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ECOLOGICAL MODELLING

Ecological Modelling 135 (2000) 311–325

www.elsevier.com/locate/ecolmodel

Sensitivity of soil acidification model to deposition and forest growth

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Received 12 October 1999; received in revised form 7 August 2000; accepted 17 August 2000

Abstract

We report an investigation concerning the impacts of acid deposition and forest growth scenarios on simulated soil effective base saturation for a forested catchment in eastern Finland. These forests have not been managed during the last 150 yr and the area receives low levels of acidifying deposition. The fluxes of sulphur, nitrogen and base cations were assessed with models simulating historic and future deposition, stand uptake and leaching. We tested the effects of calibrating the modelled deposition time series to high and low estimates of current levels of deposition. The highest future soil base saturation was predicted when using the year with the lowest observed sulphur and nitrogen deposition (1993) as representative of the present deposition. The lowest historical and future soil base saturation resulted for using the year with the highest observed deposition of sulphur and nitrogen (1988). All scenarios concerning nutrient uptake, emission reduction levels and timing of the reductions resulted in simulated future soil base saturation values located between those predicted with the high and low observed present deposition. The standard deviation in the soil base saturation introduced by varying the present forest biomass and growth was smaller than that produced by varying the present deposition values.

Keywords: Acidification; Model; Forest growth; Soil base saturation

1. Introduction

Soils and surface waters in forested catchments are subject to the effects of anthropogenic activity through atmospheric deposition and forest management. Over the last 20 yr, scientific and political efforts have focussed on the evaluation and abatement of harmful changes induced by anthropogenic acidifying deposition (Jenkins et al., 1998). The importance of other factors regulating soil and water acidification has been recognised (Johnson et al., 1988). Forest growth itself contributes to soil acidification (Nilsson et al., 1982). In areas that have low acid buffering capacity or receive only moderate amounts of acid deposition the effects of forest growth and forest management can be appreciable.

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PII: S0304-3800(00)00387-2

Tiktak and van Grinsven (1995) reviewed 16 forest-soil-atmosphere models and reported that only a few of those models were well-balanced, meaning that they describe the complete cycling of water, carbon and nutrients with the same degree of detail for all relevant compartments. This lack of balance is part of the reason why so few model studies report work on the combined impact of atmospheric acid deposition and forest growth, although forest canopy defoliations has been partly attributed to soil chemistry (UN/ECE and EC, 1999) and decreasing forest growth has been observed in combination with changes in soil chemistry resulting from acid deposition (Hovmand and Bille-Hanssen, 1999).

Models originally developed to simulate the dynamics of soil processes in atmospheric acidification have been coupled to reconstructions and scenarios of forest growth to illustrate that afforestation may accentuate surface water acidification (Ferrier et al., 1993) and that the sustainability of tree stands depends on the relative rates of deposition, weathering and nutrient uptake (Fichter et al., 1998). Regional applications of soil acidification models coupled to nutrient uptake scenarios have also been reported (Kurz et al., 1998), and the calculation of critical loads of acidity involves the estimation of nutrient uptake rates (Posch et al., 1999). The feedback between soil status and forest performance is not, however, accounted for in these applications, and the relative importance of forest growth compared with atmospheric deposition remains unclear.

In this paper, we use models to explore the relative importance of forest growth compared with acid deposition at a site that is unmanaged and receives low deposition loads. The response of the soil to changes in the driving variables used in the models was evaluated on the basis of simulated soil base cation saturation. The family of models available did not allow for an analysis of the feedback of soil status on forest growth, but we believe the sensitivity of the simulated soil base cation saturation to variations in the different driving variables merits a study.

We compared three sets of driving variables or scenarios concerning (i) the reduction of deposition; (ii) the timing of the reduction; and (iii) the growth of the forest. In addition, the sensitivity of the simulations to uncertainty in the current deposition level was investigated. The impact of each set of driving variables was analysed in terms of the standard deviation calculated for the annual effective base saturation values.

The anthropogenic deposition of sulphur and nitrogen to the site was calculated with the Deposition, AIr Quality and Integrated Regional Information (DAIQUIRI) model. The DEPUPT model was used to estimate the total deposition of sulphur, nitrogen, chloride and base cations, and the uptake of nitrogen and base cations by the forest. The development of soil effective base saturation was estimated with the soil acidification model, Simulation Model for Acidification's Regional Trends (SMART). The future emission scenarios were chosen from those proposed by the European Commission in its so called Acidification Strategy (Amann et al., 1997). The work reported in this paper was carried out in parallel with model based assessment at six other European sites (Forsius et al., 1998a).

2. Methods

2.1. Study site

The forested catchment Hietajärvi in eastern Finland (63°10′N 30°43′E) extends over 4.6 km^2 and lies between 165 and 214 m a.s.l. (Table 1). The catchment is one of the UN-ECE Integrated Monitoring Programme areas (EDC, 1993; Bergström et al., 1995; Starr et al., 1998; Ukonmaanaho et al., 1998). The bedrock of the area is formed of Archaean granitoids (Korsman et al., 1997). Over a third of the catchment is covered by Fibric Histosols, the remaining being Haplic and Ferric Podzols. Gleyic properties indicate a shallow fluctuating water table (Bergström et al., 1995).

