Experiences with liming in European countries – results of long-term experiments

W. Schaaf, R. F. Hüttl

Brandenburg University of Technology, Cottbus, Germany

ABSTRACT: Liming was and still is carried out in many European forests since the beginning of the 1980s. Main objectives of compensation liming is to neutralize actual acidic deposition in order to prevent further soil acidification, to increase base saturation and acid buffering capacity of the soil. This paper evaluates results from long-term liming experiments throughout Europe using different amounts and forms of liming materials. Differences in forest ecosystem reaction to liming can be caused by amount, form and frequency of the application, and by the solubility, reaction kinetics, contents and composition of the material used. Furthermore, other factors like climate, site history, management, environmental conditions (e.g. actual and historical deposition rates), humus form, C/N ratio, nitrogen stores, and the soil chemical properties (e.g. pH, base saturation, CEC) may strongly influence the effects of liming treatments. Element budgets of forested ecosystems on a plot or watershed scale provide a very useful tool to integrate the various effects of additional Mg inputs via liming/fertilization.

Keywords: acid deposition; soil acidification; element budget; lime; dolomite; fertilization; soil pH; base saturation; buffer capacity; site history

Liming was and still is carried out in many European forests since the beginning of the 1980s. Reasons and objectives of liming treatments in forest ecosystems can be manifold. Main objectives of compensation liming is to neutralize actual acidic deposition in order to prevent further soil acidification, to increase base saturation and acid buffering capacity of the soil and to reduce possible toxic or antagonistic effects of elevated aluminium concentrations in acidified soils (Beese, Meiwes 1995; Moravčík, Cienciala 2003). In addition, since magnesium was found to be one of the most deficient elements in tree nutrition (Zech, Popp 1983), mainly dolomitic lime is used to mitigate foliage deficiency (Hüttl, Schaaf 1997). Other liming treatments, often called amelioration liming, mainly aim at improving the humus form of a site, at re-supplying nutrient losses either by leaching or by biomass harvesting, at deepening of the root system or are carried out as a site amelioration tool to mitigate natural or silvicultural regeneration or transformation of forest stands (Andersson, Persson 1988). These types of liming experiments date back to the beginning of last century (Lundström et al. 2003).

RESULTS AND DISCUSSION

In many studies the main effects of forest soil liming are found with respect to soil pH, cation exchange capacity and base saturation (Hüttl 1989; Kreutzer 1995). In most cases these effects are restricted to the organic surface layer and the uppermost mineral soil (Beese, Meiwes 1995; Fig. 1). Kreutzer (1995) in the well documented Höglwald project in southern Germany, could show that dolomitic lime was completely dissolved after six years and 60 to 80% of the applied calcium and magnesium, respectively, was adsorbed to the exchange sites mainly of the humus layer, but even in the mineral soil down to 40 cm depth (Fig. 2). In an evaluation of 22 European liming studies, Lundström et al. (2003) generally found an pH increase of 0.1–2.4 units, base saturation increases by 9–58%
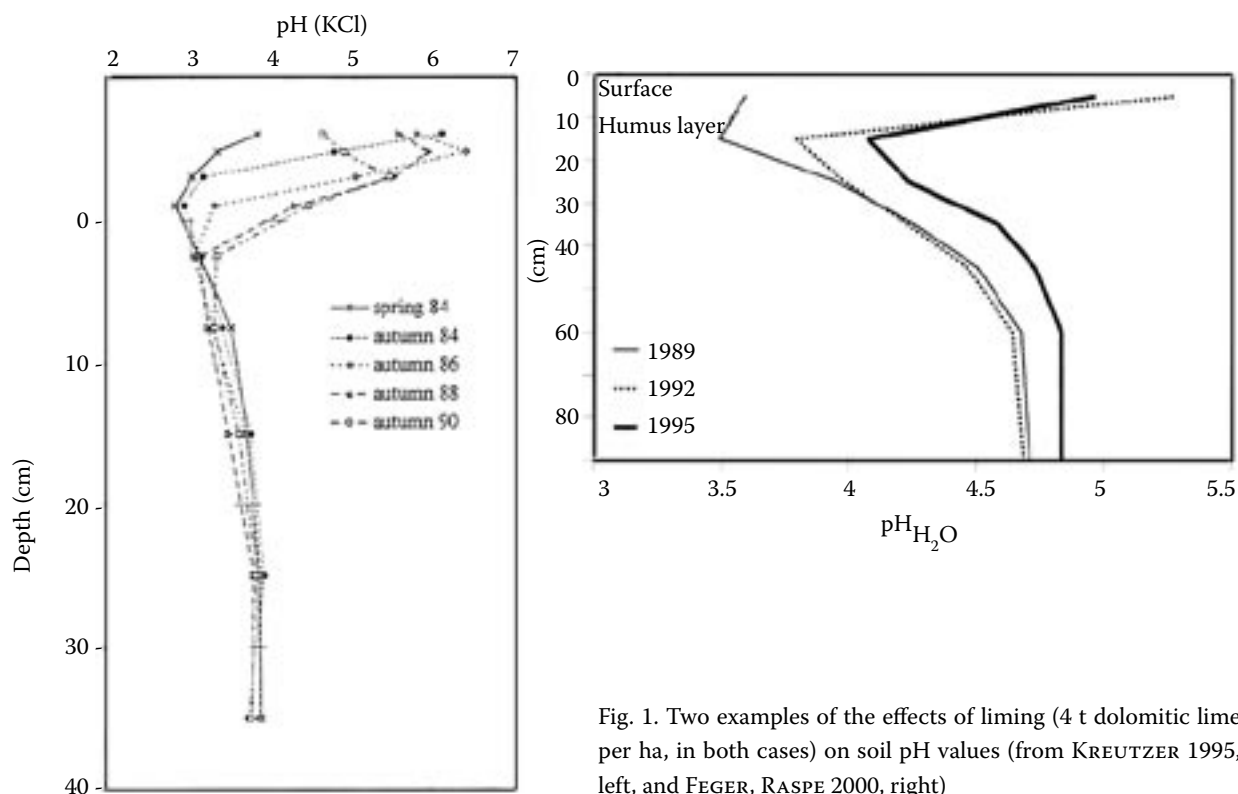
Fig. 1. Two examples of the effects of liming (4 t dolomitic lime per ha, in both cases) on soil pH values (from Kreutzer 1995, left, and Feger, Raspe 2000, right)

