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# Modeling of the long-term effect of tree species (Norway spruce and European beech) on soil acidification in the Ore Mountains

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ABSTRACT

The MAGIC model was applied to estimate soil and soil water (−90 cm) chemistry between 1854 and 2094 at two experimental stands (Načetín, Ore Mountains), one covered by a Norway spruce monoculture and the other by a natural European beech forest. The primary aims were to evaluate the influence of tree species on long-term acidification and to predict future development under different forest management scenarios.

Depletion of base cations from the soils, caused by high acidic deposition, resulted in low base saturations of 8.2% at the spruce stand and of 6.4% at the beech stand in 2003. The concentration of aluminum in soil water was 135 $\mu$mol $l^{-1}$ and pH 4.32 at the spruce stand, and 70 $\mu$mol $l^{-1}$ and pH 4.4 at the beech stand, respectively, in 2005. During the peak of acidification in the mid 1980s, modeled aluminum concentrations contributed 70% to neutralizing acidity at the spruce stand, and 55% at the beech stand. In addition, $SO_4$ concentration was significantly higher at the spruce stand (525 $\mu$equiv. $l^{-1}$) compared to the beech stand (330 $\mu$equiv. $l^{-1}$) as a result of higher dry deposition onto the spruce canopy. The enhanced leaching of base cations was comparable at both stands (191 $\mu$equiv. $l^{-1}$ at the spruce stand and 215 $\mu$equiv. $l^{-1}$ at the beech stand). The higher deposition of base cations onto the spruce canopy was able to partially mitigate the effect of high leaching.

The model results suggest that future recovery of soil water will be significantly better at the beech stand (higher pH, ANC, Bc/Al ratio and lower $SO_4^{2-}$ and Al concentrations). Interestingly, modeled soil base saturation for 2094 will be lower at the beech stand. Alternative scenarios such as clear-cutting and new re-forestation resulted in more favorable soil chemistry for the beech plantations compared to the spruce. The best regeneration of the soil environment is predicted for the scenario with an absence of forest. This suggests, that future soil recovery from acidification will be delayed by the removal of base cations through harvesting.



## 1. Introduction

The upper parts of the Ore Mountains, Czech Republic, were almost completely converted from European beech dominated forests to managed Norway spruce (*Picea abies* /L./ Karst.) monocultures during the 19th century, as a result of wood shortage at this time. After World War II, coal mining in a nearby basin was rapidly stepped up. The S-rich coal, in

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which the sulfur content ranges from 1% to 15% (Moldan and Schnoor, 1992), has been combusted in several local power plants, which lacked abatement technology until 1994. Such intensive coal combustion resulted in extremely high $SO_2$ concentrations in ambient air (average $SO_2$ for the 1980s >100 $\mu g\,m^{-3}$) (Peters et al., 1999). Air pollution together with acid sensitive bedrocks resulted in massive acidification of soil and stream waters and in direct damage of tree assimilatory tissue. By 1990, 25,000 ha of spruce forests had become dead or severely damaged (Kubelka et al., 1993).

The acidification of such damaged ecosystems has been intensively monitored at the Načetín mature spruce stand since the late 1980s (Černý and Pačes, 1995). In the Czech Republic, acidic deposition and occurrence of spruce plantations are two major factors causing the acidification and nutrient degradation of forest soils (Hruška and Cienciala, 2003). To evaluate the hypothesis that deciduous forest was able to significantly mitigate the long-term acidification of the soil environment, similar monitoring, as at the spruce forest, was started at a nearby mature beech forest in 2003 (Oulehle and Hruška, 2005). The results from both stands were used for historical estimation and future prediction of soil chemistry under different forest management scenarios using the biogeochemical model MAGIC (Model of Acidification of Groundwaters In Catchments) (Cosby et al., 1985, 2001). In this paper we focus on:

- Reconstruction of historical soil acidification over the past 140 years at both stands using the MAGIC model.
- Predicting future soil status at the spruce and beech stands under different scenarios of forest management for the next century.

## 2. Material and methods

### 2.1. Site description

We studied spruce and beech stands located near the border between the Czech Republic and Germany in the Ore Mountains, close to the villages of Kienhaide (Germany) and Načetín (Czech Republic) (Fig. 1). The distance between the stands is only 700 m. Thus, the stands have experienced similar past climatic and pollution conditions. Average annual temperature is 6.3 °C (1991–2004), average annual precipitation is 842 mm (1991–2004), and the slope is to the north-west. Paragneiss underlies both stands. The dominant soils are dystric cambisols (Černý and Pačes, 1995). The spruce stand (50°35′26″N, 13°15′14″E) lies at an elevation of 784 m a.s.l. and is covered by 68-year-old Norway spruce (*P. abies*). The spruce forest at this site is one of the very few remaining mature spruce stands in the Ore Mountains that has never been limed. The beech stand (50°35′22″N, 13°16′07″E) lies at an elevation of 823 m a.s.l. The site is a part of a nature reserve and was partially affected by liming of large clear cuts in the neighbourhood. This reserve is predominantly covered by beech (*Fagus sylvatica*) that is older than 120 years. The spruce site at Načetín has been studied since the late 1980s (Dambrine et al., 1993), very intensively in 1994–1999 (Schulze, 2000) and together with the beech stand since 2003.

### 2.2. Sampling program

Two bulk precipitation collectors situated in clearings in the middle of each stand were sampled monthly from May 1991 and fortnightly in 1995 and 1996 (applies to throughfall and soil water sampling as well). The sampling was stopped in March 1999 and started again in May 2003. In April 1994, a sampling network of 25 samplers spaced in a regular 10 m × 10 m grid was installed for throughfall measurements in the spruce stand. Polyethylene (PE) collectors (area of 122 $cm^2$) were replaced in winter by open plastic buckets (area of 380 $cm^2$) with PE bags. This sampling was finished in March 1999. Since May 2003, a new set-up for throughfall measurements consisting of a 15 m × 15 m grid with nine samplers has been installed at the same place as the former (winter plastic buckets were changed by the PE collectors with area of 167 $cm^2$). Since May 2003, the same set up for throughfall measurements as at the spruce stand has been installed in the beech stand. Precipitation annual mean concentrations are volume-weighted.

Soil water at the 90 cm depth was collected from July 1994 using seven PRENART® suction lysimeters at the spruce stand. Each lysimeter was sampled monthly, from 1994 to 1996 fortnightly, and bulked at the end of the month. This sampling was finished in March 1999 and started again since May 2003 with the original set-up as in the 1990s with monthly sampling frequency. In May 2003, six suction lysimeters were installed at the 90 cm depth at the beech stand. Annual mean concentration was calculated as the mean of monthly samples. Reported annual mean concentrations are based on water years (November–October).

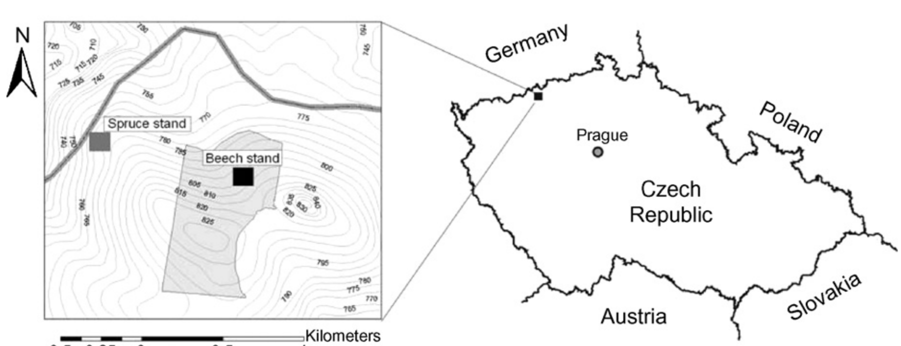


**Fig. 1 – Site locations in the Ore Mountains, Czech Republic (grey, nature reserve).**

Soils were sampled at four locations in 1994 and six locations in 2003 at the spruce stand and at four locations in 2003 at the beech stand. Soil physical properties were estimated by excavating 0.5 $m^2$ pits. Spruce tissue samples were obtained from 11 representative spruce trees in 1994. The procedures for analysis of soil, water and spruce tissue chemistry are described in detail in Oulehle et al. (2006).