The forests are mainly mature or old Scots pine (100–200 years) growing on submesic heaths. Norway spruce, birch and aspen occur among the pine (Bergström et al., 1995). The total forested area amounts to 56% (upland forests 43%) and the surface water area to 23% of the catchment area, with the remaining 21% treeless mire areas (Tuominen, 2000). The diatom-inferred pH of lake Iso-Hietajärvi indicates that the pH has been 6.4–6.8 since pre-industrial times (Simola et al., 1991). Lake Iso-Hietajärvi pH increased from 6–6.5 to 6.5–7 in 1989–1996 (Rask et al., 1998). Annual ion mass budgets calculated for 1988–1991 (Forsius et al., 1995) showed that outputs exceeded inputs of Na^{+}, K^{+}, Ca^{2+}, Mg^{2+}, HCO_3^{-} and organic anions, whereas retention was found in the case of H^{+}, NH_4^{+}, NO_3^{-} and SO_4^{2-}. For the period 1989–1995 Ukonmaanaho et al. (1998) report a reduction in the acidity of precipitation (open and through fall), due to decreasing SO_4^{2-} concentrations.

2.2. Models

2.2.1. Deposition, air quality and integrated regional information

Deposition scenarios based on domestic and international emission reductions were derived using the long-range transport matrices developed by the Meteorological Synthesising Centre-West of the Co-operative Programme for Monitoring and Evaluation of the Long-Range Transmission of Air Pollutants in Europe (Barrett et al., 1995). Estimated historical sulphur (Mylona, 1993) and nitrogen depositions (Asman and Drukker, 1988; Alveteg et al., 1998) were used. These data were used in the deposition model DAIQUIRI (Syri et al., 1998), developed for the integrated assessment of emission reduction strategies. DAIQUIRI calculates deposition for a certain location by interpolating the data from the four closest surrounding grid cells with inverse square distance weighting. To minimise the effect of meteorological variations, the averaged matrices for the years 1985–1994 were used. For consistency with the other sites (Forsius et al., 1998a), the higher-resolution mesoscale module of DAIQUIRI (Ruoho-Airola et al., 1998) for calculating deposition from domestic sources was not used in this study.

Table 1
Characteristics of the Hietajärvi site

Location	63°10′N 30°43′E
Total catchment area	461.6 ha
Elevation	165–214 m a.s.l.
Forested area	56%
Non-forested land area	21%
Lake area	23%
Area of peat	34%
Geology	Acidic granitoids
Soils	Fibric Histosols, Haplic and Ferric Podzols
Dominant vegetation	Scots pine
Forest age	100–200 years
Lake pH (present)	6.6

2.2.2. DEPUPT

The DEPUPT model derives site-specific historical and future values of annual deposition and forest nutrient uptake values (Johansson et al., 1996; Forsius et al., 1998a). It is based on the model MAKEDEP (Alveteg et al., 1998), modified concerning sulphur deposition history and the contribution of the wet, dry, anthropogenic and marine deposition components.

In order to reconstruct the deposition history at a particular site, the total observed deposition is divided into four components: the dry and wet fractions of the contribution from marine or from non-marine sources. Observations of SO_4^{2-}–S, Cl^{-}, NO_3^{-}–N, NH_4^{+}–N, Na^{+}, K^{+}, Ca^{2+} and Mg^{2+} in bulk open and through fall deposition are input to DEPUPT. All Na^{+} is assumed to be associated with marine deposition and all nitrogen in deposition is assumed to be of anthropogenic origin. The contribution of sea spray is assumed to remain constant in time. The dry components of marine SO_4^{2-}–S, Cl^{-}, Na^{+}, K^{+}, Ca^{2+} and Mg^{2+} are affected by forest filtering, resulting in a higher deposition to forest floor than in open land. The dry components of the non-marine fractions of these ions, as well as NO_3^{-}–N and NH_4^{+}–N, are also proportional to canopy biomass (Alveteg et al., 1998). All non-marine components are assumed to be proportional to standard historical deposition curves of sulphur (Mylona, 1993) and nitrogen (Asman and Drukker, 1988; Alveteg et al., 1998).

The uptake due to forest growth is based on the biomass density, element content and annual increment of each tree compartment: stem over bark, branches and needles. The forest growth of the whole catchment is described with the use of an average stand. The average biomass of stemwood including bark W_s (kg ha^{-1}) is given by Eq. (1)

$$W_s = \rho_s \cdot V_s, \tag{1}$$

where ρ_s (kg m^{-3}) is stemwood density, and the volume V_s (m^3 ha^{-1}) of stemwood is calculated in Eq. (2)

$$V_s = \frac{V_{s\,max}(t-t_0)^a}{(t-t_0)^a + b^a}, \tag{2}$$

in which $t(a)$ is time; $V_{s\,max}$ (m^3 ha^{-1}) is the maximum stem volume, a, and b are parameters, which determine the form of the growth curve, and $t_0(a)$ is the year in which the growth of the stand is assumed to begin. Potential annual growth is calculated by differentiating Eq. (2). The volume of branches is assumed to be a constant fraction (c) of the volume of the stem including bark. The average canopy biomass W_n (kg ha^{-1}) is given by

$$W_n = \frac{W_{n\,max}(d(t-t_0))^a}{(d(t-t_0))^a + b^a}, \tag{3}$$

in which $t(a)$ is time; $W_{n\,max}$(kg ha^{-1}) is the maximum needle biomass, and d is an additional parameter, which allows needle growth to achieve its maximum earlier than stem growth. Potential annual canopy growth is calculated by differentiating Eq. (3). The root compartment is not described and no interaction with soil N stores is assumed to take place.