and an increase in cation exchange capacity of up to 51 $kmol_c$/kg in the humus layers of the investigated sites. In the upper mineral soil changes were much smaller, in some cases even negative. In soil solutions pH values increased by up to two units after liming (Lundström et al. 2003). Accordingly, aluminum concentrations decreased generally, in some cases liming lead to a short-term increase, when cation exchange reactions released higher amounts of adsorbed Al. Liming generally resulted in clearly increased concentrations of nitrate and dissolved organic carbon (DOC) in soil solutions. This finding is mainly due to increased microbial acitivity, soil respiration and mineralization of soil organic matter (SOM; Persson et al. 1991) and could result in a risk for further mineral soil acidification and pollution of ground and surface waters as well as for substantial losses of SOM stores from the soil. Kreutzer (1995) reported losses of up to 23% of total carbon and 13% of total nitrogen from the humus layer after liming (6.5–7 t C/ha and 170 kg N/ha, respectively). A major part of the carbon losses was contributed to increased respiration losses and release from the soil as CO_2. Effects of surface liming on mineralization and NO_3 leaching seem to be highly depending on the N-status of the site. An important indicator seems to be the C/N-ratio (Persson et al. 1989). Another possible risk of surface lime treatments is the mobilization of heavy metals from polluted

humus layers and leaching from the soil in form of metal-organic complexes.

In general, pH increases are correlated with the applied amount of dolomite. In the Hoeglwald experiment the application of 4,000 kg/ha dolomitic lime increased pH in the organic surface layer from values of 2.8–3.5 to 3.5–6.5 (Kreutzer 1995).

Positive effects of dolomitic liming also have been shown with respect to increasing exchangeable Mg stores and base saturation (Aldinger 1987), although there seems to be a need for an appropriate determination method of exchangeable Mg and CEC if the soil sample contains undissolved lime material. Base saturation was increased from 50–80% to 100% in the organic surface layer and in the upper mineral soil from 6–10% to 17–33% in the Hoeglwald experiment (Kreutzer 1995). An application of 4 t/ha lime plus 2 t/ha dolomitic lime increased the base saturation from 5.0–6.0% to 18.5–13.0% in the upper 20 cm of the profile (Wiedey 1991).

A comparison of results from two dolomite fertilization experiments reported by Schüler (1991) demonstrates the influence of grain size. Two Norway spruce stands were treated with 3 t/ha of dolomite, at the site Linz the grain size was 0–0.09 mm and at the site Pruem it was 0.1–2 mm. While the contents of CaO and MgO where almost the same, the reactivity, as tested in short-time studies with HCl, was three to four times higher for the fine

ground dolomite. Correspondingly, the acid neutralization capacity (ANC) of the amounts applied to the forest were different. Two years after application of the two fertilizers the pH-values of the organic layers were increased by 1.5 and 0.3 units, respectively, compared to the control. Base saturation in the upper mineral soil was increased from 6% to 12% in Linz and from 3% to 5.5% in Pruem. Similar results were obtained by HILDEBRAND and SCHACK-KIRCHNER (1990) in soil column studies.

Almost all surface liming experiments agree that these main positive effects on soil chemistry are restricted to a few cm depth of the profile, in most cases to the organic surface layer or the upper few cm of the top mineral soil (KREUTZER 1995). Only with very high dosages or after long times effects in deeper soil layers are reported (KREUTZER 1995; MARSCHNER et al. 1989; MATZNER et al. 1985; REITER et al. 1986; WENZEL 1989).

Effects of liming on root distribution and intensity have been frequently reported. The results, however, can not easily be compared and seem to be strongly dependend on site conditions. Liming often results in increased rooting depth in the mineral soil and decreased root intensity in the humus layer. But similar to soil chemical effects there is a high temporal variation after treatment. FEGER and RASPE (2000) report decreasing of an initially elevated fine root intensity reaching the value of the control plot seven years after liming. As a short-term reaction several studies reported a decreased fine root density and/or vitality in the humus layer (SCHAAF, ZECH 1991). 25 years after the application of 9 t/ha

lime, KAUPENJOHANN (1995) found a lower fine root biomass compared to the control down to 20 cm mineral soil depth in a 120 year old Norway spruce stand in the Fichtel Mountains, Germany. Below fine root biomass was higher at the limed plot. The soil chemical status before liming seems to be of much importance for its reaction to liming and subsequently for the reaction of the root system.

