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**Effects of various deposition scenarios on the aluminium hydroxide  
content of Dutch forest Soils**

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## ABSTRACT

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Effects of four deposition scenarios up to 2010 (two scenarios) and 2050 (two scenarios) on the Al hydroxide content of non-calcareous sandy forest soils in the Netherlands were calculated with a simple model, using a 10 x 10 km grid. The scenarios reflected the effects of policy measures to reduce sulphur and nitrogen emissions. Compared to expected emission reductions, additional measures only had a small effect on the predicted change in Al hydroxide content, except for forest soils near farms with intensive animal husbandry. Results showed that the relative decrease in Al hydroxide content was nearly always less than 50% for all scenarios. The median value was approximately 10%. Depletion of Al hydroxides was largest in areas with intensive animal husbandry but even here a depletion above 50% hardly occurred. However, Al depletion can be very fast (< 10 to 20 years) in forest soils located near farms with intensive animal husbandry.

Keywords: aluminium depletion, acid deposition, modelling, emission reductions

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## PREFACE

Model simulations and laboratory experiments performed earlier showed that depletion of the pool of readily available aluminium is a real threat within several decades at present deposition levels. The question remained what will happen with this pool when emission reductions, as announced in several policy documents, are realized. This report gives an answer to this question by analysing the impacts of various deposition scenarios on the content of Al hydroxides in non-calcareous sandy forest soils in the Netherlands. The research was carried out by the DLO Winand Staring Centre during the period March to May 1992.

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## SUMMARY

Nearly all Dutch forests on non-calcareous sandy soils occur in the aluminium buffer range. Depletion of the readily available pool of Al hydroxides may have serious ecological consequences. Previous studies showed that this may occur within several decades at present deposition levels. This report gives results of the impact of various deposition scenarios on the Al hydroxide content of non-calcareous sandy forest soils in the Netherlands. Planned emission reductions are evaluated on their effect regarding Al depletion. The research was carried out by the DLO Winand Staring Centre during the period March to May 1992.

The decrease in Al hydroxide amount was calculated from the accumulated exceedance of present acid loads over critical acid loads. The critical acid load was calculated by requiring a negligible depletion of Al hydroxides. Effects of dynamic processes such as cation exchange and mineralization/immobilization dynamics were not included. Calculations were made for forest-soil combinations in a 10 x 10 km grid, distinguishing 12 tree species and 18 non-calcareous sandy soil types.

Calculations were made with four deposition scenarios: two scenarios from 1990 to 2010 and two scenarios from 1990 to 2050. In each case, one scenario reflected planned emission reductions (generic policy measures) and one scenario reflected the effect of additional policy measures on a regional scale. Critical acid loads were derived from an earlier study. In order to calculate the relative decrease in Al hydroxide amount, present values were derived from an inventory of the chemical soil composition below 150 forest stands in 1990. The variation in Al hydroxide amounts that was found for each soil type was included in the calculations.

Results showed that the decrease in Al hydroxide amount was relatively low compared to the present amounts. A depletion of 50% hardly occurred in any of the scenarios even in areas of intensive animal husbandry. The median depletion ranged between 8 and 12% depending on the scenario. This implies that planned emission reductions seem sufficient to avoid complete depletion of Al hydroxides. The effects of additional regional measures are relatively small. However, Al depletion can be very fast (less than 10 to 20 years) in forest soils located near farms with intensive animal husbandry. Here additional policy measures could have an important effect.

## 1 INTRODUCTION

In this report, results are given of the effects of four deposition scenarios (two scenarios from 1990 to 2010 and two scenarios from 1990 to 2050) on the depletion of Al hydroxides in the topsoil (0-30 cm) of non-calcareous sandy forest soils in the Netherlands. In each case, one scenario reflected planned emission reductions (generic policy measures) and one scenario reflected the effect of additional policy measures on a regional scale. The major aim of this research, that was carried out by the DLO Winand Staring Centre during the period March to May 1992, was to evaluate whether additional measures (regarding NH<sub>3</sub> emissions) are necessary to prevent Al depletion on a regional scale. Depletion of Al hydroxides is a real danger in Dutch forest soils, since nearly all Dutch forests on non-calcareous sandy soils occur in the aluminium buffer range (Heij et al., 1991). This implies that, apart from weathering and nitrogen retention (i.e. immobilization, uptake and denitrification), acid atmospheric deposition is buffered by mobilization of Al from hydroxides in these soils. Both laboratory experiments (Mulder et al., 1989) and model simulations (De Vries and Kros, 1989) show that the pool of readily available Al in forest topsoils can be depleted in several decades at present deposition levels. This causes a further decrease in pH and furthermore the availability of phosphate may decrease. In order to avoid this effect, critical deposition levels of acidity have been derived (De Vries, 1993; De Vries et al., 1992).

The effect of various deposition scenarios, aiming to reach these critical loads between 2010 and 2050, on non-calcareous sandy forest soils have been assessed before by dynamic model calculations (De Vries et al., 1991). These calculations showed that depletion of Al hydroxides in such forest topsoils, is not a real threat in the Netherlands when necessary emission reductions are taken before 2010. However, the deposition data used in these calculations were average values for 20 deposition areas. Furthermore, the Al hydroxide content used was an average value for the various soil types considered in the application (De Vries et al., 1991). On a smaller scale, depletion of Al hydroxides may still occur because of a higher deposition level and/or a lower Al hydroxide content, compared to the average values used in the previous study (De Vries et al., 1991). Especially in areas with intensive animal husbandry high present loads may cause a relative fast depletion. In this report, these aspects were accounted for by using a grid system of 10 km x 10 km and including the variability in Al hydroxide content in forest soils.



