SENSITIVITY ANALYSIS OF A REGIONAL SCALE SOIL ACIDIFICATION MODEL

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PREFACE

The IIASA "Acid Rain" project started in 1983 in order to provide the European decision makers with a tool which can be used to evaluate policies for controlling acid rain. This modeling effort is part of the official cooperation between IIASA and the UN Economic Commission for Europe (ECE). The IIASA model currently contains three linked compartments: Pollution Generation, Atmospheric Processes and Environmental Impact. Each of these compartments can be filled by different substitutable submodels. The soil acidification submodel is part of the Environmental Impact compartment.

A model which is intended for use in decision making, deserves a vigorous testing program to strengthen the confidence of model users in its estimates. Such a program is currently underway at IIASA to test the model system. Part of the approach involves conventional model validation and verification. A less conventional approach is also being taken by acknowledging that model uncertainty exists and that it should be incorporated explicitly in the model. This paper describes results of sensitivity tests on the soil acidification submodel.

Leen Hordijk
Project Leader

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ABSTRACT

A dynamic model has been introduced for describing the acidification of forest soils. In one-year time steps the model calculates the soil pH as a function of the acid stress and the buffer mechanisms of the soil. Acid stress is defined as the hydrogen ion input into the top soil. The buffer mechanisms counteract acidification by providing a sink for hydrogen ions. The concepts buffer rate and buffer capacity are used to quantify the buffer mechanisms. The model compares (i) the rate of acid stress (annual amount) to the buffer rate, and (ii) the accumulated acid stress (over several years) to the buffer capacity. These two types of comparisons produce an estimate of the soil pH.

The model has been incorporated into the RAINS model system of the International Institute for Applied Systems Analysis for analyzing the acidic deposition problem in Europe. The data on acid stress, entering the soils, is obtained from the other submodels. Data on buffer rate and buffer capacity has been collected from soil maps and geological maps.

The sensitivity of the model to the forcing functions, parameter values and initialization of the soil variables is evaluated in this paper. The model's sensitivity to initial base saturation appears to be crucial. Base saturation varies widely in forest soils, while the variation of, e.g., total cation exchange capacity is normally not more than ± 50% of the average. Whenever possible, recent measurements about the status of the soil should be used.

The difference of acid stress and the buffer rate of silicates determines whether the soil alkalinizes or acidifies. The sensitivity of the model to that difference varies in time and space, being highest in areas where the deposition rate nearly equals the silicate buffer rate, e.g. at present in Scandinavia.

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1. Introduction

Soil acidification is considered as one important link between air pollution and forest damage. The ability of the soil to buffer acid deposition is also a key factor in controlling the surface water and groundwater acidification. Therefore soil acidification was considered a suitable starting point for IIASA's Acid Rain Project for evaluating environmental impacts of acid precipitation in Europe. The overall objective of the project is to develop a framework, which would assist in comparing the cost and effectiveness of different pollution control strategies (Alcamo et al., 1985).

The aim of this paper is to test the sensitivity of the soil acidification model. The modeling of soil acidification in the IIASA RAINS (*R*egional *A*cidification *In*formation and *S*imulation) model system is based on the description of proton consumption reactions presented by Ulrich (1981, 1983). The uncertainty in the model structure, i.e. in the underlying theory, is not considered in this paper. We restrict ourselves to the evaluation of the sensitivity in the forcing functions, parameter values and initialization of the soil variables.

2. Soil Acidification

Soil acidification has been defined as a decrease in the acid neutralization capacity (van Breemen et al., 1984). The acidification is caused by acid stress, which is defined as the input of hydrogen ions into the top soil. The acid stress due to air pollution can result from the direct deposition of hydrogen ions or from the indirect effect of acid producing substances, such as the dry deposition of sulfur compounds.

A consecutive series of chemical reactions has been documented in soils, in which acidification proceeds. Information regarding the dominant reactions counteracting acid stress has been used for defining categories, called buffer ranges. Buffering in each range can be described using two variables, buffer capacity (*BC*, *kmol ha* $^{-1}$), the gross potential, and buffer rate (*br*, *kmol ha* $^{-1}$ *yr* $^{-1}$) for the rate of the reaction. They can be quantified for any volume of the soil. In the following paragraphs we briefly describe the different buffer ranges. The original description can be found in Ulrich (1981, 1983).

Calcareous soils are classified into the carbonate buffer range (pH $\geq$ 6.2). Its buffer capacity is proportional to the amount of $CaCO_3$ in the soil. The buffer rate, i.e. the dissolution rate of $CaCO_3$, is high enough to buffer any occurring rate of acid stress.

If there is no $CaCO_3$ in the fine earth fraction and the carbonic acid is the only acid being produced in the soil, the soil is classified into the silicate buffer range (6.2 > pH $\geq$ 5.0). Buffering is based on weathering of silicates. The buffer capacity is high (practically infinite considering a time horizon of hundreds of years), but the buffer rate is quite low. The weathering of silicates occurs throughout all buffer ranges. The switch to lower buffer ranges implies, that the weathering rate of silicates is not sufficient to buffer all the incoming stress.

When the cation exchange reactions play the major role in the acid buffering, the soils are classified into the cation exchange buffer range (5.0 < pH $\leq$ 4.2). The acid stress not buffered by the silicate buffer range is adsorbed in the form of H^+- or Al^{3+}-ions at the exchange sites, thus displacing the base cations. The buffer rate (= rate of the cation exchange reactions) is high, effectively counteracting any occurring acid stress. The buffer capacity, CEC_{tot}, is generally rather low, depending mainly on the soil texture. The remaining buffer capacity at any given time is expressed by base saturation, the percentage of base cations of the total cation exchange capacity.

When base saturation decreases below 5-10%, the soils are classified into the aluminum buffer range (4.2 < pH < 3.8). H^+-ions are consumed by releasing aluminum, mainly from clay minerals. High Al^{3+}-concentrations characterize the soil solution and may cause toxic effects to bacteria and plant roots. The buffer capacity is almost infinite due to the abundance of aluminum compounds in the soil. The decrease of pH below 3.8 implies increasing solubility of iron oxides and the soil is classified into the iron buffer range, although in quantitative terms aluminum may still act as the dominant buffer compound.

3. The Model

The model describes soil acidification in terms of a sequence of buffer ranges. The model compares (i) the amount of acid stress accumulated over the course of time to the buffer capacity, and (ii) the stress rate, the time derivative of the amount of stress, to the buffer rate. As the buffer capacity of silicates is very large, only the buffer rate is compared in that range. The buffer rates of carbonate and cation exchange range are always high enough to counteract any occurring stress rate. Thus, only the capacities of these ranges have to be considered.

Within one time step the capacity of the cation exchange buffer system, BC^t_{CE}, is depleted by the difference of the acid stress rate, as^t, and the buffer rate of silicates, br_{Si} (Eq.1). At pH-values between 5.6 and 4.0 a non-linear relationship is assumed between base saturation and soil-pH within the silicate, cation exchange and the upper aluminum buffer range, as long as $BC^t_{CE} \geq 0$ (Eq.2)

$$BC^t_{CE} = BC^{t-1}_{CE} - (as^t - br_{Si}) \tag{1}$$

$$pH = 4.0 + 1.6\ (BC^t_{CE} / CEC_{tot})^{3/4} \tag{2}$$

The shape of the pH - base saturation relationship has been adopted from results of an equilibrium model by Reuss (1983).

If $BC^t_{CE} = 0$, equilibrium with gibbsite is assumed. As precipitation infiltrates into the soil and mixes with the soil solution, disequilibrium concentrations $[Al^{3+}]_s$ and $[H^+]_s$ are obtained

$$[Al^{3+}]_s = V_f [Al^{3+}]^{t-1} / \left[V_f + (P-E)\right] \tag{3}$$

$$[H^+]_s = \left[V_f [H^+]^{t-1} + (as^t - br_{Si})\right] / \left[V_f + (P-E)\right] \tag{4}$$

where V_f is the volume of soil solution at field capacity and P and E mean annual precipitation and evapotranspiration, respectively. The soil solution volume is simply defined by

$$V_f = \theta_f z \tag{5}$$

The soil thickness, z, is fixed to 50 cm and the volumetric water content value at field capacity, θ_f, is estimated separately for each soil type based on the grain size distribution in the soil. Aluminum is dissolved or precipitated until the gibbsite equilibrium state is reached (Eq.6). This process involves a change from disequilibrium concentrations as defined in Eq.7

$$[Al^{3+}]^t / \left[[H^+]^t\right]^3 = K_{so} \tag{6}$$

$$3\left[[Al^{3+}]_s - [Al^{3+}]^t\right] = [H^+]^t - [H^+]_s \tag{7}$$

Combining Eqs.6 and 7 yields a third order equation which has a single real root

$$3K_{so}\left\{[H^+]^t\right\}^3 + [H^+]^t - 3[Al^{3+}]_s - [H^+]_s = 0 \qquad (8)$$

The main characteristics of the model are summarized in the flow chart given in Figure 1 and described as well as demonstrated in more detail in Kauppi et al. (1985a,b).

4. Screening

In this section we will screen all the input variables, parameters and forcing functions in order to find out, which of them should be looked at in more detail.

4.1. Dominant Soil Types

IIASA's soil acidification model deals with forest soils only. To focus the sensitivity analysis on the most important soil types, the soils were ranked according to their coverage of the total forest area in Europe Three soil types -- Orthic Podzol (Po), Eutric Podzoluvisol (De) and Orthic Luvisol (Lo) -- are estimated to comprise over 50% of the total forested area of Europe (see Table 1). These three soil types will be used for testing the sensitivity of the model to varying parameter values and forcing functions.

4.2. Soil Parameters

The model requires initial values for the following soil parameters: carbonate buffer capacity, BC_{Ca}, silicate buffer rate, br_{Si}, total cation exchange capacity, CEC_{tot}, base saturation, β, and volumetric water content at field capacity, θ_f. Since all the three dominant soil types (Po, De and Lo) are non-calcareous, BC_{Ca} can be neglected. θ_f is used in the model only when calculating equilibrium concentrations in the aluminum buffer range. Testing the sensitivity of the H^+-concentrations (given by Eqs.3-8) for a range of θ_f-values of 0.05-0.30 it was found that the effect of varying θ_f on the result is negligible. The soil parameters to be looked at are therefore br_{Si}, CEC_{tot}, and β.

4.3. Atmospheric Parameters

The model is driven by two forcing functions: acid deposition and net precipitation. The above mentioned three main soil types (Po, De and Lo) occur in quite different parts of Europe. Orthic Podzols dominate in Scandinavia, but are almost absent elsewhere, while Eutric Podzoluvisols and Orthic Luvisols are typical forest soils in Central Europe (Figures 2,3 and 4). Because acid deposition in Scandinavia is generally lower than in Central Europe the typical acid stress on Orthic Podzols is lower than on the two other soil types (Figure 5). Thus for Po 2 $kmol\ ha^{-1}yr^{-1}$ was used as a high stress rate, while 4 $kmol\ ha^{-1}yr^{-1}$ was used for De and 6 $kmol\ ha^{-1}yr^{-1}$ for Lo. The low values used were 0.5 $kmol\ ha^{-1}yr^{-1}$ for Orthic Podzol and 1 $kmol\ ha^{-1}yr^{-1}$ for Eutric Podzoluvisol and Orthic

Table 1. Dominant forest soil types in Europe. The soils are ranked according to their coverage of the total forest area in Europe.

