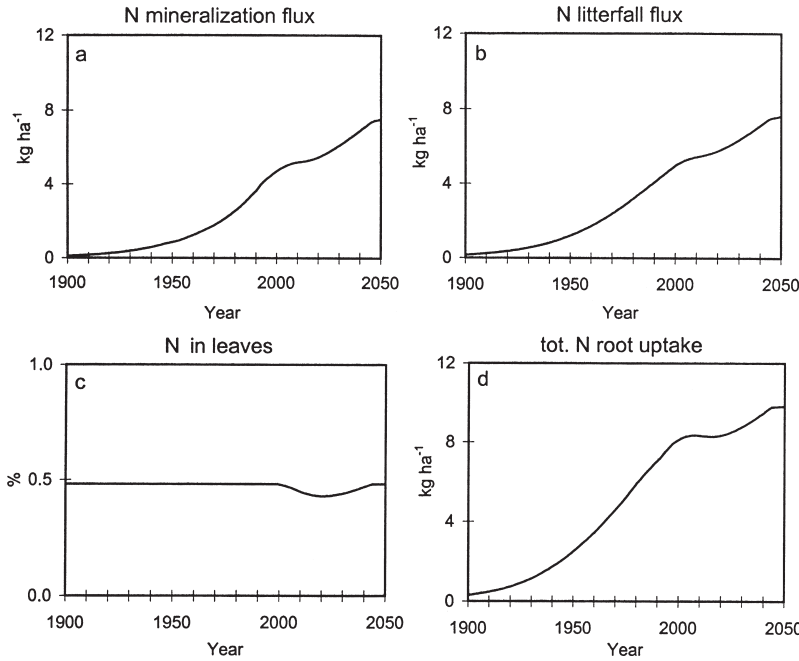
The mineralization of fresh litter turned out to be problematic in Hietajärvi. It is dependent on the C to N ratio of the leaves and microorganisms (internal parameter). Because the simulation was started at the point when all the forest was assumed to be burnt, we had to assume that the soil organic nutrient pool had been fully depleted. This was done in order to prevent the consequent occurrence of the very high nutrient leaching from the soil because the model does not calculate the growth of ground vegetation that strongly binds the nutrients released from the trees to the soil. For growing leaves, constant (average) nitrogen (0.8%) and carbon (50%) contents were used. This led to a high C to N ratio that impeded the mineralization of fresh litter. Therefore, litter was treated as one pool with one single mineralization rate. The mineralization from litter was therefore simulated starting from the second year after litterfall. To prevent a too high accumulation of nitrogen in the organic nitrogen pool, the old litter mineralization rate was set high (Table 4).

With SMART2, nitrogen concentrations in both the soil and runoff water were very dynamic

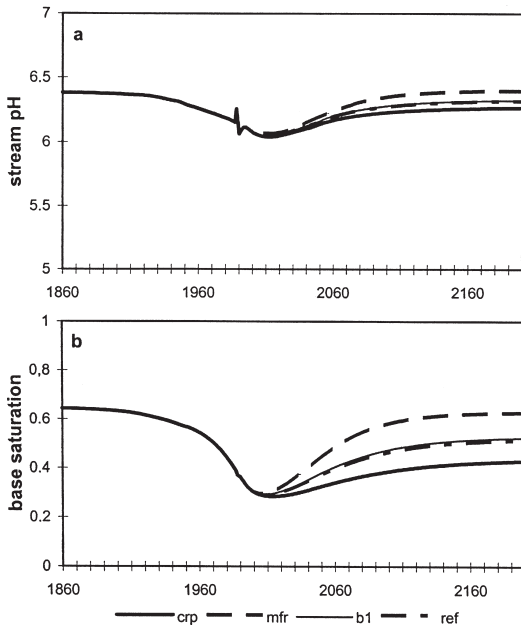
throughout the simulation period. This was partly due to the pH and C to N ratio dependent processes in the soil, and partly due to the sensitivity to changes in nitrogen deposition. A decrease in deposition after the year 1990 was reflected in all nitrogen related processes. This had an immediate influence on the nitrate concentration in runoff water (Fig. 4a) and a delayed influence on nitrogen related to the vegetation. The mineralization flux, litterfall flux, and nitrogen root uptake all slowed down, and the nitrogen content in the leaves decreased between the years 2000 and 2040, after the start of the decrease in nitrogen deposition (Fig. 5).