The growth of the forest is assumed to be nitrogen limited, and the only source of nitrogen considered is atmospheric deposition. If the available atmospheric nitrogen is not enough to satisfy the potential growth, the annual calculated growth of stem, branches and needles is reduced in proportion to the deficiency in nitrogen. DEPUPT is calibrated with forest age and volume by choosing the values of the parameters t_0, a, b and d such that the observed stem growth and a feasible value of V_{max} are obtained, and that the growth curves of stem and needles are realistic (Koivisto, 1959).

The uptake of base cations and nitrogen by the forest is estimated by multiplying the annual growth increment with the element concentrations in biomass. Growth increment and element concentrations are given separately for each species and each biomass compartment. The symbol a_j denotes the fraction of forested area allocated to each species j = 1, …, 3 for pine, spruce and birch, and g_i is the annual growth of each compartment i = 1, …, 3 for stem including bark, branches including bark, and needles. Using the index x = Ca, Mg, K, N to denote each element, and the symbol c_{xij} for the uptake of each element x by any compartment i of any species j, we can write the equation for the uptake u_x of each element

$$u_x = \sum_{j=1,\ldots,3} \left(a_j \sum_{i=1,\ldots,3} g_i c_{xij} \right). \tag{4}$$

The annual values of uptake of base cations and nitrogen are then used to drive the model SMART, together with annual deposition values.

2.2.3. Simulation model for acidification's regional trends

The SMART model (De Vries et al., 1989) is a single layer soil model, which calculates the dynamics of soil effective base saturation and concentrations of major anions and cations in soil solution and catchment runoff. The model is based on the charge balance principle and the concept of anion mobility (Reuss et al., 1987). Mineral weathering is included at a constant rate, Gaines-Thomas equations regulate the exchange reactions, and sulphate adsorption is described by a Langmuir isotherm. SMART simulates the exchange of protons and Al^{3+} with the total amount of Ca and Mg in soil solution and on the exchange sites. It has been used to estimate long-term chemical changes in soil and soil water in response to changes in atmospheric deposition (Posch et al., 1993; Kämäri et al., 1995; Forsius et al., 1997, 1998b).

2.3. Data used by the models

2.3.1. Emissions

The year 1990 is used as the base year in the development of emission control strategies within the UN/ECE and the EU, and 1990 emission data especially for the EU is considered the most reliable year because of the completed CORINAIR'90 inventory (European Environmental Agency, 1996). Therefore, 1990 emissions were used in DAIQUIRI to calculate present-day deposition levels. The domestic and European emissions of 1990 approximate the average for the period studied (1988–1994). Average meteorological data for 1985–1994 were used in order to reduce inter-annual variability caused by weather conditions.

The emissions of SO_2, NO_x and NH_3 in the year 2010 were taken from the different future alternatives investigated in the strategy to combat acidification by the European Commission. The emissions corresponding to the scenario of Joint Optimisation of Acidification and Ozone (called C12) were given by Amann et al. (1997) for the 15 countries belonging to the European Union. For the other European countries and for the Current Reduction Plans (CRP) and the Maximum Feasible Reductions (MFR) scenarios, the emissions given by Amann et al. (1996) were used.

2.3.2. Deposition

The modelled historical and future depositions to the Hietajärvi region were derived from the country-level emissions of the EU Acidification Strategy by using the long-range transport matrices provided by EMEP/MSC-W (Barrett et al., 1995) as described in Section 2.2. There are no emission sources in the vicinity of the Hietajärvi region that would have a significant effect on deposition to the site (Ruoho-Airola et al., 1998). Deposition of all acidifying compounds at Hietajärvi originates mainly from sources outside Finland (Barrett et al., 1996; Ruoho-Airola et al., 1998). Future deposition to Hietajärvi is also considered to be mainly determined by the development of emissions from abroad, and long-range transport modelling can be expected to give a reliable estimate of average future deposition levels under various emission control alternatives.

Deposition at a certain site, however, can deviate from the average regional estimates predicted by larger-scale models because of local-scale geographical features or vegetation patterns, for example. Therefore, when deriving site-specific historical and future deposition, it was assumed that the observations would give a more representative estimate for present-day deposition at the site than the large-scale deposition model used in this study. All deposition time series were thus adjusted to observed present-day deposition.

The level of present-day deposition was estimated from measurements of bulk through fall and open deposition (Ukonmaanaho et al., 1998). Annual bulk deposition values were averaged over the years 1988–1994 and through fall at one representative Scots pine (*Pinus sylvestris*) stand over 1988 –1993. The highest values of sulphur and nitrogen deposition were observed in 1988, and the lowest in 1993. The total depositions to the area, calculated by weighting the bulk deposition by the open land area and the through fall by the forest area are given in Table 2.

2.3.3. Leaching

Stream water quality was sampled and analysed according to methods outlined by the Integrated Monitoring Manual 1993–1996 (EDC, 1993). Annual average concentrations of major cations and anions in runoff water were calculated by weighing monthly concentration values with their corresponding discharge values. The annual concentrations were used to calibrate SMART. Annual leaching values were calculated by multiplying the annual average concentrations by total annual discharge. Table 2 gives the averages of the annual values for the years 1988–1993.

