



Acidification of Forest Soils: A Model for Analyzing Impacts of Acidic Deposition in Europe - Version II

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**ACIDIFICATION OF FOREST SOILS:
A MODEL FOR ANALYZING IMPACTS
OF ACIDIC DEPOSITION IN EUROPE
VERSION II**

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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 modelling effort is part of the official cooperation between IIASA and the UN Economic Commission of 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 submodels currently available are Sulfur Emissions, the EMEP Long Range Transport Model, Forest Soil pH and Lake Acidity. In addition, two submodels are under development: the NO_x Emissions submodel and the Direct Forest Impacts submodel. The first version of the Forest Soil pH submodel was presented in May 1984. Since then several changes have been implemented following the advice of experts. This paper describes the Forest Soil pH model as it stands in March 1985.

Leen Hordijk

Acid Rain Project Leader



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ABSTRACT

Acidification is an unfavorable process in forest soils. Timber logging, natural accumulation of biomass in the ecosystem, and acidic deposition are sources of acidification. Acidification causes a risk of damage to plant roots and a subsequent risk of a decline in ecosystem productivity.

A dynamic model is introduced for describing the acidification of forest soils. In one-year time steps the model calculates the soil pH as function of 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 the acid stress (annual amount) to the buffer rate, and (ii) the accumulated acid stress (over several years) to the buffer capacity. The comparisons produce an estimate of the soil acidity as the output.

Since the first version in May 1984 several changes have been implemented following the advice of the experts. For aluminum and iron buffer ranges an equilibrium approach has been introduced. The pH of the silicate, cation exchange and upper aluminum buffer ranges is now a function of base saturation. In the current version of the model forests are assumed to absorb sulfur compounds more effectively than agricultural lands and, moreover, forests are assumed to grow on poor soil types rather than on the average soil type of a grid.

The model system as a whole is now available for analyzing the impact of different emission scenarios. The soil acidification model assumes sulfur deposition estimates from the other submodels as input, and as output it computes the total area of forests in Europe with the estimated soil pH lower than any selected threshold value. Additionally it produces estimates of the acidity of European forest soils in a map format.



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1. Introduction

Forest damage has been observed in rural areas in Central Europe to a large extent since the 1970's. It was first reported on silver fir (Schütt, 1977) and later on Norway spruce, Scots pine, beech, and other tree species as well (Schütt et al., 1983). In 1984, in the Federal Republic of Germany damage was reported for a forest area of 2,549,000 ha (Lammel, 1984). Forest damage is a result of many factors such as direct impact of air pollutants on tree foliage, soil acidification, and climate. In this study we address one of them, soil acidification, which has been demonstrated as an important link between air pollution and forest damage. It is intended that other factors contributing to forest damage will be incorporated into the model at a later stage.

The first version of the soil pH model was presented in May 1984 (IIASA CP-84-16). Several soil scientists were then asked to review the model. According to their suggestions the model structure was substantially changed in the description of aluminum and iron buffer ranges. Instead of assuming a certain buffer rate, an equilibrium approach was introduced. In addition, the pH of the soil in the silicate, cation exchange and upper aluminum range is now calculated as a function of base saturation. Besides these structural changes, suggestions concerning the large scale application to Europe have been incorporated into the model: forests are known to grow on poor soils rather than on the average soil type. There is also strong evidence on the filtering effects of forests, i.e. the deposition velocity over forests is larger than that over open land. Some other suggestions, although considered correct, could not be taken into account, because of the project's focus on a large spatial scale.

2. Soil Acidification

Soil acidification has been defined as being a decrease in the acid neutralization capacity of the soil (Van Breemen et al., 1984). Such a decrease may coincide with a decrease in soil pH. It may also take place in conditions of a relatively constant pH assuming efficient buffering processes. In such a case the buffering of the soil counteracts the factors tending to decrease the soil pH so that over long periods of time the soil pH stabilizes at a constant level. Yet the neutralization capacity is being consumed and the soil is subject to acidification.

2.1. Acid Stress

Acid stress is defined as the input of hydrogen ions (protons) into the top soil. Acid stress can result from acidic deposition of air pollutants, from biomass utilization, and from the natural biological activity of ecosystems (Ulrich, 1983a; Van Breemen et al., 1984). Any one of these sources can dominate the flux of protons entering the 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.

Acid stress has two important aspects. One is the accumulating amount of stress and the other is the instantaneous rate of the stress. The variable *amount of stress* refers to the load, and involves accumulation over several years. The unit for the amount of stress is kilomoles of acidity per hectare ($kmol\ ha^{-1}$). The variable *stress rate* refers, in principle, to the time derivative of the amount of stress although in practice it is given as annual hydrogen ion input. The unit for the stress rate is kilomoles of acidity per hectare and year ($kmol\ ha^{-1}\ yr^{-1}$).

2.2. Buffering Processes

Soil reacts to the acid stress depending on the soil properties. Acid stress implies the flux of hydrogen ions into the soil, and in the corresponding way the *buffering properties of the soil* imply the consumption of hydrogen ions within the soil profile. Buffering is described using two variables, one for the gross potential and the other for the rate of the reaction. Both variables refer to the intrinsic properties of the soil. They can be quantified for any volume of the reacting soil.

Buffer capacity, the gross potential, is the total reservoir of the buffering compounds in the soil. The unit for the buffer capacity is the same as that for the amount of acid stress ($kmol\ ha^{-1}$).

Buffer rate, the rate variable, is defined as the maximum potential rate of the reaction between the buffering compounds and the hydrogen ions. This variable is needed because the reaction kinetics is sometimes of importance. Although the buffer capacity is high, the rate sometimes limits hydrogen ion consumption. The buffer rate is expressed in units which are comparable to those of the stress rate ($kmol\ ha^{-1}\ yr^{-1}$).

The proton consumption reactions in soils have been systematically described by Ulrich (1981, 1983b). A consecutive series of chemical reactions has been documented in soils in which the acidification proceeds. Information regarding the dominant reactions has been used for defining

categories, called *buffer ranges*. They are briefly described in the following paragraphs and summarized in Table 1. The name of each buffer range refers to the dominant buffer reaction and the typical pH ranges given refer to the pH of a soil/water suspension ($pH(H_2O)$).

Table 1: Classification of the acid buffering reactions in forest soils (Ulrich, 1981,1983b)

Buffer range	pH range	Base saturation	Buffer reaction
Carbonate	8.0-6.2	1.00	$CaCO_3 + H_2CO_3 \rightarrow Ca^{2+} + 2HCO_3^-$
Silicate	6.2-5.0	0.70-1.00	$CaAl_2Si_2O_8 + 2H_2CO_3 + H_2O \rightarrow Ca^{2+} + 2HCO_3^- + Al_2Si_2O_5(OH)_4$
Cation exchange	5.0-4.2	0.05-0.70	clay mineral=Ca + $2H^+$ -> H-clay mineral-H + Ca^{2+}
Aluminum	4.2-3.0	0.00-0.05	$AlOOH + 3H^+ \rightarrow Al^{3+} + 2H_2O$
Iron	<3.8	0.00	$FeOOH + 3H^+ \rightarrow Fe^{3+} + 2H_2O$

Carbonate buffer range

Soils containing $CaCO_3$ in their fine earth fraction (calcareous soils) are classified into the carbonate buffer range ($pH \geq 6.2$). Ca^{2+} is the dominant cation in the soil solution and in the exchange surfaces of the soil particles. The buffer capacity of soils in this range is proportional to the amount of $CaCO_3$ in the soil. In case $CaCO_3$ is evenly distributed 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.