## Scenario analysis

Our model results show the start of a decline in soil base saturation and runoff water pH in the 1960s when the level of acidifying deposition starts to increase rapidly (Fig. 6). The decline continues through the years of high deposition and even into the 21st century. According to the SMART2 model, a delayed recovery of the ecosystem will take place after the implementation of the reduction strategies, the response being dependent on the stringency of the applied strategy. Model outcomes that give the shortest lag to the turning point (with MFR strategy around the



**Fig. 5.** Calculation results for nitrogen (a) mineralization, (b) litterfall, and (c) root uptake fluxes and (d) nitrogen content in leaves according to the CRP scenario.



**Fig. 6.** (a) Stream water pH and (b) soil base saturation scenarios calculated by SMART2.

year 2020) suggest that the effects of reduction strategies will not be observed before 2010, the year when the chosen reductions should be attained.

If the ecosystem response is studied for an extended time period, with a hypothesis that atmospheric deposition and nutrient uptake would remain constant, it is evident that pH and base saturation would stabilise (Fig. 6a and b). With the MFR strategy they would return to the level of 1860, but, with the B1, REF and CRP strategies ecosystem deterioration would be permanent to a certain extent.

### Conclusions

Calibration of the SMART2 model to the desired accuracy was achieved. Compared to the previous SMART model calibration, neither version of the model appeared to be preferable for the purposes of this study. However, the SMART model requires the determination of a considerably smaller number of parameters. On the other hand, if the soil nitrogen status or the relationship between the soil and nutrient uptake by the growing vegetation are of interest, only SMART2 is applicable. In general, SMART2 behaved more responsively and can be considered to be more sensitive to changing conditions.

The SMART2 model results show a decline in base saturation and runoff water pH in response

to the rapid increase of acidifying deposition that started in the 1960s. The decline is stopped and reversed assuming any of the reduction scenarios. If future trends are studied on very long time scale (up to the year 2300), assuming constant input and uptake shows that the environmental improvement will be stabilised, but the final level of BS and pH will depend on the chosen strategy. The more ambitious the strategy is, the faster and more complete will be the recovery.

*Acknowledgements:* We thank Sophia Mylona and Erik Berge at EMEP/MSC-W for providing the transfer matrices for atmospheric pollutant and data on historical sulphur deposition. We also thank the Finnish Forest Research Institute (METLA) for providing their unpublished data for the model calibration. The present study was carried out as part of the project 'Development of Assessment of Monitoring Techniques at Integrated Monitoring Sites in Europe', financed by the Financial Instrument for the Environment (LIFE) of the EU (project LIFE95/FIN/A11/EPT/387).