2.3.4. Vegetation variables

In the DEPUPT runs, the forests in Hietajärvi were treated as one mixed upland forest stand, consisting of pine (90%), spruce (5%) and birch (5%). Present-day values for stem volume (194 $m^3\,ha^{-1}$) and stem growth (3.5 $m^3\,ha^{-1}\,a^{-1}$) were based on unpublished biomass estimates carried out by the Finnish Forest Research Institute (Starr and Hartman, personal communication). Since the element concentrations of the biomass

Table 2
Annual fluxes in precipitation, bulk deposition and runoff at Hietajärvi

	Bulk deposition			Runoff	
	Average[a]	1988	1993	Observed[b]	Simulated[c]
Precipitation (mm)	620	770	570	422 ± 127	
Na (mg m^2 a^{-1})	76	76	78	474 ± 149	467
K (mg m^2 a^{-1})	86	44	89	182 ± 94	266
Ca (mg m^2 a^{-1})	93	107	85	546	776[d]
Mg (mg m^2 a^{-1})	20	15	20	140	–
NH_4 (mgN m^2 a^{-1})	138	220	65	–	–
NO_3 (mgN m^2 a^{-1})	168	230	114	11 ± 3	31
SO_4 (mgS m^2 a^{-1})	441	638	291	283 ± 83	316
Cl (mg m^2a^{-1})	134	123	116	176 ± 46	191
TOC (gCm^2a^{-1})	–	–	–	2.6	
ANC ($mmol_c$ m^2 a^{-1})	–	–	–	41	
pH (range of obs.)	–	–	–	5.96–6.28	6.28

[a] Estimated total deposition to the catchment area, averaging bulk deposition values over the years 1988–1994 and through fall over the years 1988–1993.
[b] Observed runoff water quality averaged over the years 1988–1993, except for Ca and Mg 1988–1992, and TOC given for 1992.
[c] Simulated runoff water quality for the year 1991.
[d] The sum of Ca and Mg is given as the output of the SMART model.

compartments of the different tree species were not known, we used the latitude-dependent formulation by Olsson et al. (1993) to estimate the content of Ca, Mg and K in stemwood, branches and foliage. The biomass of branches was assumed to be 15% and the biomass of needles 10% of that of stemwood (Kauppi et al., 1995; Vanninen et al., 1996).

2.3.5. Soil variables

The soil acidification model SMART is a so-called lumped parameter model, which means that it uses only one value for each of the parameters cation exchange capacity, soil thickness and bulk density as representative of the whole catchment. Also the variable soil effective base saturation is described in the model by only one value for the whole catchment. Observed values of these variables are available for four plots in the catchment, for the O-horizon and for mineral soil from 0–5, 5–20 and 20–40 cm depth. For the purpose of applying the lumped model, these parameter values were calculated as volume-weighted averages for the solum, including the O-horizon. The average profile values were weighted with the areas the plots were assumed to represent to give an average value for the whole catchment. The estimated present-day catchment average base saturation was 47%. The layer-specific values of effective base saturation ranged from a minimum of 15% for the uppermost mineral soil to 76% for the O-horizon (Starr, 1995). The weathering rate was estimated as described by Johansson and Tarvainen (1997). The value of 34 meq m^2 a^{-1} for the weathering of Ca and Mg is in the upper end of the range obtained by Starr et al. (1998) for Hietajärvi with the Zr reference method.

2.4. Calibration of the models

The total anthropogenic deposition of SO_4–S, NO_3–N and NH_4–N to the site was calculated by DAIQUIRI for the year 1990 and for the year 2010 with the CRP, MFR and C12 emission scenarios (see Section 2.3). The ratio between the anthropogenic fraction of the observed and the 1990 modelled deposition was 0.85 for SO_4–S, 0.75 for NO_3–N and 0.80 for NH_4–N (Table 3). For the year 2010, the modelled deposition values were multiplied by this ratio, based on the assumption that the observed deposition values were more representative of the real deposition than the modelled ones.

The deposition time series 1900–2010 were calibrated to present measured open and through fall deposition values (Table 2) in DEPUPT. Three different sets of deposition observations were used for the calibration to present-day conditions: (i) average observed values for the years 1988–1993 for through fall and 1988–1994 for open deposition; (ii) values for the year 1993, showing the lowest sulphur and nitrogen deposition; and (iii) values for the year 1988, showing the highest sulphur and nitrogen deposition. The purpose of using these different calibration points for the deposition time series was to investigate how the choice of present-day deposition estimates effects the outcome of the simulation. The results would indicate whether it is necessary to use a more accurate value of the present deposition level, and thus what deposition data were required for reliable dynamic modelling analysis.

The parameters (*a*, *b* and *d*) of the growth curve in DEPUPT were chosen to reproduce present-day volume and growth of stem, by looking at the form of the resulting growth curves, taking into account the maximum stem volume estimated for this particular site. Five different assumptions concerning present-day stem volume and growth were used to calibrate DEPUPT: (i) observed volume and growth; (ii) observed volume but a little slower growth; (iii) observed volume but much slower growth; (iv) stem volume 20% larger than observed, with observed growth; and (v) stem volume 20% smaller than observed, with observed growth. The annual uptake of base cations was calculated in DEPUPT from the demand resulting from the varying annual growth and the constant estimated element concentrations in biomass.

Table 3
Total anthropogenic deposition to Hietajärvi for 1990 calculated with DAIQUIRI, compared to the average observed deposition for the years 1988–1994.