Soil	Symbol	%	$\sum$ %	$\beta \cdot CEC_{tot}$	BC_{Ca}
Orthic Podzol	Po	26.39	26.39	78.0	0.
Eutric Podzoluvisol	De	14.22	40.61	136.5	0.
Orthic Luvisol	Lo	10.72	51.34	107.3	0.
Dystric Cambisol	Bd	8.13	59.47	165.8	0.
Dystric Histosol	Od	5.84	65.31	72.0	0.
Gleyic Luvisol	Lg	5.44	70.75	146.3	0.
Leptic Podzol	Pl	3.13	73.88	68.3	0.
Dystric Podzoluvisol	Dd	2.80	76.69	136.5	0.
Eutric Cambisol	Be	2.41	79.09	1824.0	500.0
Haplic Chernozem	Ch	1.76	80.86	390.0	0.
Humic Cambisol	Bh	1.66	82.51	136.5	0.
Chromic Cambisol	Bc	1.25	83.76	1225.0	500.0
Calcic Cambisol	Bk	1.18	84.94	1470.0	25000.0
Chromic Luvisol	Lc	1.16	86.10	170.7	3000.0
Humic Podzol	Ph	1.03	87.13	49.0	0.
Gleyic Podzol	Pg	0.81	87.94	180.0	0.
Haplic Kastanozem	Kh	0.75	88.68	136.5	0.
Lithosol-...	I-U	0.68	89.36	136.5	0.
Vertic Cambisol	Bv	0.64	90.00	2210.0	0.
...-Gelic Regosol	I-Re-Rx	0.62	90.62	106.8	0.
Rendzina	E	0.46	91.08	2600.0	20000.0
Eutric Regosol	Re	0.45	91.53	136.5	0.
	I-Bc-Lc	0.43	91.96	469.1	1500.0
Luvic Chernozem	Cl	0.41	92.36	419.3	0.
Ranker	U	0.41	92.77	136.5	0.
Mollic Gleysol	Gm	0.40	93.17	183.8	0.
Calcaric Regosol	Rcc	0.40	93.57	857.5	500.0
Luvic Phaeozem	Hl	0.38	93.95	312.0	0.
	I-Bd	0.38	94.32	151.2	0.
	I-Be	0.37	94.69	765.6	0.
Calcaric Fluvisol	Jc	0.37	95.06	315.0	8000.0

Luvisol.

Also precipitation, P, and evapotranspiration, E, which enter as a driving function in the aluminum buffer range, vary significantly over Europe. In our data base, which was derived from a 30 year climatic mean statistics of 253 stations in Europe, the Sovjet Union and Northern Africa (Müller, 1982), the annual net precipitation, $P - E$ received by the soil type Po, ranges from 95 *mm* to 1950 *mm* (mean: 430 *mm*), while for De the range is 127-365 *mm* (mean: 263 *mm*) and for Lo 67-1590 *mm* (mean: 285 *mm*). In the sensitivity tests 300 *mm* was considered a typical net precipitation for Po, and the range used was 100-700 *mm*; for De 270 *mm* (range: 100-400 *mm*) and for Lo 200 *mm* (range: 100-600 *mm*) were chosen as typical values (Figure 6).

The local stress rate in forests resulting from a given regional mean of sulfur deposition may vary significantly. Forests are known to absorb air pollutants more effectively than open land, and estimates of this filtering factor, φ, vary from 1.1 to 4.0 (Table 2). Secondly, part of the acid stress deposited is accompanied by airborne dust, and other impurities which contain significant amounts of base cations. Depositing base cations contribute to the exchangeable base cations in the soil and therefore the estimated base cation equivalents have to be subtracted from the calculated sulfate equivalents. This phenomenon is especially important in areas where dry deposition comprises a significant part of the total sulfur deposition. According to the literature the acid stress parameter, σ, expressing the fraction of acid stress that is not counteracted by base cation deposition, varies between 0.56 and 0.78 (Table 3).

Table 2. φ-values calculated from local observations on bulk deposition and total deposition to forest floor measured as throughflow+stemflow. Bulk deposition is assumed to represent deposition to open field.

Species	φ	Location	Reference
Quercus-Carya	4.0	Walker Branch, USA	Schriner & Henderson (1978)
Fagus-Acer	2.4	Hubbard Brook, USA	Likens et al. (1977)
Fagus	2.1	Solling, FRG	Ulrich (1984)
Quercus	1.6	France	Rapp (1973)
Quercus	1.1	Poland	Karkanis (1976)
Quercus-Betula	3.0	Netherlands	van Breemen et al. (1982)
Pinus	3.5	Netherlands	-"-
Pinus	1.3	Sweden	Bringmark (1977)
Pinus	1.2	Jädraas, Sweden	Andersson et al. (1980)
Picea abies	3.9	Solling, FRG	Ulrich (1984)
Picea sithcensis	1.5	Kilmichaer, UK	Miller & Miller (1980)
-"-	1.8	Leanachan, UK	-"-
-"-	1.7	Strathyre, UK	-"-
-"-	1.8	Kershope, UK	-"-
-"-	2.1	Elibank, UK	-"-
-"-	2.0	Fetteresso, UK	-"-

The atmospheric transport models provide the average total sulfur deposition, d_{tot}, in each grid as input to the soil model. The deposition on forests within one grid square, d_f, is assumed to be φ times larger than the deposition on open land, d_0, i.e.

$$d_f = \varphi d_0 \tag{9}$$

$$f\, d_f + (1-f) d_0 = d_{tot} \tag{10}$$

Table 3. σ-values calculated from local observations on $Ca^{2+}+Mg^{2+}$-deposition and SO_4^{2-}-deposition (see text for further details).

Location	σ	Reference
Birkenes, Norway	0.68	Wright & Johannessen (1980)
Fyresdal-Nissedal, Norway	0.75	Johannessen & Joranger (1976)
Langtjern, Norway	0.78	Henriksen (1976)
Fillefjell, Norway	0.70	Dovland (1976)
Beech, Solling, FRG	0.56	Matzner (1983)
Spruce, Solling, FRG	0.68	-"-
Oak, Solling, FRG	0.70	-"-
Pine, Solling, FRG	0.69	-"-
Heath, Solling, FRG	0.70	-"-
Jädraas, Sweden	0.77	Andersson et al. (1980)

where f is the fraction of forests within the grid. From this we get

$$d_f = d_{tot} \cdot \varphi / (1 + (\varphi - 1) f) \qquad (11)$$

The acid stress, as, on the forests within the grid is then given by

$$as = \sigma \cdot d_f \qquad (12)$$

The sensitivity of the model to the parameters φ and σ was tested by looking at the changes in the area of soils below a critical pH-value in Europe.

4.4. Critical pH

The concept "critical pH" refers to an increased risk for forest damage due to changes in soil chemistry. The value 4.2 has been used in the model application to a European scale. This is the value, which — according to Ulrich (1981, 1983) — implies the change from the cation exchange range to the aluminum range. The connection between forest damage and increased dissolved aluminum-ion concentrations in the soil solution is not, however, straightforward. It does not mean therefore, that there would be no risk above the critical pH, nor that there definitely occurs damage below it. Some criteria have been proposed by Ulrich et al. (1984) for the evaluation of risks caused by soil acidity (Table 4). This information can also assist in interpreting results from our model. Concerning the European application the effect of varying the critical pH-value on the estimate of forest area under risk was tested.

Table 4. Criteria for relating risk of forest damage to some chemical characteristics of soils (cf. Ulrich et al., 1984).

	Increasing risk	High risk	Very high risk
pH_{H_2O}	≥ 4.2	4.0-4.2	< 4.0
β	≥ 0.05	< 0.05	0.0
$[Al^{3+}]$ $\mu eq / l$	< 80	80-320	> 320
Ca / Al	> 0.4	0.1-0.4	< 0.1

5. Sensitivity Analysis

5.1. Soil Parameters

The values for the buffer capacities and buffer rates were initialized based on the International Geological Map of Europe and the Mediterranean Region (UNESCO, 1972) and the Soil Map of the World (FAO-UNESCO, 1974). The Geological Map provided information about the parent material of the soils and the Soil Map about the dominant soil types. These sources, however, do not give too much direct information about the buffering properties of the soils. So the silicate buffer rate was related to the Ca+Mg content of the parent material following the buffer rate values given by Ulrich (1981). The estimation of the total cation exchange capacity as well as the base saturation of a certain soil type was based on (i) information given by the definition of the soil type (FAO-UNESCO, 1974) and (ii) descriptions and results of analyses of typical soil profiles given in the Appendix of the Soil Map.

According to Ulrich (1983), the silicate buffer rate may vary from 0.1 to 1.0 $kmol\ ha^{-1}yr^{-1}$ for a 50 cm soil layer. This range was also used in the sensitivity runs (see Table 5 for the values and ranges used in the sensitivity runs). The acid stress rate, as (in $kmol\ ha^{-1}yr^{-1}$), which is compared to br_{St}, varies at present between zero in some remote areas and over ten in some parts of Europe. If as and br_{St} are at the same level, the model is highly sensitive to changes in br_{St} (see Figures 7a-10a). In a case of a high stress rate, on the contrary, a change of br_{St} from 0.1 to 1.0 $kmol\ ha^{-1}yr^{-1}$ has only a marginal effect on the results, because in any case it can buffer only a minor part of the stress (Figures 11a,12a). Due to the temporal and spatial variation of the acid stress the sensitivity of the model to the buffer rate of silicates varies also in time and space. At present the model is sensitive to br_{St} only in remote areas like Scandinavia. If, however, emissions are going to decrease considerably in the future, new areas will occur, where the value of br_{St} is important.

The effect of the total cation exchange capacity is quite straightforward: the higher the capacity of the soil, the longer it takes to consume it for the incoming proton flux. Doubling CEC_{tot} results in doubling the time needed to exhaust it, when other parameters are kept constant (Figures 7b-12b). CEC_{tot}, however, is quite strongly related to the soil type, i.e. CEC_{tot} of a certain soil has only a limited range of variation, typically not

Table 5. Values of the variables, soil parameters and forcing functions used for testing the sensitivity of the model (see Figs.7-12,15). The bold values give the minimum, increment and maximum for the range of the variable tested, while the other variables are kept constant (normal print).

Fig.	Soil	br_{St}	CEC_{tot}	β	$P-E$	as
7a	Po	**0.1(0.15)1**	780	0.1	300	0.5
7b	Po	0.5	**200(200)1000**	0.1	300	0.5
7c	Po	0.5	780	**0.1(0.1)0.5**	300	0.5
7d	Po	0.5	780	0.1	**100(150)700**	0.5
8a	De	**0.1(0.15)1**	910	0.15	270	1
8b	De	0.5	**600(200)1200**	0.15	270	1
8c	De	0.5	910	**0.1(0.1)0.8**	270	1
8d	De	0.5	910	0.15	**100(100)400**	1
9a	Lo	**0.1(0.15)1**	715	0.15	200	1
9b	Lo	0.5	**700(200)1700**	0.15	200	1
9c	Lo	0.5	715	**0.1(0.2)0.9**	200	1
9d	Lo	0.5	715	0.15	**100(100)600**	1
10a	Po	**0.1(0.15)1**	780	0.1	300	2
10b	Po	0.5	**200(200)1000**	0.1	300	2
10c	Po	0.5	780	**0.1(0.1)0.5**	300	2
10d	Po	0.5	780	0.1	**100(150)700**	2
11a	De	**0.1(0.15)1**	910	0.15	270	4
11b	De	0.5	**600(200)1200**	0.15	270	4
11c	De	0.5	910	**0.1(0.1)0.8**	270	4
11d	De	0.5	910	0.15	**100(100)400**	4
12a	Lo	**0.1(0.15)1**	715	0.15	200	6
12b	Lo	0.5	**700(200)1700**	0.15	200	6
12c	Lo	0.5	715	**0.1(0.2)0.9**	200	6
12d	Lo	0.5	715	0.15	**100(100)600**	6
15a	Po	0.5	780	0.1	300	**0.5(0.3)2**
15b	De	0.5	910	0.15	270	**1(0.5)4**
15c	Lo	0.5	715	0.15	200	**1(1)6**

more than ± 50% of the average. The rate of consumption, $as - br_{St}$, on the contrary, may vary tenfold, mostly due to the variation in the acid stress. Knowing the difficulties involved in the determination of acid stress (see Section 4.3) the uncertainty in CEC_{tot} seems to be a less important problem.