### 2.3. *Water budged modeling*

Because the studied stands are not a part of monitored watershed with precise water balance calculations, we used a hydrological model for estimation of water fluxes. We choose the one-dimensional soil water model SIMPEL (www.hydrology.uni-kiel.de/simpel) for calculating seepage flux at the 90 cm depth. This model is based on interception storages (leaves, litter), soil and groundwater storages. Input data are precipitation, potential evapotranspiration, leaf area index and soil physical parameters. Output is given as flow between the reservoirs, actual evapotranspiration as the sum of interception, evaporation and transpiration, infiltration to the groundwater and surface runoff. The input dataset for calculating evaporation consists of daily climatic data (mean air temperature, duration of sunshine, mean air humidity, mean wind velocity, temperature at 2 p.m. and relative humidity at 2 p.m.), geographical position, plant parameters and LAI time series (data for spruce measured between 1994 and 1997, for the beech stand derived from the Solling beech data set). The climatic data for the period 1992–2004 were taken from a climatologic station at Nová Ves (765 m a.s.l.), 15 km westward from the Načetín stands.

Mean annual precipitation measured between 1992 and 1998 was 911 mm, which is in good agreement with precipitation data from Nová Ves for this period (882 mm), differing by only 3%. The mean annual seepage flux at 90 cm was calculated as 373 ± 110 mm at the spruce stand and 364 ± 113 mm at the beech stand for the period 1992–2004. The ratio between precipitation and runoff was 0.44 at the spruce stand and 0.42 at the beech stand. These results were in good relationship with data measured at the Lysina watershed (spruce forest) in the Slavkov Forest (about 80 km SW from Načetín, elevation of 884 m a.s.l.), where mean annual precipitation is 953 mm, and discharge 432 mm with a precipitation/discharge ratio of 0.45 (Hruška and Krám, 2003).

### 2.4. *Emission history*

Burning of locally mined lignite has been the major source of $SO_2$ emissions, and consequently the major source of S deposition in the Czech Republic over the last 150 years. There is a close relationship between coal mining and emissions of $SO_2$ in the Czech Republic (Hruška et al., 2002). Coal burning decreased from 73 Mt in the peak year 1984 to 51 Mt in 1993 and then decreased only slightly to 42 Mt in 2002 (Fig. 2). Emissions of $SO_2$ peaked in 1982 and have declined since then. However, there is a distinct break point in 1993 when the first power plants in the Czech Republic were equipped with flue-gas desulfurization, a process that was completed in the Czech Republic in 1999.

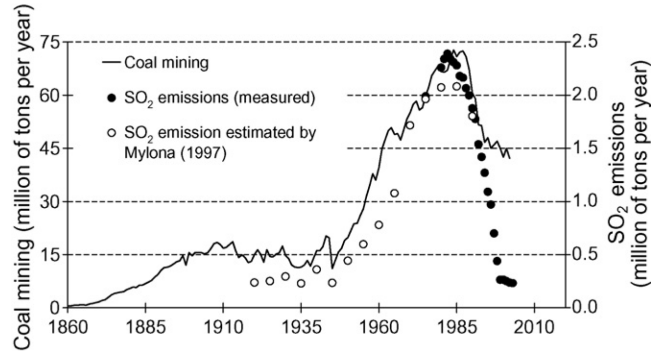


**Fig. 2 – Coal (lignite) mining (1860–2002) and annual $SO_2$ emissions (1986–2003) in the Czech Republic (Mylona, 1997).**

### 2.5. *Model description*

MAGIC (Model of Acidification of Groundwater in Catchments) was designed to estimate stream, soil water and soil chemistry (Cosby et al., 1985, 2001), using a lumped representation of physical and chemical properties (Table 1). The model simulates soil solution and surface water chemistry to predict concentrations of the major ions. MAGIC calculates for each time step (in this case yearly) the concentrations of major ions on the assumption of simultaneous reactions involving sulfate adsorption, cation exchange, dissolution–precipitation–speciation of aluminum and dissolution–speciation of inorganic and organic carbon. MAGIC accounts for the mass balance of major ions in the soil by book keeping the fluxes from atmospheric inputs, chemical weathering, net uptake in biomass and loss to runoff.

Data inputs required for calibration of MAGIC comprise soil chemical and physical characteristics, input and output fluxes of water and major ions, and net uptake of base cations by vegetation (Wright and Cosby, 2003).

### 2.6. *Input data description*

The parameters used in the modeling at the Načetín spruce and beech stands are summarized in Table 1. Concentration of exchangeable cations and cation exchangeable capacity were weighted according to the measured fine soil pools (<2 mm for the mineral soil, <5 mm for the organic soil) of individual soil horizons. The concentration of organic acids in soil was assumed from the mean concentration of soil water TOC (total organic carbon) in 2004 (Hruška et al., 2002). The $Al(OH)_3$ solubility constant was obtained from a long-term data record of Al concentration in soil water (Oulehle and Hruška, 2005). Sulfate adsorption parameters and weathering rates were fitted in the MAGIC calibration procedure. Net uptake of base cations fixed to above ground tree biomass was calculated from the tissue chemistry of stem wood and bark multiplied by increment per year. Data based on 11 trees from the spruce stand were extrapolated to the beech stand, because available tissue chemistry data are lacking (it is not allowed to cut trees in the natural reserve). A time-constant base cation uptake by vegetation was used. Absolute values of the vegetation uptake