## 2 CALCULATION METHOD

In order to calculate the decrease in the Al hydroxide content in the topsoil (0-30 cm), the dynamic model RESAM (Regional Soil Acidification Model) was used in previous studies (De Vries and Kros, 1989; De Vries et al., 1991). However, use of this model for a regional application for a 10 x 10 km grid gives rise to problems due to the number of data required and the computation time. Consequently, we used a simple approach by calculating the absolute decrease in Al hydroxide amount for each forest-soil combination in each grid during a certain time period according to:

$$\Delta Al_{ox} = \sum_{t=1}^{t=n} EL(Ac_{pot})_t - CL(Ac_{pot}) \cdot n \quad (1)$$

where  $\Delta Al_{ox}$  stands for the absolute decrease in Al hydroxide amount ( $\text{mol}_c \text{ ha}^{-1}$ ),  $EL(Ac_{pot})_t$  for the expected load of potential acidity at year t ( $\text{mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$ ),  $CL(Ac_{pot})$  for the critical load of potential acidity ( $\text{mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$ ) and n for the considered time period (yr). The relative decrease in the Al-hydroxide amount in the forest topsoil was calculated by dividing the absolute decrease in Al hydroxide amount (Eq. 1) by the present amount of Al hydroxides in this compartment.

The (present) load of potential acidity was calculated as:

$$EL(Ac_{pot}) = EL(SO_x) + EL(NO_x) + EL(NH_x) - BC_{dw}^* \quad (2)$$

where  $EL(SO_x)$ ,  $EL(NO_x)$  and  $EL(NH_x)$  stand for the present loads of  $SO_x$ ,  $NO_x$  and  $NH_x$  respectively and  $BC_{dw}^*$  for the seasalt corrected bulk deposition of base cations. The load of potential acidity is based on the assumption that leaching of  $NH_4$  from the rootzone of forest soils is negligible (Sverdrup et al., 1990). This implies that  $NH_3$  is not counted as a base but as a (potential) acid that should be added to the acidifying effect of  $SO_x$  and  $NO_x$ .

Critical acid loads were derived using a steady-state soil acidification model according to (De Vries, 1991):

$$CL(Ac_{pot}) = BC_{dd}^* - BC_{gu} + BC_{we} + N_{gu} + N_{im}(\text{crit}) + Ac_{le}(\text{crit}) \quad (3)$$

where  $BC_{dd}^*$  is the seasalt corrected dry deposition of base cations,  $BC_{gu}$  and  $N_{gu}$  are the growth uptake (net uptake needed for forest growth) of base cations and nitrogen respectively,  $BC_{we}$  is the base cation weathering,  $N_{im}(\text{crit})$  is a critical long-term nitrogen immobilization, and  $Ac_{le}(\text{crit})$  is a critical leaching flux of acidity. The element fluxes in Eq. (3) are all given in  $\text{mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$ . An overview of the various assumptions to derive Eq. (3) with a justification of it is given in De Vries (1991).

The critical acidity leaching flux was calculated as the sum of Al leaching and H-leaching. The critical Al leaching flux ( $Al_{le}(crit)$ ) was calculated by requiring a negligible depletion of Al hydroxides:

$$Al_{le}(crit) = r \cdot BC_{we} \quad (4)$$

where  $r$  is a stoichiometric equivalent ratio of Al to BC weathering. For  $r$  a value of 2 was used (De Vries, 1991). In a standard calculation of critical acid loads, the value of  $Al_{le}(crit)$  is calculated by requiring that

- the Al concentration stays below  $0.2 \text{ mol}_c \text{ m}^{-3}$
- the molar Al/Ca ratio stays below 1.0
- Al depletion does not occur (cf De Vries, 1991; De Vries et al., 1992).

However, up to a soil depth of 30 cm the requirement of negligible Al hydroxide depletion appears to be most stringent (De Vries et al., 1993).

The critical H leaching flux ( $H_{le}(crit)$ ) was calculated as:

$$H_{le}(crit) = FW \cdot [H](crit) \quad (5)$$

where  $[H](crit)$  is a critical H-concentration ( $\text{mol}_c \text{ m}^{-3}$ ) and  $FW$  is the water flux ( $\text{m}^3 \text{ ha}^{-1} \text{ yr}^{-1}$ ) that is taken equal to the precipitation surplus leading from the rootzone. The critical H concentration was related to the critical Al concentration according to:

$$[H](crit) = ([Al](crit) / KAl_{ox})^{1/6} \quad (6)$$

where  $KAl_{ox}$  is the gibbsite equilibrium constant in  $\text{mol}_c^{-2} \text{ m}^6$ . For  $KAl_{ox}$  we used a value of  $3 \times 10^2 \text{ mol}_c^{-2} \text{ m}^6 (=10^8 \text{ mol}^{-2} \text{ l}^2)$ . The value of the critical Al concentration was determined by the critical Al leaching flux divided by the water flux.