Silicate buffer range

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 ($6.2 > pH \geq 5.0$). In this range the only buffer process acting in the soils is the weathering of silicates and the associated release of base cations, since the dissolution of aluminum compounds does not start in significant amounts until at pH less than 5.0. The buffer rate is often quite low. The buffer capacity, in turn, is high as it is formed by the massive storage of the silicate material. 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 the acid stress completely.

Cation exchange range

The soils are classified into the cation exchange buffer range when the cation exchange reactions play the major role in the acid buffering. This implies that the silicate buffer range is not capable of buffering the acid stress completely. The excess stress, not buffered by the reactions of the silicate buffer range, is adsorbed in form of H^+ - or Al-ions at the exchange sites, thus displacing the base cations. The cation exchange reactions are fast and, therefore, the buffer rate of soils in this range effectively counteracts any occurring rates of the acid stress. The total buffer capacity (= cation exchange capacity, CEC_{tot}) is generally rather low depending mainly on the soil texture. The remaining buffer capacity at any given time is quantified by *base saturation*, the percentage of base cations of the total CEC. As long as the base saturation stays above 5-10 percent, the excess stress is buffered by the cation exchange reactions and the soil pH takes a value between 5.0 and 4.2, the actual value depending on the base saturation.

Aluminum and iron buffer ranges

Below the critical value of the base saturation soils are classified into the aluminum buffer range. Hydrogen ions are consumed in releasing aluminum mainly from clay minerals. These reactions merely change the form of acidity from hydrogen ions to Al^{3+} . The leachate thus has a potential of acidifying the adjacent ecosystems. High aluminum ion concentrations characterize the soil solution and may cause toxic effects to bacteria and plant roots.

Aluminum compounds are abundant in soils, so that the buffer capacity hardly ever restricts the reaction. The soil pH is determined by the equilibrium with solid phases of aluminum compounds. As long as the soil pH stays within the range 4.2-3.8, the soil is classified into the aluminum buffer range.

At the extreme stage of acidification ($pH < 3.8$) soil may be classified into the iron buffer range. Increasing solubility of iron oxides is observed. This leads to visible (colour) symptoms in the soil profile, which is not the case for aluminum, although in quantitative terms aluminum may still act as a dominant buffer compound. The pH values as low as 3.8 indicate toxicity and nutrient deficiency to living organisms.

3. Model Development

3.1. Basic Assumptions

The requirement of a large spatial scale necessitates several simplifications in the model. The assumptions affecting the model structure itself are briefly described here, whereas the additional assumptions included in the model application at its present stage are discussed in a subsequent chapter.

The soil is considered as a homogeneous box. It is, however, possible to divide the soil into several layers if it is considered important when estimating the effects of soil acidification. In fact, this has already been done in connection with the IIASA surface water acidification model (Kämäri et al., 1984). In that case two layers were introduced.

The ion exchange and buffering properties of organic matter are not taken into account separately from the inorganic buffer systems. The information about the humus content of the soil or the thickness of the moor layer is not commonly available from different parts of Europe. At least in Northern Europe, where the accumulation of organic matter is significant, it would be important to take the buffering properties of organic matter into account.

The model was designed to focus on the year-to-year changes of soil acidity. Seasonal, monthly or even daily patterns of soil acidity are potentially very important as they may effectively act as key situations triggering biological effects. Our model describes the annual baseline level instead of the short term peaks of low or high acidity. In this way it does not directly focus on the potentially crucial events but it estimates trends of increasing probabilities of such events. This restriction of focus made it possible to omit redox processes and sulphate adsorption processes from the model. It was assumed that these processes generate seasonal variability in soil acidity which levels out in the long run without affecting the year-to-year trend.

The weathering rate of silicates and the connected release of base cations is assumed independent from the pH of the soil. In some laboratory experiments it has been shown that the release of silicates increases with decreasing pH (e.g. Wollast, 1967; Busenberg and Clemency, 1975; Stumm et al., 1983). However, the release of silica does not necessarily imply that base cations are released at the same rate. They may precipitate with aluminum compounds to form clay minerals. Increased base cation leaching is usually due to cation exchange reactions, not necessarily to increased weathering rate. In Solling (FRG), no deviation in the weathering rate of silicates from the long term average has been observed, although the pH of the soil has decreased (Matzner, unpublished).

3.2. Model structure

The model describes soil acidification in terms of the sequence of the buffer ranges. The model compares (i) the amount of stress (cumulative value over the time period of interest) to the buffer capacity, and (ii) the stress rate (year-to-year basis) to the buffer rate. The comparisons are made separately for the carbonate, silicate and cation exchange buffer ranges. The model thus assumes, that values for the buffering variables — buffer capacity and buffer rate — are determined separately for each of these buffer ranges.

All the buffering variables do not have to be considered in the model. The buffer rates of the carbonate range and the cation exchange range are so high that in practice they can not be exceeded by any occurring rate of acid stress. Moreover, the buffer capacities of silicate and aluminum ranges can not be exhausted in the time scale of hundreds of years. For the aluminum and iron ranges, an equilibrium approach was chosen. The soil pH

is assumed to stay in equilibrium with solid phases of aluminum compounds. Thus, a buffer rate is not needed. The iron range is also assumed to be quantitatively irrelevant for buffering at pH-values above 3.0. In this way the number of buffering variables actually included into the model reduces to three: buffer capacity of the carbonate range (BC_{Ca}), buffer rate of the silicate range (br_{Si}) and buffer capacity of the cation exchange range (BC_{Σ}).

The model is used by taking the given pattern of acid stress as the input variable. The program compares the (annual) acid stress to the buffer rate determined for the prevailing buffer range. It also compares the accumulated amount of acid stress to the buffer capacity. With these comparisons the program calculates which buffer range prevails each year, and then computes the approximation of the prevailing soil pH.

Acid stress to the top soil is partly or as a whole neutralized by the weathering of carbonate or silicate minerals. It is assumed that soils containing free carbonates (calcareous soils) always have a buffer rate high enough to neutralize any rate of acid stress. In this case the soil pH is assumed to stay at 6.2 as long as the buffer capacity of this range is not exhausted. In non-calcareous soils, neutralization depends on the intensity of silicate weathering (silicate buffer rate). As long as this buffer rate is larger than the acid stress no decrease in soil pH is assumed to occur.

If the acid stress exceeds the actual buffer rate of the silicates, the soil shifts into the cation exchange buffer range. Then the hydrogen ions gradually replace the base cations on the exchange sites of the soil particles thus decreasing the base saturation of the soil. The capacity of the cation exchange buffer system is depleted with a rate that equals the difference between the acid stress rate and the buffer rate of silicates. This has to do with the equilibrium between the ions attached to the soil particles and those dissolved in the soil solution. The gradual character was introduced also for the recovery. The soil pH is then estimated on the basis of the prevailing base saturation within the cation exchange range and the upper aluminum range at pH from 5.6 to 4.0. If the cation exchange capacity is totally exhausted the hydrogen ion concentration is assumed to be determined by equilibrium with solid phases aluminum which implies dissolution or precipitation of aluminum until an equilibrium state is reached. The specific equations incorporated into this model structure are presented in Appendix. The main characteristics of the model are summarized in the flow chart (Figure 1).