**Table 1 – Main parameter values for the MAGIC model calibration at the Načetín stand**

| | Units | Value for soil | |
|---|---|---|---|
| | | Spruce forest | Beech forest |
| Fixed parameters | | | |
| Discharge, annual | m | 0.37 | 0.36 |
| Precipitation, annual | m | 0.89 | 0.89 |
| Soil depth | m | 0.9 | 0.9 |
| Bulk density | kg $m^{-3}$ | 706 | 510 |
| CEC | mequiv. $kg^{-1}$ | 37.9 | 49.6 |
| $SO_4$ adsorption half saturation | mequiv. $m^{-3}$ | 500 | 500 |
| $SO_4$ maximum adsorption capacity | mequiv. $kg^{-1}$ | 15 | 15 |
| $p_{CO_2}$ | atm | 0.022 | 0.022 |
| Temperature | °C | 5 | 5 |
| $pK_1$ of organic acids | −log 10 | 2.5 | 2.5 |
| $pK_2$ of organic acids | −log 10 | 4.1 | 4.1 |
| $pK_3$ of organic acids | −log 10 | 6.7 | 6.7 |
| Organic acids | mmol $m^{-3}$ | 30 | 30 |
| Ca saturation 1994 | % of CEC | 4.4 | 4.2[a] |
| Mg saturation 1994 | % of CEC | 1.6 | 1.7[a] |
| Na saturation 1994 | % of CEC | 0.4 | 0.1[a] |
| K saturation 1994 | % of CEC | 1.1 | 0.4[a] |
| Total base saturation 1994 | % of CEC | 7.5 | 6.4[a] |
| Vegetation uptake Ca 1994 | mequiv. $m^{-2}$ | 7.5 | 7.5 |
| Vegetation uptake Mg 1994 | mequiv. $m^{-2}$ | 2.5 | 2.5 |
| Vegetation uptake Na 1994 | mequiv. $m^{-2}$ | 0.2 | 0.2 |
| Vegetation uptake K 1994 | mequiv. $m^{-2}$ | 2.3 | 2.3 |
| Optimised parameters | | | |
| $Al(OH)_3$ solubility constant | log 10 | 9.05 | 9.05 |
| Weathering Ca | mequiv. $m^{-2}$ | 12.5 | 12.5 |
| Weathering Mg | mequiv. $m^{-2}$ | 18.5 | 18.5 |
| Weathering Na | mequiv. $m^{-2}$ | 3.5 | 3.5 |
| Weathering K | mequiv. $m^{-2}$ | 3 | 3 |
| Weathering of Σ(Ca + Mg + K + Na) | mequiv. $m^{-2}$ | 37.5 | 37.5 |
| Weathering F | mequiv. $m^{-2}$ | 3 | 3 |
| Selectivity coefficient Al–Ca | log | −1.4 | −0.44 |
| Selectivity coefficient Al–Mg | log | −0.34 | 0.68 |
| Selectivity coefficient Al–Na | log | −2.25 | −1.85 |
| Selectivity coefficient Al–K | log | −5.23 | −4.81 |
| Ca saturation 1854 | % | 6.4 | 6.4 |
| Mg saturation 1854 | % | 5.7 | 5.7 |
| Na saturation 1854 | % | 1 | 1 |
| K saturation 1854 | % | 1.2 | 1.2 |
| Total base saturation 1854 | % | 14.3 | 14.3 |

[a] Data for 2003.

and immobilization of nitrogen species were calculated by MAGIC using the fitted (for 1994) or estimated (before 1970) uptake of the available N as a fraction of the atmospheric N deposition. We supposed, according to the development of annual stem increments, that leaching of nitrate began in 1970s, when trees started to reduce growth. After 1999, almost complete retention of nitrogen in spruce ecosystem was observed (Oulehle et al., 2006). The future N uptake was set to the same value as for the calibration years 2004 and 2005.

In a deterministic model with multiple input parameters, many different combinations of input parameters may give similar output parameters, and hence similar fits to measurements of the output parameters. Detailed MAGIC model testing against observations and uncertainty could be found in Jenkins et al. (2003) or estimating uncertainty using Bayesian techniques in Larssen et al. (2006).

### 2.7. *Management scenarios*

To determine the influence of forest management practices on the recovery of soil environment from acidification, four different management scenarios were used. Each scenario was implemented at the spruce and beech stands separately. Projections were made for the period 2007–2094 and were based on following assumptions:

Scenario (1) Continuing of the present conditions (hypothesis of no change). Growth of spruce or beech forest without cutting, with future deposition corresponding to the average measured in 2004 and 2005.

Scenario (2) Clear-cut of mature forest (beech or spruce) and reforestation with spruce. Between 2007 and 2023 the forest conditions were set the same as for a clearing absent of trees. Between 2023 and 2037 the forest gradually reached the conditions used in Scenario (1) for a mature spruce stand. For the whole period stable nutrient uptake was assumed.

Scenario (3) Clear-cut of mature forest (beech or spruce) and reforestation with beech. Future development was modeled as in Scenario (2), but with beech forest conditions.

Scenario (4) Clear-cut of forest in 2007 without reforestation. Deposition was equal to bulk deposition. The water flux at 90 cm was increased to 500 mm, which corresponds to the value without vegetation transpiration. The uptake of nutrients by vegetation was not taken into account.

Even if all scenarios in the predictions are hypothetical, they should provide an insight into processes, which may occur under forest management applied in the Czech Republic.

## 3. Results

### 3.1. *Atmospheric deposition*

The daily average concentration of airborne $SO_2$ from station Zinwald (Germany, 35 km NE from Načetín) was $76 \pm 12\,\mu g\,m^{-3}$ for the period 1979–1989. From 1990 to 1998, when power plants were desulfurized, $SO_2$ declined sharply to annual average of $37 \pm 14\,\mu g\,m^{-3}$. From 1999, the concentration of $SO_2$ was stable, with an average concentration of $9 \pm 1\,\mu g\,m^{-3}$.

Modeled atmospheric deposition of $SO_4$ and all major ions increased over the historical period from 1854 up to the peak values around 1980 at both stands, and then declined (Fig. 3). The steepest part of the decline was partly covered by the period of deposition measurements at the spruce stand from 1992 to 1998 and since 2003 for bulk precipitation, and from 1994 to 1998 and since 2003 for throughfall precipitation. Total S deposition (wet plus dry) measured in 1994 was 220 mequiv. $m^{-2}$. Dry deposition contributed 60% of total deposition. After 11 years, the S deposition was 74 mequiv. $m^{-2}$ (2005), which was a decline of 65%. The dry deposition fraction still contributed 50% in 2005 at the spruce stand. At the beech stand, S total deposition was 42 mequiv. $m^{-2}$ (2004 and 2005) and the dry deposition fraction was only 15%.

Bulk deposition of $NO_3$ showed no significant trend, with an average of $38 \pm 6.6$ mequiv. $m^{-2}$ $year^{-1}$ during the observation period. Similarly, bulk deposition of $NH_4$ was $43 \pm 11$ mequiv. $m^{-2}$ $year^{-1}$ on average.

Atmospheric deposition of the sum of base cations (Ca + Mg + K + Na) was stable during the observed period with an average of $40 \pm 10$ mequiv. $m^{-2}$. Nevertheless, a significant decrease of Ca was observed (Oulehle et al., 2006). Dry deposition was estimated using the approach published by Bredemeier (1988) assuming negligible internal flux of Na in a spruce stand. The ratio of Na in throughfall to Na in bulk flux was used to estimate dry deposition for Ca, Mg and K. Dry deposition was estimated to be $47 \pm 7$% of bulk deposition of these elements at the spruce stand, with no significant trend during the observed period. At the beech stand, the dry deposition contributed only 14% to bulk deposition in 2004 and 2005.

Deposition of $H^+$ in bulk precipitation decreased from an average 69 mequiv. $m^{-2}$ $year^{-1}$ (1994–1995) to 33 mequiv. $m^{-2}$ $year^{-1}$ (2004–2005). Precipitation pH increased from 4.2 to 4.5. Spruce throughfall pH increased even more significantly from 3.61–3.66 (1994–1995) to 4.35–4.24 (2004–2005). Consequently, the throughfall $H^+$ flux declined from 244–220 mequiv. $m^{-2}$ (1994–1995) to 45–57 mequiv. $m^{-2}$ (2004–2005), a decline of ca. 80%. At the beech stand, throughfall pH was 4.90 and 4.62 with $H^+$ flux of 13 mequiv. $m^{-2}$ $year^{-1}$ and 24 mequiv. $m^{-2}$ $year^{-1}$ in 2004 and 2005, respectively.

### 3.2. *Soil water chemistry*

The seepage flux at 90 cm varied between 227 mm and 567 mm (average 373 mm), with no temporal trend observed between 1992 and 2004 at the spruce stand.

Sulfate concentration decreased from 815 μequiv. $l^{-1}$ in 1995 to 525 μequiv. $l^{-1}$ in 2005 at the spruce stand (decrease of 35%). The sulfate concentration was 330 μequiv. $l^{-1}$ at the beech stand in 2005 (Fig. 4). Nitrate concentration at the spruce stand was markedly enhanced between 1995 and 1998 (average of $195 \pm 40$ μequiv. $l^{-1}$). In 2004 and 2005, the nitrate concentration decreased to 22 μequiv. $l^{-1}$ and 5 μequiv. $l^{-1}$, respectively. The concentration of nitrate was 14 μequiv. $l^{-1}$ in 2004 and 7 μequiv. $l^{-1}$ in 2005 at the beech stand.