$$[Al](crit) = \frac{Al_{le}}{FW} \quad (7)$$

### 3 DISTRIBUTION OF NON-CALCAREOUS SANDY FOREST SOILS

In calculating the effects of various deposition scenarios, we used a grid system of 10 km x 10 km. A 10 km x 10 km gridcell was considered an optimum between the number of areas and the spatial variability in atmospheric deposition within each area. Detailed information regarding tree species and soil types exists at this scale. Information on the area (distribution) of each specific forest-soil combination in each gridcell was derived by a grid-overlay of a digitized 1 : 250 000 soil map with a spatial resolution of 100 m x 100 m and a data base with tree species information with a spatial resolution of 500 x 500 m. A distinction was made in 12 tree species and 18 soil types. Tree species included were *Pinus Sylvestris* (Scotch Pine; 38.2%), *Pinus Nigra* (Black Pine; 5.9%), *Pseudotsuga Menziesii* (Douglas Fir; 5.5%), *Picea Abies* (Norway Spruce; 5.1%), *Larix Leptolepis* (Japanese Larch; 5.7%), *Quercus Robur* (Oak; 17.4%), *Fagus Sylvatica* (Beech; 4.1%), *Populus Spec* (Poplar; 4.6%), *Salix Spec* (Willow; 2.4%), *Betula Pendula* (Birch; 7.4%), *Fraxinus Nigra* (Ash; 1.9%) and *Alnus Glutinosa* (Black Alder; 1.9%). Soil types were differentiated in 18 non-calcareous sandy (mainly podzolic) soils on the basis of a recent 1 : 250 000 soil map of the Netherlands. The total number of forest/soil combinations for all grids on non-calcareous sandy soils was 12514. The number of forest/soil combinations in a gridcell varied between 1 and 125.

An overview of the percentage of forests on the various non-calcareous sandy soils in the Netherlands thus derived is given in Table 1. Approximately 85% of all forests are located on non-calcareous sandy soils, i.e. nearly all coniferous forests (60%) and about 2/3 of the deciduous forests (40%). For all soil types given in Table 1, a distinction was made between fine textured and coarse textured variants, because of differences in weathering and transpiration rate.

*Table 1 Area of the non-calcareous sandy forest soils in the Netherlands as a percentage of the total forested area<sup>1)</sup>*

Soil type (FAO, 1988)	Area (%)		
	fine textured	coarse textured	all
CambicPodzol	2.7	8.7	11.4
Gleyic Podzol	14.9	0.3	15.2
Carbic Podzol	11.3	8.3	19.6
Fimic Anthrosol <sup>2)</sup>	7.5	0.7	8.2
Umbric Gleysol	4.1	0.1	4.2
Haplic Arenosol <sup>3)</sup>	13.6	1.2	14.8
Associations	11.2	0.2	11.4
	$\Sigma=65.3$	$\Sigma=19.5$	$\Sigma=84.8$

<sup>1)</sup> The total forested area in the Netherlands is about 320 000 ha, which is approximately 9.5% of the total area of the Netherlands.

<sup>2)</sup> Including so called loopodzols, i.e. podzols with an anthropogenic humus layer. There is no suitable equivalent in the FAO classification.

<sup>3)</sup> Including Gleyic Arenosols (1.2%).

## 4 ASSESSMENT OF MODEL INPUTS

### 4.1 Acid deposition scenarios

Calculations were made with four deposition scenarios: two scenarios from 1990 to 2010 and two scenarios from 1990 to 2050. Total deposition estimates for SO<sub>2</sub>, NO<sub>x</sub> and NH<sub>3</sub> for each grid between 1990 and 2000 were derived by linear interpolation of calculations with the deposition model TREND for the years 1985, 1994 and 2000 (Schutter and De Leeuw, 1991). Values for each grid between 2000 and 2050 were derived by linear interpolation between the years 2000 (TREND model estimates), 2010 and 2050 (Table 2).

*Table 2 Deposition levels used for each grid in 2010 and 2050 for four scenarios*

Scenario	Deposition level per grid	
	2010	2050
1	TREND 2000 x reduction fraction	
2	Critical acid load	
3	TREND 2000 x reduction fraction	TREND 2000 x reduction fraction
4	TREND 2000 x reduction fraction	Critical acid load

In three of the four scenarios (scenario 1, 3 and 4), the deposition level on each grid in 2010 was taken equal to the value calculated with the TREND model for the year 2000, multiplied by a reduction fraction. The reduction fraction used was calculated by dividing an average acid deposition target for the Netherlands of 1230 mol<sub>c</sub> ha<sup>-1</sup> yr<sup>-1</sup> for the year 2010 (1400 mol<sub>c</sub> ha<sup>-1</sup> yr<sup>-1</sup> on Dutch forests) by an average acid deposition level in the Netherlands calculated with the TREND model for the year 2000 (2204 mol<sub>c</sub> ha<sup>-1</sup> yr<sup>-1</sup>). The reduction fraction used thus equalled 0.55.

The deposition levels used in the scenarios 1, 3 and 4 up to 2010 reflect the assumed effect of planned emission reductions. A fixed reduction of the deposition level in 2000 by the year 2010 still caused acid loads that exceeded critical acid loads in areas with intensive animal husbandry. Therefore, in scenario 2 expected acid loads on all grids in 2010 were taken equal to the critical acid loads for all forest-soil combinations. This reflects the effect of additional policy measures on a regional scale before 2010. Scenario 3 was included to investigate the long term effects of an average deposition level of 1400 mol<sub>c</sub> ha<sup>-1</sup> yr<sup>-1</sup> on Dutch forests. Scenario 4 was included to investigate the effect of additional policy measures after 2010, i.e. a further reduction to critical acid loads for each forest-soil combination within each grid.

The influence of tree species by filtering of dry deposition was included by multiplying the deposition on each grid by filtering factors for spruce forests (Douglas Fir and Norway Spruce), pine forests (Scotch Pine and Black Pine) and deciduous forests (including Japanese Larch). New values for these filtering factors were calculated when the total deposition on open land became less than the (wet) bulk

deposition, according to a procedure described in De Vries (1991). However, recalculation of the filtering factors was hardly ever necessary since forests in the Netherlands generally cover a small fraction of the land area in a grid.

Values for the filtering factors were derived from a comparison of throughfall data below spruce, pine and deciduous forests in the Netherlands and total deposition estimates with the TREND model using 1985 emission data (cf De Vries, 1991). Values thus derived are given in Table 3. These values are much higher than the filter factors for each deposition area (cf Erisman, 1991), that were used in the previous study with RESAM (De Vries et al., 1991).