	Modelled	Observed/Modelled
SO_4 (mgS m^2 a^{-1})	513	0.85
NO_3 (mgN m^2 a^{-1})	224	0.75
NH_4 (mgN m^2 a^{-1})	172	0.80

The SMART model was calibrated to present-day soil and stream water quality by visual comparison of the model results with the observed soil effective base saturation for the year 1990 and observed annual average concentrations of major cations and anions in stream water for the years 1988–1994 (Bleeker et al., 1994; Ahonen et al., 1998). The calibration procedure involved adjusting values of 14 parameters relating to weathering, SO_4-adsorption, dissociation of organic acids, cation exchange and nitrification. Measurements were available of present-day soil base saturation and C:N ratio in organic matter together with observations of 6 yr of annual volume-weighted average stream water pH and concentrations of SO_4, NO_3, Cl, Ca, Mg, Na and K. Observed TOC was available only for 1992. The modelled 1991 pH value was 6.28 while the observed pH values (1988–1983) were in the range 5.96–6.28 (Table 2). Past simulated pH values were in agreement with only slightly declining pH trends reported as the result of a paleolimnological study at Hietajärvi (Simola et al., 1991). Table 4 summarises the parameters of the calibrated model SMART.

In the runs for calibrating SMART, the deposition history used to drive SMART was adjusted to average present-day deposition observations. The removal of base cations and nitrogen was driven by past forest growth calibrated to observed stem volume and observed forest growth.

2.5. Future deposition and forest growth scenarios

A set of scenarios concerning future deposition and forest growth was explored (Table 5). There were 12 scenarios in all labelled A through K. Five scenarios concern deposition reductions and level of present deposition, three scenarios concern the timing of reductions, and five scenarios concern forest growth and present volume.

Uncertainty in the estimates of present deposition levels was examined by evaluating three sets of deposition values as representative of present conditions. First, the year of the lowest observed sulphur and nitrogen deposition (1993) was chosen as the calibration point for the simulation ‘A’. Second, the year of the most acid deposition (1988) was used to calibrate past and future depo-

Table 4
Parameters of the SMART (De Vries et al., 1989) model application at Hietajärvi

Variable	Unit	Value
Thickness of the soil compartment	M	0.85
Bulk density of soil	g cm^{-3}	1.291
Volumetric water content of the soil	m m^{-1}	0.35
Initial amount of carbonates	meq kg^{-1}	0
Cation exchange capacity of the soil	meq kg^{-1}	3.76
Organic matter content in mineral topsoil	kg kg^{-1}	0.014
Initial C:N ratio in organic matter	–	40
Selectivity coefficient for cation exchange (Al, Ca+Mg)	–	1
Selectivity coefficient for cation exchange (H, Ca+Mg)	–	$10^{7.0}$
Gibbsite equilibrium constant	–	$10^{8.2}$
Initial effective base saturation		0.83
Nitrification factor		1
Denitrification factor		0.5
Initial Al buffer capacity	meq kg^{-1}	10^{6}
Maximum SO_4 adsorption capacity	meq kg^{-1}	3
Half-saturation coefficient for SO_4 adsorption	eq m^{-3}	0.1
Total concentration of organic acids	eq m^{-3}	0.048
Three parameters for modelling pK_a		4.5/0/0
Net precipitation	m a^{-1}	0.4224
CO_2 pressure in soil solution (multiple of pCO_2 [atm] in air)		20
Weathering rate of Ca+Mg	eq m^{-3} a^{-1}	0.040
Weathering rate of K	eq m^{-3} a^{-1}	0.008
Weathering rate of Na	eq m^{-3} a^{-1}	0.023

sition values (simulation 'E'). For both scenarios, the deposition in the year 2010 was calculated by multiplying the deposition corresponding to the C12 emissions with the Observed/Modelled ratio (Table 3). For all other simulations, the average observed values (1988–1994) were used for the present conditions.

The low future deposition scenario 'C' was based on the MFR emissions, the high future scenario 'D' on the CRP emissions and the medium future deposition scenario 'G' on the C12 emissions. For these scenarios, the Observed/Modelled ratio was also used to scale the future deposition to the observed present level. All five deposition reduction scenarios (A, C, D, E, and G) were calculated assuming that the reductions began immediately and become totally effective by the year 2010, with no further reductions taking place thereafter. The base case forest growth scenario (growth curve that was calibrated to the present forest growth and stem volume) was used with all these deposition scenarios.

To explore the impact of the timing of the reductions, the target year of the C12 emission scenario was varied. In the early reduction scenario 'F', all reductions become effective immediately (1998). The late reduction scenario 'H', assumes constant present deposition until the year 2020, when the reductions take place. The linear reduction scenario 'I' assumes a linear decrease in deposition between 1998 and 2020. The growth curve that was calibrated to the present forest growth and stem volume was used also with these three reduction timing scenarios.

In five different forest growth reconstructions, the form of the growth curves and the present volume were varied. The objective was to represent extreme cases of present fast and slow growth (Fig. 1). The fast growth scenario (F1) was represented by the growth curve that was calibrated to present-day stem volume and growth. At the other extreme, the slow growth scenario (F3), was based on a low maximum stem volume and form parameters giving a small annual increment (Table 6). To investigate the impact of variability in the estimates of present forest stem volume, the growth curve 'F4' used observed stem volume +20% and 'F5' observed −20% to represent present conditions. All five forest growth scenarios were combined with the medium deposition scenario, with linear decrease of deposition from 1998 to 2010.