The sum of base cations declined from 325 μequiv. $l^{-1}$ in 1995 to 191 μequiv. $l^{-1}$ in 2005, representing a decline of 40% (Fig. 4). The steepest decrease was observed for $Ca^{2+}$, which decreased from 136 μequiv. $l^{-1}$ to 45 μequiv. $l^{-1}$ (decrease of

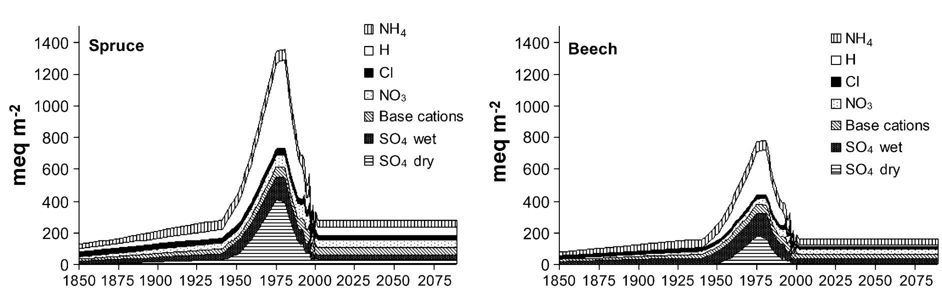


**Fig. 3 – Atmospheric deposition at Načetín used for MAGIC modeling. For the period 1854–1994 estimated chemistry, for 1994–1998 and 2004–2005 measured chemistry and for 2006–2094 the average chemistry measured in 2004–2005.**

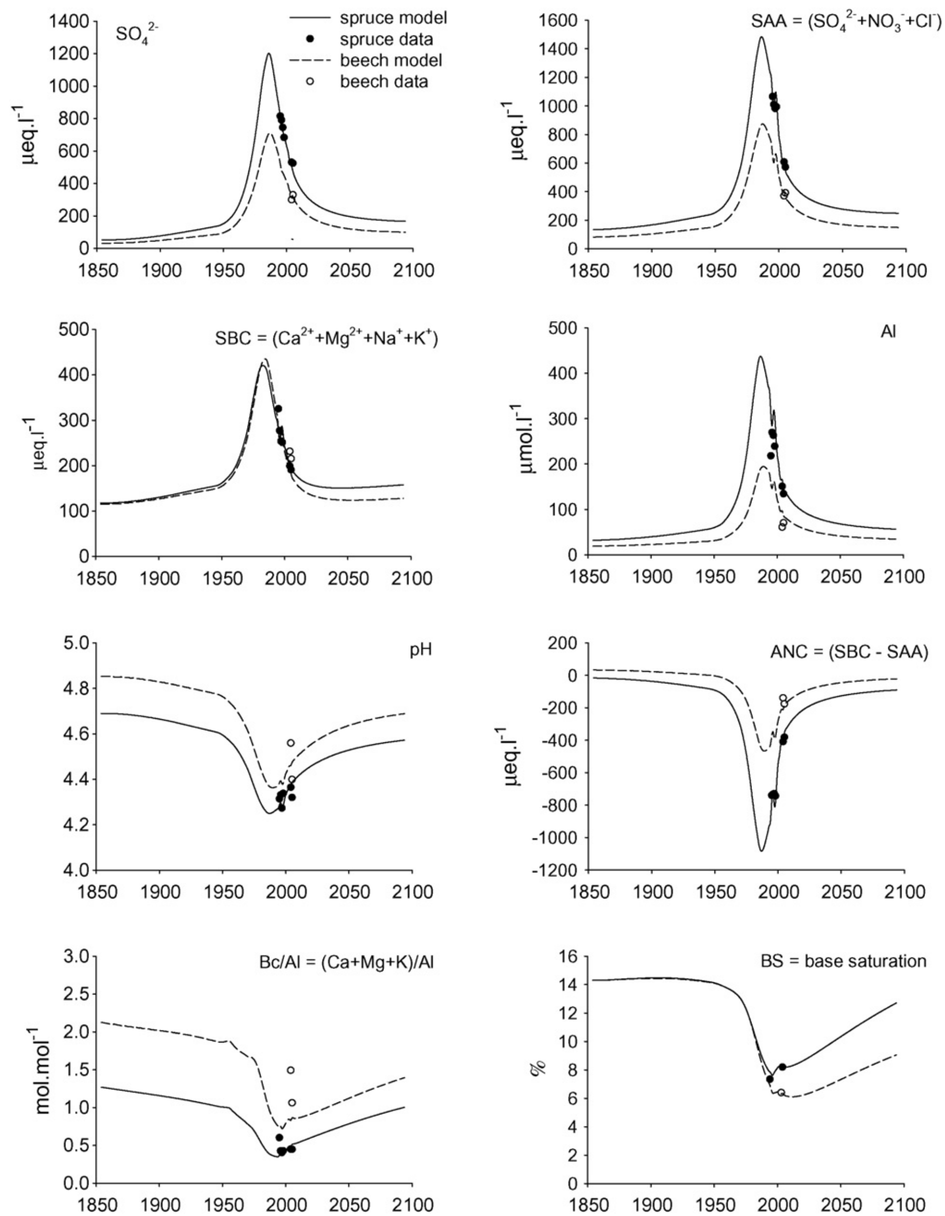


**Fig. 4 – Measured (dots, full for spruce and open for beech) and simulated (1854–2094, full line for the spruce stand and dashed line for the beech stand) changes in soil water chemistry at Načetín. Deposition measured in 2004–2005 was used for the forecast.**

about 70%). $Mg^{2+}$ decreased from 89 μequiv. $l^{-1}$ to 56 μequiv. $l^{-1}$ (40%) and $K^+$ from 19 μequiv. $l^{-1}$ to 10 μequiv. $l^{-1}$ (50%). $Na^+$ showed no significant trend. The concentrations of Ca, Mg, K and Na were 27 μequiv. $l^{-1}$, 116 μequiv. $l^{-1}$, 3 μequiv. $l^{-1}$ and 69 μequiv. $l^{-1}$ in 2005 at the beech stand. Soil water pH at the spruce stand increased only slightly, but significantly ($p < 0.05$; Mann–Kendall test). The pH values were 4.56 and 4.4 at the beech stand in 2004 and 2005, respectively (Fig. 4). Total Al declined by 40% (from 218 μmol $l^{-1}$ in 1995 to 134 μmol $l^{-1}$ in 2005) at the spruce stand. At the beech stand the concentration of aluminum was significantly lower (70 μmol $l^{-1}$ in 2005).

Acid neutralizing capacity (ANC), calculated as the difference between the sum of base cations (Ca + Mg + K + Na) and the sum of strong inorganic anions ($SO_4 + NO_3 + Cl + F$), increased from −762 μequiv. $l^{-1}$ in 1995 to −397 μequiv. $l^{-1}$ in 2005 at the spruce stand. At the beech stand, the ANC was −186 μequiv. $l^{-1}$ in 2005.

### 3.3. *Soil properties*

The soil profiles at both the Načetín plots had chemistry typical for acidified soils in Central Europe (Table 1). The soil mass-weighted base saturation for the depth of 90 cm increased only slightly from 7.5 to 8.2 between 1994 and 2004 at the spruce stand. The base saturation was 6.4 at the beech stand in 2003.