**Table 3** *Forest filtering factors for SO<sub>4</sub>, NH<sub>4</sub> and NO<sub>3</sub> for spruce -, pine - and deciduous forests in the Netherlands.*

Forest type	Filtering factors (-)		
	SO <sub>4</sub>	NH <sub>4</sub>	NO <sub>3</sub>
Spruce forest	1.6	1.5	1.0
Pine forest	1.4	1.3	0.85
Deciduous forest	1.15	1.1	0.7

Bulk deposition data for base cations and Cl were derived from 22 weather-stations using the inverse distance interpolation technique to get values for each grid (cf De Vries, 1991).

#### 4.2 Critical acid loads

Data that are needed to map critical acid loads are the dry deposition, weathering and uptake of base cations, the nitrogen uptake and the precipitation surplus. Critical N immobilization rates related to the formation of stable C-N compounds were neglected (cf Schulze et al., 1989). Results thus obtained are given in De Vries et al. (1992). Here, we shortly summarize the approach.

Dry deposition of base cations were calculated by multiplying the bulk deposition with a dry deposition factor. This is based on the implicit assumption that dry deposition is linearly related to wet deposition. Dry deposition factors for base cations and Cl on each tree species were derived from available data on Na in throughfall and bulk deposition at 42 sites. Values thus derived were about 2.0 for spruce forests, 1.5 for pine forests and 1.0 for deciduous forests.

Weathering rates were based on information on base cation depletion rates in soil profiles, budget studies and on column and batch experiments, which have been conducted during five years on the most relevant non-calcareous sandy soils in the Netherlands (Hootsmans and Van Uffelen, 1991). Data are given in De Vries et al. (1992).

Uptake rates were calculated by multiplying forest growth rates with element contents in stems. Forest growth estimates for all relevant combinations of forest and soil type and contents of the elements N, K, Ca and Mg in stems were based on a literature survey for all tree species included (De Vries et al., 1990).

Precipitation surpluses were determined by the precipitation rate minus the sum of interception evaporation, soil evaporation and transpiration (evapotranspiration). Precipitation estimates were derived from 280 weather stations in The Netherlands, using interpolation techniques to obtain values for each grid. Interception fractions, relating interception to precipitation, were derived from literature data for all tree species considered. Data for evaporation and transpiration were calculated for all combinations of tree species and soil type with a separate hydrological model (De Visser and De Vries, 1989). More information on the various model inputs is given in De Vries et al. (1992).

#### 4.3 Aluminium hydroxide pools

Present amounts of Al hydroxides were based on an inventory of the chemical soil and soil solution composition of 150 non-calcareous sandy forest soils in 1990 (De Vries and Leeters, 1993). Data related to the forest topsoil are given in Table 4.

**Table 4** Ten percentile values (areal weighted) of the amount of Al hydroxides in the topsoil (0-30 cm) of the various sandy soils used in the calculations

Soil type	N <sup>1)</sup>	Al hydroxide amount (kmol <sub>e</sub> ha <sup>-1</sup> )									
		5%	15%	25%	35%	45%	55%	65%	75%	85%	95%
Cambic Podzol	17	351	456	489	545	563	580	610	644	686	850
Carbic Podzol	17	266	304	353	437	513	649	662	672	701	1015
Gleyic Podzol	57	200	350	383	440	490	528	564	613	671	848
Fimic Anthrosol	11	155	178	191	199	208	238	260	269	500	702
Umbric Gleysol	14	137	175	268	300	332	361	436	485	569	732
Haplic Arenosol	26	99	207	231	265	277	314	337	399	439	564

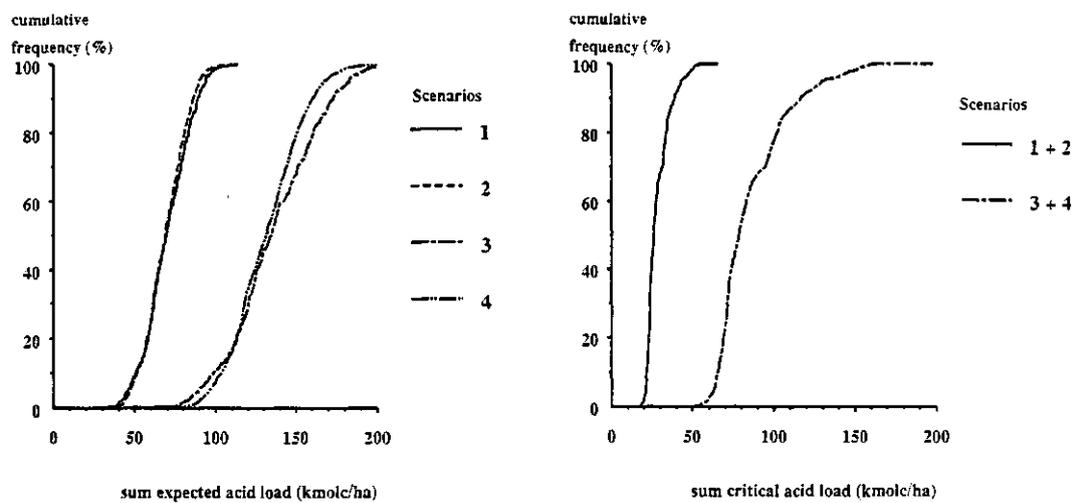
<sup>1)</sup> N is the number of sites

The variation in Al hydroxide amounts found for each soil type was included in the calculations by quantifying the range with ten percentile values (5, 15, ..., 95 percentile; cf Table 4). It was assumed that each percentile represented 10 percent of the area of the soil occurring in a grid.