To illustrate the relative impact of the different sets of scenarios, the standard deviation of the simulated soil effective base saturation values in each set was determined for each year. The annual standard deviation, σ_s, corresponding to each set, s, of effective base saturation values was calculated according to

Table 5
Scenarios used in the SMART model

Driving variables	Scenario[a]											
	A	B	C	D	E	F	G	H	I	J	K	L
Forest growth	F1	F3	F1	F1	F1	F1	F1	F1	F1	F2	F4	F5
Deposition	C12_93	C12	MFR	CRP	C12_88	C12	C12	C12	C12	C12	C12	C12
Timing	2010	2010	2010	2010	2010	1998	2010	2020	2020	2010	2010	2010

[a] A: Low depos. Deposition scaled to low observations (1993), C12 from 2010 on, base case forest growth; B: Slow growth Deposition scaled to average obs. (1988–1994), C12 from 2010 on, slow forest growth; C: MFR Deposition scaled to average obs. (1988–1994), MFR from 2010 on, base case forest growth; D: CRP Deposition scaled to average obs. (1988–1994), CRP from 2010 on, base case forest growth; E: High depos. Deposition scaled to high observations (1988), C12 from 2010 on, base case forest growth; F: Early red. Deposition scaled to average obs. (1988–1994), C12 from 1998 on, base case forest growth; G: C12 Deposition scaled to average obs. (1998–1994), C12 from 2010 on, base case forest growth; H: Late red. Deposition scaled to average obs. (1988–1994), no reductions until 2020 (C12), base case forest; I: Linear red. Deposition scaled to average obs. (1988–1994), linear reductions 1998–2020 (C12), base forest; J: Medium gr. Depos. scaled to average obs. (1988–1994), C12 from 2010 on, medium forest growth; K: Vol. +20% Depos. scaled to average obs., C12 from 2010 on, forest volume scaled to observed +20%; L: Vol. −20% Depos. scaled to average obs., C12 from 2010 on, forest volume scaled to observed −20%.

$$\sigma_s = \sqrt{\frac{n_s \sum_{i=1,\ldots,n_s} \mathrm{BS}_i^2 - \left(\sum_{i=1,\ldots,n_s} \mathrm{BS}_i\right)^2}{n_s}}, \qquad (5)$$

where BS_i is the annual simulated soil effective base saturation value calculated for each scenario (i) in the set s of different scenarios (s = *deposition scenarios*; *forest growth scenarios*; *timing of reduction scenarios*).

3. Results

3.1. Historical and future deposition estimates

The deposition of sulphur (Fig. 2) and nitrogen compounds at Hietajärvi shows a clear peak around 1970–1980 and decreases towards 2000. The lowest deposition scenario (A) leads after 2010 to sulphur deposition values that are as low as the estimates for the beginning of the 20th century. The difference between the highest and lowest level of sulphur deposition in 2050 (18 meq $m^2 a^{-1}$) is large compared with the estimated weathering rate for Hietajärvi of 34 meq $m^2 a^{-1}$, given by Starr et al. (1998).

3.2. Simulated soil effective base saturation

The simulated soil effective base saturation decreases throughout the 20th century in response to forest growth and acid deposition (Fig. 3). The simulated BS values do not all coincide at present. Only the BS values that were obtained with the

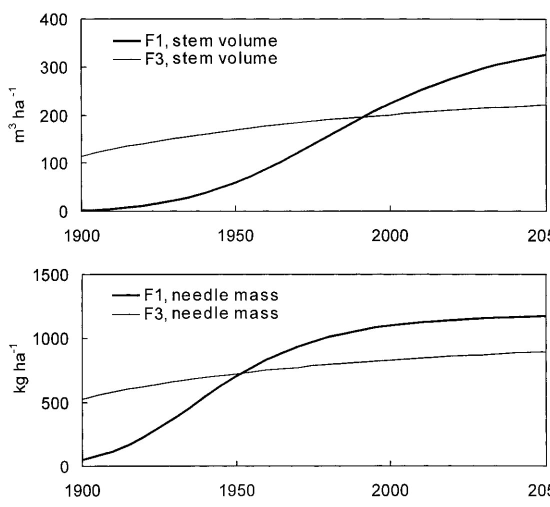

Fig. 1. Stem volume and needle biomass simulated at Hietajärvi with DEPUPT model.

Table 6
Parameters of forest growth equations in DEPUPT applied to Hietajärvi for five forest growth scenarios

	F1	F2	F3	F4	F5
Volume of stemwood V_s (m^3 ha^{-1}) in 1994	194	194	194	233	155
Assumed density of stemwood ρ_s (kg m^{-3})	400	400	400	400	400
Assumed year of beginning of growth t_0	1850	1850	1850	1850	1850
Assumed max. vol. of stemwood $V_{s\,max}$ (m^3 ha^{-1})	381	381	320	381	381
Assumed max. needle biomass $W_{n\,max}$ (1000 kg ha^{-1})	12	12	12	12	12
Assumed volume of branches as % of stem volume	15	15	15	15	15
Parameter a	5	2	1	5	5
Parameter b	140	140	90	140	140
Parameter d	1.5	1.5	1.5	1.5	1.5

scenarios 'C'and 'D' match the observed catchment average BS at present. This is because only the most realistic deposition history and the most probable forest growth reconstruction were used to calibrate SMART. The values obtained for the SMART model parameter in the calibration were then used in all other runs as well. Running SMART with the calibrated parameter values but different past deposition and past forest growth than what was used in the calibration yielded a variation in past and present soil base saturation values.