To enhance the liming effect both in time and in a larger soil depth also more soluble materials were used. In a field trial in the Fichtel Mountains, Germany, 2,650 kg/ha of granulated magnesium hydroxide were applied at a 60 year Mg-deficient old Norway spruce stand on a sandy typic Dystrochrept derived from granite (SCHAAF 1995). Magnesium concentrations of the soil solution increased three months after treatment down to 70 cm soil depth. As an average over three years, Mg concentrations were increased by the factor 10 compared to control plots. Even 4.5 years after application these relations still were found. $Mg(OH)_2$ application had also a marked effect on solution pH in the humus layer, but also in 25 cm mineral soil depth, where pH values stabilized about 1 year after fertilization with a mean value of 4.0 compared to 3.5 of the control plots (Fig. 2). This pH effect was still detectable after 4.5 years. Nitrate concentrations were elevated after application. However, after some time they decreased and even fell below the NO_3 concentrations of the control (Fig. 3). In 25 cm soil depth the NO_3-values remained constantly below the control values after 15 months. It was concluded that due to its high solubility, effectivity of $Mg(OH)_2$ in soil
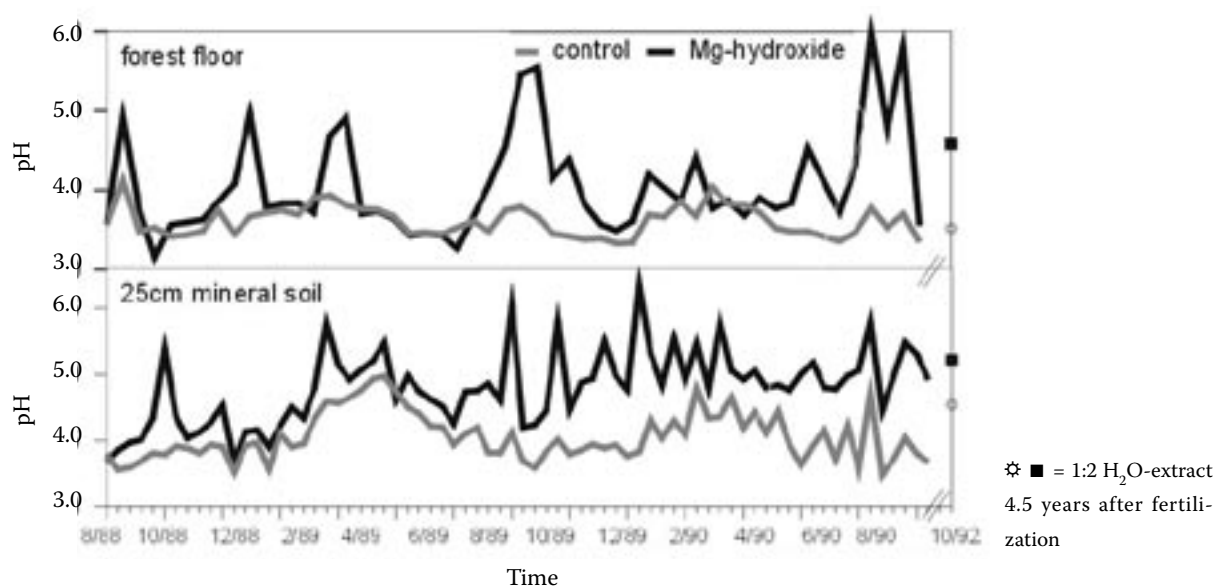

Fig. 2. pH values in soil solution from two soil depths over the period August 1988 to October 1990 and results of water extracts in October 1992 (from SCHAAF 1995)
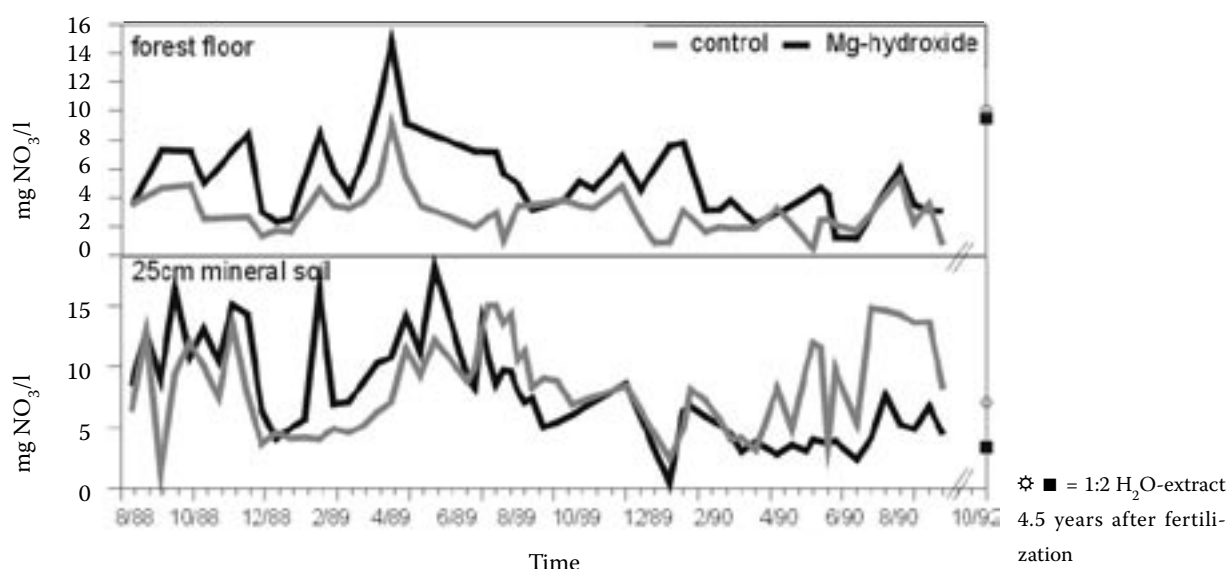
Fig. 3. Nitrate concentrations in soil solution from two soil depths over the period August 1988 to October 1990 and results of water extracts in October 1992 (from SCHAAF 1995)

depth is increased compared to results from surface application of lime and could be used to combine the positive effects of liming and e.g. MgSO$_4$ fertilization.

Element budgets of forested ecosystems on a plot or watershed scale are very useful to integrate the various effects of additional Mg inputs via liming/fertilization. If continued over sufficiently long periods, i.e. at least several years after the treatment, these budget data can be used for an evaluation of the measures on the ecosystem level. That includes all different processes at different compartment levels and enables to integrate over all factors in-

fluencing the treatment itself like management and deposition history as well as site conditions (soil, climate, etc.). In an attempt to evaluate liming effects on the ecosystem level, SCHAAF (1997) compared the element budgets from plot or watershed studies of control and limed forest stands.