Base saturation, β, expresses the proportion of CEC_{tot} occupied by base cations and may vary from 0.1 to 1.0. Even in poor forest soils (like Podzols) it may go up to 0.5. This range (0.1-0.5) is very important, since it implies, that the amount of exchangeable base cations at the exchange sites in the beginning of the simulation period may vary fivefold and thus also the time needed to exhaust it. For Podzols (with $as = 0.5\ kmol\ ha^{-1}yr^{-1}$) increasing β from 0.1 to 0.2 raises the final pH as much as changing br_{St} from 0.1 to 1.0 $kmol\ ha^{-1}yr^{-1}$ (Figure 7a,c). The same relation exists with other soil types, when the stress rate is low (Figures 8a,c and 9a,c). When the stress is high (6 $kmol\ ha^{-1}yr^{-1}$), only the highest β-values (0.7 ... 0.9) are able to keep the soil in the cation exchange range throughout the whole simulation period (Figure 12c).

In summary, initial base saturation appears to be crucial; it varies at least from 0.1 to 0.5 for typical forest soils, while the variation of the total cation exchange capacity is normally not more than ± 50% of the average.

5.2. Atmospheric Parameters

The acid stress entering the soil is determined by (a) the total sulfur deposition, (b) forest filtering effect and (c) base cation deposition originating, e.g., from canopy and dust. Thus the same sulfur deposition into a grid may cause quite a different acid stress depending on the fraction of forests within a grid and the values of the filtering and base cation factors.

As can be seen from Eq.11 the fraction of forests, f, within a grid as well as the filtering factor, φ, enter the calculation of the acid stress, as, nonlinearly. This relation is depicted in Figure 13 for a unit deposition value. For $\varphi = 1$ the percentage of forests within a grid has no influence on as, while for $\varphi = 4$ f strongly determines the actual acid stress.

The effect of the base cation factor is linear, because it enters the equation as a multiplication factor (see Eq.12), and does not depend on the location.

The influence of φ on the forested area below a certain pH in Europe is not as dramatic as in the case of a single grid with little forest ($f = 0.5$, see Figures 13 and 14). Changing φ from 1.0 to 4.0 about doubles this area. Varying σ from 0.5 to 0.8 (which means a relative increase of 60% only compared to 400% change in φ!) has about the same effect as the fourfold change in φ. This holds true for the whole critical pH-range 4.0 to 4.2.

However, both factors vary spatially depending on climate, tree species, etc. (Tables 2 and 3). It is an important issue for future deposition studies to determine their actual variation.

As it is the difference $as - br_{St}$, which determines the response of the soil, the shape of the curves (Figure 15) with varying acid stress is similar as in Figures 7a-12a, in which br_{St} is varied. If the acid stress is higher than the buffer rate of silicates, the variation of as is only reflected in the length of time needed to exhaust the cation exchange capacity and in the equilibrium pH of the aluminum buffer range. Still the crucial question is which one, as or br_{St}, is higher, because it determines whether the soil acidifies or alkalinizes. It implies that the sensitivity of the model with respect to acid stress varies in time and space, as it does in case of br_{St}. The location of the areas where the difference $as - br_{St}$ for the year 1980 is below zero is shown in Figure 16.

5.3. Precipitation and Evapotranspiration

Precipitation, P, and evapotranspiration, E, enter the model only in the aluminum buffer range, where the net precipitation, $P - E$, together with the acid stress rate determines the pH-value of the soil (Figures 11d-12d). For the same stress rate the soil pH in the aluminum buffer range stays higher in humid areas than in areas with a drier climate.

5.4. Critical pH

In the previous description of the soil model (Kauppi et al., 1984 and 1985a) the decision about the critical pH is left to the user. As the area of forests under risk varies depending on the critical pH value selected, the user should be aware of the significance of this decision. Therefore the sensitivity of the results to this value is also presented here, although the critical pH is not really a part of the model.

With the low energy scenario the proportion of forests with the soil pH below the critical value reaches 23.6% when using pH 4.2 as a limit and decreases from 4.6% in 1990-2000 to 2.8% in the year 2030 when taking a lower limit pH 4.0 (Figure 17a).

With the high scenario the forest area with the soil below the critical pH varies from 26.2% to 48.5%, when the critical pH changes from 4.0 to 4.2 (Figure 17b). This shows that the critical pH values chosen significantly affect the forest area under risk.

6. Conclusions

The sensitivity tests clearly pointed out how necessary it is to know the status of the soil at the time of initialization. Base saturation, i.e. the fraction of cation exchange sites occupied by base cations, is the variable which needs special attention. So far the initial values in the European application of the model were based on the information given in the soil map of the World (FAO-UNESCO, 1974). As this map has originally been prepared for agricultural purposes it does not necessarily fulfill the requirements of the acidification studies. Whenever possible, recent measurements about the status of the soil should be used. This requires cooperation with those responsible for soil surveys in European countries.

The only important forcing function in the model is acid stress which is derived from sulfur deposition. The acid stress is partly or completely counteracted by the buffer rate of silicates, i.e. silicate weathering. As a matter of fact it is the difference of the two factors which determines whether the soil is going to alkalinize or acidify. Thus both variables should be estimated with special care when they are about equal, as they are at present, e.g., in Scandinavia. If the emissions are going to decrease drastically in the future this area might increase substantially.

Acid stress is estimated from the regional mean of sulfur deposition, the filtering effect of forests and the base cation deposition counteracting part of the acidic deposition. The forest filtering factor becomes important in areas where forest coverage is small. The information concerning both the forest filtering factor and the base cation factor is sparse. More research is needed on both of these factors.

The low soil pH is known to pose a threat to forests by generating a predisposing stress on ecosystems, which implies an increasing risk for forest damage. There is no certain pH value, however, below which damage occurs. In the European application of the soil acidification model, a pH of 4.2 was selected as a critical pH to demonstrate the model behavior. However, instead of using one particular value of critical pH, the user should evaluate the model output for a range of pH-values.

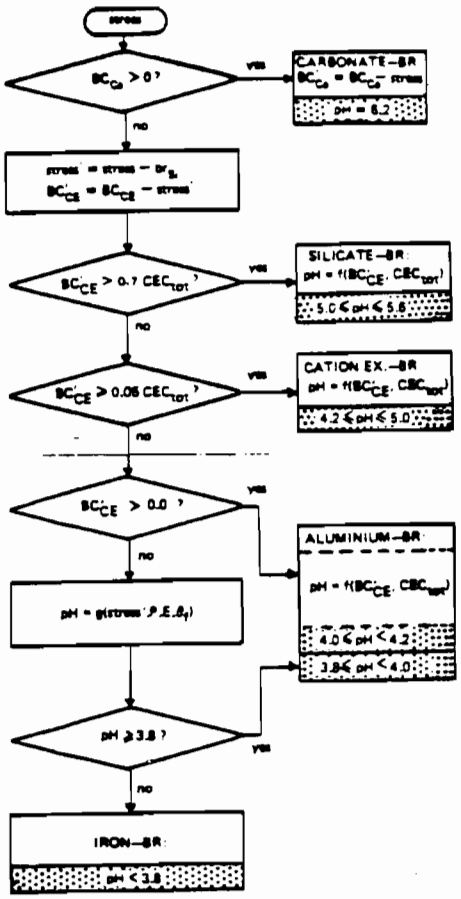


Figure 1: Flow diagram of the soil acidification model.

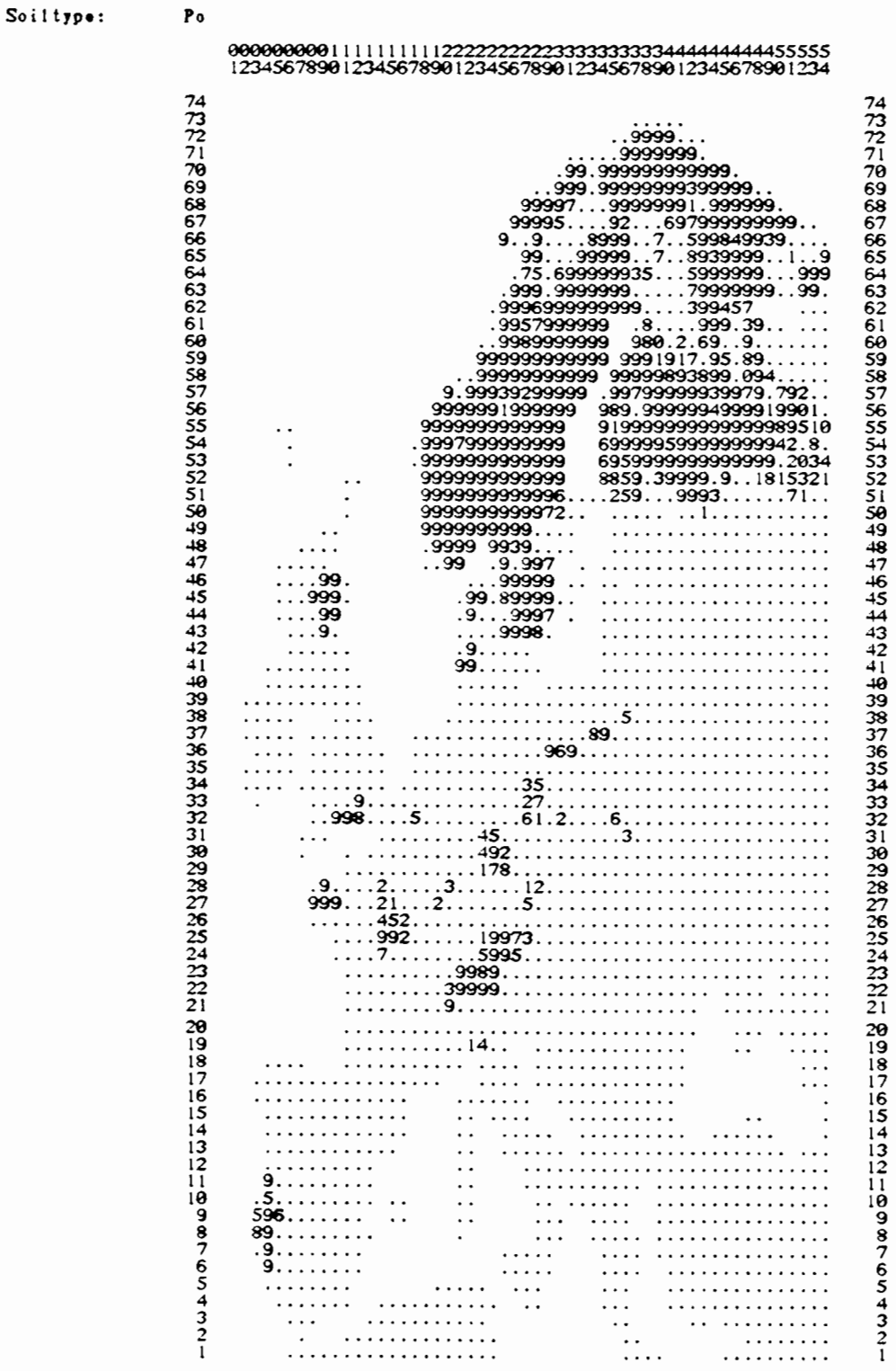


Figure 2: Spatial distribution of Orthic Podzols (Po) on European forest soils. The number in each grid gives the percentage class: n in a grid means that the percentage of Po in this grid lies between $10n$ and $10(n+1)$.