## 5 RESULTS

### 5.1 Present and critical acid loads

The decrease in Al hydroxide contents is determined by the difference between expected and critical acid loads (Eq. 1). The sum of the expected acid loads up to 2010 hardly differed between scenario 1 and 2, and the sum of the acid loads up to 2050 was nearly similar for scenario 3 and 4 (Fig. 1A). Obviously there was no difference in the sum of the critical acid loads between scenario 1 and 2 and between scenario 3 and 4 (Fig 1B). These values were simply calculated by multiplying the annual average critical acid load by the simulation period, i.e. 20 years for scenario 1 and 2 and 60 years for scenario 3 and 4.



**Fig. 1** Cumulative frequency distributions of the summed expected acid load (A) and the summed critical acid load (B) of four scenarios

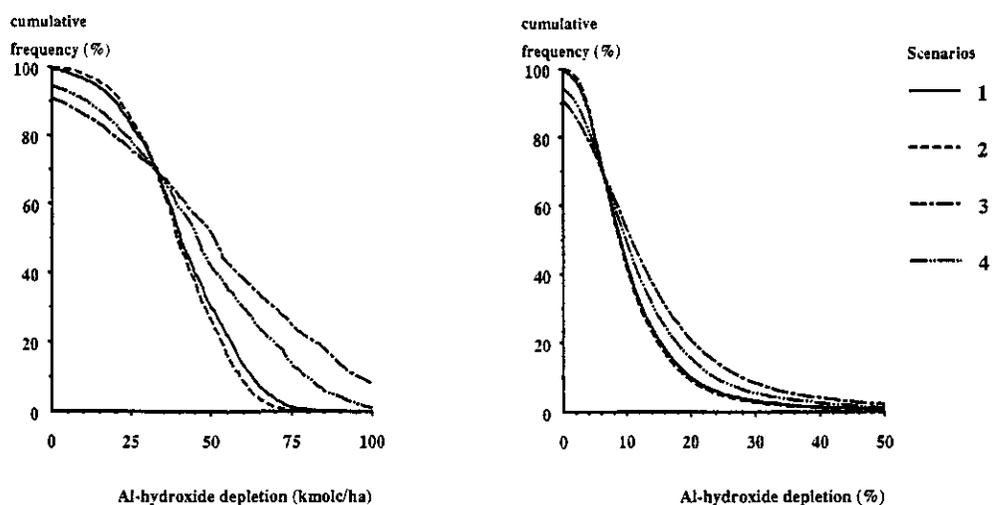
A comparison between Fig. 1A and 1B shows that the frequency distributions of the summed expected acid loads exceeds that of the summed critical acid loads for all scenarios thus causing a decrease in Al hydroxide contents.

### 5.2 Aluminium-hydroxide depletion

#### *Absolute and relative depletion*

Results of Al hydroxide depletion are presented in Fig. 2, by inverse cumulative frequency distributions, which directly give the percentage of the total forested area on non-calcareous soils (Y-axis) exceeding a certain absolute or relative decrease in Al hydroxides (X-axis). For all scenarios the absolute decrease in Al hydroxide

amount stayed nearly always below  $100 \text{ kmol}_c \text{ ha}^{-1}$  (Fig. 2A) whereas the relative decrease hardly ever exceeded 50% (Fig. 2B).



**Fig. 2** Inverse cumulative frequency distributions of the absolute (A) and relative Al hydroxide depletion (B) in response to four scenarios

The median value for relative Al depletion was about 10% for all scenarios. As with the summed present loads, the difference between the scenarios 3 and 4 was somewhat larger than between the scenarios 1 and 2, although small in all cases. The largest difference occurred between scenario 2 and 3. This difference illustrates the the long-term (up to 2050) impact of additional policy measures taken before 2010. Note that extension of scenario 2 up to 2050 does not influence the cumulative frequency distribution of Al hydroxide depletion since expected and critical acid loads are equal after 2010.

Figure 2 shows that Al depletion was not always lower for the scenarios 2 and 4 as compared to 1 and 3. For several forest soil combinations the critical acid load was sometimes higher than the acid load related to a national average value of  $1400 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$ . Consequently, sometimes a negative depletion (Al hydroxide accumulation) was even found, especially for scenario 3 and 4. This is illustrated in Figure 2 by a cumulative frequency at zero Al depletion of less than 100%. Accumulation of Al hydroxides (due to incongruent silicate weathering; cf De Vries and Kros 1989) mainly occurred below deciduous forests with a relatively low acid load, on relatively strongly buffered soils, such as Fimic Anthrosols and Umbric Gleysols, with a high critical acid load. At these forest-soil combinations, the summed critical acid load sometimes exceeded the summed expected acid load.

#### *Influence of tree species and soil type*

Both tree species and soil type influenced the occurrence of Al hydroxide depletion as described above. This is illustrated more specifically by presenting the 95 percentile of the relative Al depletion below major forest types (Table 5) and in major soil types (Table 6). The 95 percentile value of a cumulative frequency distribution

means that 95% has a value less or equal to that value (whereas 5% has a value larger than that value). The 95 percentile value is often used in mapping necessary critical load reduction to ensure protection of 95% of the area.

*Table 5 95 percentile values of the relative Al depletion below coniferous and deciduous forests as a function of the deposition scenario*

Forest type	Al hydroxide depletion (%)			
	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Coniferous	29	28	43	36
Deciduous	18	18	21	19
All	26	25	37	31

Al hydroxide depletion was larger below coniferous trees mainly because of a larger acid input induced by forest filtering (cf Table 2) and to a lesser extent because these trees occur more often on poorly buffered soils with a relatively low critical acid load.