MEYER (1992) investigated the effects of two increasing doses of dolomitic lime in Northern Germany. The soils were characterized as typic haplorthods derived from sand. The control plot showed slightly positive budgets for the base cations Ca and Mg (Fig. 4). For aluminum a strongly negative budget of 2.6 kmol$_c$/ha/year was calculated. Also sulphate
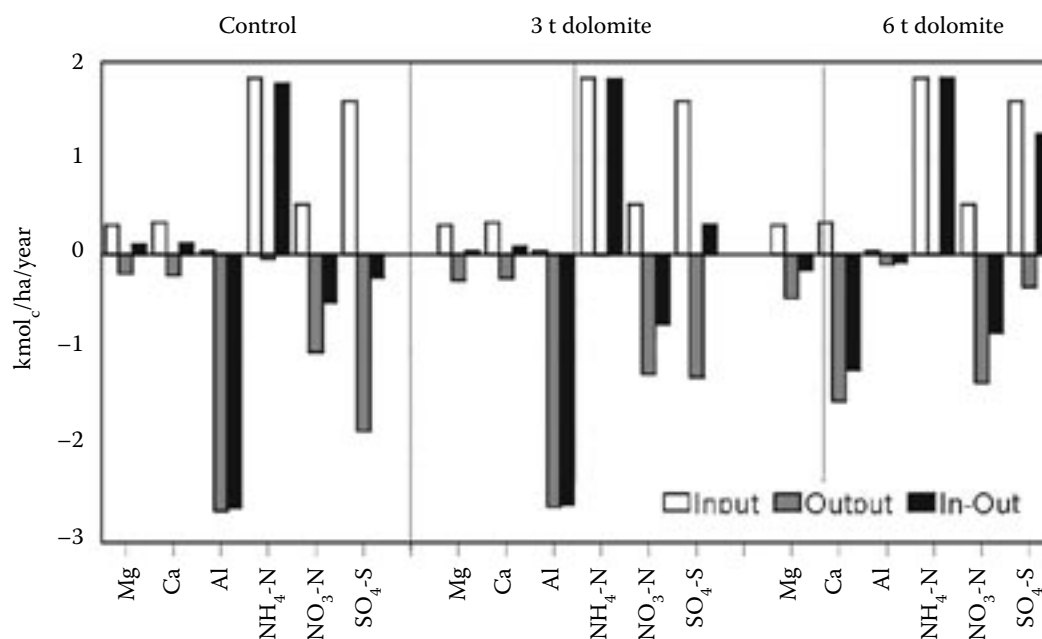

Fig. 4. Element budgets of control and limed plots at the site Wingst, NW-Germany (data from MEYER 1992)

and nitrate output exceeded total deposition. The application of 3 t/ha dolomitic lime resulted in an increased NO_3-N-output. Ca- and Mg-outputs were also increased, but were still below the inputs (not including the fertilizer inputs). S-output was reduced. After the treatment with 6 t/ha dolomitic lime the budgets of Mg and especially Ca became negative while Al-output was almost zero. Also sulphate output was significantly reduced. Within the first two years only 0.2–4.2% of the fertilized Ca- and Mg-amounts were lost from the soil.

40-year-old Scots pine ecosystems on dystric cambisols in Berlin, Germany were studied by MARSCHNER et al. (1989). Element budgets were calculated over a period of 3½ years on a control plot and one plot treated with 6.1 t/ha dolomitic lime + 145 kg/ha "Patentkali" (K_2SO_4). The total amounts fertilized were 2,450 kg Ca, 144 kg Mg, 34 kg K, and 20 kg S. Besides for N, all budgets on the control plot were negative (Fig. 5). The outputs of all fertilized basic cations were increased, but outputs of aluminum and sulphate were almost doubled, leading to strongly negative budgets.

From a watershed study in the Black Forest, Germany, FEGER and RASPE (2000) reported element budgets of two 60–80-year-old Norway spruce stands growing on iron-humus podzols. One of the watersheds was limed with 4 t/ha of dolomite. In the first year after application the treatment resulted in elevated Ca- and Mg-outputs (Fig. 6). The budgets of these elements were negative on the control. Outputs of sulphate and nitrate were slightly decreased and increased, respectively. Al showed no effect.

TÜRK (1992) reported budget data for control and limed plots in a 40 year old Norway spruce stand in the Fichtel Mountains, Germany, over a period of six years. The very high application of 10 t dolomite, corresponding to 1,715 kg Mg/ha increased the Mg-output, but not the Ca-output at the site within the period of six years after treatment (Fig. 7). There were only minimal increases in Al- and S-output, however, nitrate output was tripled.

The comparison of element budget data confirmed the statement that there is no uniform reaction of forest ecosystems or forest soils to different liming or fertilization treatments. Liming, especially using dolomite, does not necessarily result in increased nitrate losses, as shown for example by FEGER and RASPE (2000) in their catchment study. One of the major keys in understanding the effects of liming on the N-budget of a site seems to be its N-status (KREUTZER 1995) including atmospheric deposition as well as site and management history.