## References

- Alveteg M., Walse C. & Warfvinge P. 1998. Reconstructing historic atmospheric deposition and nutrient uptake from present day values using MAKEDEP. *Water, Air and Soil Poll.* 104: 269–283.
- Amann M., Bertok I., Cofala J., Gyarmas F., Heyes C., Klimont Z. & Schöpp W. 1996. Cost-effective control of acidification and ground-level ozone. *Second Interim Report to the European Commission*, DG-XI. IIASA, Laxenburg, Austria. 112 pp.
- Asman W. & Drukker B. 1988. Modelled historical concentrations and depositions of ammonia and ammonium in Europe. *Atmos. Environ.* 22: 725–735.
- Barrett K., Seland Ø., Foss A., Mylona S., Sandnes H., Styve H. & Tarrason L. 1995. European transboundary acidifying air pollution; ten years calculated fields and budgets to the end of the first Sulphur protocol. *EMEP/MSC-W Report 1/95*, Norwegian Meteorological Institute, Oslo, Norway.
- Bergström I., Mäkelä K. & Starr M. 1995. Integrated Monitoring Programme in Finland. *First National Report*. Ministry of the Environment, Environmental Policy Department, Helsinki. 138 pp.
- Bleeker A., Posch M., Forsius M. & Kämäri J. 1994. Calibration of the SMART acidification model to integrated monitoring catchments in Europe. *Mimeograph Series of the National Board of Waters and the Environment*, Helsinki, Finland. 52 pp.
- Cosby B.J., Hornberger G.M., Galloway J.N. & Wright R.F. 1985. Modeling the effects of acid deposition: Assessment of a lumped parameter model of soil water and stream water chemistry. *Water Resour. Res.* 21: 51–63.
- De Vries W., Kros J. & Voogd J.C.H. 1994. Assessment of critical loads and their exceedance on Dutch forests using a multi-layer steady-state model. *Water, Air and Soil Poll.* 76: 407–448.
- De Vries W., Posch M. & Kämäri J. 1989. Simulation of the long-term soil response to acid deposition in various buffer ranges. *Water, Air and Soil Poll.* 48: 349–390.
- EC 1997. *Draft Communication to the Council and Parliament on a: Community Strategy to Combat Acidification*. COM(97)88/4 European Commission, Brussels.
- Finér L. 1989. Biomass and nutrient cycle in fertilized and unfertilized pine, mixed birch and pine and spruce stands on a drained mire. *Acta Forestalia Fennica* 208, Helsinki, Finland. 63pp.
- Forsius M., Alveteg M., Jenkins A., Johansson M., Kleemola S., Lükewille A., Posch M., Sverdrup H. & Walse C. 1998. MAGIC, SAFE and SMART Model Applications at Integrated Monitoring Sites: Effects of Emission Reduction Scenarios. *Water, Air and Soil Poll.* [In press].
- Helmsaari H.-S. 1990. *Nutrient Retranslocation within Pinus sylvestris*. Academic dissertation. University of Joensuu. Scientific Publication.
- Jenkins A., Whitehead P.G., Musgrove T.J. & Cosby B.J. 1990. A regional model of acidification in Wales. *J. Hydrology* 116: 403–416.
- Johansson M., Alveteg M., Walse C. & Warfvinge P. 1996. Derivation of deposition and uptake scenarios. In: Knoflach M., Schneider J. & Soja G. (eds.), International workshop on exceedance of critical loads and levels. 22–24 Nov 95, Vienna. *Conference papers* Vol.15, Federal Ministry of Environment, Youth and Family, Vienna, Austria, pp. 318–324.
- Kleemola S. & Forsius M. (eds.) 1996. International Cooperative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems. 5th Annual Report. *The Finnish Environment* 27. Finnish Environment Institute, Helsinki. 82 pp.
- Kros J., Reinds G.J., de Vries W., Latour J.B. & Bollen M.J.S. 1995. Modelling of soil acidity and nitrogen availability in natural ecosystems in response to changes in acid deposition and hydrology. *Report* 95. DLO Winand Staring Centre, Wageningen, The Netherlands. 88 pp.
- Kurka A.-M., Vanhala P., Karsisto M. & Starr M. 1995. Needle litter decomposition. In: Bergström I., Mäkelä K. and Starr M. (eds.), Integrated Monitoring Programme in Finland. *First National Report*. Ministry of the Environment, Environmental Policy Department, Helsinki. Report 1, p. 101.
- Kämäri J., Posch M., Kähkönen A.-M. & Johansson M. 1994. Modeling potential long-term responses of a small catchment in Lapland to changes in sulfur deposition. *Sci. Tot. Environ.* 160/161: 687–701.
- Nilsson, J. & Grennfelt, P. 1988. Critical loads for S and N. Nord 1998:97. Nordic Council of Ministers, Copenhagen. 418 pp.
- Mylona S. 1993. Trends of sulphur dioxide emissions, air concentrations and depositions of sulphur in Europe since 1880. *EMEP/MSC-W Report 2/93*, Oslo, Norway.
- Oliver B.G., Thurman E.M. & Malcolm R.L. 1983. The contribution of humic substances to the acidity of