3.3. Model Demonstration

The dynamic features of the model are demonstrated in this section by producing two input-output patterns. These figures describe the reactions of only one soil type, Dystric Cambisol (Bd). Table 2 indicates the characteristics of this soil type assumed to prevail in the beginning of the 100 year study period. When fixing these values the reacting soil layer was assumed to be 50 cm. BC_{Ca} being zero indicates that Dystric Cambisol is free of lime. The input for this model demonstration consists of two hypothetical time patterns of the acid stress for the period of 100 years. The output is the time pattern of the soil pH, corresponding to the mean

hydrogen ion concentration of the soil layer of 50 cm.

Table 2: Initial conditions and parameter values for model demonstration (Soil type: Dystric Cambisol, Bd). Soil thickness of 50 cm is assumed.

Carbonate buffer capacity	BC_{Ca}	0.0	$kmol\ ha^{-1}$
Silicate buffer rate	br_{St}	1.0	$kmol\ ha^{-1}yr^{-1}$
Total cation exchange capacity	CEC_{tot}	1105.0	$kmol\ ha^{-1}$
Base saturation	β	0.15	
Volumetric water content at field capacity	θ_f	0.27	
Precipitation; Central Europe	P	0.90	$m\ yr^{-1}$
Evapotranspiration; Central Europe	E	0.50	$m\ yr^{-1}$

Figure 2 indicates that for this soil the soil pH is gradually declining from 4.6 down to 4.0 in 100 years when the soil is subject to a growing stress from 1 to 8 $kmol\ ha^{-1}yr^{-1}$. The silicate buffer range accounts for the buffering of 1 $kmol\ ha^{-1}yr^{-1}$ of the acid stress. The excess stress is buffered by the processes of the cation exchange range. The buffering within the silicate buffer range, essentially due to the weathering of the silicate mineral, is acting through all the buffer ranges. After 60 years the buffer capacity of the cation exchange range is decreased to a base saturation level of 5%. At this point, none of the higher buffer ranges is capable of buffering the stress, and the soil pH declines to the level which corresponds to the pH range of the aluminum buffer system. The acid stress, partly buffered within the silicate buffer range, finally determines the new equilibrium pH in the soil solution according to the aluminum solubility assumed. This process results in a slowly decreasing soil pH due to the growing stress rate.

A dramatic pattern of the acid stress was selected to summarize the dynamic behavior of the model (Figure 3). The pattern includes a constant stress of 8 $kmol\ ha^{-1}yr^{-1}$ for 30 years, a linear decline to zero in the subsequent 40 years, and a constant zero stress over the remaining 30 years.

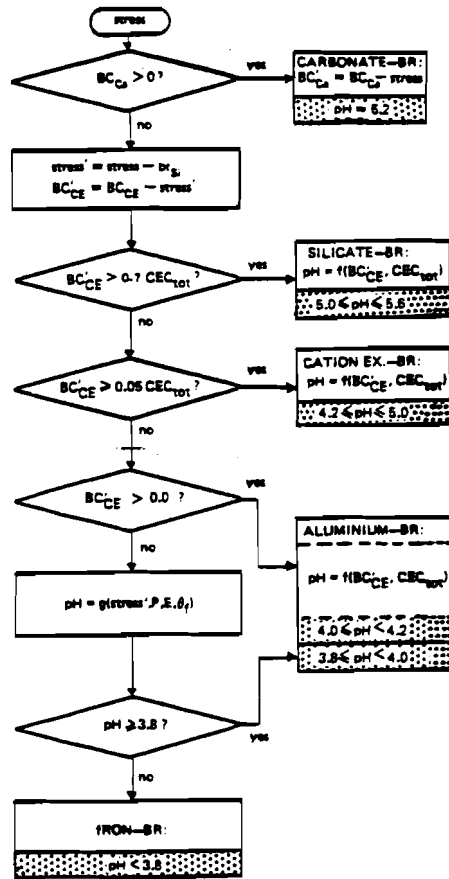


Figure 1. Flow diagram of the soil acidification model

The soil with initial conditions as in Table 2 reacts in the following way: First, there is a gradual but accelerating decline in pH from 4.6 to 4.2. Next, there is a rapid decline of pH near to the pH value 3.7. The buffer capacity of the cation exchange range is exhausted and the buffer rate of the aluminum range cannot keep the pace with the acid stress rate. Next, there is an increase of the soil pH to 4.0. At that point the acid stress has declined so that the joint buffering of the silicate and the aluminum range is capable of increasing the pH. Finally, a recovery starts from pH 4.0 upwards. This is possible because the acid stress declines to levels where the silicate buffer rate is sufficient for buffering the stress alone. During the gradual recovery in the soil, weathering slowly replaces hydrogen ions from the cation exchange sites. The cation exchange capacity is refilled, starting at pH 4.0, with a rate equal to the difference of the buffer rate of silicate buffer range and the rate of acid stress. A base saturation level of 4% will be reached by the end of the 100 year period.

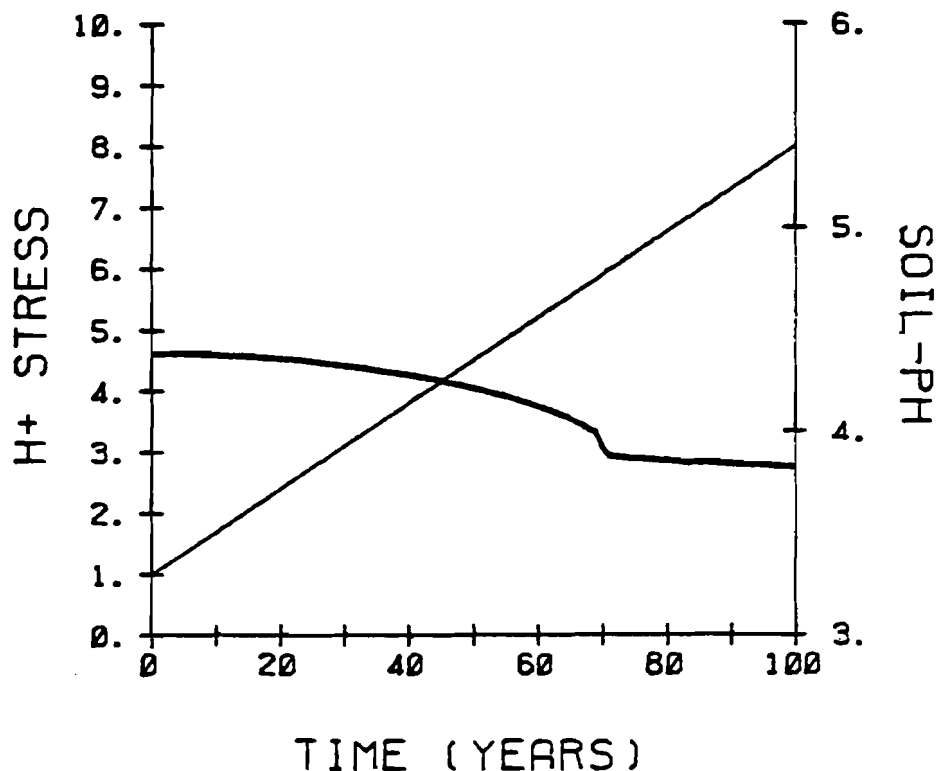


Figure 2. Input-output relationship: response of the soil to an increasing stress

4. Model Application

This application is part of the IIASA Acid Rain Project which has the general objective of analyzing alternative control strategies of the European sulfur emissions. The focus of the application is hence restricted to the stress due to air pollution. The IIASA framework sets the prerequisite of a large spatial scale. The project has provided an energy-emission model for generating scenarios of future sulfur emissions in Europe assuming optional programs for energy development and sulfur control (Alcamo et al. 1984). The computed emissions are converted into sulfur deposition scenarios by using the long-range transport model for air pollutants developed within the EMEP-program (see Eliassen and Saltbones, 1983). This model has been applied at IIASA by reducing it to a source receptor matrix (Alcamo et al. 1984). Sulfur deposition is then transformed into an approximation of the acid stress, and this information is used as the driving variable of the soil acidification model (Figure 4).