One further important aspect of forest soil liming was intensively investigated in the eastern part of Germany. Here, large areas were affected by long-term deposition of large amounts of alkaline dust (SCHAAF et al. 2004; HÜTTL, BELLMANN 1998). Comparable situations also existed in western Europe until the 1960s and still exist in many East-European countries. At these sites, despite high sulphur deposition rates, an alkalinization of the forest soils is found that only now re-acidify after the discontinuation and modernization of existing emission sources like e.g. lignite-fired power plants.
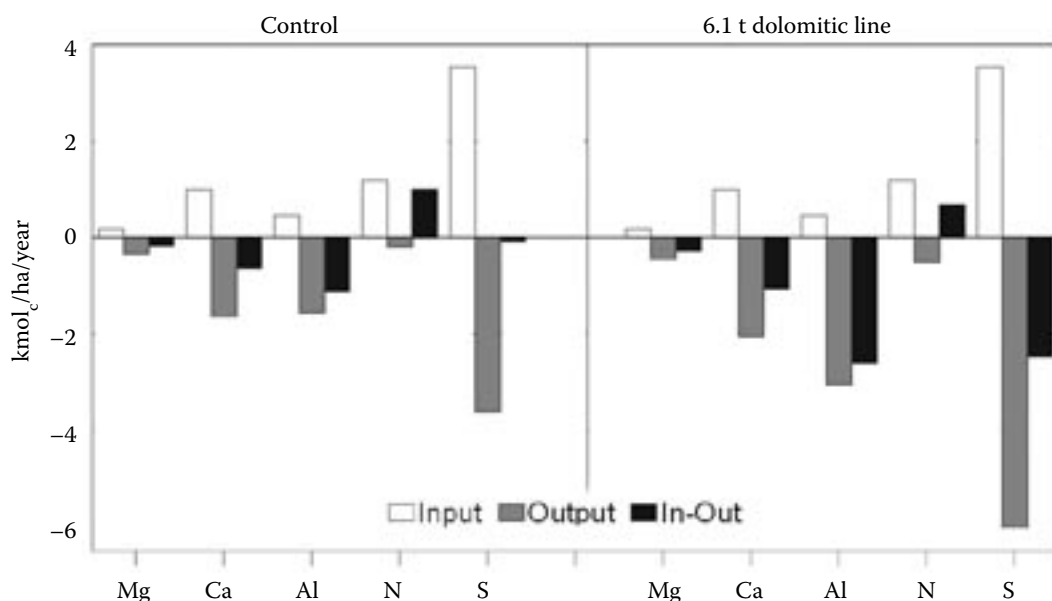

Fig. 5. Element budgets of control and limed plots in Berlin, Germany (data from MARSCHNER et al. 1989)
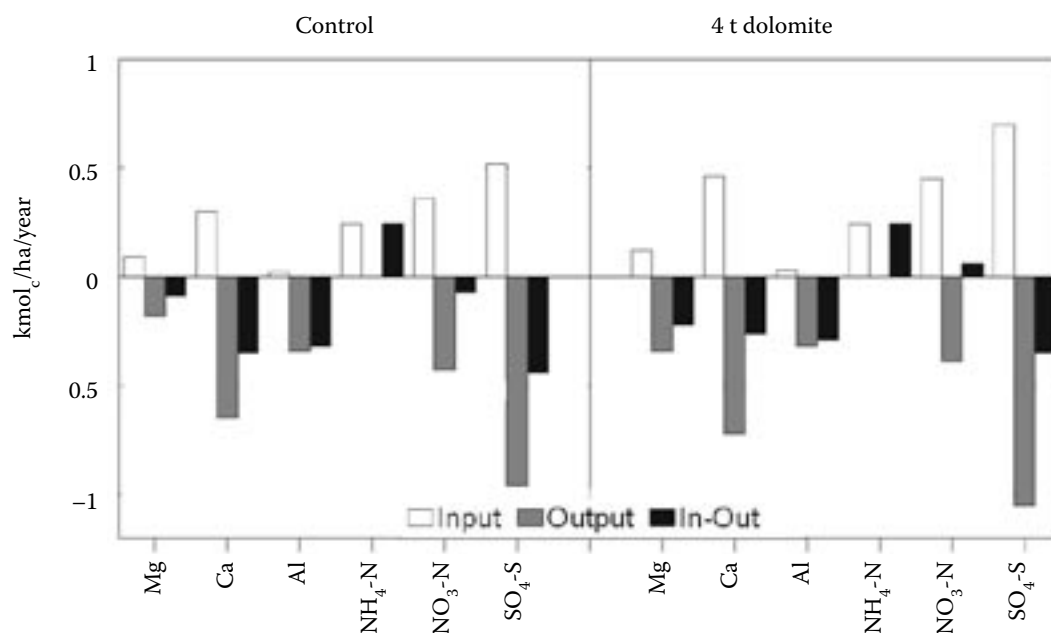
Fig. 6. Element budgets of control and limed watersheds in the Black Forest, Germany (data from Feger, Raspe 2000)

In contrast to western Germany and Europe, emission rates of pollutants in the area of the GDR remained high until 1990 or even increased during the 1980s. Due to the prevailing industrial production conditions and environmental standards (e.g. in the GDR 80% of the total energy production was based on combustion of lignite with relatively high sulphur contents, and in power plants missing effective filter systems) high emission rates not only of SO_2 but also of alkaline dust were recorded. This situation led to area specific deposition patterns. However, the high amount of alkaline dust emission contributed to a considerable neutralization of acid pollutants already in the atmosphere and even resulted in an alkalinization of top soil layers at the most impacted areas (Konopatzky, Freyer 1998). During the former deposition regime mainly sulphur and, at sites close to emission sources, also Ca accumulated in high amounts in the organic surface layer.
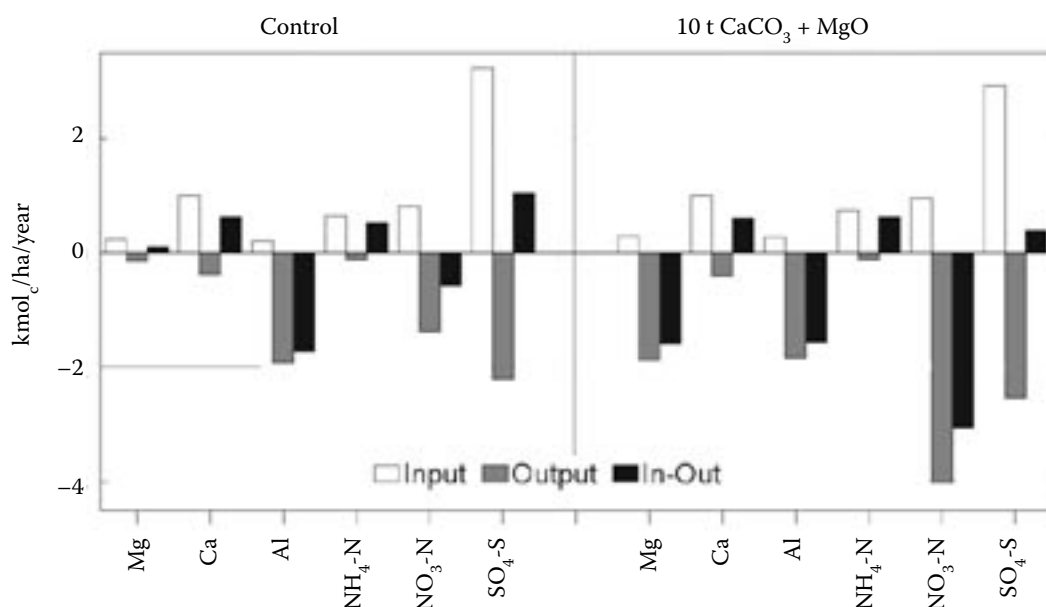