The influence of soil type on relative Al depletion appeared to be larger than that of tree species. Largest changes occurred in the relatively poorly buffered Arenosols with low Al hydroxide contents (Table 6). Generally the differences in relative Al depletion between soil types coincided with differences in the initial Al hydroxide amounts (cf Table 4 and 6). Table 6 also illustrates that at a national average acid load of 1400 mol<sub>c</sub> ha<sup>-1</sup> yr<sup>-1</sup>, obtained by generic policy measures, Al hydroxide

*Table 6 95 percentile values of the relative Al depletion in different sandy forest soils as a function of the deposition scenario*

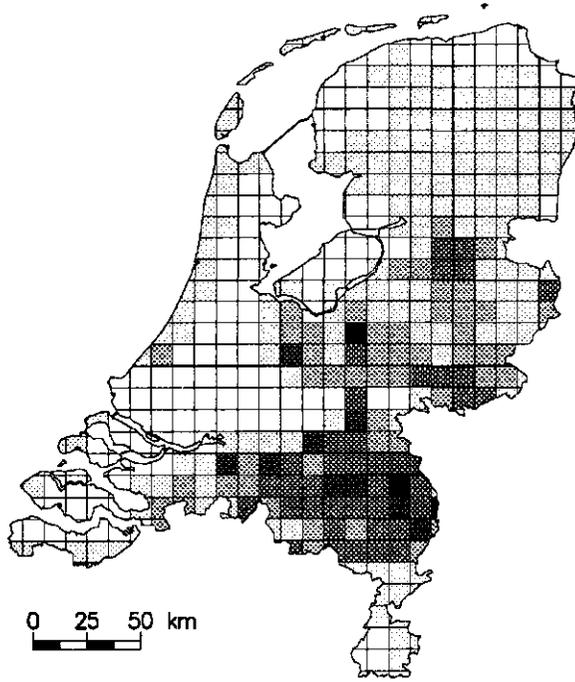
Soil type	Al hydroxide depletion (%)			
	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Cambic Podzol	13	12	20	16
Carbic Podzol	17	17	23	20
Gleyic Podzol	27	26	41	34
Fimic Anthrosol	25	25	28	27
Umbric Gleysol	25	25	26	25
Haplic Arenosol	46	44	64	55

depletion increases for relatively poorly buffered soils, such as Arenosols and Podzols, whereas it hardly increases for relatively strongly buffered soils such as Fimic Anthrosols and Umbric Gleysols (cf difference between scenario 1 and 3).

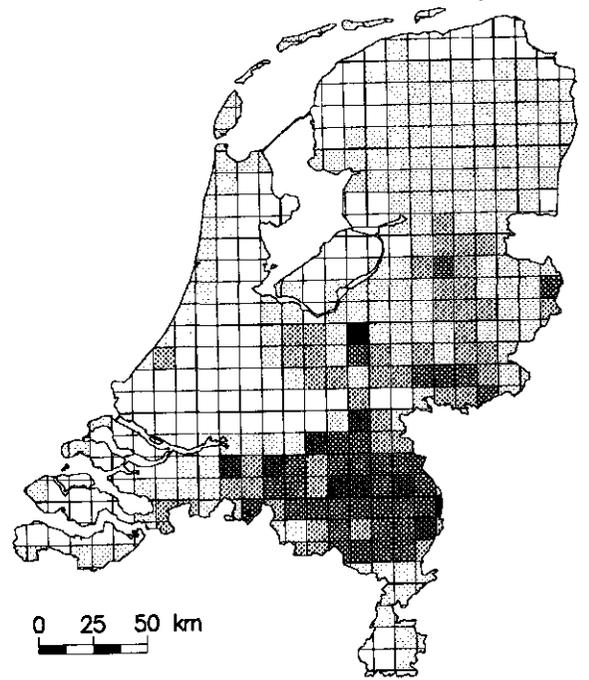
#### *Geographic distribution*

The geographic distribution of forest soils in which more than 50% of the initial Al hydroxide amount is depleted is illustrated in Figure 3.