- colored natural waters. *Geochim. Cosmochim. Acta* 47: 2031–2035.
- Posch M., Falkengren-Grerup U. & Kauppi P. 1989. Application of two soil models to historical soil chemistry data from Sweden. In: Kämäri J., Brakke D.F., Jenkins A., Norton S.A. & Wright R.F. (eds.), *Regional Acidification models: Geographic Extent and Time Development*. Springer Verlag, New York, pp. 241–252.
- Posch M., Reinds G.J. & de Vries W. 1993. SMART- A simulation model for acidification's regional trends: Model description and user manual. *Mimeograph Series of the National Board of Waters and the Environment* 477, Helsinki, Finland. 43 pp.
- Posch M. 1997. Averaging and other simplifications in dynamic soil models. *Report of the Workshop on: Data analysis for modelling and assessment of biogeochemical effects of air pollution in temperate ecosystems*. CIEMAT, Madrid (SP), 8–11 October 1997, pp. 61–67.
- Posch M., Kämäri J., Forsius M., Henriksen A. & Wilander A. 1997. Exceedance of Critical Loads for Lakes in Finland, Norway, and Sweden: Reduction Requirements for Acidifying Nitrogen and Sulfur Deposition. *Environmental Management* 21/2, pp. 291–304.
- Reuss J.O., Cosby B.J. & Wright R.F. 1987. Chemical processes governing soil and water acidification. *Nature* 329: 28–31.
- Schmidt E.L. 1985. Nitrification in Soil. In: Stevenson F.J. (ed.), *Nitrogen in agricultural soils*. *Agronomy* 22. American Society of Agronomy, Madison, Wisconsin. pp. 253–283.
- Simola H., Huttunen P., Rönkkö J. & Uimonen-Simola P. 1991. Palaeolimnological study of an environmental monitoring area, or, Are there pristine lakes in Finland? *Hydrobiologia* 214: 187–190.
- Sverdrup H., Warfvinge P., Blake L. & Goulding K. 1995. Modelling recent and historic soil data from the Rothamsted experimental station, UK, using SAFE. *Agriculture Ecosystems and Environment* 53: 161–177.
- Syri S., Johansson M. & Kangas L. 1998. Application of nitrogen transfer matrices for integrated assessment. *Atmos. Environ.* 32/3: 409–413.
- UN/ECE 1994. Protocol to the 1979 Convention on Long-Range Transboundary Air Pollution on Further Reduction of Sulphur Emissions. *Document ECE/EB.AIR/40*, New York and Geneva, 106 pp. [In English, French and Russian].
- Warfvinge, P. & Sverdrup, H. 1992. Calculating critical loads at acid deposition with PROFILE — a steady state soil chemistry model. *Water, Air and Soil Poll.* 63: 119–143.

Received 8 December 1997, accepted 30 June 1998

#### Appendix. Equations added to the SMART2 model

##### Mass balances

$$\begin{aligned} d/dt(\theta \times T_{rz} \times [SO_4] + r_{rz} \times T_{rz} \times ctSO_{ac})(z) &= SO_{4td} - P \times [SO_4](z) \\ d/dt(\theta \times T_{rz} \times [NO_3])(z) &= NO_{3td} + fr_{ru}(z) \times (NH_{4ni} - NO_{3ru} - NO_{3de}) - P \times [NO_3](z) \\ d/dt(\theta \times T_{rz} \times [NH_4])(z) &= NH_{4td} - NH_{4fu} + N_{mi} - fr_{ru}(z) \times (NH_{4ni} - NH_{4ru}) - P \times [NH_4](z) \\ d/dt(\theta \times T_{rz} \times [BC^{2+}] + r_{rz} \times T_{rz} \times CEC \times frBC_{ac}^{2+})(z) &= \\ BC_{td}^{2+} + BC_{fe}^{2+} + BC_{mi}^{2+} + BC_{ru}^{2+} - fr_{ru}(z) \times BC_{ru}^{2+} + BC_{we}^{2+} \times z - P \times [BC^{2+}](z) \\ d/dt(\theta \times T_{rz} \times [K])(z) &= K_{td} + K_{fe} + K_{mi} - fr_{ru}(z) \times K_{ru} + K_{we} \times z - P \times [K](z) \\ d/dt(\theta \times T_{rz} \times [Na])(z) &= Na_{td} + Na_{we} \times z - P \times [Na](z) \end{aligned}$$

##### Nitrification and denitrification

$$\begin{aligned} NH_{4ni} &= fr_{nimx} \times rf_{nipH} \times (NH_{4in} - NH_{4ru} - NH_{4im} + NH_{4mi}) \\ NO_{3de} &= fr_{demx} \times rf_{dePH} \times (NO_{3in} - NO_{3ru} - NO_{3im} + NH_{4ni}) \end{aligned}$$

##### Nutrient uptake

$$\begin{aligned} X_{ru} &= (N_{if} - N_{fu} + N_{gu}) \times X_{in} / N_{in} & X &= NH_4, NO_3 \\ X_{ru} &= X_{if} + X_{fe} + X_{gu} & X &= BC^{2+}, K \\ X_{gu} &= (Am_{st}(t) - Am_{st}(t-i)) \times ctX_{st} & X &= NH_4, NO_3, BC^{2+}, K \\ Am_{st}(t) &= Am_{stmx} / (1 + \exp(-k_{gj} \times (t - t_{0.5}))) \\ Am_{if} &= Am_{ifmx} \times Am_{st} / Am_{stmx} \end{aligned}$$