4.1. Specific Assumptions

For the time being, the acid stress was estimated on the basis of sulfur deposition only, simply by assuming acid stress to be proportional to sulfate ion equivalents in the water entering the soil. The actual acid stress associated with sulfur deposition depends on the neutralization intensity of, e.g. atmospheric dust and canopy. The spatial variation of these processes was

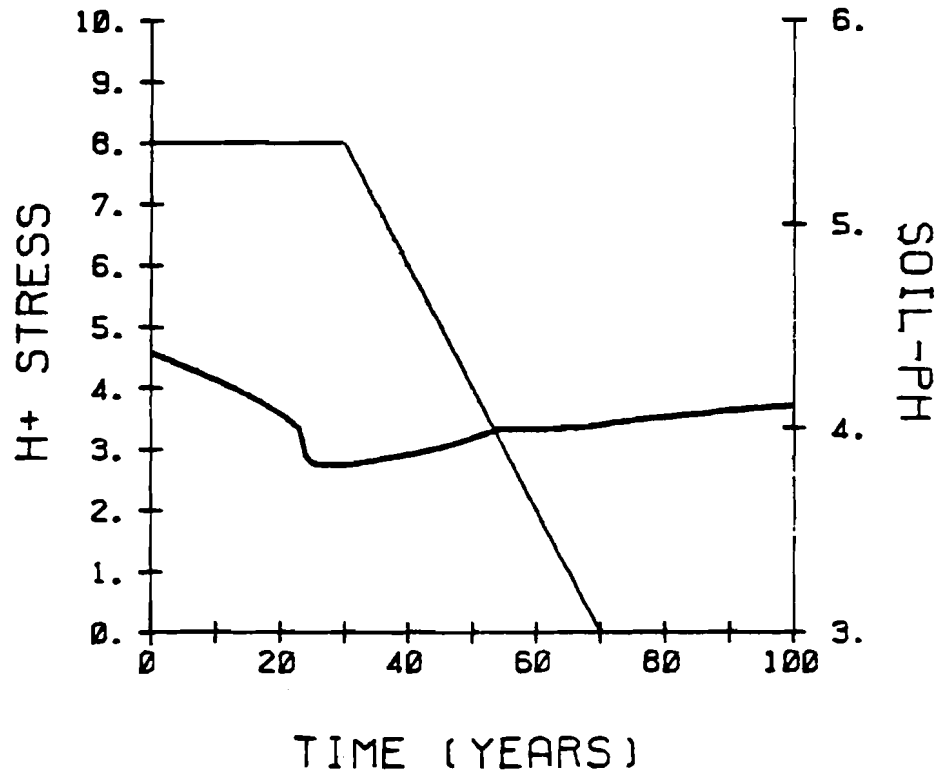


Figure 3. Input-output relationship: response of the soil to a declining stress

not taken into account. A single relationship was assumed as the first step for the whole of Europe. Internal proton production, i.e. proton production resulting from the excess accumulation of cations to the biomass and humus was not included in the estimates of acid stress.

The EMEP model assumes constant deposition velocity over all land surfaces (Eliassen & Saltbones, 1983). This assumption is necessary as the model covers the whole of Europe; it would be an enormous task to describe the spatial variability of the deposition velocity in detail. Model validation suggests that, in general, the assumption of constant deposition velocity can be supported when aiming at modeling the concentrations of sulfur compounds on a large spatial scale. From local experiments it appears, however, that forests have a rather strong filtering effect on air pollutants, so that the deposition velocity over forests is larger than that of open land by a factor of two to three, depending on the tree species. We believe that the apparent controversy between model validations and local experiments can be explained by assuming that within any of the large grid squares the average deposition velocity is the same as that selected for the EMEP model. In this way the EMEP model produces quite satisfactory results as far as the variability between the grid squares is concerned. In local scale within the grid square, however, it underestimates the deposition on forest land. As forests were the main target ecosystems for our model it was considered

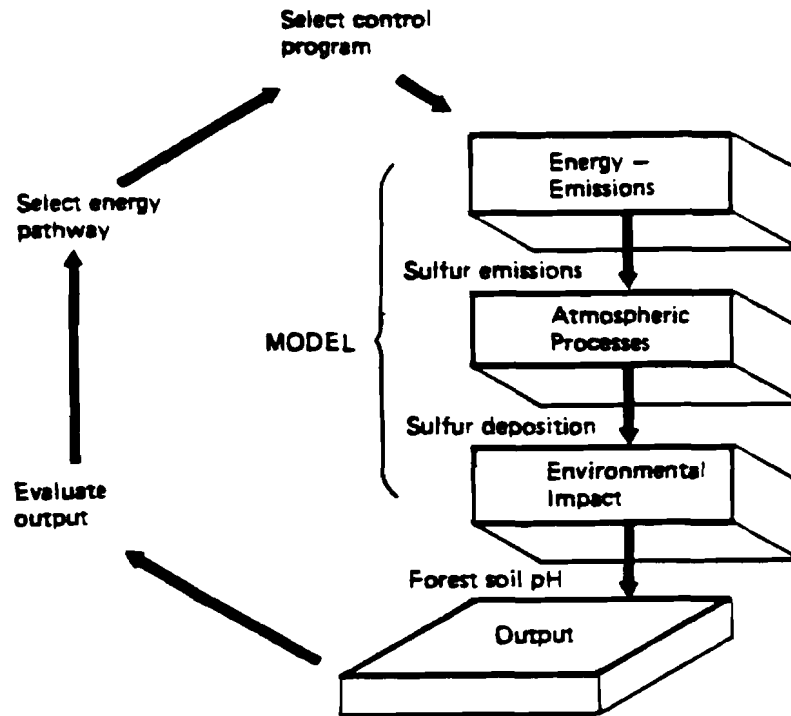


Figure 4. The IIASA acid rain framework and procedure for using the model

necessary to include the filtering effect into the model.

Based on the validation experiments of the EMEP model the average total deposition of a grid square, d_{tot} , was assumed correct. The deposition on the forest within this grid, d_f , was then assumed to be φ times larger than the deposition on open land, d_o

$$d_f = \varphi d_o \quad (1)$$

Since

$$f d_f + (1 - f) d_o = d_{tot} \quad (2)$$

where f is the fraction of forest within the grid, we get for d_f

$$d_f = d_{tot} \cdot \varphi / (1 + (\varphi - 1) f) \quad (3)$$

from which acid stress, as , was derived.

$$as = \sigma \cdot d_f \quad (4)$$

The factor σ implies that part of the sulfur deposition is neutralized before it enters the soil. This holds especially for dry deposition, which may be neutralized by dust, canopy, etc. The above calculation procedure takes into account (i) the estimated gross deposition on each grid square, (ii) the filtering factor φ , (iii) the fraction of forests in each grid square, f , digitalized from the World Forestry Atlas (Weltforstatlas, 1975), and (iv) the acid stress factor, σ . As an output it produces the allocation of deposition between forests and the agricultural land within each grid square. This specific feature of the IIASA model gives the first priority to the long range transport model as far as large scale variability of deposition is concerned and yet describes the filtering effect of forests by including small scale information on the distribution of forests vs. open land within the grid square. A factor $\varphi = 2$ is used as long as detailed information on the spatial distribution of φ is not available. The acid stress coefficient, σ , seems to have values between 0.5 and 0.75 in some European forests (e.g. Matzner, 1983; Wright and Johannessen, 1980). The value $\sigma = 2/3$ was chosen as a tentative approximation.