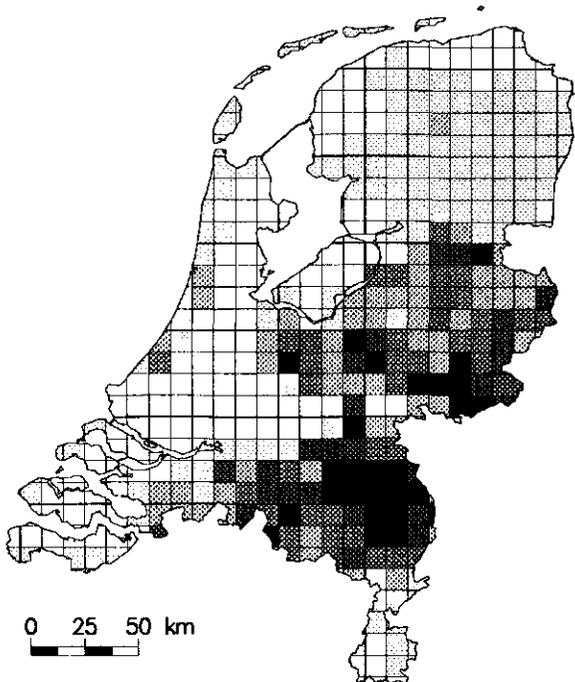
Scenario 1



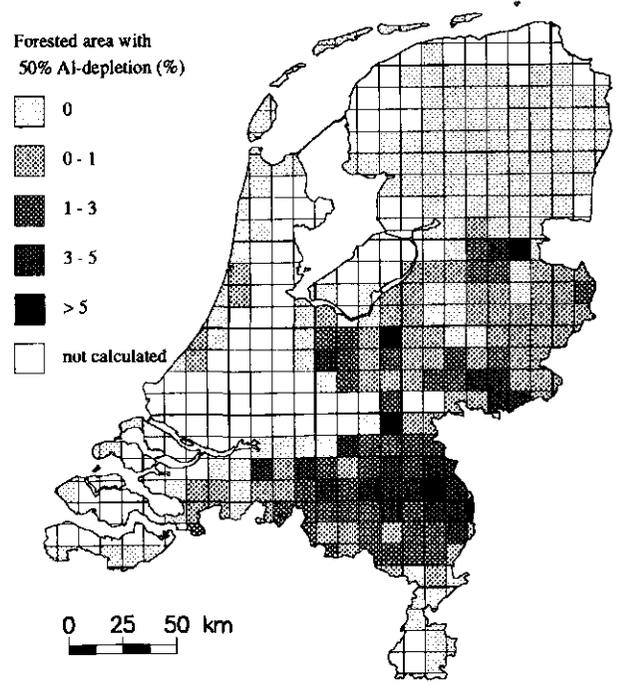
Scenario 2



Scenario 3



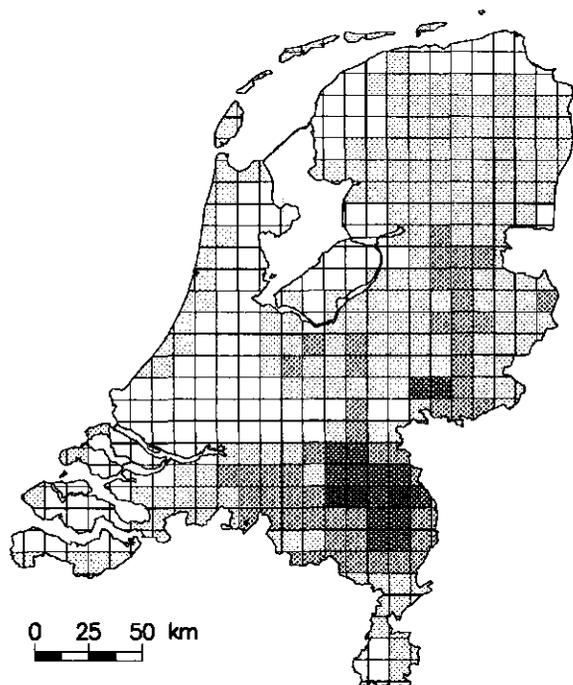
Scenario 4



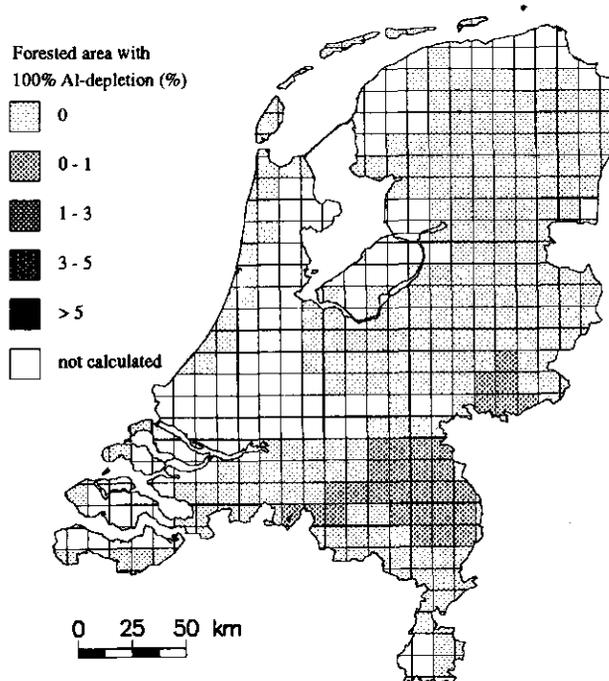
**Fig. 3** Maps of the forested area (in percent of the total forest area) exceeding a relative AI-depletion of 50% for scenario 1 (A), scenario 2 (B), scenario 3 (C) and scenario 4 (D)

Figure 3 shows that an exceedance of 50% Al depletion mainly occurred in the southern and eastern part of the country, i.e. in the areas with intensive animal husbandry. Except for scenario 3, the forested area exceeding 50% Al depletion was nearly always less than 5%. Complete (100%) Al depletion did not occur for the scenarios 1 and 2 but it was predicted for the scenarios 3 and 4 (Fig. 4).

Scenario 3



Scenario 4

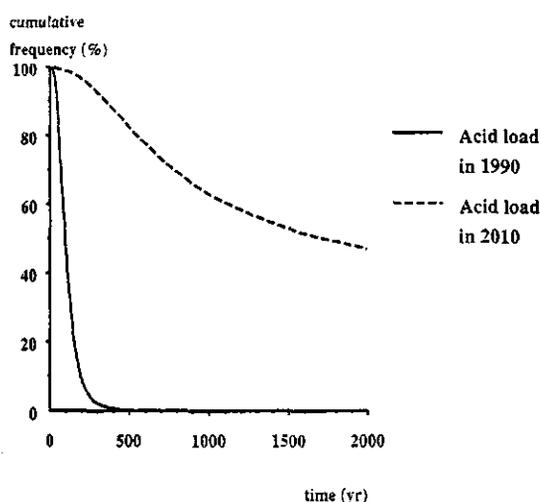


**Fig. 4** Maps of the forested area (in percent of the total forested area) with 100% Al depletion for scenario 3 (A) and scenario 4 (B)

Figure 4 shows that total depletion of Al hydroxide never occurred in more than 5% of the forested area. For scenario 4 it was even less than 1% in all gridcells (Fig. 4B).