The deposition time series 'A', obtained by assuming the lowest measured annual depositions to be representative of the present level, gave the highest effective base saturation values throughout the simulation period. Future effective base saturation values obtained with this scenario are substantially higher than those given by any other scenario. The highest deposition time series 'E' gave the lowest effective base saturation values over the whole simulation period. In the future, the result of this scenario coincides with that of the 'D' scenario, using average observed present deposition and highest future emissions (CRP). The results of all other scenarios lie between these extremes.

The future recovery predicted for soil base saturation with the low future emissions (MFR) and the future decline resulting from the high future emissions (CRP) encompass the stabilisation following the Joint Optimisation of Acidification and Ozone (C12) emissions in scenario 'G'. The three scenarios concerning the timing of reductions (F, H, I) also give soil base saturation values that fall between those obtained by varying the level of reduction.

The very slow and the medium growth scenarios 'B' and 'J', both gave higher effective base saturation values in the near future than those obtained with 'C' using the lowest future emissions (MFR). The lowest future emissions, however, led to a

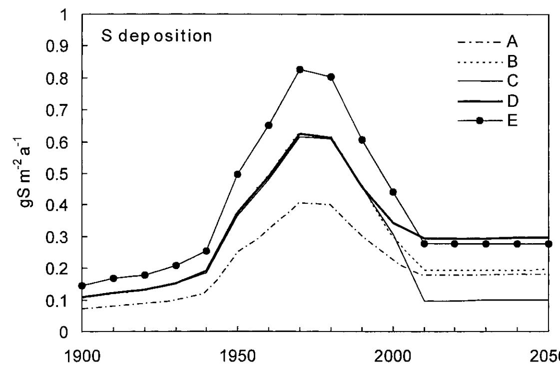

Fig. 2. Deposition of sulphur simulated for Hietajärvi with DEPUPT.

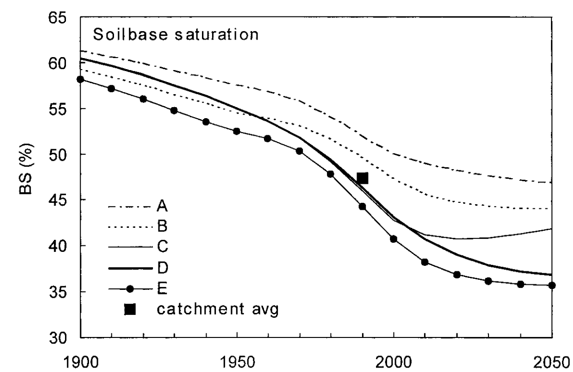

Fig. 3. Soil effective base saturation simulated for Hietajärvi with SMART.

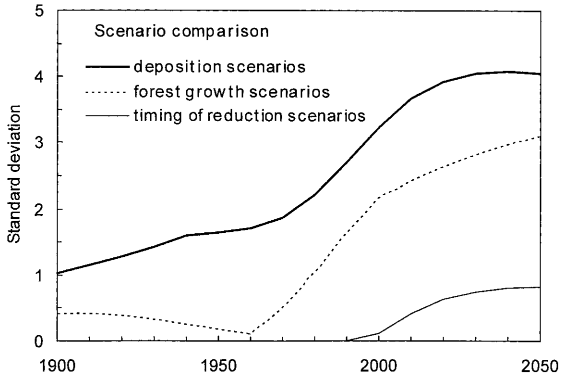

Fig. 4. Standard deviation of soil effective base saturation for five deposition scenarios (A, C, D, E, G), five forest growth scenarios (B, G, J, K, L) and three timing scenarios (F, H, I) (see Table 5 for scenario descriptions).

faster recovery than the slow growth scenarios, and would stabilise at a higher soil base saturation.

3.3. Comparison of impacts

In terms of the standard deviation of the soil effective base saturation calculated for each set of driving variables, the assumptions concerning present and future deposition at the site introduced the largest source of variability (scenarios A, C, D, E, G; Fig. 4). Because of the temporal pattern assumed for the driving variables, the future variability is larger than the past variability for all the sets.

The five forest growth scenarios (B, G, J, K, L) had a smaller impact than the deposition scenarios, but larger than the timing scenarios. Varying the present stem volume with $\pm 20\%$ (K, L) hardly showed any difference on the future effective base saturation values.

The impact of the timing of the reductions (F, H, I) was smaller than the impact of the level of deposition reduction or the impact of forest growth.

4. Discussion

The weakest link in the DAIQUIRI-DEPUPT-SMART chain may be the lack of feedback from soil status to forest growth. Without the dynamic link between soil and vegetation any long-term predictions of soil or forest properties are bound to lack in realism. The models used in this work are not easily modified to account for this link, but there are other strategies that may be useful. Competition among trees for light and nutrients was used by Hauhs et al. (1995) to model adaptation to external input fluxes over several tree generations, who conclude that neglecting adapted variability in growth response to soil conditions may mislead an assessment of acid deposition effects. Forest growth and element cycling in forest, soil and aquatic ecosystems were successfully simulated by Krám et al. (1999) with a forest productivity model coupled to a model of equilibrium processes in the soil.

Although it may not be possible to demonstrate the predictive reliability of any model of a complex natural system in advance of its actual use (Oreskes et al., 1994; Oreskes, 1998), it is clear that much information can be gained by a thorough assessment of model quality. Guidelines for testing model results against the real world have been given by several authors (Klepper and Hendrix, 1994), with recommendations of careful use and interpretation of quantitative techniques together with qualitative estimation (Janssen and Heuberger, 1995), and suggestions of how to overcome the uncertainty of experimental data (Monte et al., 1996), as well as methods for testing hypotheses instead of the goodness-of-fit against experimental data (Loehle, 1997). Tiktak and van Grinsven (1995) recommend that simulation results from simple, lumped models should be compared with those obtained by detailed, mechanistic models that, in turn, can be compared with results from field observations.