It is conceivable that forests, as they represent areas neglected by agriculture, grow on particularly susceptible soils. Soils which have low specific weathering rates and low levels of base saturation are more susceptible to acidification than soils on the average. The concentration of forests on poor soils, although hypothetical, was considered so obvious that it was included as part of the model. Rather than assuming the fraction of forests constant on all soil types we used the following calculation procedure: Forests of a given grid square were allocated starting from soil types with the lowest weathering rates and cation exchange capacity values and continuing until all forests were distributed. In this way agriculture was located on the most fertile soils whereas poor soils of a grid were assumed for forests.

In the presentation of results an important indicator is the "critical acidity". At present the switch to aluminium buffer range (base saturation 0.05, pH 4.2) is assumed to imply an increased risk for forest damage. There are several reasons why this degree of acidity was assumed to be critical: soil chemistry changes quite drastically; Al-concentration in the soil solution increases and Ca/Al-ratio reaches the level that implies the risk of soil borne toxicity to tree roots (Matzner and Ulrich, 1984; Ulrich et al. 1984). More research, however, would be needed for relating the risk of forest damage to the soil acidity. The final decision about the 'critical pH' is left to the model user.

4.2. Initialization of Buffering Variables

Initialization of the soil variables was based on the chemistry information available on European soils, and on the soil thickness selected to approximate the rooting zone. The buffer capacity of the carbonate range is proportional to the lime content of the soil; the buffer rate of the silicate range is related to the chemical weathering rate of the silicate minerals; the buffer capacity of the cation exchange rate depends on the clay content and on the base saturation of the soil; and the buffer rate of the aluminum range depends on the accessibility of aluminum compounds. Although such relationships, especially those regarding the aluminum accessibility are

only partially understood, they can be used as a guideline in quantifying the susceptibility of the soils to acidification. The values for the buffer capacities and buffer rates were initialized accordingly based on the International Geological Map of Europe and the Mediterranean Region (1972) and the FAO-UNESCO Soil Map of the World (1974). The depth of the reacting soil was assumed 50 cm throughout the study area. The year 1960 was selected as being the baseline year.

All information regarding soils was stored into a computerized grid-based format. Each grid square had the extension of 1 degree longitude times 0.5 degrees latitude. In this way the size of a grid was fixed at 56 km in the south-north direction, but in the east-west direction it varied from 91 km to 38 km depending on the latitude. The number of the grid squares was 2304.

Detailed soil chemistry information regarding the other soil variables was available from the Soil Map. The fraction of each soil type within the grid square was computerized with an accuracy of 5 percent units. The resolution of the map is such that the standard grid square was composed of 1-7 soil types. The number of different soil types was 80. The soil data base consists of 5212 soil units, the mean number of soil types per grid square being 2.2. One 70 year simulation for the whole of Europe requires then about 365,000 model runs.

Initial values for the soil variables were given for every soil type (Table 3). The Soil Map, however, could not provide the information regarding the buffer rate of the silicate buffer range which is equal to the weathering rate of the parent material. The approximation of this variable was based on other sources. Ulrich (1983b) reports a range of variation in European soils from 0.2 to 2.0 $kmol\ ha^{-1}\ yr^{-1}\ m^{-1}$. Four classes for the reacting 50 cm soil layer were introduced with the following buffer rates (in $kmol\ ha^{-1}\ yr^{-1}$):

class	1	2	3	4
buffer rate	0.25	0.50	0.75	1.00

The Geological Map was used to determine parent materials of soils in each grid square. Depending on the dominant parent material the soil of each grid square was classified into one of the above categories.

Based on this information the model is applicable for producing acidification scenarios for forest soils. The model is run separately for each soil type within each grid square. An estimate of the soil pH is produced as the output.

4.3. Results of Model Runs

Two example scenarios were introduced using the IIASA energy-emission model, and the long range transport model supplied by the EMEP programme. From 1960 until 1980 the scenarios were identical. From that on the scenarios departed so that the 'high' deposition scenario assumed high rates of energy development throughout Europe, as defined by the ECE 'trends continued' scenario (ECE, 1983) linearly extrapolated to 2030. The 'low' deposition scenario was constructed according to the ECE

Table 3. Buffer capacities of the carbonate and cation exchange buffer ranges estimated for the year 1960 for soil types of the FAO-UNESCO Soil Map of the World (1974). Soil thickness of 50 cm is assumed.

soil type	BC_{Ca}	BC_{CE}	soil type	BC_{Ca}	BC_{CE}	soil type	BC_{Ca}	BC_{CE}
	$kmol\ ha^{-1}$			$kmol\ ha^{-1}$			$kmol\ ha^{-1}$	
Ao	200.0	910.0	Kk	8000.0	1170.0	Vc	32000.0	1170.0
Bc	500.0	1225.0	Lc	3000.0	170.7	Vp	9000.0	3640.0
Bd	0.	165.8	Lf	0.	138.8	Wd	0.	47.3
Be	500.0	1824.0	Lg	0.	146.3	We	500.0	1410.5
Bg	500.0	180.0	Lo	0.	107.3	Xk	43000.0	1170.0
Bh	0.	136.5	Lv	3000.0	1225.0	Xy	40000.0	1225.0
Bk	25000.0	1470.0	Mo	0.	1495.0	Zg	15000.0	1225.0
Bv	0.	2210.0	Od	0.	72.0	Bc-Lc	3000.0	685.6
Ch	0.	390.0	Oe	0.	168.8	I-Bc	200.0	1050.0
Ck	19000.0	2535.0	Pg	0.	180.0	I-Bc-Lc	1500.0	469.1
Cl	0.	419.3	Ph	0.	49.0	I-Bd	0.	151.2
Dd	0.	136.5	Pl	0.	68.3	I-Be	0.	765.6
De	0.	136.5	Po	0.	78.0	I-Be-Lc	1500.0	533.9
Dg	0.	468.0	Pp	0.	239.2	I-Bh-U	0.	136.5
E	20000.0	2600.0	Qc	100.0	227.5	I-C	500.0	910.0
Gd	0.	126.8	Ql	0.	117.0	I-E	10000.0	1750.0
Ge	0.	302.3	Rca	0.	47.3	I-L	0.	149.3
Gh	0.	146.3	Rcb	0.	136.5	I-Lc	1500.0	153.6
Gm	0.	183.8	Rcc	500.	857.5	I-Lo-Bc	0.	408.5
Hc	7000.0	1170.0	Re	0.	136.5	I-Lc-E	10000.0	1500.0
Hg	500.0	1820.0	So	500.0	1183.0	I-Po	0.	126.8
Hh	1000.0	321.8	Sm	0.	236.3	I-Po-Od	0.	108.5
Hl	0.	312.0	Th	0.	127.5	I-Rc-Xk	20000.0	1500.0
I	0.	136.5	Tm	0.	136.5	I-Re-Rx	0.	106.8
Jc	8000.0	315.0	To	0.	183.8	I-U	0.	136.5
Je	200.0	1008.0	Tv	0.	120.0	Lo-Lc	1500.0	139.1
Kh	0.	136.5	U	0.	136.5			

'conservation' scenario, assuming lower rates of energy use and, in addition to that, effective measures taken for the control of the sulfur emissions (Figure 5). The specific method of generating different scenarios is presented by Alcamo et al. (1984).