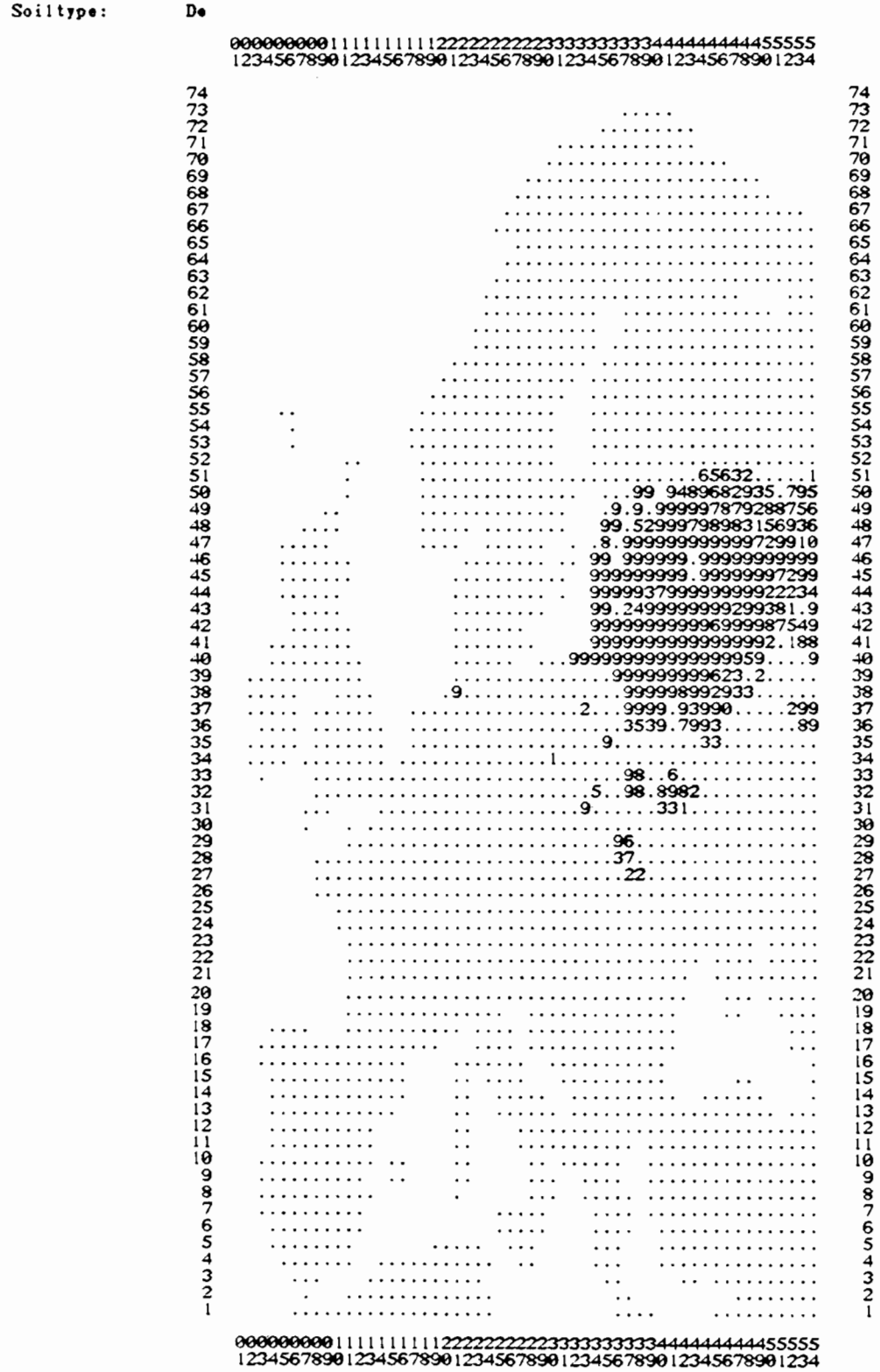


Figure 3: Spatial distribution of Eutric Podzoluvisols (De) on European forest soils. The number in each grid gives the percentage class: n in a grid means that the percentage of Po in this grid lies between $10n$ and $10(n+1)$.

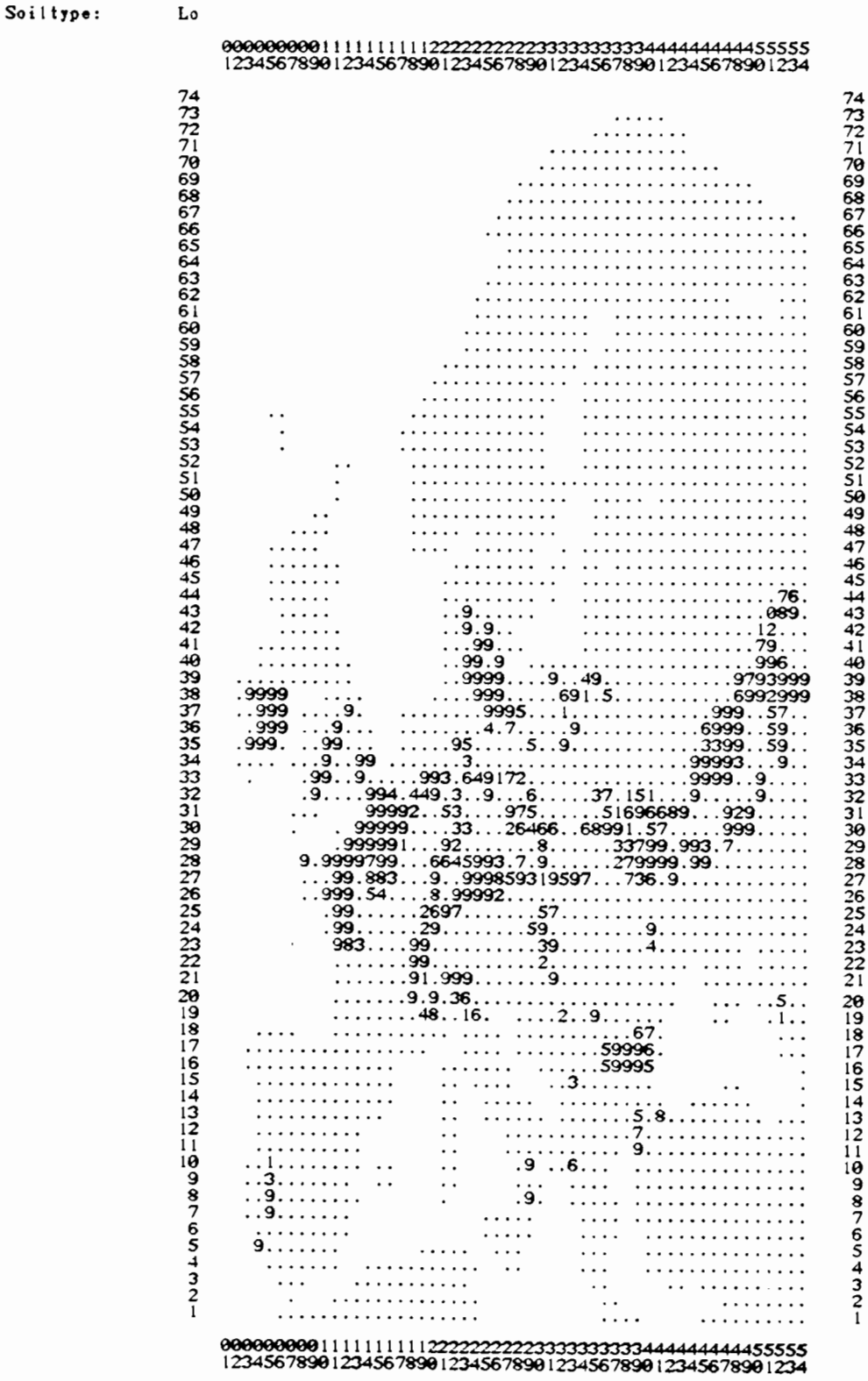


Figure 4: Spatial distribution of Orthic Luvisols (Lo) on European forest soils. The number in each grid gives the percentage class: ***n*** in a grid means that the percentage of Po in this grid lies between $10n$ and $10(n+1)$.

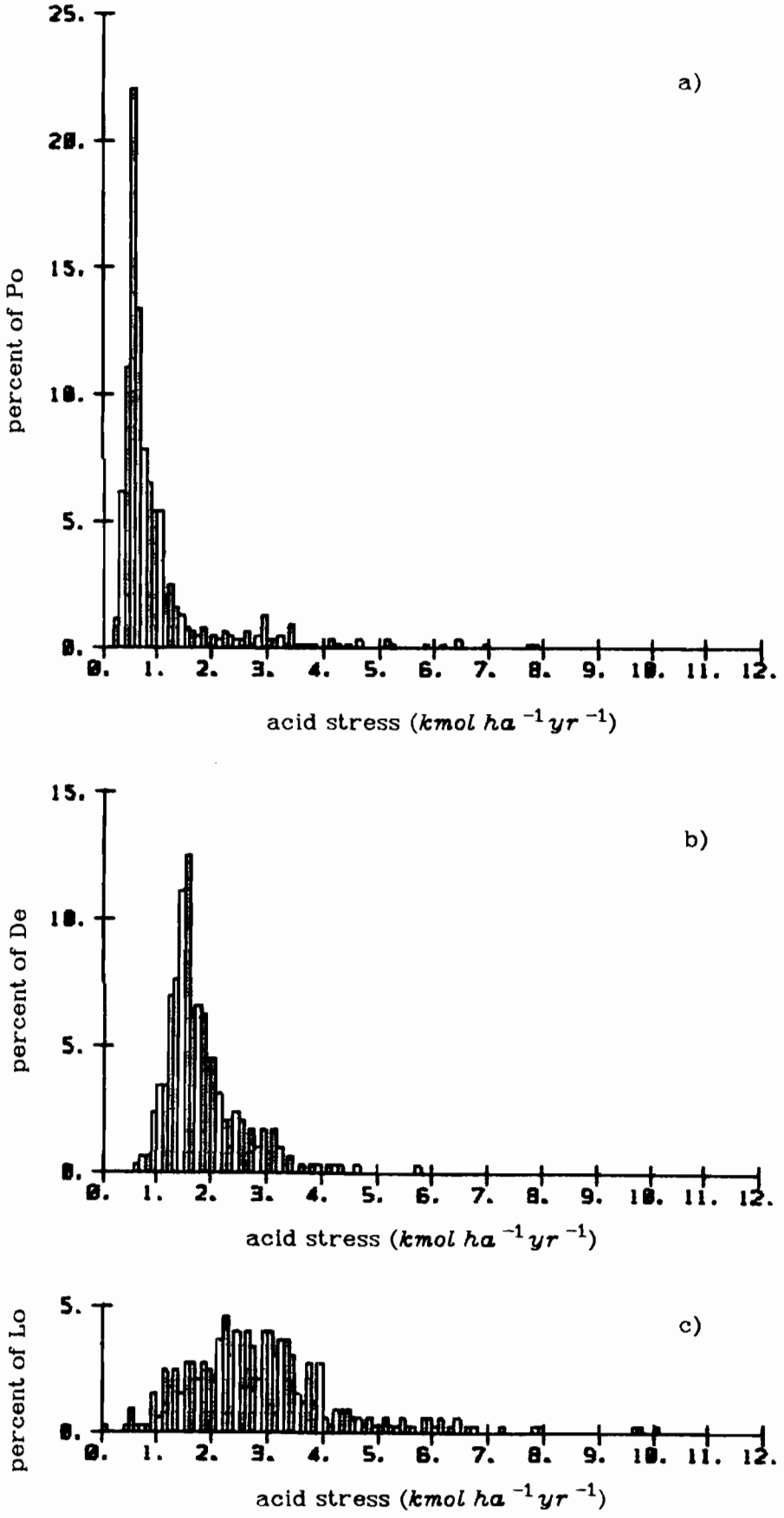


Figure 5: Frequency distribution of the exposure of a) Orthic Podzols (Po), b) Eutric Podzoluvisols (De), and c) Orthic Luvisols (Lo) to acid stress in Europe due to sulfur deposition in 1980.