Decisions to use models for predictive or management purposes are, however, often based on other factors than objective analysis of model performance (Vanclay and Skovsgaard, 1997). In acid rain modelling, uncertainty and sensitivity analyses are not standard procedures (Hordijk and Kroeze, 1997), not even for integrated assessment models that are developed as tools to assist policy makers in evaluating different abatement options. Acidification models are often poorly identifiable, because of the complex nature of the processes involved. The application of Bayesian techniques for the identification of model parameters may provide a solution (Reichert and Omlin, 1997; Omlin and Reichert, 1999).

Here, the soil acidification model SMART was calibrated by adjusting the values of 14 parameters in order to obtain a visually satisfying fit with observations from six consecutive years of the values of eight components of stream water quality and one observation of soil base saturation. The model performance was judged to be useful for this application because the simulated pH history accorded with the diatom-inferred pH history of the lake in the catchment. The year to year variation in the 6 years of observed stream water quality had only a small impact on the calibration over the time horizon 1900–2050. It might therefore be argued, that too few observations were available to thoroughly evaluate the performance of SMART at Hietajärvi. We believe, however, that the application is useful for analysing the sensitivity of the model to the driving variables deposition and nutrient uptake.

Although the site receives low deposition in comparison with less remote areas and the forests have been growing unmanaged, the assumptions concerning anion mobility and cation exchange incorporated into SMART yields a declining soil base saturation in response to the combined effect of acid deposition and forest growth.

The year chosen to represent present deposition had a large effect on the results. This is understandable in the light of the large year-to-year variation in deposition observations at this site. Such large variation is not uncommon, and emphasises the importance of long-term monitoring. In this case, the observed 6-years average deposition values were closer to the modelled values for 1990, given by the long-range transport matrices provided by EMEP/MSC-W, and incorporated into DAIQUIRI. This is a point in favour of using modelled deposition values rather than observations of only 1 or 2 years.

Calibrating the deposition time series on the basis of only one years' measured deposition caused significant variations in the modelled soil effective base saturation, depending on the choice of year. Using the average of all deposition observations in calibrating model runs that were produced with average meteorological data yielded deposition histories between these extremes. The use of observations averaged over several years as representative of the current deposition level reduces bias in the estimated deposition time series caused by single, extreme years. A further calibration of the modelled deposition could be to calculate the modelled depositions with emissions and meteorological variables corresponding to the same year. If this was done for all the years for which observations are available, the modelled deposition time series could then be adjusted according to the average of the annual ratios between observations and model results.

Lumped-parameter models that describe the whole catchment in terms of only one soil profile are sensitive to the way in which the soil variables are derived. The base case deposition runs were calibrated to a catchment-average value of present-day soil effective base saturation. In this case, this value was 47%, which compares with an observed minimum value of 15% for the uppermost mineral soil (0–5 cm) and an observed maximum of 76% for the O-horizon. Another procedure for calculating the catchment-average present-day base saturation, for instance, including values for peatland, would have resulted in a different estimate for the present-day value to which SMART was to be calibrated. The effect of different present-day base saturation values on the modelled results was not quantified in this exercise. Varying the present-day deposition estimates (in runs 'A' and 'E'), resulted in simulated present-day base saturation values which varied from 40% to 52%.

Without the assumed past differences in deposition and forest growth the future soil base saturation variability would be smaller. The past variability in soil base saturation, introduced by varying past deposition and forest growth, is not, however, as large as the future variability caused by assuming different future depositions.

This exercise focussed on the impacts of deposition and forest growth on soil effective base saturation values only. Results concerning the effects on stream water pH and ANC would probably have been similar, except that the timing of reductions may be expected to have had a larger impact because of faster reaction times to changes in the driving functions compared to soil variables.

5. Conclusions

The response of a forested catchment to changes in deposition and forest growth was studied in terms of the simulated soil effective base saturation. We compared three sets of driving variables: deposition reduction scenarios, reduction timing scenarios, and forest growth scenarios. The impact of each set of driving variables was analysed in terms of the standard deviation calculated for the annual effective base saturation values.

The simulated soil effective base saturation values obtained under varying future emissions and the timing of the emission reductions all lay between the two extreme simulations that resulted from using the year corresponding to the highest and the lowest observations of sulphur and nitrogen depositions. The variability introduced by varying forest growth scenario and present stem volume was smaller than that given by choosing different years to represent present deposition. In the light of these results, it seems preferable to calculate present average deposition levels using observations for as many years as possible. When predictions are made on the basis of only a few years of deposition observations, careful consideration should be given to whether the calibration of future deposition to the observed present values ought to be done or not.

Acknowledgements

We thank Michael Starr and two anonymous readers for reviewing the manuscript and Sirkka Vuoristo for drawing the graphs. The EU Financial Instrument for the Environment (LIFE) is acknowledged for financial support of this work (LIFE95/FIN/A11/EPT/387). The monitoring of the catchment was performed under the UN/ECE Integrated Monitoring programme. Country-specific emissions were provided by IIASA in their Interim Reports to the European Commission concerning the European Acidification Strategy. The EMEP/MSC-W is gratefully acknowledged for the provision of the long-range transport matrices and the historical sulphur deposition data.

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