## 4. Discussion

### 4.1. *Modeling of sulfur*

#### 4.1.1. *Historical development*

The historical changes in atmospheric deposition of sulfur were the main factor causing changes in the biogeochemistry of the forest soil environment at the Načetín stands. The amount of historical sulfur deposition was estimated to be proportional to the regional coal mining (Fig. 2) for the period 1854–1994. Estimated S deposition of 6.5 kg ha$^{-1}$ year$^{-1}$ and 3.5 kg ha$^{-1}$ year$^{-1}$ in the mid of 19th century at spruce and beech stands, respectively, resulted in soil water concentration of sulfate simulated to be 50 μequiv. l$^{-1}$ at the spruce stand and 30 μequiv. l$^{-1}$ at the beech stand.

The modeled concentration of sulfate in soil water peaked in the mid 1980s with 1200 μequiv. l$^{-1}$ at the spruce stand and 700 μequiv. l$^{-1}$ at the beech stand. The higher concentration at the spruce stand was caused by a higher total sulfur deposition under the spruce canopy (Fig. 3). For estimation of dry deposition at the beech stand we multiplied the estimated historical spruce dry deposition scale factor (DDF) by the dry deposition ratio of spruce and beech forest in 2004 and 2005. This approach was applied for estimation of deposition of all elements at the beech stand in the period 1854–1994. The modeled high sulfur retention capacity of soil caused a relatively slow decrease of sulfate in soil water in the period after the rapid decline of sulfur deposition in 1990s. A new steady state with modeled S deposition will have approached the soil water sulfate concentration by the year 2050 (Fig. 4). It has been also confirmed by $\delta^{34}$S isotopes. Novák et al. (2000) estimated new steady state between reduced S deposition and runoff within 60 years.

#### 4.1.2. *Future progress*

If the spruce and beech forest remains at both stands until 2094, with identical deposition as measured in 2004 and 2005 (Scenario (1)), the predicted soil water $SO_4$ concentration will decrease to ca. 170 μequiv. l$^{-1}$ at the spruce stand and to 100 μequiv. l$^{-1}$ at the beech stand in 2094. These values will be similar to concentrations estimated at the beginning of the 1950s, but approximately 15% of the highest $SO_4$ simulated for 1986. Despite this pronounced decline, $SO_4$ will be still the dominant soil water anion (Fig. 5).

Scenario (2) represents stand clear-cut and reforestation with spruce for both stands. The decrease in soil water sulfate concentration, when stands behave similarly to conditions in the absence of forest, the spruce plantation will quickly (∼30 years) reach the soil water $SO_4$ concentration similar to values for spruce (Scenario (1)) at both spruce and beech stands (Fig. 5). This will occur due to the increase of sulfur dry deposition in the spruce canopy.

Scenario (3) takes reforestation with beech into account. Compared to the spruce reforestation scenario, the sulfate concentration will decrease steadily until 2094, when the estimated $SO_4$ concentration decreases to 111 μequiv. l$^{-1}$ at the spruce stand and to 102 μequiv. l$^{-1}$ at the beech stand (Fig. 5). The values will be similar to the concentration estimated for the beech stand in Scenario (1) (100 μequiv. l$^{-1}$).

If Scenario (4) was applied (clear-cut without reforestation), the soil water $SO_4$ will decrease markedly (Fig. 5). The estimated sulfate concentration will decrease to 80 μequiv. l$^{-1}$ at the spruce stand and to 70 μequiv. l$^{-1}$ at the beech stand by 2094, which corresponds to the values modeled in the 1900s at the spruce stand and the 1920s at the beech stand. Two important factors influence the model predictions within Scenario (4): exclusion of dry deposition and the higher water flux through the soil profile after de-forestation.

### 4.2. *Nitrogen status*

#### 4.2.1. *Historical development*

Načetín received only moderate amounts of N deposition (between 8.8 kg ha$^{-1}$ and 13.8 kg ha$^{-1}$ annually, based on bulk precipitation chemistry). The C/N ratio in the humus layer was 21 in 2003 (Oulehle et al., 2006). Some studies have suggested that the C/N ratio of organic horizons and the N input could serve as indicators of $NO_3^-$ leaching from forested areas (Dise et al., 1998; Gundersen et al., 1998; Dise and Wright, 1995). In general, when N deposition is low (<10 kg ha$^{-1}$ year$^{-1}$), no significant nitrogen leaching occurs, while with higher N deposition, the expected risk of nitrate leaching can be estimated by the C/N ratio in the O horizon. For a C/N ratio >30 the risk for nitrate leaching is considered as low, for the range of 25–30 as moderate and for the levels <25 as high. Between 1995 and 1998, the observed average nitrate concentration varied between 160 μequiv. l$^{-1}$ and 250 μequiv. l$^{-1}$ at the spruce stand, which corresponded to a loss of 6–13 kg N year$^{-1}$. After the re-start of sampling in 2003, the average nitrate concentration was 20 μequiv. l$^{-1}$ and 5 μequiv. l$^{-1}$ in 2004 and 2005, respectively. We have not been able to satisfactorily model this recent rapid change. The causes behind these changes (decrease N deposition, increase of biotic soil immobilization) were hypothesized in detail in Oulehle et al. (2006). In MAGIC, we fitted the concentrations between 1995 and 1998 as 90% of the atmospheric N deposition. After 1999 we shifted the ecosystem to a state with the ability to retain all incoming nitrogen. According to changes in radial stem increments, we supposed that the ability to retain nitrogen in the forest ecosystem had been disrupted during the 1970s and subsequent substantial leaching of nitrate had occurred.

This approach was applied to the beech forest as well. Nevertheless, the dominant anion in soil water was sulfate during the whole modeled period.

#### 4.2.2. *Future progress*

After clear-cutting, mineralization and nitrification are often enhanced, causing "excess nitrification", further acidification in the soil, $NO_3$ and Al-leaching, and losses of nutrient—cations (Dahlgren and Driscoll, 1994). After forest dieback caused by a bark beetle attack in Bavarian Forest National Park near the Czech border, $NO_3$ concentrations were significantly elevated in soil water at 40 cm for 5 years, with the highest annual mean flux-weighted concentration of 579 μequiv. l$^{-1}$ in the fifth year after dieback (Huber, 2005). Katzensteiner (2003) reported that in the first and second growth periods after a clear-cut, inorganic N fluxes with seepage had increased to 20 kg ha$^{-1}$ and 30 kg ha$^{-1}$. Mellert et al. (1996) detected maximum $NO_3$ concentrations of more than

4500 μmol and average concentrations up to 3500 μmol in most of 13 investigated wind-blown spruce stands in Bavaria. On the other hand, much lower $NO_3$ concentrations have also been observed, especially in N-limited or non N-saturated sites (Ring, 1995; Berden et al., 1997). The Načetín spruce stand is located in mountain area, with chronically elevated N deposition and until 2003 also with high nitrate leaching, suggesting N-saturation. Therefore we modeled the seepage flux of N-$NO_3$ in the year of clear-cut (2007) as 11 kg ha$^{-1}$, with the highest flux in the third year after clear-cut (47 kg ha$^{-1}$). The $NO_3$ increase was the main reason for the short-term increase in modeled SAA (Fig. 5). The peak of nitrate concentrations in soil water lasts for a period of 5 years after the clear-cut, and after that we expect the ecosystem to sufficiently retain the incoming nitrogen.