Fig. 7. Element budgets of control and limed plots at the site Oberwarmensteinach in the Fichtel Mountains, Germany (data from Türk 1992)

Table 1. Soil chemical parameters of the top soils at three Scots pine sites in eastern Germany

Site/horizon	Depth (cm)	pH (H$_2$O)	pH (KCl)	ECEC (mmol$_{(c)}$/kg)	BS (%)	Al$_{ex}$ (%)	C$_t$ (%)	N$_t$ (%)	C/N	S$_t$ (mg/kg)
Rösa										
Oa	10–4	5.2	4.3	251	67.8	24.6	36.64	2.01	18	2,943
Oe	4–0	5.7	4.6	291	92.0	6.2	29.43	1.36	22	2,883
E	0–7	5.1	4.1	42	73.8	19.4	9.72	0.42	23	1,420
Taura										
Oa	7–2	4.1	3.2	260	66.1	22.5	46.51	1.74	27	3,006
Oe	2–0	3.8	3.1	167	43.4	38.8	40.32	1.55	26	3,052
E	0–6	4.0	3.2	25	24.2	38.8	5.58	0.20	28	754
Neuglobsow										
Oa	5–0	3.8	2.7	188	48.0	17.6	39.19	1.30	30	2,203
E	0–3	3.9	3.1	43	15.5	51.6	4.09	0.14	29	545

BS – base saturation, Al$_{ex}$ – exchangeable aluminium in % of ECEC

Although throughfall deposition rates of Ca and SO$_4$-S show a considerable decrease from 1993 to 1999, even the higher values at the beginning of our investigation were already manifold lower compared to input rates during GDR times. Nagel et al. (1991) reported deposition loads of 5.9 kmol S/ha/a and 8.0 kmol Ca/ha/a in throughfall for the period 1986–1989 in Scots pine stands close to the emission sources (lignite power plants and industrial complexes), as e.g. for the Bitterfeld area, and 1.9 kmol S/ha/a and 1.3 kmol Ca/ha/a in more remote areas.
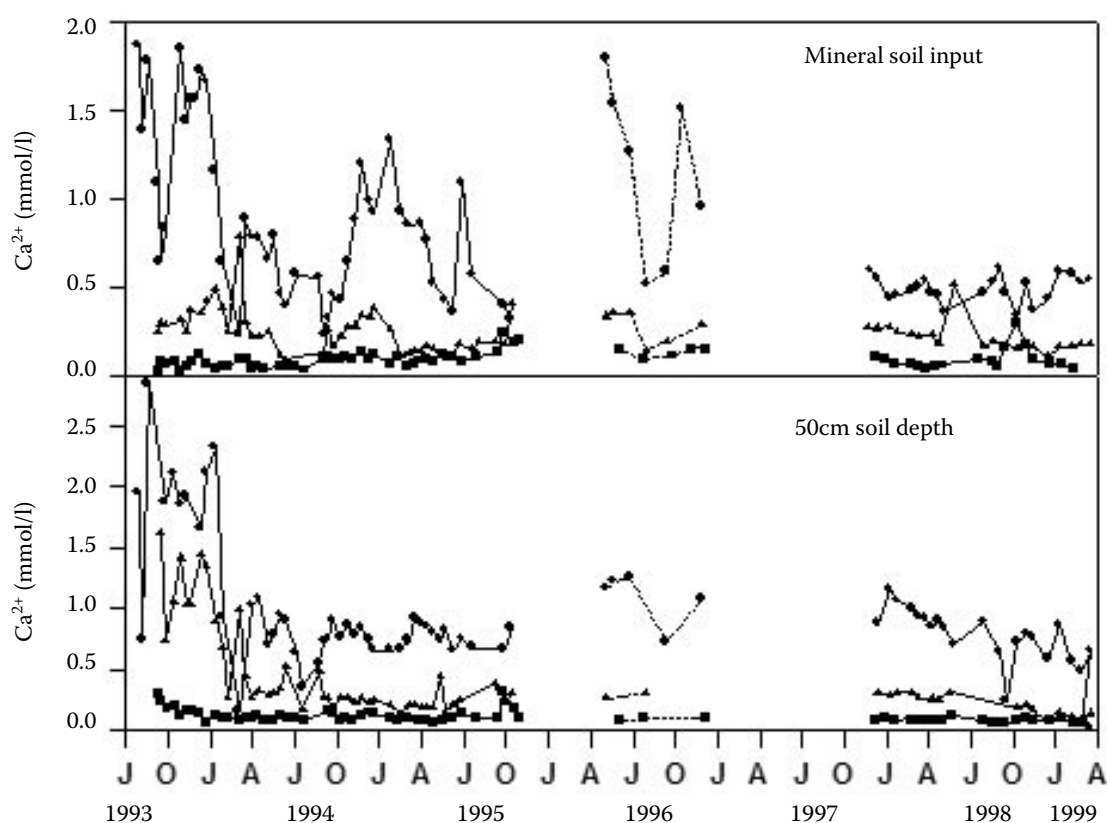