##### Litterfall and root decay

$$\begin{aligned} X_{if} &= (1 - frX_{re}) \times Am_{if} \times ctX_{lv} & X &= N, BC^{2+}, K \\ X_{rdms}(z) &= fr_{ru}(z) \times X_{if} \times nfc \times (1 - fr_{rtl}) & X &= N, BC^{2+}, K \\ X_{mi} &= k_{mimx} \times rf_{mipH} \times rf_{miCN} \times Am_{if} \times ctX_{st} & X &= N, BC^{2+}, K \\ rf_{miCN} &= 1 - (CN_l - CN_{mo}) / DA_{mo} \times CN_{mo} & & CN_{mo} < CN_l < (1 + DA_{mo}) \times CN_{mo} \\ rf_{miCN} &= 1 & & CN_l < CN_{mo} \\ rf_{miCN} &= 0 & & CN_{mo} > (1 + DA_{mo}) \times CN_{mo} \\ X_{mi}(z) &= X_{mi} + X_{rdms}(z) & X &= N, BC^{2+}, K \\ RCOO_{mi} &= N_{mi} + BC_{mi}^{2+} + K_{mi} \end{aligned}$$

##### Canopy interactions

$$\begin{aligned} X_{fu} &= frX_{fu} \times X_{td} & X &= NH_4, H \\ X_{fe} &= frX_{fe} \times (NH_{4fu} + H_{fu}) & X &= K, BC^{2+} \end{aligned}$$

Continues

## Appendix. Continued.

Parameter	Description	Unit
$\theta$	Volumetric moisture content of soil	$\text{m}^3 \text{m}^{-3}$
$r_z$	Bulk density in root zone	$\text{kg m}^{-3}$
$Am_{\text{stm}}x$	Max. amount of stems and branches	$\text{kg ha}^{-1}$
$Am_{\text{lifm}}x$	Max. amount of litterfall	$\text{kg ha}^{-1} \text{a}^{-1}$
$Am_{\text{li}}$	Amount of litter	$\text{kg ha}^{-1}$
$ctSO_{4ac}$	Sulphate concentration at adsorption complex	$\text{mmol kg}^{-1}$
$ctX_{\text{st}}$	Nutrient content in stems and branches	%
$ctX_{\text{lv}}$	Nutrient content in leaves	%
$CN_{\text{mo}}$	C/N ratio of micro-organisms	–
$CN_{\text{l}}$	C/N ratio of litter	–
$DA_{\text{mo}}$	Dissimilation/assimilation ratio of microbes	–
$fr_{\text{demx}}$	Maximum denitrification fraction	–
$fr_{\text{nimx}}$	Maximum nitrification fraction	–
$fr_{\text{rlt}}$	Fraction of roots in litter layer	–
$fr_{\text{ru}}$	Cumulative transpiration fraction	–
$frX_{\text{fe}}$	Foliar exudation fraction of ion X	–
$frX_{\text{fu}}$	Foliar uptake fraction of ion X	–
$frX_{\text{re}}$	Reallocation fraction of ion X for leaves	–
$k_{\text{gl}}$	Logistic growth rate constant	$\text{kg ha}^{-1} \text{a}^{-1}$
$k_{\text{mimx}}$	Maximum mineralization rate of litter	$\text{a}^{-1}$
$nfc$	Nutrient cycling factor	–
$P$	Precipitation surplus	$\text{m a}^{-1}$
$RCOO_{\text{mi}}$	Flux of organic anions	$\text{mol m}^{-2} \text{a}^{-1}$
$rf_{\text{depH}}$	Reduction factor of denitrification for pH	–
$rf_{\text{miCN}}$	Red. factor of mineralization rate for N conc.	–
$rf_{\text{mipH}}$	Reduction factor of mineralization rate for pH	–
$rf_{\text{nipH}}$	Reduction factor of nitrification for pH	–
$t$	Time	a
$t_{0.5}$	Half-life time of logistic growth function	a
$T_{\text{rz}}$	Thickness of the root zone	m
$[X]$	Concentration of ion X	$\text{mol m}^{-3}$
$X_{\text{de}}$	Denitrification flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{fe}}$	Foliar exudation flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{fu}}$	Foliar uptake flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{gu}}$	Growth uptake flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{im}}$	Immobilization flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{in}}$	Input flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{lf}}$	Litterfall flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{mi}}$	Mineralization flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{ni}}$	X nitrification flux	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{rdms}}$	Root decay flux in mineral soil layer	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{ru}}$	Root uptake flux of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{td}}$	Total deposition of ion X	$\text{mol m}^{-2} \text{a}^{-1}$
$X_{\text{we}}$	Weathering flux of base cation X	$\text{mol m}^{-2} \text{a}^{-1}$
$z$	Depth	m