### 4.3. *Modeling of pH, aluminum and acid neutralizing capacity*

#### 4.3.1. *Historical development*

Chronic soil acidification caused a decline in pH and enhanced Al mobilization at both the spruce and beech stands (Fig. 4). As a result of sulfur deposition and consequently base cation depletion, modeled pH declined to a minimum of 4.25 at the

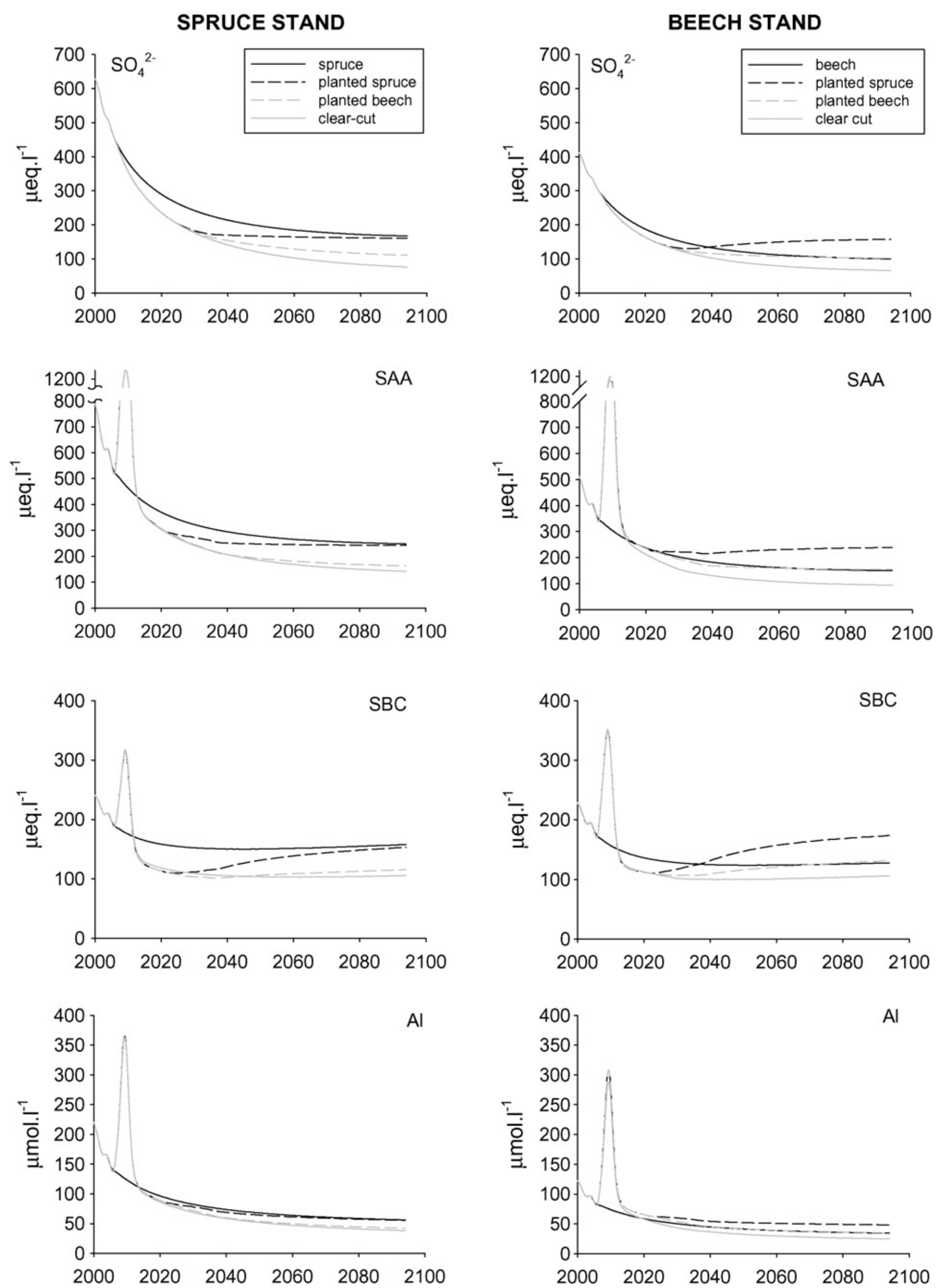


**Fig. 5 – Simulated concentrations of sulfate, sum of strong anions, sum of base cations, aluminum, pH, acid neutralizing capacity, Bc/Al molar ratio in soil water and base saturation at the spruce and beech stands from 2000 to 2094. Solid black line represents Scenario (1), black dashed line represents Scenario (2), grey dashed line represents Scenario (3) and solid grey line represents Scenario (4).**

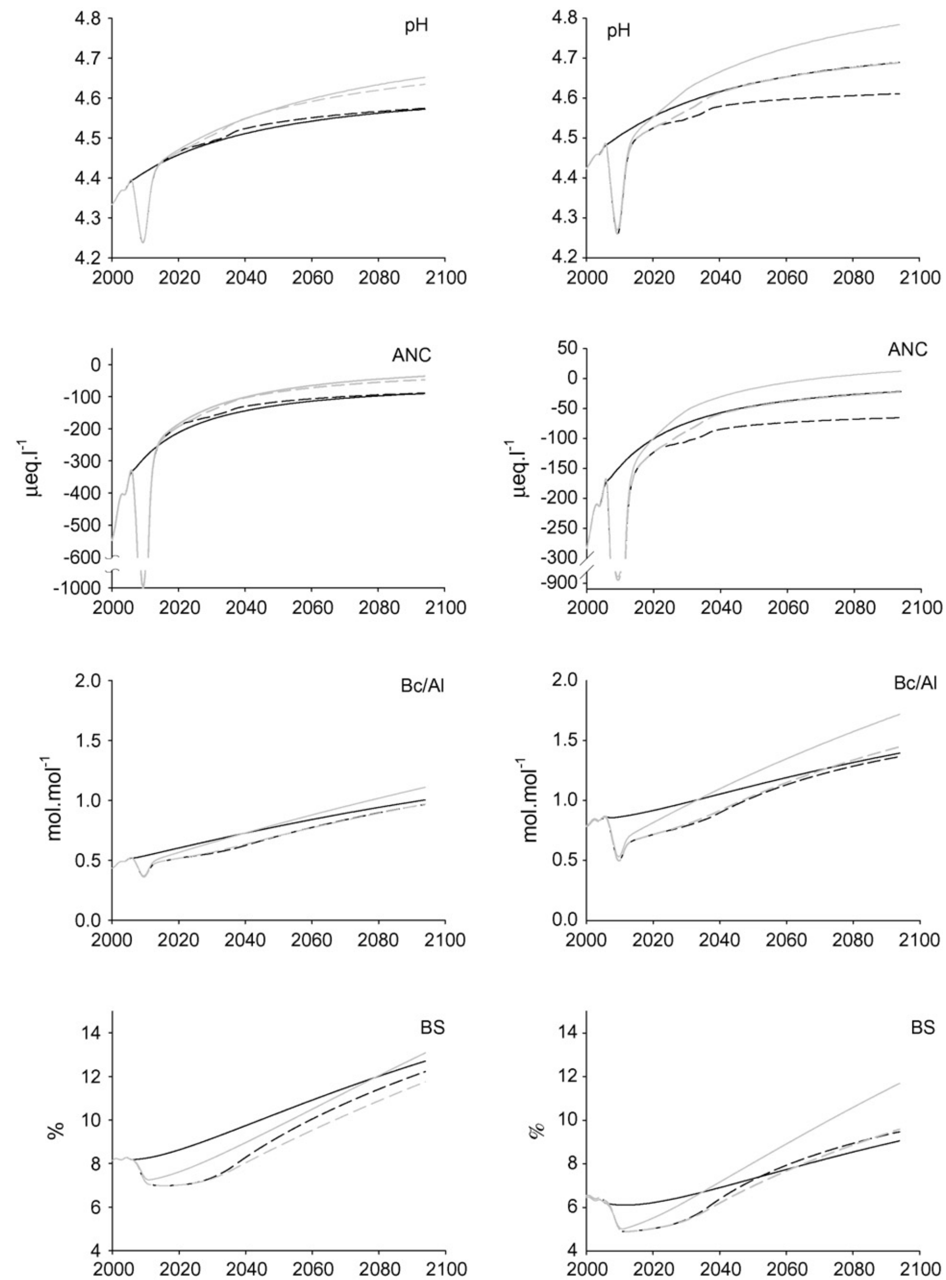


**Fig. 5 – (*Continued*).**

spruce stand and 4.36 at the beech stand during the 1980s. The decrease was only of about 0.5 pH compared to the values estimated for 1854. After a decade of declining deposition, the pH increased only slightly (Fig. 4).