Fig. 8. Ca concentrations in soil solution concentrations the mineral soil input and in 50 cm soil depth at the three sites for the period July 1993 to April 1999

The soil properties at three sites along a gradient in pollution impact are clearly affected by the former deposition regimes. Especially the pH values, the base saturation data, and the total sulphur contents clearly reflect the gradient in former deposition with regard to the input of alkaline dust and sulphur (Table 1). The Ca concentration in soil solution at the site Rösa, closest to emission sources, are clearly elevated compared to the other sites (Fig. 8)

Whereas soil respiration and N mineralization measurements revealed no significant differences for the humus layers at the three sites, nitrification is clearly affected by the altered soil properties. At Rösa, almost all nitrogen is transformed to nitrate, at Taura ammonium and nitrate make up half each and at Neuglobsow all nitrogen is released as NH_4 (FISCHER et al. 1995). Incubation studies with undisturbed soil columns showed strong increase with rising temperature both in nitrate and sulphate release from the humus layer at Rösa. This potentially acidifying process is completely buffered by Ca. In contrast, at the other two sites mineralization of N and S lead to significant Al release from the top soil (SCHAAF et al. 2004).

CONCLUSION

Differences in forest ecosystem reaction to liming can be caused by amount, form and frequency of the application, and by the solubility, reaction kinetics, contents and composition of the material used. Furthermore, effects of the same treatment at different sites or different treatments at the same site may lead to quite different results due to the important influence of many other factors like climate, site history, management, environmental conditions (e.g. actual and historical deposition rates), humus form, C/N ratio, nitrogen stores, and the soil chemical properties (e.g. pH, base saturation, CEC). This finally leads to the conclusion that it is hardly possible to predict the effects of lime applications at sites where these factors are not known and it explains the variety of different findings of many experiments and studies.

Important to keep in mind for an evaluation of treatment effects is the factor "time". Most of the studies were carried out only over few years after application and, thus, cover only the initial effects of liming in forest ecosystems. Any evaluation has to take into account the objectives of the treatment, changes in environmental conditions, management options, socio-economic factors, and, last but not least, the balance between possible risk by the treatment and possible effects of not acting.

References

ALDINGER E., 1987. Elementgehalte in Boden und Nadeln verschieden stark geschädigter Fichten-Tannen-Bestände auf Praxiskalkungsflächen im Buntsandstein. Freiburger Bodenkundliche Abhandlungen, *19*: 266.

ANDERSSON F., PERSSON T. (eds.), 1988. Liming as a measure to improve soil and tree condition in areas affected by air pollution. National Swedish Environmental Protection Board, Report 3518: 131.

BEESE F., MEIWES K.J., 1995. Stand und Perspektiven. Allgemeine Forstzeitschrift, *17*: 946–949.

FEGER K., RASPE H., 2000. Verbundprojekt ARINUS im Schwarzwald: Auswirkungen von atmogenen Einträgen und Restabilisierungsmaßnahmen auf den Wasser- und Stoffhaushalt von Fichtenwäldern. In: FRÄNZLE O., MÜLLER F., SCHRÖDER W. (eds.), Handbuch der Umweltwissenschaften 2. Stuttgart, Ecomed, V-4.10: 1–55.

FISCHER T., BERGMANN C., HÜTTL R.F., 1995. Auswirkungen sich zeitlich ändernder Schadstoffdepositionen auf Prozesse des Kohlenstoff- und Stickstoffumsatzes im Boden. In: HÜTTL R.F., BELLMAN K., SEILER W. (eds.), Atmosphärensanierung und Waldökosysteme. Cottbus, Umweltwissenschaften, *4*: 143–160.

HILDEBRAND E.E., SCHACK-KIRCHNER H., 1990. Der Einfluß der Korngröße oberflächlich ausgebrachter Dolomite auf Lösungsverhalten und vertikale Wirkungstiefe. Forst- und Holzwirt, *45*: 139–142.

HÜTTL R.F., 1989. Liming and fertilization as mitigation tool in Germany's declining forests. Water, Air, and Soil Pollution, *44*: 93–118.

HÜTTL R.F., BELLMANN K. (eds.), 1998. Changes of Atmospheric Chemistry and Effects on Forest Ecosystems – A Roof Experiment without Roof. Nutrients in Ecosystems, 3. Dordrecht, Kluwer Academic Publishers: 324.

HÜTTL R.F., SCHAAF W. (eds.), 1997. Mg Deficiency in Forest Ecosystems. Nutrients in Ecosystems, 1. Dordrecht, Kluwer Academic Publishers: 362.

KAUPENJOHANN M., 1992. Mehrjährige Erfahrungen mit der Magnesiumdüngung in Waldökosystemen des Fichtelgebirges. In: GLATZEL G., JANDL R. (eds.), Magnesiummangel in mitteleuropäischen Waldökosystemen. Schriftenreihe Universität für Bodenkultur Wien, Band 5: 122–131.

KONOPATZKY A., FREYER C., 1998. The long-term change of soil status in pine forests in the Duebener Heide area as a model region for territories strongly influenced by industrial deposits in eastern Germany. In: HUETTL R.F., BELLMAN K. (eds.), Effects of Changes in Atmospheric Chemistry on Forest Ecosystems – a Case Study from East Germany. Nutrients in Ecosystems, 3. Dordrecht, Kluwer Academic Publishers: 324.