The model can be used for producing estimates of the time patterns of the total forest area with soils below a selected critical pH for any scenario (Figure 6). The area of the forest in each grid square is calculated and the time evolution of the area of European forests with soil pH below a selected critical value is then displayed. Another option is to display the areas with soils below a critical pH for a selected year in a map format. Different shadings indicate the percentage of the total forest area with soil pH below the selected value (see Figures 7 and 8a, b).

As part of the IIASA study this application of the soil acidification model is designed for quick comparisons of sulfur emission scenarios. It is up to the model user to decide what kind of scenarios should be compared. The two examples were selected to demonstrate the model behaviour. Therefore, the examples are relatively useless as far as selecting feasible policy options is concerned. The following paragraphs discuss the effects of the 'low' vs. the 'high' scenario but this discussion is intended merely to demonstrate the properties of the model.

By the year 1980, that is, assuming the historical deposition pattern, the model predicts a decline in the forest soil pH in relatively large regions of Central Europe (Figure 7). Continuing with the 'high' deposition scenario the area of low pH substantially enlarges by the year 2010 and much of the soils in Central Europe and Southern Scandinavia reach the aluminum buffer range (Figure 8a). When the 'low' scenario is used as the input, the results indicate much less risk of forest damage by the year 2010. As indicated by Figure 8b the forest area with more acidic soils than the threshold is estimated two times larger with the 'high' scenario than with the 'low' scenario.

5. Discussion

The model developed in this study can be used for quantifying some aspects of the acidification problem of forest soils which have earlier been discussed using qualitative terms. The soil acidification model and the application to the European overview are simplifications, which necessarily include uncertainties. Many solutions, as they stand now, are crude approximations which need clarification in future research. It is the hope of the authors, however, that the model structure would act as a tool for organizing the data and for identifying research needs. Even in its present stage

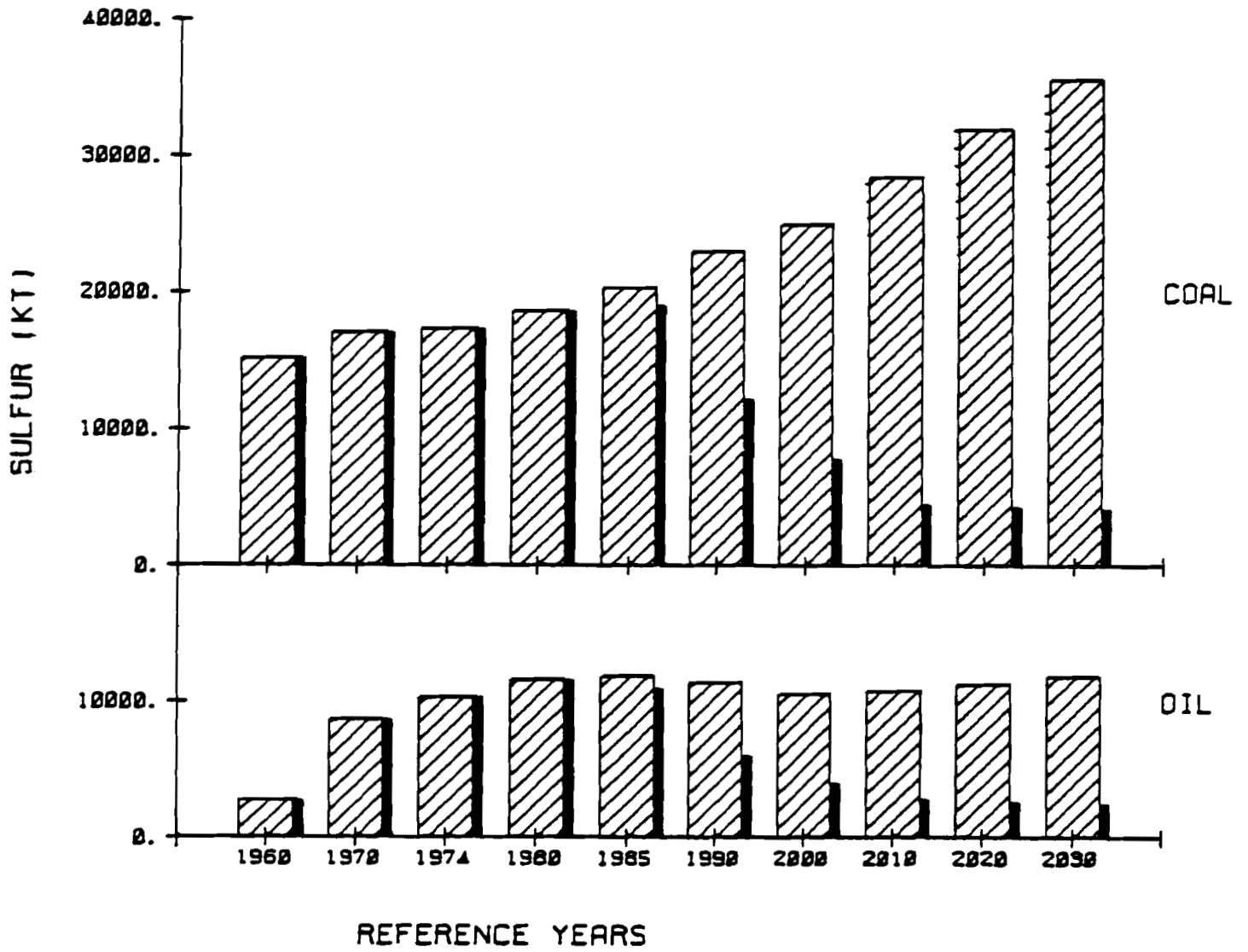


Figure 5. Total sulfur emitted in Europe according to the 'high' and 'low' emission scenario from coal and oil sectors