Total Al concentrations in the MAGIC model are controlled by $Al(OH)_3$ solubility. The $Al(OH)_3$ solubility constant for mineral soil at the Načetín stands was optimised using soil water measurements between 1994 and 2005 to $-pK_{Al(OH)_3} = 9.05$. This value is in good agreement with results of Oulehle and Hruška (2005) who found an equilibrium of $Al^{3+}$ concentration in soil water between gibbsite ($-pK = 8.04$) and amorphous $Al(OH)_3$ ($-pK = 9.66$). The highest modeled Al concentrations were observed during the 1980s (Fig. 4). The concentration of Al peaked with 440 $\mu$mol l$^{-1}$ in 1987 at the spruce stand and with 195 $\mu$mol l$^{-1}$ in 1989 at the beech stand. The higher acidic deposition, lower pH and reduced ability of base cations to neutralize the acidity were the main factors affecting the unfavorable high aluminum concentration at the spruce stand. One of the results was a decline in the Bc/Al ratio to the measured value of 0.45 at the spruce stand and 1.1 at the beech stand in 2005, compared with much higher values estimated for the 19th century (Fig. 4).

The charge balance ANC (Fig. 4) was in inverse relationship with Al concentration. The estimated historical ANC was around $-20$ $\mu$equiv. l$^{-1}$ at the spruce stand and around 30 $\mu$equiv. l$^{-1}$ at the beech stand. The steep decrease after 1950 was caused by changes in $H^+$ and particularly $Al^+$ concentrations, because the concentration of organic anions is negligible in the deep mineral soil water. The lowest values were modeled in 1987 with $-1080\,\mu$equiv. $l^{-1}$ at the spruce stand and in 1989 with $-470\,\mu$equiv. $l^{-1}$ at the beech stand.

The more pronounced pH decrease at spruce stand resulted in Al mobilization and consequently to decrease of Bc/Al ratio <1 in 1950. It could be an important stress factor contributing to the spruce decline in the Ore Mts. between 1960s and 1980s.

#### 4.3.2. *Future progress*

Keeping original trees on the stands (Scenario (1)) will result in a slow increase of pH to the predicted values of 4.57 at the spruce stand and 4.69 at the beech stand in 2094. The concentration of Al will consequently decrease to about $60\,\mu$mol $l^{-1}$ at the spruce stand and to about $30\,\mu$mol $l^{-1}$ at the beech stand. The predicted value at the beech stand in 2094 corresponds to the modeled value for spruce in 1854 ($32\,\mu$mol $l^{-1}$) and is only slightly enhanced compared with $19\,\mu$mol $l^{-1}$ modeled for the beech stand in 1854. Also, predicted concentrations at the spruce stand will be higher compared with modeled concentrations in 1854 (Figs. 5 and 4). Predicted ANC will increase to $-90\,\mu$equiv. $l^{-1}$ at the spruce stand and to $-20\,\mu$equiv. $l^{-1}$ at the beech stand in 2094.

Spruce plantation after harvesting of stands (Scenario (2)) will result, after a drop of pH, Al and ANC caused by the clear-cut and enhanced $NO_3^-$, in prompt convergence to predicted values as used in Scenario (1) at the spruce stand. The planting of spruce at the originally beech stand causes lower pH, higher Al and lower ANC compared to predicted values in Scenario (1) for the beech stand (Fig. 5).

The planting of beech (Scenario (3)) at the beech stand leads to the convergence of predicted values of pH, Al concentration and ANC with values estimated in Scenario (1), similarly as to planted spruce at the spruce stand. On the other hand, planting of beech at the spruce stand leads to an increase in pH, decrease in Al and increase in ANC compared with spruce planting (Fig. 5).

The clear-cutting of Scenario (4) results in an increase of pH to 4.65 at the spruce stand and to 4.69 at the beech stand, which is close to modeled values for the spruce stand in 1854. The Al concentration decreases to $40\,\mu$mol $l^{-1}$, close to the beech scenario at the spruce stand. At the beech stand, clear-cutting resulted in a decrease in Al close to the modeled value in 1854 (Fig. 5).

The predicted chemistry of soil water under different types of forest management resulted to similar conclusions. Planting of a new generation of spruce will result in the deterioration of soil status at the beech stand and in preservation of status quo at the spruce stand. Conversely, planting of beech leads to an improvement in the spruce soil environment. The rapid drop of pH, increase of Al and decrease of ANC (Fig. 5) will occur only in the short-term period, when the pulse of acidification after the clear-cut is in progress.

### 4.4. *Modeling of base cations*

#### 4.4.1. *Historical development*

The modeled sum of base cations (Ca + Mg + Na + K) in soil water increased gradually, while $SO_4$ increased over the historical period from 1854 to the mid 1980s. The SBC then declined following the decrease of $SO_4$ (Fig. 4). Weathering rates (Table 1) were estimated within the MAGIC calibration procedure simultaneously with historic soil base saturation. Due to the similar geographic and climatic conditions, as well as historical land use (we supposed that conversion of the spruce stand to spruce monoculture had occurred in the mid 19th century, real data are lacking), an annual weathering rate of 37.5 mequiv. $m^{-2}$ year$^{-1}$ and initial base saturation of 14.3% for both stands was used. According to Reuss (1994), weathering is accelerated by elevated input of $H^+$ from the atmosphere and this hypothesis is supported by laboratory experiments (Sverdrup, 1990). Assuming Si concentration as an indicator of silicate mineral weathering rate, our data supported the hypothesis, that decline of $H^+$ input resulted in decrease of Si concentration. The concentration of $SiO_2$ in soil water at the spruce stand was stable with an average of $9.6 \pm 1.6$ mg $l^{-1}$ between 1995 and 1997, and since 2003 has decreased to $6.6 \pm 0.6$ mg $l^{-1}$. In contrast, Na as a second potential weathering indicator, increased slightly from $65 \pm 9.4\,\mu$mol $l^{-1}$ between 1995 and 1999 to $74 \pm 9.3\,\mu$mol $l^{-1}$ since 2003. This is opposite to the results of Hruška and Krám (2003). They observed an increase of Si and decrease of Na in runoff from the acidic Lysina watershed. Due to the contradictory signals from these two major weathering indicators, constant weathering rates were used in MAGIC for the whole modeled period 1854–2094 at Načetín.

The simulated concentration of SBC was $120\,\mu$equiv. $l^{-1}$ at the spruce stand, similar to $115\,\mu$equiv. $l^{-1}$ at the beech stand in 1854. The concentration peaked in 1983 at the spruce stand and in 1985 at the beech stand with $420\,\mu$equiv. $l^{-1}$ and $435\,\mu$equiv. $l^{-1}$, respectively. The contribution of base cations to neutralizing acidity was estimated to be about 75% at the beech stand and 62% at the spruce stand in 1854. During the increasing acidic deposition the ratio of base cations contributing to neutralization decreased to 40% at the beech stand and to only 25% at the spruce stand in the 1980s. An increasing role of aluminum in neutralizing of soil water acidity was modeled for that period. In the 1980s, aluminum contributed 55% at the beech stand and 70% at the spruce stand to neutralizing acidity (Fig. 6).