### 5.3 Time periods to reach complete aluminium depletion

It was shown above that emission reductions up to a national average acid load of  $1400 \text{ mol}_e \text{ ha}^{-1} \text{ yr}^{-1}$  hardly caused complete Al depletion within 60 years (Scenario 3). Since the acid load was assumed constant after 2010 for this scenario, we also calculated the time period to reach complete Al depletion. This was done by dividing the present Al hydroxide amount by the difference between present and critical acid load at 2010. Results show that it took about 2000 years to deplete 50% of the forested area (Fig. 5). To show the effectiveness of the expected emission reductions up to 2010 ('generic policy') the time period to reach complete Al depletion was also calculated when using the acid load at 1990. Results show that in this situation, 50% of the forested area would be depleted in approximately 100 years whereas nearly all forest soils would be depleted within 400 years (Fig. 5).

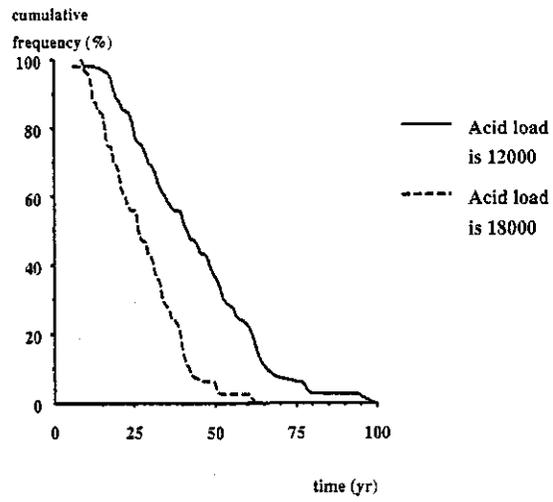


**Fig. 5** *Inverse cumulative frequency distributions of the time period to reach complete Al depletion at present acid loads and expected acid loads in 2010*

It should be stressed that the results described above do not include effects that may occur on a local scale. For example, a very fast depletion of Al hydroxides can occur in forests located nearby farms with intensive animal husbandry. A recent inventory on  $\text{NH}_3$  emissions (Marseille, Ministry of Housing, Public Health and Environment, pers. comm.) showed that approximately 200 of such farms cause an  $\text{NH}_3$  deposition above  $12000 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$  on nearby forests and sometimes (about 50 farms) even above  $18000 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$ . Assuming that these  $\text{NH}_3$  loads cause an equivalent acid production in the soil (this occurs when nitrification is complete and N uptake, denitrification and N immobilization are negligible) the calculated time period to reach complete Al depletion was nearly always less than 100 years (Fig. 6). Again, the time period differed as a function of soil type as illustrated in Table 7. Forest type hardly influenced the results. At extreme acid inputs, it was predicted that soils can totally be depleted within 10 to 20 years (Table 7).

**Table 7** *Minimum, average and maximum time periods to reach complete depletion of Al hydroxides in areas with extreme  $\text{NH}_3$  loads as a function of forest type and soil type*

Soil type	Time period (yr)					
	Acid load is $12000 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$			Acid load is $18000 \text{ mol}_c \text{ ha}^{-1} \text{ yr}^{-1}$		
	minimum	average	maximum	minimum	average	maximum
Carbic Podzol	25	54	106	16	34	65
Cambic Podzol	32	55	88	21	35	54
Gleyic Podzol	18	47	84	12	30	53
Fimic Anthrosol	15	30	71	10	19	47
Umbric Gleysol	13	38	80	8	24	48
Haplic Arenosol	9	29	56	6	19	34



**Fig. 6** *Inverse cumulative frequency distributions of the time period to reach complete Al depletion at acid loads nearby farms with intensive animal husbandry*

## 6 CONCLUSIONS AND DISCUSSION

From this study it can be concluded that:

- 1 Expected emission reductions (generic policy measures) strongly reduce the occurrence of Al hydroxide depletion. Compared to present acid loads the predicted time period to reach complete Al depletion in 50% of all forest soils increases from approximately 100 to 2000 years. In the near future (the year 2050), a relative Al depletion above 50% is mainly to be expected in the southern and eastern part of the Netherlands, i.e. in areas with large acid (ammonia) inputs, especially in Haplic Arenosols that have low present amounts of Al hydroxides.
- 2 Compared to expected emission reductions (generic policy measures) additional policy measures only have a small effect on the predicted change in Al hydroxide content. Largest effects occur when critical acid loads for each forest/soil combination are already reached in 2010.
- 3 Al depletion can be very fast (less than 10 to 20 years) in forest soils located near farms with intensive animal husbandry. Here additional policy measures could have an important effect.

It should be stressed that the results presented in this study are only rough estimates of the actual occurrence of Al hydroxide depletion. First of all, the influence of nutrient cycling processes on acid production and consumption was not included. Critical acid loads were derived with a simple one-layer model, assuming a closed internal nutrient cycle. Critical acid loads derived with the multi-layer MACAL model showed that values increased with decreasing soil depth (De Vries et al., 1993). Consequently, the predicted Al depletion in the forest topsoil is likely to be overestimated. For the total soil profile, it might be a reasonable estimate although the relative depletion is here overestimated as well since the present Al hydroxide amount is larger in the rootzone as compared to the first 30 cm.

Second, the impact of dynamic processes, such as cation exchange, adsorption/desorption of  $\text{SO}_4$  and  $\text{NH}_4$  and mineralization/immobilization dynamics are not included. Dynamic simulations with the RESAM model showed that in a situation of deposition reductions, these processes lead to net H producing effects due to exchange of bases against H,  $\text{SO}_4$  desorption and net N mineralization (De Vries et al., 1991). This may give rise to a (small) underestimation of the depletion of Al hydroxides. However, the H leaching rate, calculated with RESAM, is likely to be larger than the rate calculated for the situation at critical load (cf Eqs. 5 and 6), since present loads are higher. Those effects are counteracting, so that the overall effect is likely to be small. In summary, the presented estimates are likely to be reasonable.

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