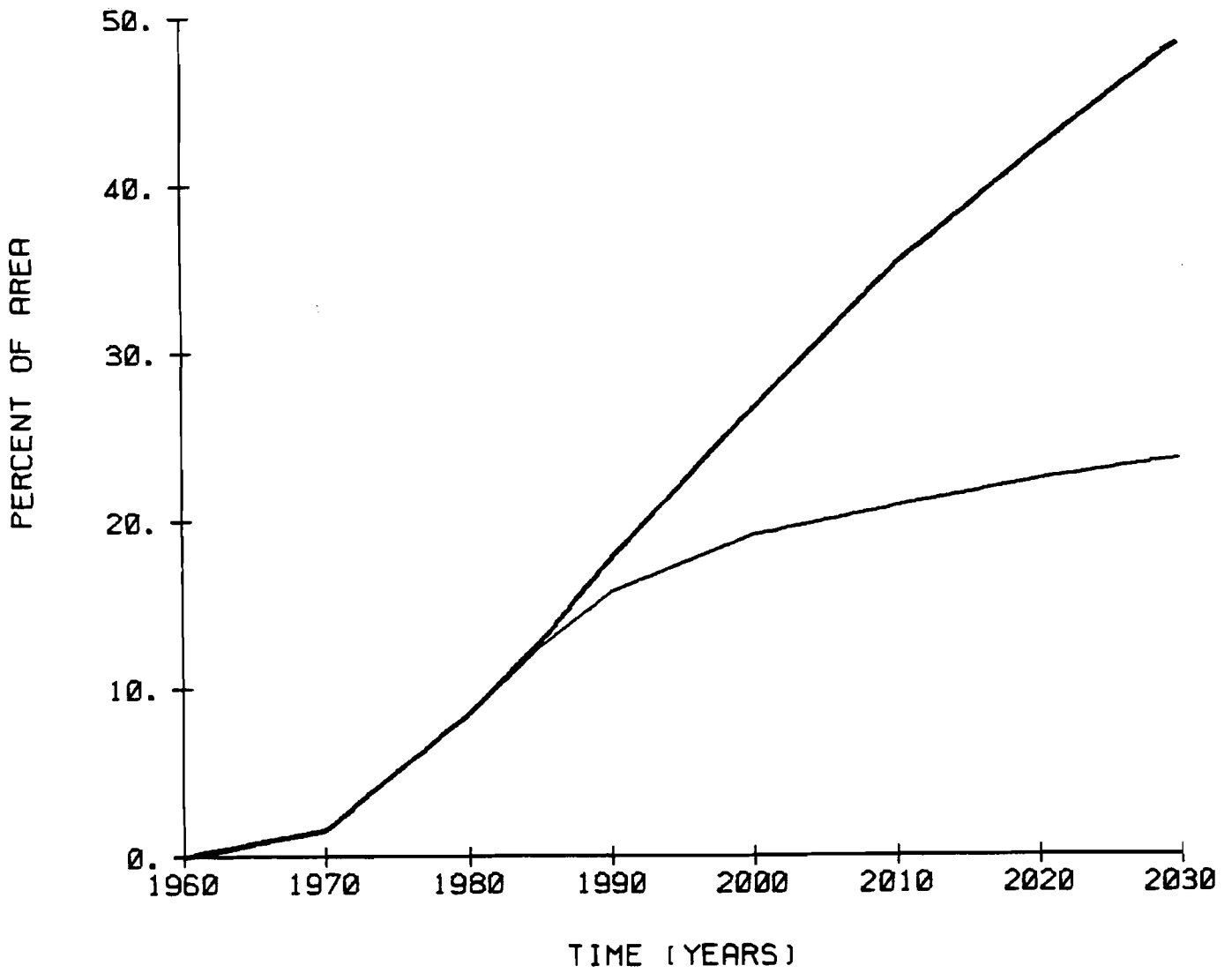


Figure 6. Time evolution of the total forest area with soils in aluminum and iron buffer ranges (pH less than 4.2) in Europe assuming the two emission scenarios

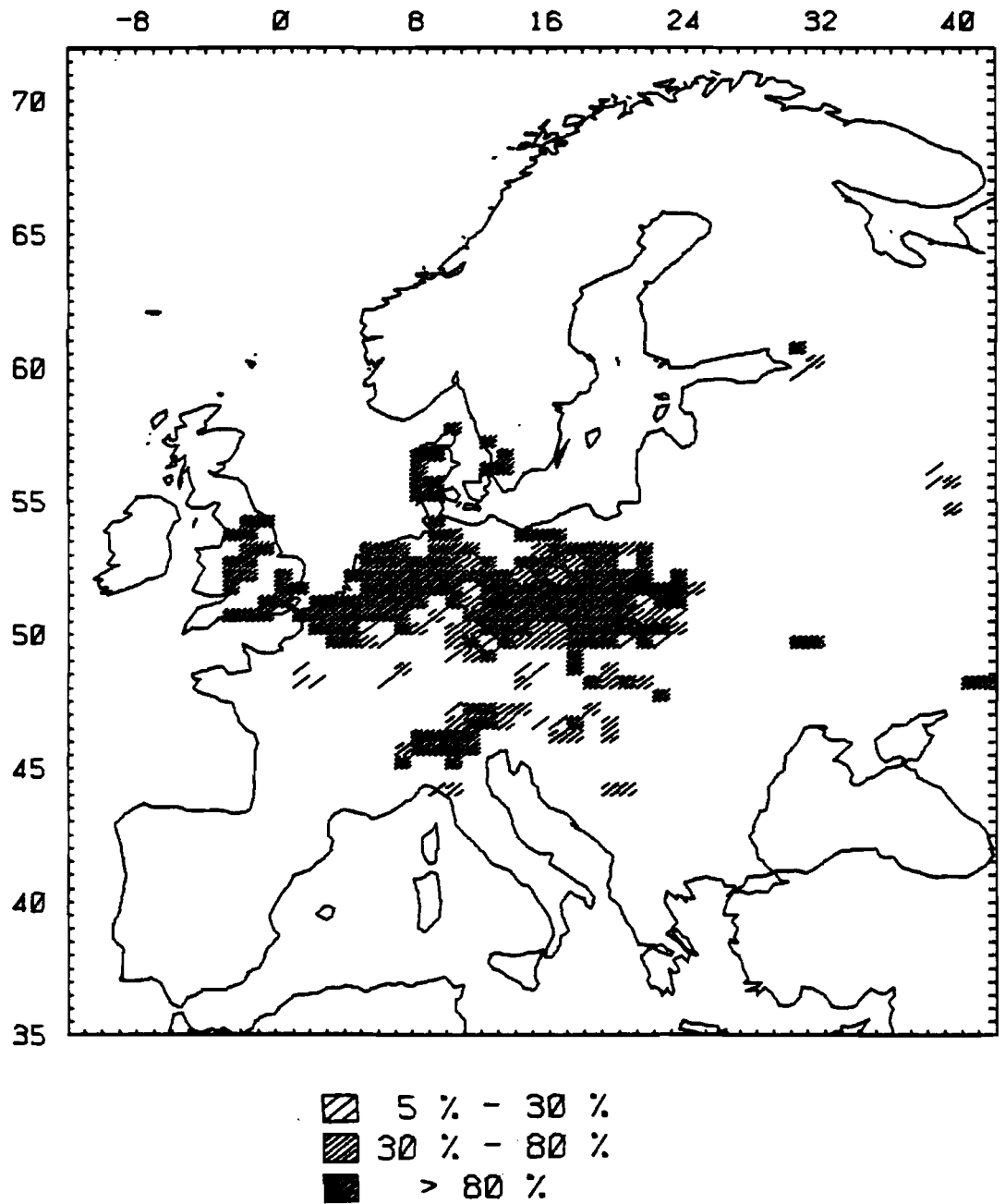


Figure 7. Model estimates of forest soils below pH 4.2 in 1980. The shading determines the fraction of forest soils below the threshold pH in each grid.