#### 4.4.2. *Future progress*

Implementation of Scenario (1) in the MAGIC prediction will lead to a substantial decrease of base cation concentration at the spruce stand until 2020 and at the beech stand until 2040 (Fig. 5). This decrease corresponds with an $SO_4$ decrease in soil water. Further progress will be characterized by equilibrium between inputs (deposition, weathering) and outputs (leaching). Estimated future higher deposition of base cations results in a faster establishment of this equilibrium at the spruce stand.

Spruce planting (Scenario (2)) will result in a higher concentration after clear-cut for approximately 6 years due to enhanced leaching of $NO_3^-$. During tree growth, concentration of base cations gradually increases with an increase in dry deposition. The modeled concentrations in 2094 were $150\,\mu$equiv. $l^{-1}$ and $170\,\mu$equiv. $l^{-1}$ at the spruce and beech stands, respectively (Fig. 5). Higher values at the beech stand correspond with the important role of base cations in neutralizing of acidity.

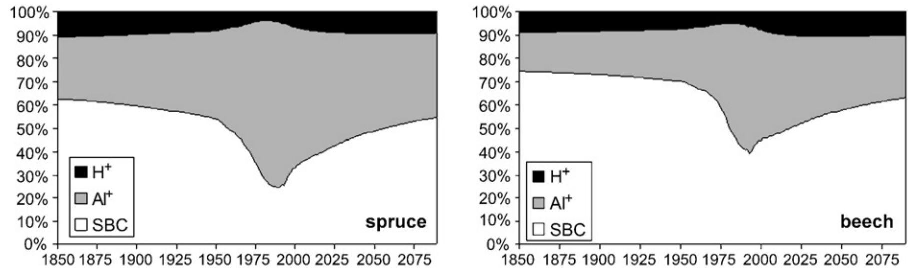


**Fig. 6 – Modeled relative contributions of hydrogen ion, aluminum and base cations on neutralizing capacity in soil water at Načetín.**

Modeled beech planting (Scenario (3)) will result in lower base cation concentrations compared with spruce planting, as a result of lower SAA concentration. The clear-cutting scenario predicted strongest decrease of base cations until 2094. At the spruce stand the concentration decreases to 106 μequiv. $l^{-1}$, which is close to estimated value for the beech forest (Fig. 5).

### 4.5. *Modeling of base saturation*

#### 4.5.1. *Historical development*

The simulated soil base saturation at the spruce stand decreased from an estimated 14.3% in 1854 to the 7.3% and 8.2% measured in 1994 and 2003, respectively (Fig. 4). A more marked decrease was observed at the beech stand, where soil base saturation decreased from modeled 14.3% in 1854 to measured 6.4% in 2003. Depletion of base cations from cation-exchange sites was an important mechanism for neutralizing incoming acidity. During the peak of acidification, soil cation exchange was the dominant source of base cations in soil water. The long-term monitoring of soil water and precipitation chemistry confirmed the hypothesis of a close relation between changes of deposition and changes of the exchangeable pool at sites with low base cations pools. The decreasing trend of Ca deposition resulted in a decrease of the Ca pool in humus and the top mineral (0–10 cm) horizon, whereas stable deposition of Mg resulted in an insignificant change of the exchangeable pool between 1994 and 2003 (Oulehle et al., 2006). Moreover, between 1994 and 2003 the soil base saturation increased from 7.3% to 8.2% at the spruce stand, which was caused namely by the increase of base saturation in the organic and upper mineral soil horizons. The decrease of soil base saturation, as a result of acidic deposition, was modeled for forest soils elsewhere in Europe (Hruška and Krám, 2003; Malek et al., 2005; Belyazid et al., 2006).

#### 4.5.2. *Future progress*

The predicted increasing trend in base saturation, as observed in 1994 and 2003, will continue until 2094 at the spruce stand (Fig. 4). The predicted decrease of base saturation at the beech stand will continue until 2010, when base saturation will have decreased to 6% (Fig. 5). After that, the soil starts to be a positive sink of base cations, and the predicted base saturation will be 9% in 2094.

In the case of the spruce scenario (Scenario (2)), the predicted soil base saturation will increase to 13% at the spruce stand and to 9.5% at the beech stand in 2094. The predicted drop of base saturation after clear-cut to 7% at the spruce stand and to 5% at the beech stand will occur until 2030 (Fig. 5). The following increase of base saturation corresponds with an increase of base cation deposition onto the spruce canopy during the simulated growing period together with decreasing water outflow.

The beech planting scenario similarly reproduces the situation as modeled for the spruce scenario (Fig. 5). The predicted increase of base saturation is not so steep at the spruce stand due to the lower deposition of base cations. On the other hand, the new planting of beech at the beech stand will result in an increase of base saturation comparable with the spruce scenario.

The future prediction of base saturation as defined in Scenario (4) will result in the highest increase: to 13% at the beech stand and to 12% at the spruce stand. This was caused by the exclusion of cation uptake by vegetation in the clear-cutting scenario. The uptake of cations by vegetation plays a more crucial role at the beech stand (Fig. 5), because SBC deposition there was lower. It is clearly evident, that uptake of base cations by vegetation is important sink in the long-term perspective. Such finding is supported also from Sweden (Sverdrup and Rosen, 1998) and from Slavkov Forest in the Czech Republic (Hruška et al., 2002).

## 5. Conclusions

Observed chemistry of soil water at the 90 cm depth at the Načetín spruce stand for the period 1994–2005 was successfully reproduced by the MAGIC model. Moreover, this calibration was also able to satisfactorily model the soil water chemistry at the beech stand, and only the difference in deposition fluxes in the beech versus spruce forest was a main driving factor controlling the soil water chemistry. This confirms the hypothesis that tree species were able to significantly influence the chemical status of the soil environment under acidic deposition.

At both stands, the depletion of exchangeable base cations from the soils played a crucial role in the acidification process, which resulted in decreases of soil base saturation. Moreover, the higher soil water anion concentration and lower pH at the spruce forest caused by higher acidic deposition under the spruce canopy resulted in more pronounced mobilization of aluminum. According to the modeled Bc/Al ratio, the aluminum stress could have been an important factor in spruce decline during 1960s and 1980s.

Different forest management scenarios with constant deposition (average 2004 and 2005) were applied to the beech and spruce forests to predict trends in soil chemical parameters until 2094. The scenario without clear-cutting and keeping of original stands will result in an increase of base saturation at both stands. Elevated concentration of aluminum and a more unfavorable Bc/Al ratio were modeled for the spruce stand compare to the beech stand. In general, planting of spruce would deteriorate the future soil environment of the beech stand. For the spruce stand, a less pronounced change was predicted. Planting of beech on the spruce stand improves the soil chemical parameters in the future predictions. For the beech stand, a less pronounced change was predicted. When the clear-cutting scenario was applied to the MAGIC predictions, the highest soil improvement was modeled for both stands. Additionally, the predicted base saturation increase was closely related to base cation vegetation uptake particularly at the beech stand, where estimated input by atmospheric deposition in the future is only two thirds of that for the spruce forest.

The MAGIC model results clearly demonstrate the importance of long-term interactions between acidic deposition and forest management on soil chemistry. This indicates that forest management practices have strongly influenced the development of soil acidification and are able to affect the future regeneration.

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