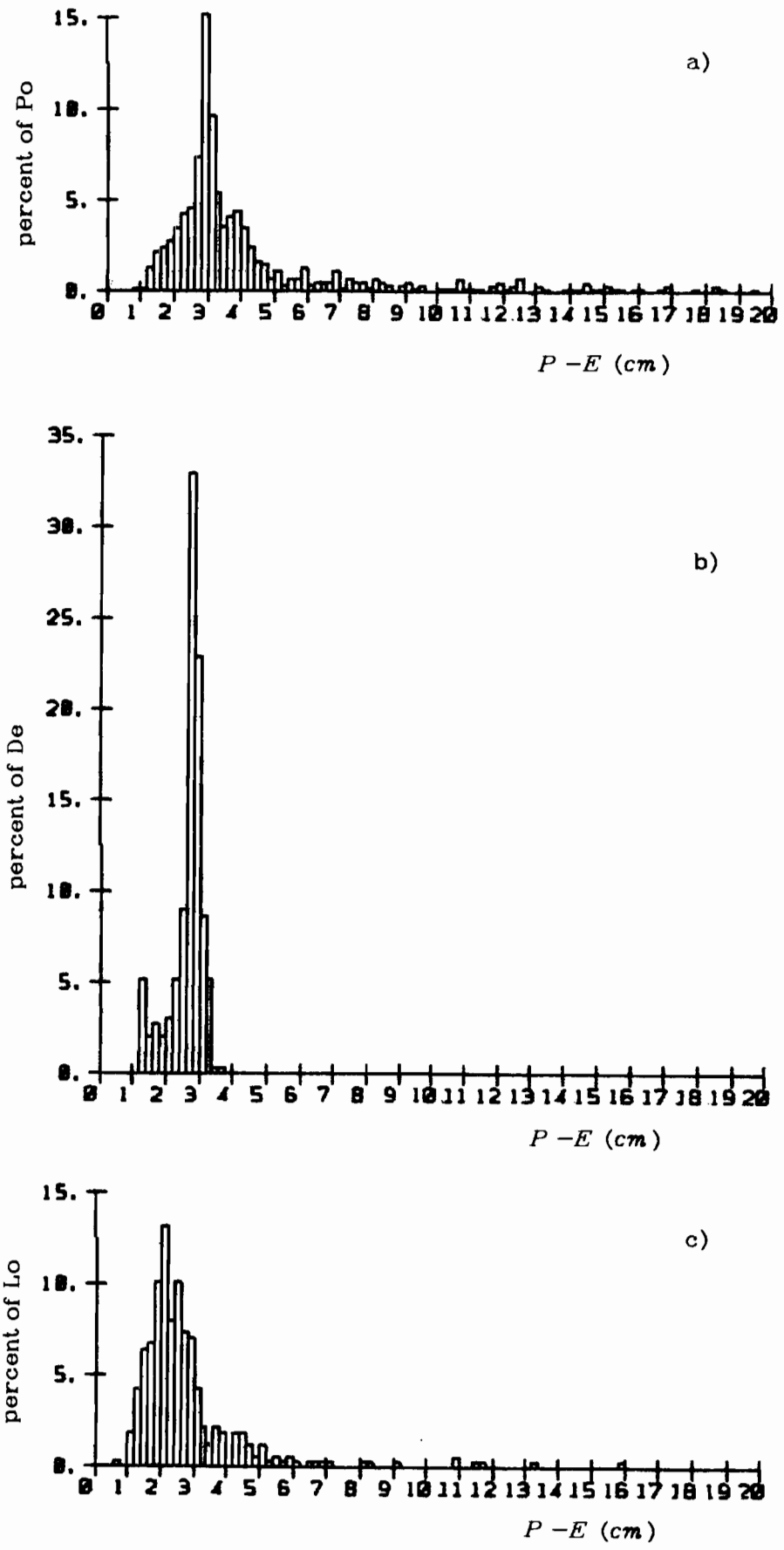


Figure 6: Frequency distribution of the exposure of a) Orthic Podzols (Po), b) Eutric Podzoluvisols (De), and c) Orthic Luvisols (Lo) to net precipitation $P-E$.

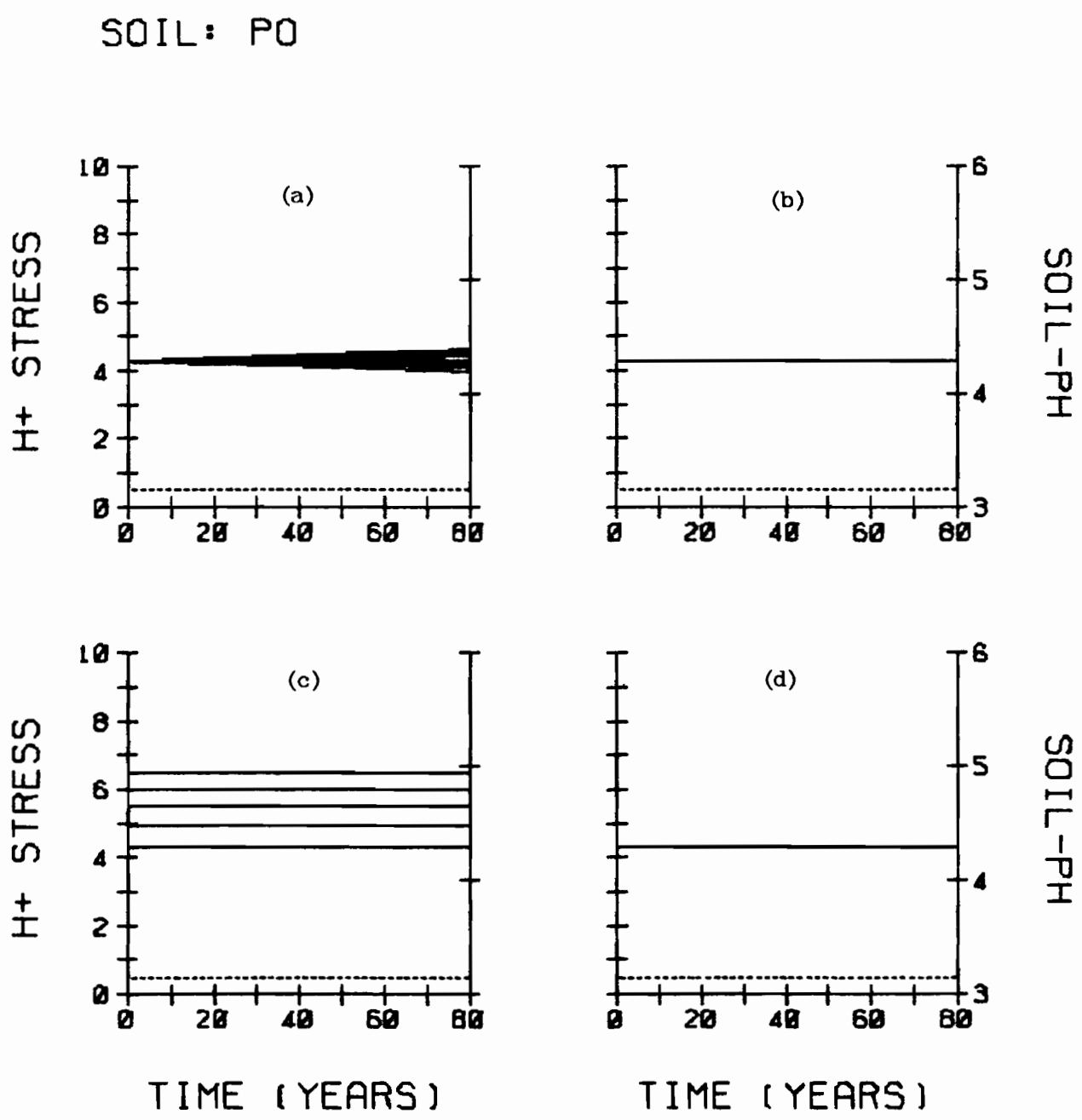


Figure 7: Soil simulation of Orthic Podzol (Po) for a "low" acid stress rate. (a) silicate buffer rate varied; (b) total cation exchange capacity varied; (c) base saturation varied; (d) net precipitation varied. The values for these variables are given in Table 5.

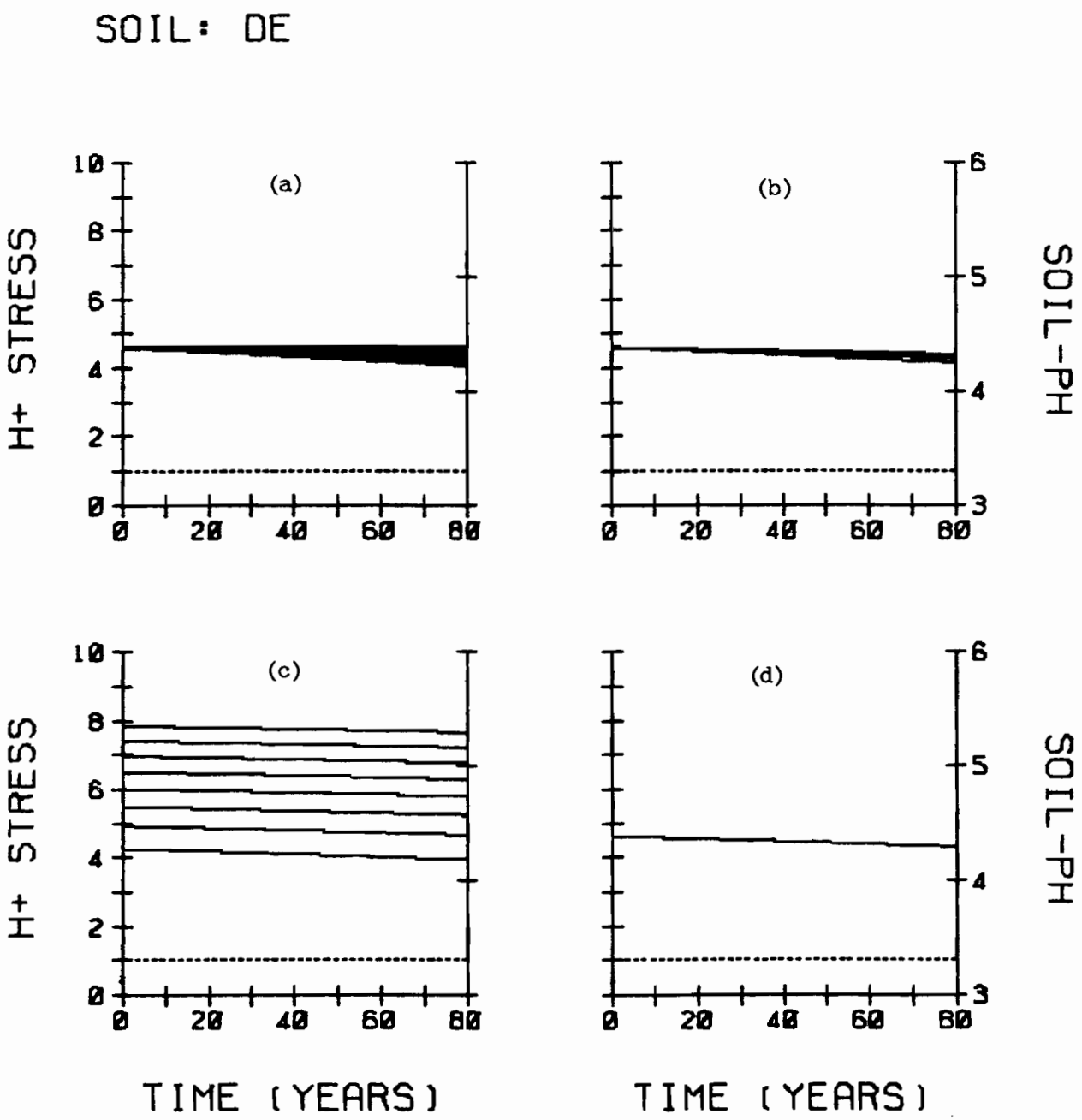


Figure 8: Soil simulation of Eutric Podzoluvisol (De) for a "low" acid stress rate. (a) silicate buffer rate varied; (b) total cation exchange capacity varied; (c) base saturation varied; (d) net precipitation varied. The values for these variables are given in Table 5.