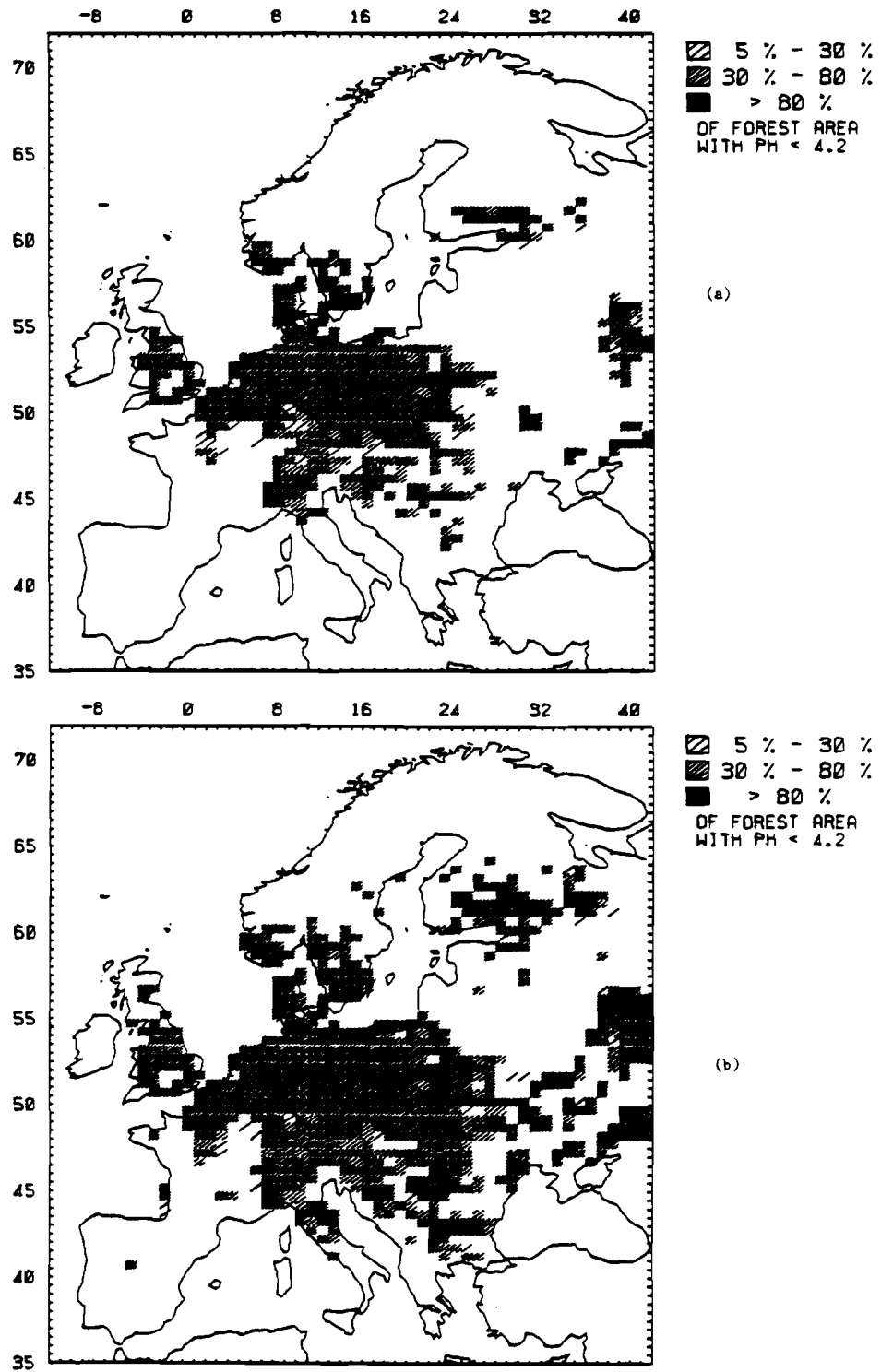


Figure 8. A comparison of the area of risk in 2010, (pH < 4.2), resulting from the high emission scenario (a) and from the low emission scenario (b).

the model might appear useful in evaluating policies to combat the acidification of forest soils.

The model makes a distinction between reversible and irreversible changes in the soil chemistry. Exhaustion of the buffer capacity is more or less irreversible. The case of an insufficient buffer rate, in turn, may be reversible: The buffer rate is again sufficient when the stress rate (annual load) is reduced below a threshold; this threshold is the value of the buffer rate variable. This feature of the model should be useful as it indicates whether a decrease in the acid stress would result in a recovery of the soil, or whether it would merely cause a delay in the acidification process.

The model, designed for studies on forest soils, appears too complex for studies on agricultural soils. Intensive agriculture maintains high pH values in soils by means of liming and other practices. In theory, the model could be used for calculating the amount of lime needed to counteract, for example, the acidic deposition. This calculation, however, can be done using more straightforward methods.

The application of the model to the problem of acidic deposition in Europe indicates that soil buffering fails in maintaining adequate pH levels in large parts of Central Europe. In Northern Europe, although the buffering is generally less efficient, the acidic deposition would cause less trouble in this respect. This does not prove that the problem of soil acidification is restricted to Central Europe. Acidification due to biomass accumulation, i.e. the so-called internal proton production, has a special role in Northern Europe where low temperatures retard biomass decomposition. High internal proton production increases the susceptibility of the environment to the acidification due to air pollutants. This additional stress needs to be addressed in future research.

The soil variables were initialized for 1960. This does not imply that no acid stress was assumed before that time. The initialization should be viewed as fixing a reference point rather than a manifestation of the state of virgin forests. The initialization should be based on field measurements; in the present application this goal was only partially fulfilled.

The reacting volume was fixed at the top 50 cm of the soil. No horizontal gradients were explicitly assumed. Including deeper layers into the reactive part of the soil would add to the reacting volume and it would thus postpone the possible problem. Including the gradients would involve faster acidification in the very top of the soil and slower acidification in the deeper layers. The above results correspond to the average situation in the volume. This average value may be inaccurate in some cases due to the nonlinearities of the model. Moreover, the model assumes that all deposition actually reacts within the top soil. This may not always be the case. If part of the deposition flows unchanged through the top soil, the soil response will be delayed and the acidification problem is transferred into the adjacent ecosystems or to the groundwater. Within the IIASA Acid Rain Project a regional lake acidification model has been developed, where the soil pH model is used for describing the soil chemistry in the catchment.

Soil acidification poses a threat to forest ecosystems and generates predisposing stress in ecosystems as defined by Manion (1981). Forest damage, however, is a multicausal phenomenon. Many factors are involved such as ozone pollution, heavy metals, exceptional climatic conditions, and

cultivation of tree species outside of their natural sites. The interactions of soil acidification and the other factors deserve concerted research effort. It does not seem possible today to describe the forest damage in satisfactory detail with any specific model. But emphasizing the complexity of the forest damage as an argument against serious modeling efforts may well cause a delay in obtaining a better understanding of the phenomenon.

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APPENDIX

The capacity of the cation exchange buffer system, BC_{CE}^t , is depleted with the rate of acid stress, as^t , minus the buffer rate of silicates, br_{Si} (A1). A non-linear relationship is assumed between the base saturation and the soil pH within the silicate, cation exchange and the upper aluminum buffer range, as long as $BC_{CE}^t \geq 0$, at pH from 5.6 to 4.0 (A2).

$$BC_{CE}^t = BC_{CE}^{t-1} - (as^t - br_{Si}) \quad (A1)$$

$$pH = 4.0 + 1.6 (BC_{CE}^t / CEC_{tot})^{3/4} \quad (A2)$$

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

If $BC_{CE}^t = 0$, 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, (A3, A4):

$$[Al^{3+}]_s = V_f [Al^{3+}]^{t-1} / [V_f + (P - E)] \quad (A3)$$

$$[H^+]_s = [V_f [H^+]^{t-1} + (as^t - br_{Si})] / [V_f + (P - E)] \quad (A4)$$

where V_f is the volume of soil solution at field capacity and P and E mean annual precipitation and evapotranspiration respectively. On annual basis the infiltrating water volume is assumed to equal $P - E$. The soil solution volume is simply defined by

$$V_f = \theta_f z \quad (A5)$$

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 soil. Aluminum is dissolved or precipitated until the gibbsite equilibrium state (A6) is reached. This process involves a change from disequilibrium concentrations as defined in equation (A7).

$$[Al^{3+}]^t / [H^+]^t = K_{so}, \quad K_{so} = 10^{-8.5} \quad (A6)$$

$$3([Al^{3+}]_s - [Al^{3+}]^t) = [H^+]^t - [H^+]_s \quad (A7)$$

Combining equations A6 and A7 yields a third order equation which has a single real root (A8).

$$3K_{so} [H^+]^t + [H^+]^t - 3[Al^{3+}]_s - [H^+]_s = 0 \quad (A8)$$