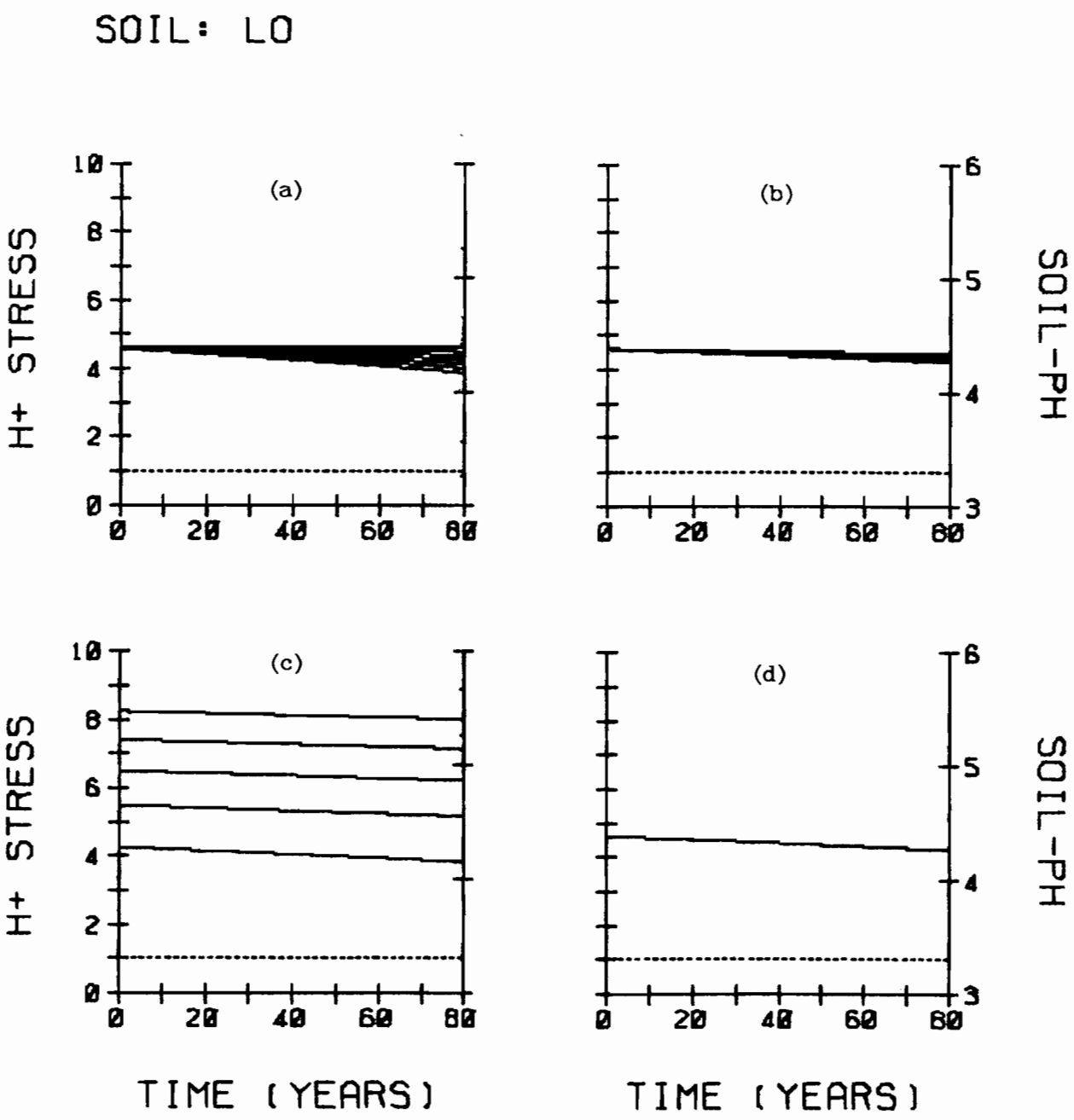


Figure 9: Soil simulation of Orthic Luvisol (Lo) for a "low" acid stress rate. (a) silicate buffer rate varied; (b) total cation exchange capacity varied; (c) base saturation varied; (d) net precipitation varied. The values for these variables are given in Table 5.

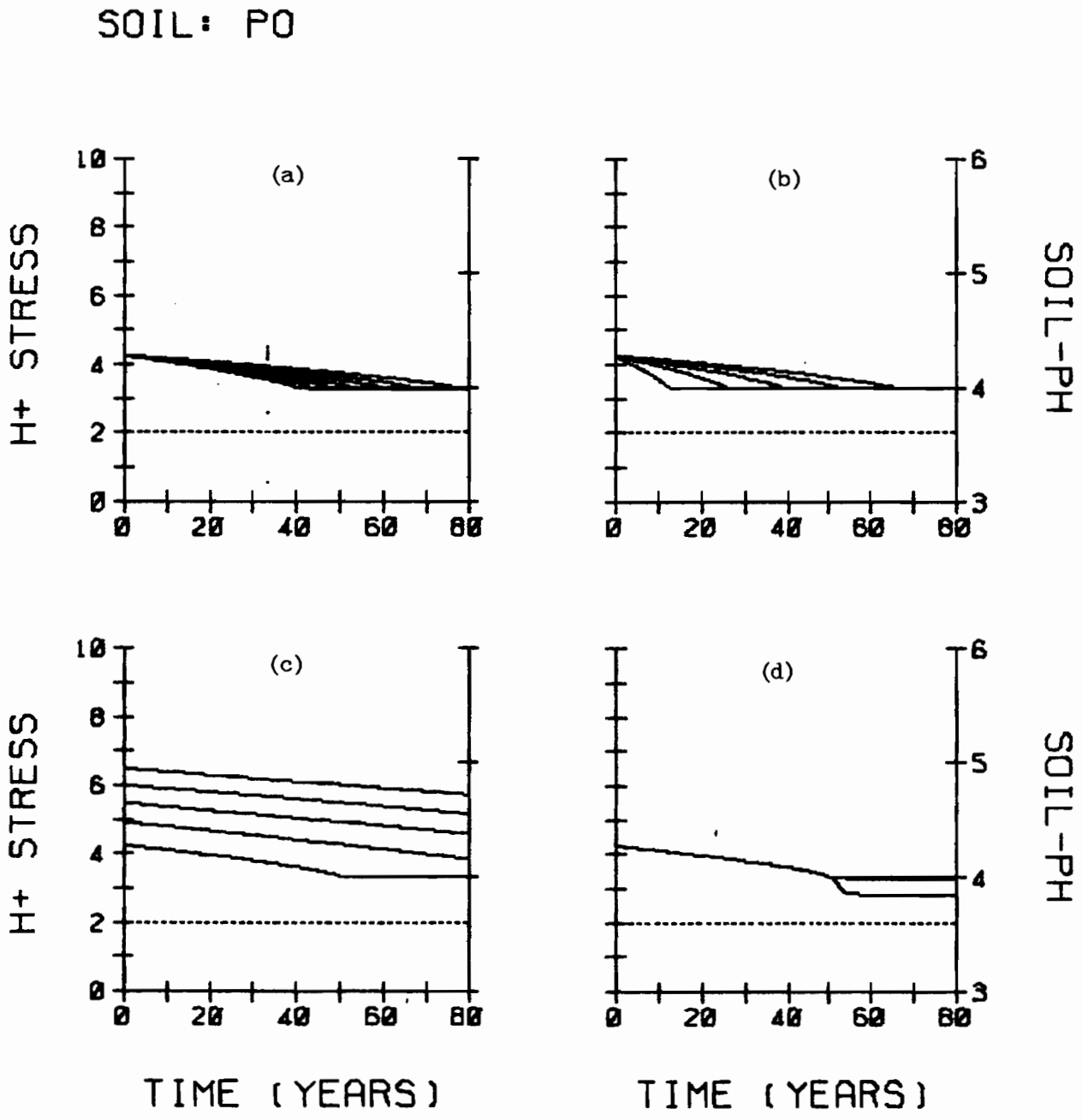


Figure 10: Soil simulation of Orthic Podzol (Po) for a "high" acid stress rate. (a) silicate buffer rate varied; (b) total cation exchange capacity varied; (c) base saturation varied; (d) net precipitation varied. The values for these variables are given in Table 5.

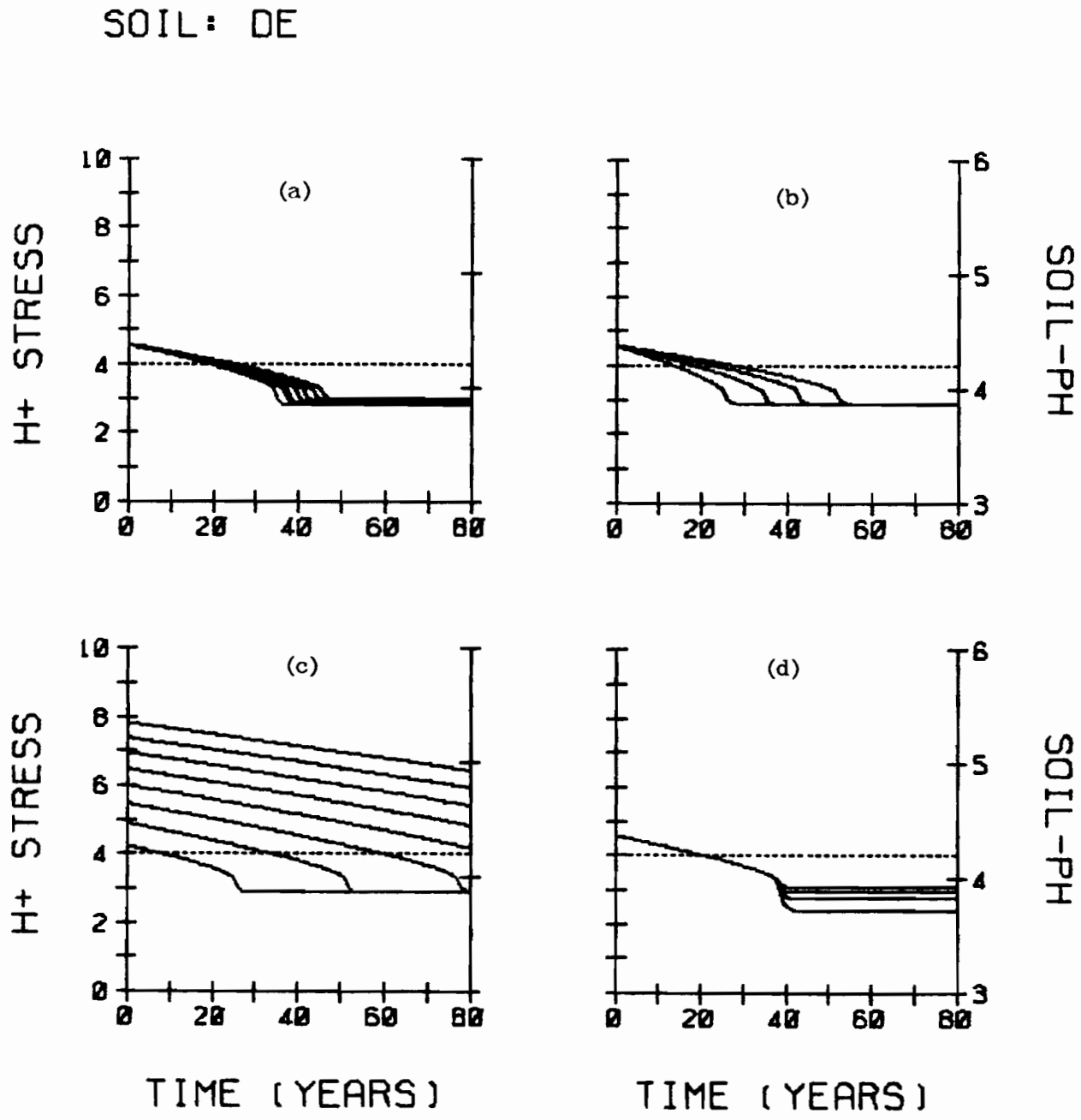


Figure 11: Soil simulation of Eutric Podzoluvisol (De) for a "high" acid stress rate. (a) silicate buffer rate varied; (b) total cation exchange capacity varied; (c) base saturation varied; (d) net precipitation varied. The values for these variables are given in Table 5.

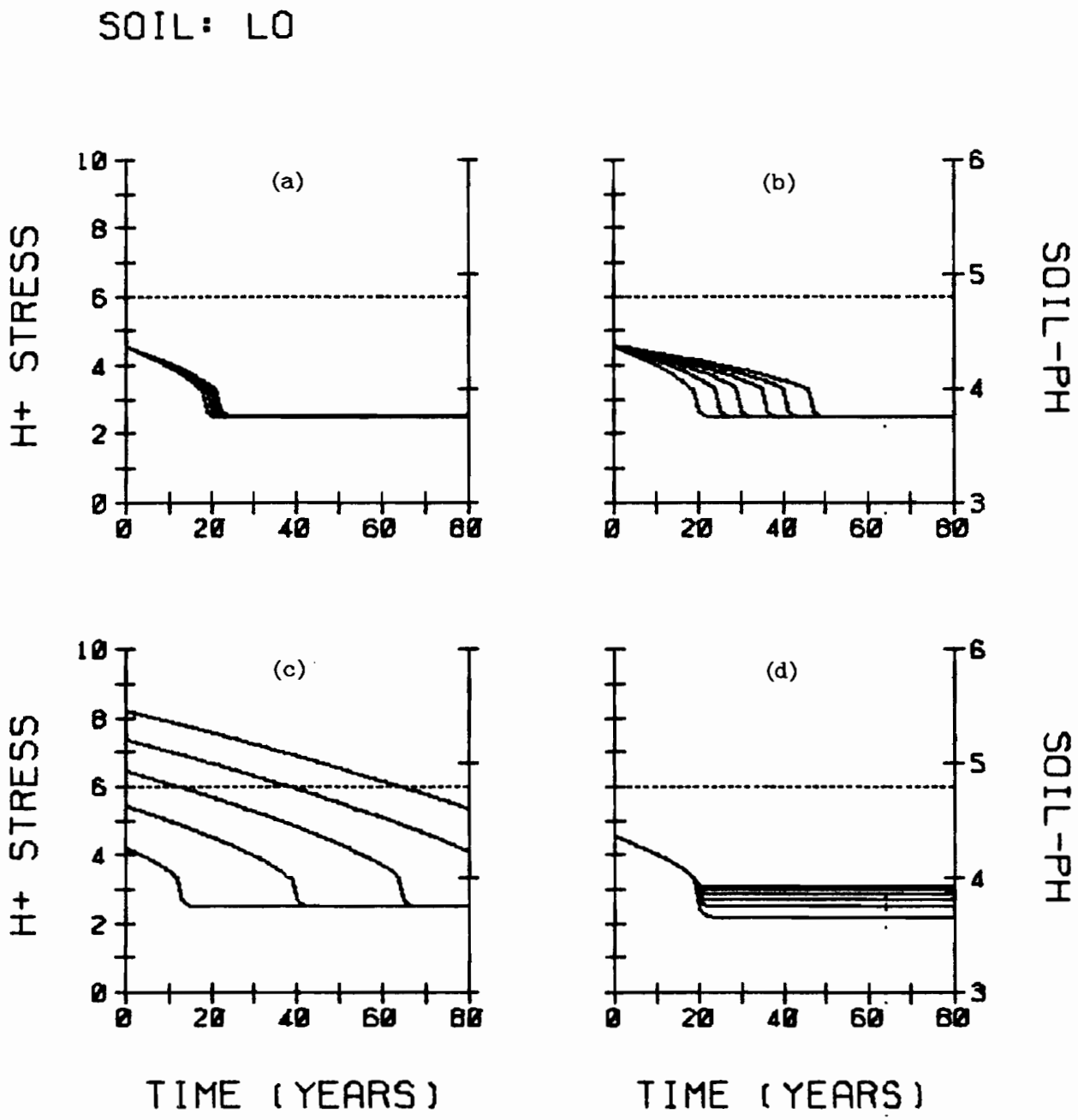


Figure 12: Soil simulation of Orthic Luvisol (Lo) for a "high" acid stress rate. (a) silicate buffer rate varied; (b) total cation exchange capacity varied; (c) base saturation varied; (d) net precipitation varied. The values for these variables are given in Table 5.

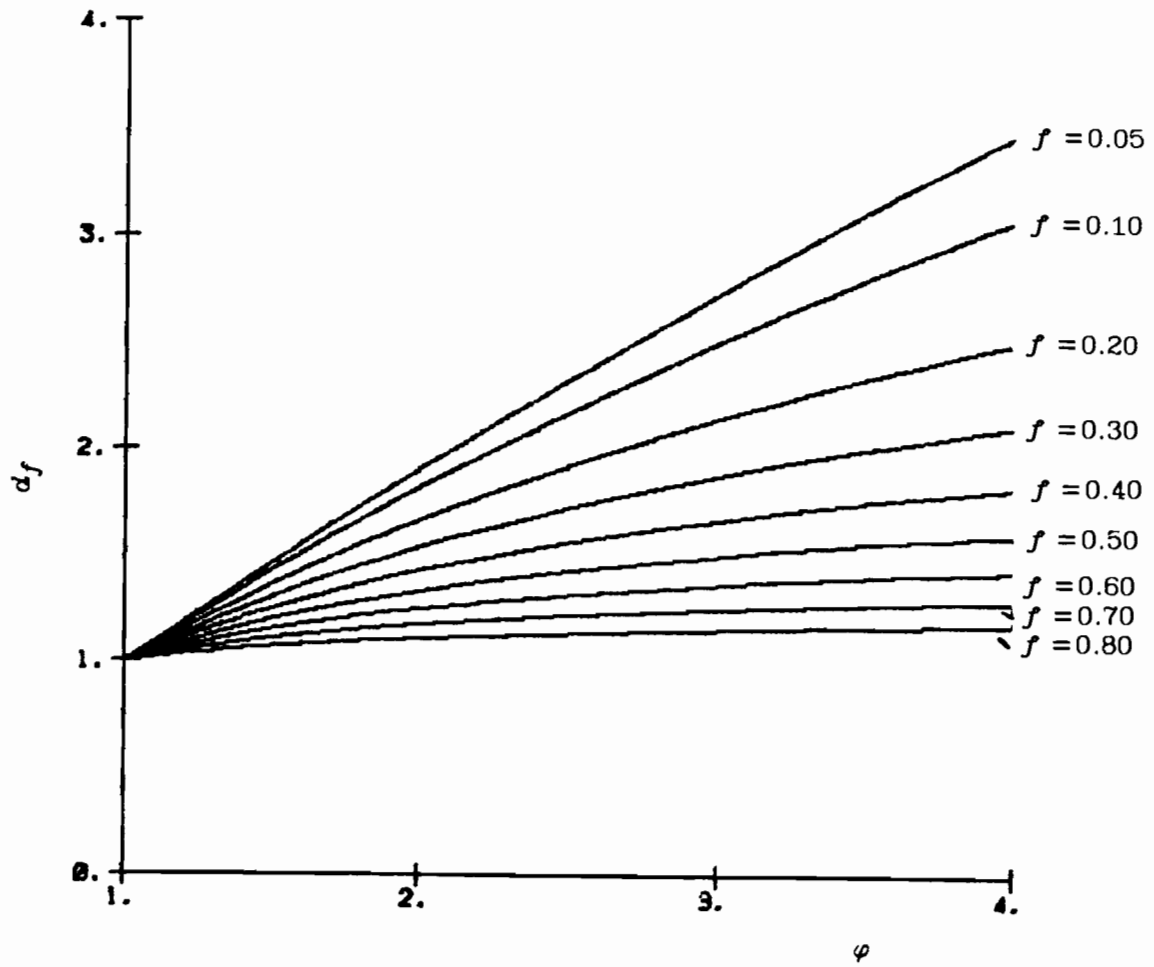
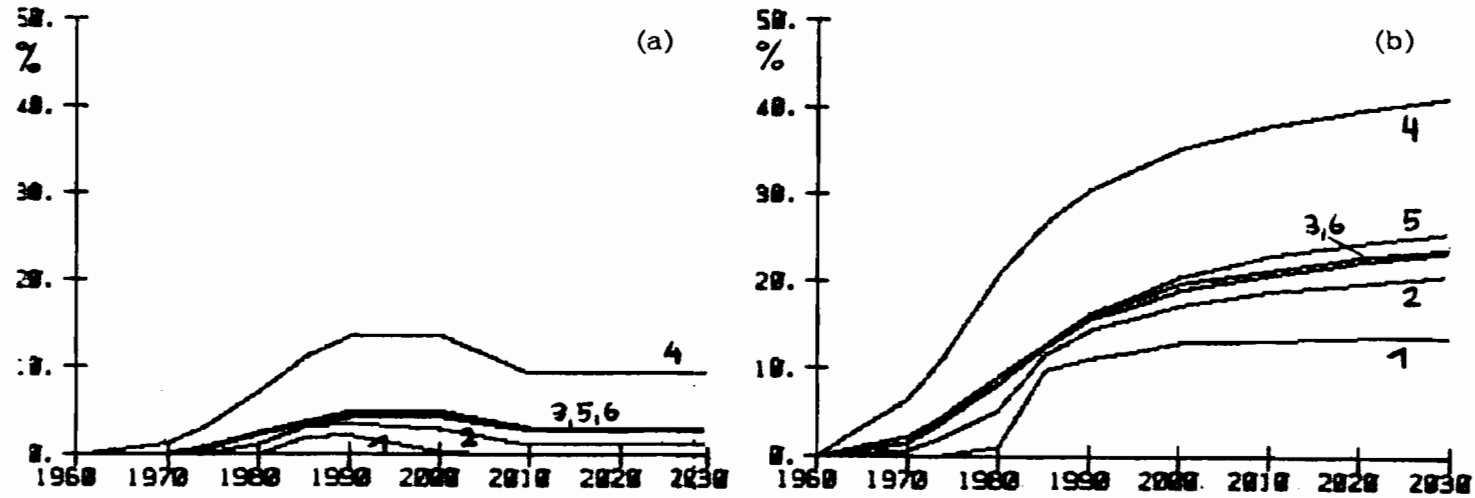


Figure 13: Acid stress as a function of the filtering factor φ for various values of the forest coverage f.



1: $\varphi = 1.$, $\sigma = 0.5$
2: $\varphi = 1.$, $\sigma = 0.8$
3: $\varphi = 4.$, $\sigma = 0.5$
4: $\varphi = 4.$, $\sigma = 0.8$
5: $\varphi = 1.$, $\sigma = 1.0$
6: $\varphi = 2.$, $\sigma = 2/3$

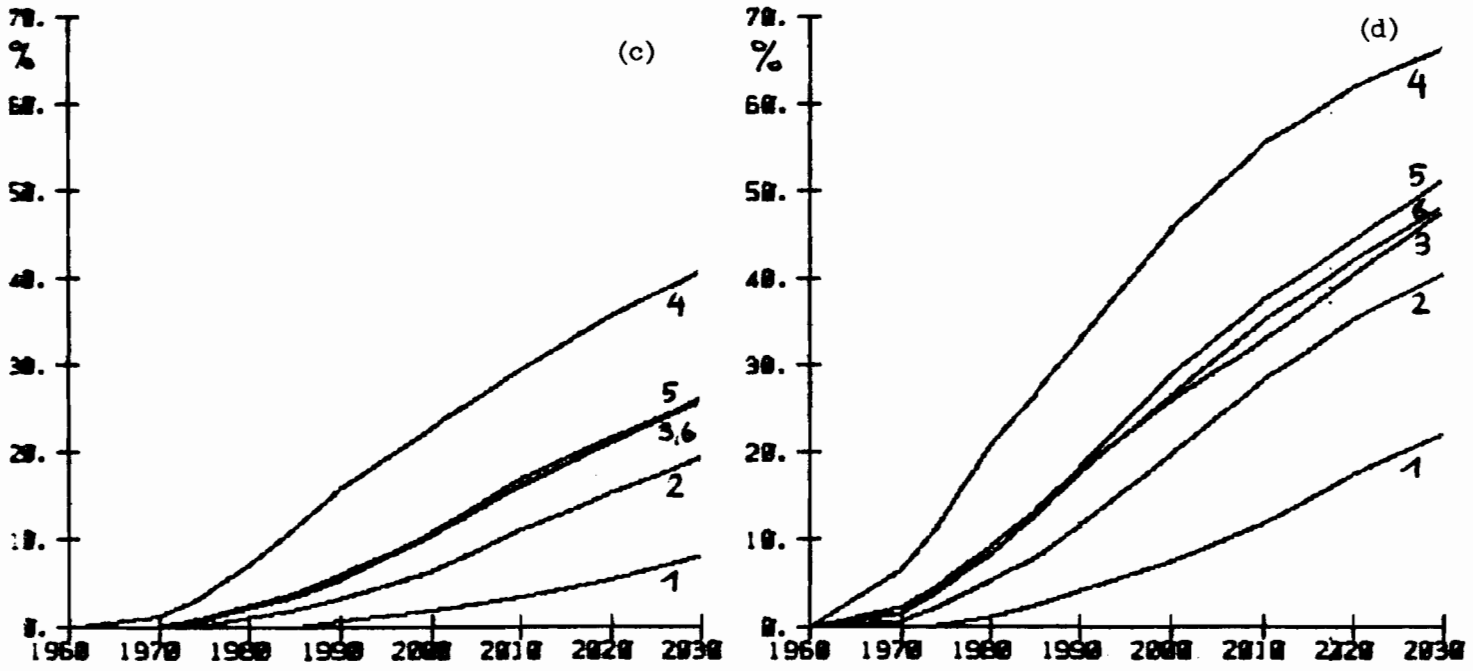


Figure 14: Time evolution of the area of European forest soils below different critical pH-values for two sulfur emission scenarios and varying φ and σ. (a) "low" scenario, pH=4.0; (b) "low" scenario, pH=4.2; (c) "high" scenario, pH=4.0; (d) "high" scenario, pH=4.2.

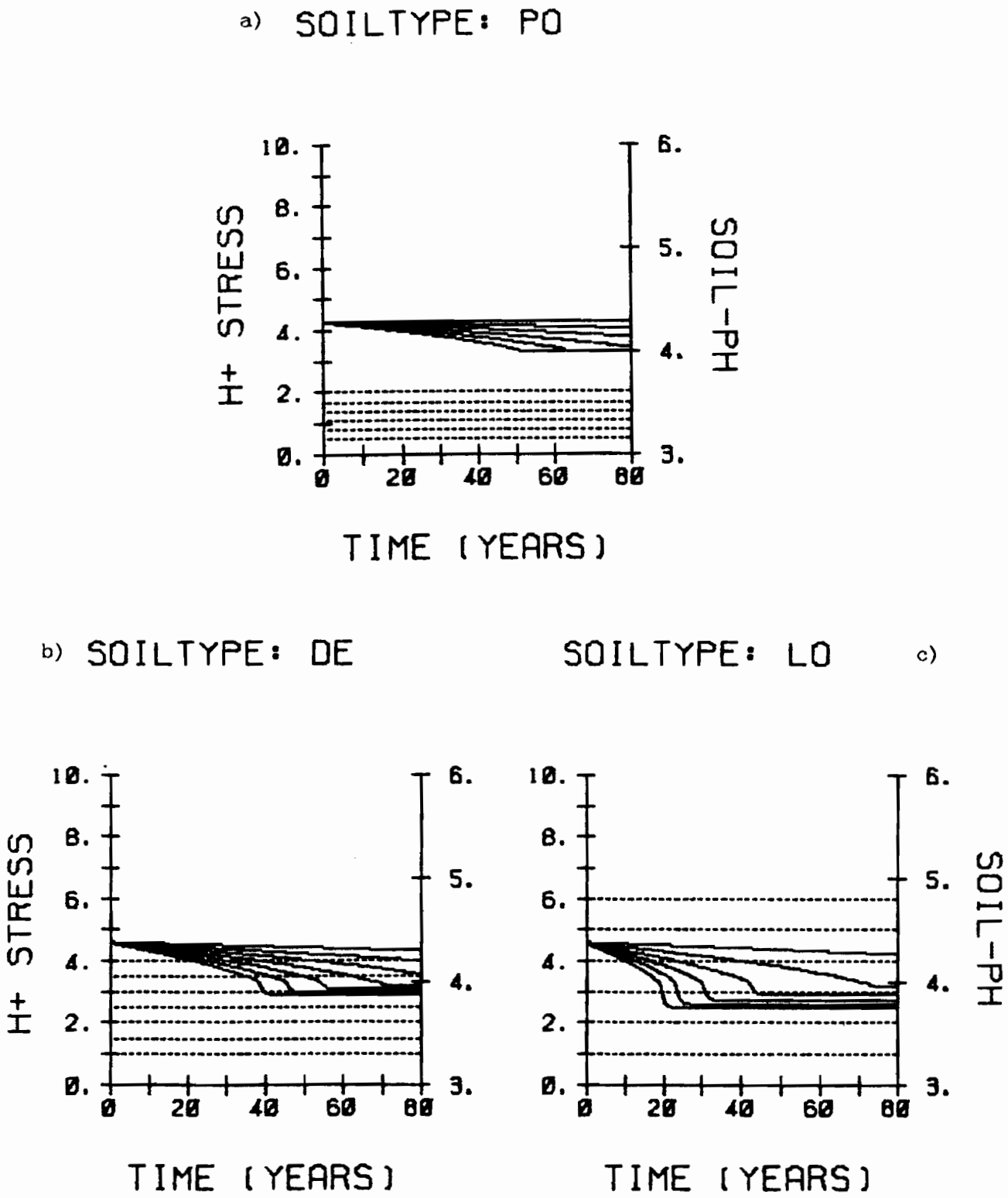


Figure 15: Soil simulation of a) Orthic Podzol (Po), b) Eutric Podzoluvisol (De), and c) Orthic Luvisol (Lo) for various stress rates. The values of the soil parameters are given in Table 5.

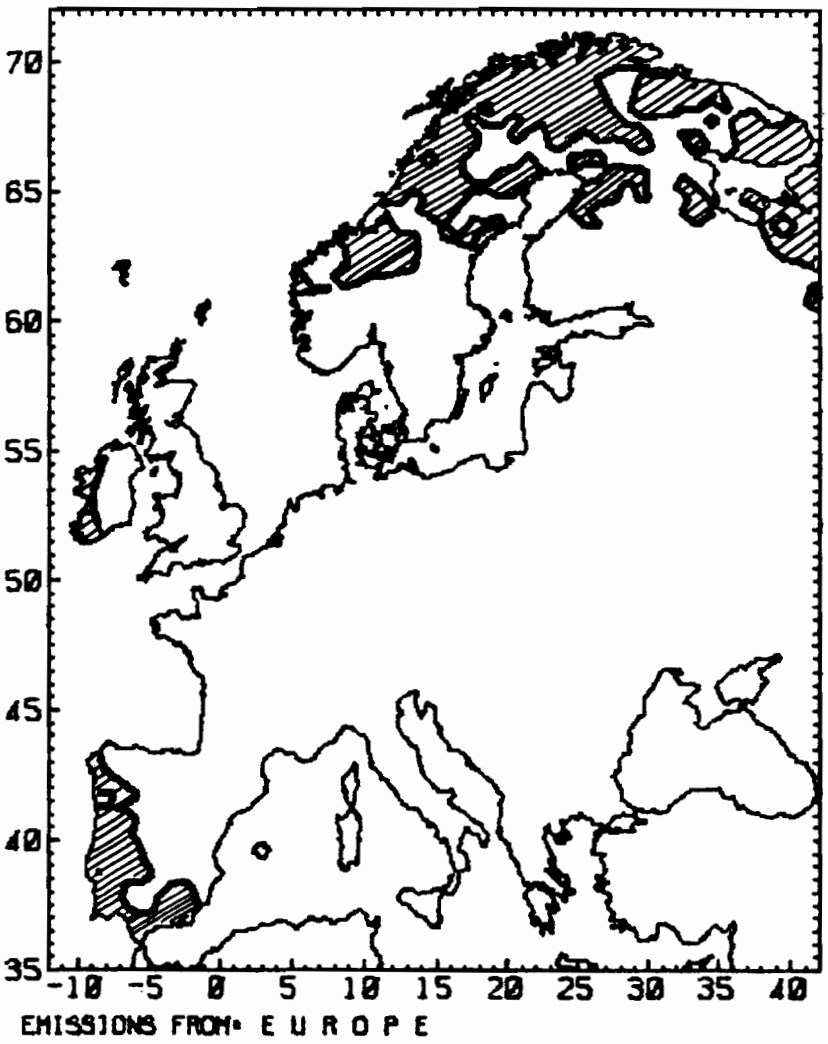


Figure 16: Spatial distribution of the "excess" acid stress $as - br_{St}$ in the year 1980 ($\varphi = 2$, $\sigma = 2/3$). Shaded areas indicate $as - br_{St} \leq 0$.

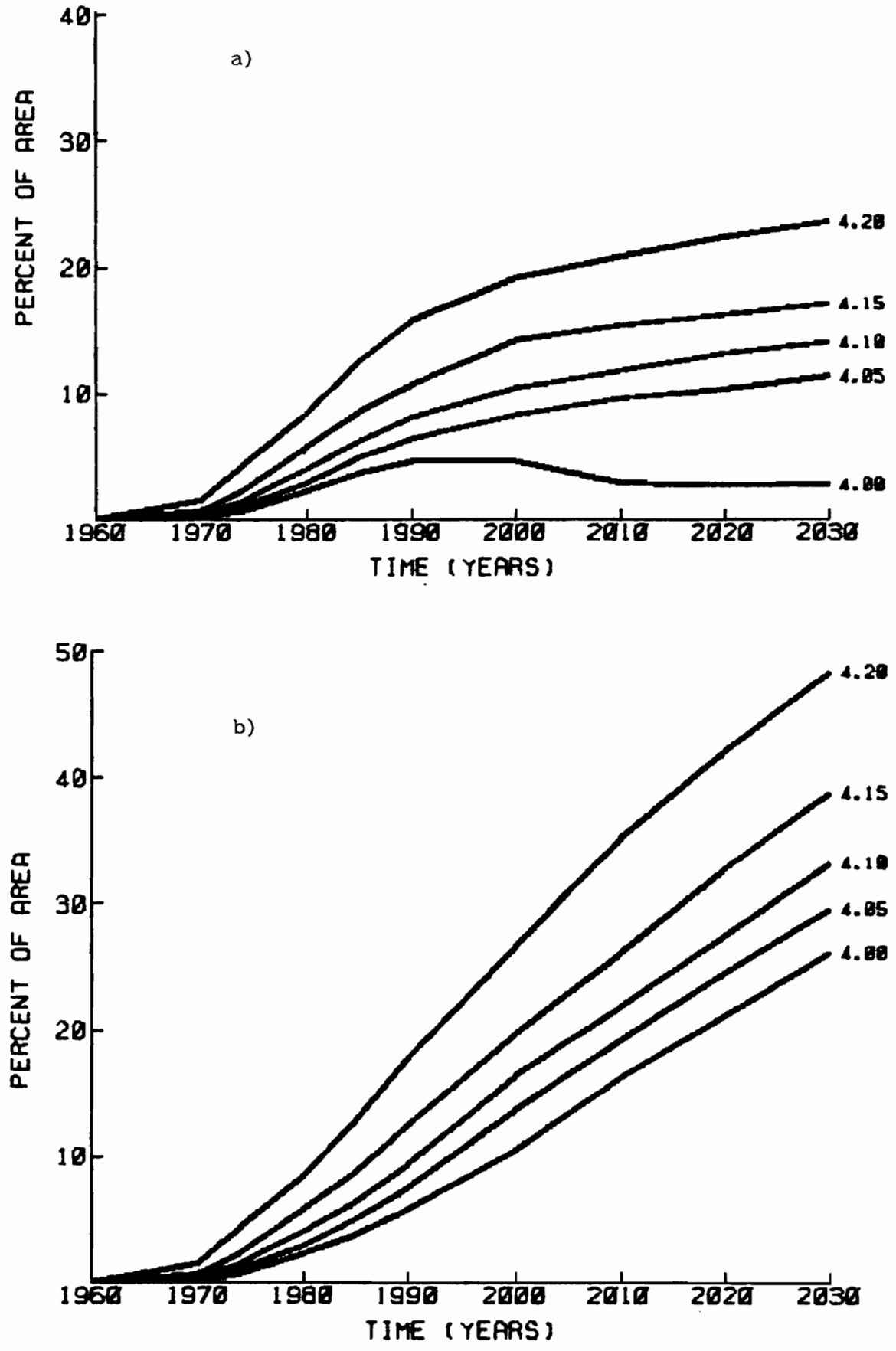


Figure 17: Time evolution of the area of forest soils in Europe with pH below different critical levels for a) a "low" sulfur emission scenario and b) a "high" sulfur emission scenario.

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