KREUTZER K., 1995. Effects of forest liming on soil processes. Plant and Soil, *168–169*: 447–470.

LUNDSTRÖM U.S., BAIN D.C., TAYLOR A.F.S., VAN HEES P.A.W., 2003. Effects of acidification and its mitigation with lime and wood ash on forest soil processes: a review. Water, Air, and Soil Pollution: Focus, *3*: 5–28.

MARSCHNER B., STAHR K., RENGER M., 1989. Potential hazards of lime application in a damaged pine forest ecosystem in Berlin, Germany. Water, Air, and Soil Pollution, *48*: 45–57.

MATZNER E., KHANNA P.K., MEIWES K.J., ULRICH B., 1985. Effects of fertilization and liming on the chemical soil conditions and element distribution in forest soils. Plant and Soil, *87*: 405–415.

MEYER M., 1992. Untersuchungen zur Restabilisierung geschädigter Waldökosysteme im norddeutschen Küstenraum (Fallstudie Wingst II). Berichte des Forschungszentrums Waldökosysteme, A94: 306.

MORAVČÍK P., CIENCIALA E., 2003. Overview of findings on mitigating the effects of immissions and soil acidifacation on forests. In: HRUŠKA J., CIENCIALA E. (eds.), Long-term acidification and nutrient degradation of forest soils – Limiting factors of forestry today. Prague, Czech Ministry of Environment: 38–60.

NAGEL C., PELK G., NIEHUS B., LOHS H., 1991. Atmosphärische Stoffeinträge und ihre Wirkungen auf den Naturhaushalt. In: Institut für Geographie und Geoökologie Leipzig (Hrsg.): Ausgewählte geoökologische Entwicklungsbedingungen Nordwest-Sachsens. Leipzig.

PERSSON T., LUNDKVIST H., WIREN A., HYVÖNEN R., WESSEN B., 1989. Effects of acidification and liming on carbon and nitrogen mineralization and soil organisms in mor humus. Water, Air, and Soil Pollution, *45*: 77–96.

PERSSON T., WIREN A., ANDERSSON S., 1991. Effects of liming on carbon and nitrogen mineralization in coniferous forests. Water, Air, and Soil Pollution, *54*: 351–364.

REITER H., BITTERSOHL J., SCHIERL R., KREUTZER K., 1986. Einfluss von saurer Beregnung und Kalkung auf austauschbare und gelöste Ionen im Boden. Forstwissenschaftliches Centralblatt, *105*: 300–309.

SCHAAF W., 1995. Effects of Mg(OH)$_2$-fertilization on nutrient cycling in a heavily damaged Norway spruce ecosystem (NE Bavaria/Germany). Plant and Soil, *168–169*: 505–511.

SCHAAF W., 1997. Evaluation of different magnesium fertilization strategies. In: HÜTTL R.F., SCHAAF W. (eds.), Mg Deficiency in Forest Ecosystems. Nutrients in Ecosystems, 1. Dordrecht, Kluwer Academic Publishers: 333–355.

SCHAAF W., ZECH W., 1991. Einfluss unterschiedlicher Löslichkeit von Düngern. Allgemeine Forstzeitschrift, *15*: 766–768.

SCHAAF W., WECKER B., TAO P., HÜTTL R.F., 2004. Changes in top soil properties of forest soils in north-eastern Germany due to long-term element accumulation. Plant and Soil, *264*: 85–95.

SCHÜLER G., 1991. Initial compensation of acidic deposition in forest ecosystems by different rock meals. Water, Air, and Soil Pollution, *54*: 435–444.

TÜRK T., 1992. Die Wasser- und Stoffdynamik in zwei unterschiedlich geschädigten Fichtenstandorten im Fichtelgebirge. Bayreuther Bodenkundliche Berichte, *22*: 203.

WENZEL B., 1989. Kalkungs- und Meliorationsexperimente im Solling: Initialeffekte auf Boden, Sickerwasser und Vegetation. Berichte des Forschungszentrum Waldökosysteme, A51: 274.

WIEDEY G.A., 1991. Ökosystemare Untersuchungen in zwei unterschiedlich exponierten Fichtenaltbeständen und in einem Kalkungs- und Düngungsversuch im Hils. Berichte des Forschungszentrums Waldökosysteme, A63: 205.

ZECH W., POPP E., 1983. Magnesiummangel, einer der Gründe für Fichten- und Tannensterben in NO-Bayern. Forstwissenschaftliches Centralblatt, *102*: 50–55.

Zkušenosti s vápněním v evropských zemích – výsledky dlouhodobých pokusů

W. Schaaf, R. H. Hüttl

Brandenburg University of Technology, Cottbus, Germany

ABSTRAKT: Vápnění se provádělo a stále se provádí v mnoha evropských zemích od počátku osmdesátých let minulého století. Hlavním cílem kompenzačního vápnění je neutralizovat aktuální kyselou depozici, aby se předešlo dalšímu zakyselování půdy, zvýšila se saturace bázemi a pufrační kapacita půdy. Práce vyhodnocuje výsledky dlouhodobých pokusů s vápněním v celé Evropě, při kterých se používala různá množství a formy materiálů k vápnění. Rozdíly v reakci lesních ekosystémů na vápnění mohou být způsobeny množstvím, formou a četností aplikace a rozpustností, kinetikou reakce, obsahem a skladbou použitého materiálu. Kromě toho ostatní faktory, jako je klima, historie stanoviště, hospodaření, podmínky prostředí (např. aktuální a minulá míra depozice), formy humusu, poměr C/N, zásoba dusíku a chemické vlastnosti

půdy (např. pH, saturace bází, CEC), mohou silně ovlivnit účinky ošetření vápnem. Bilance prvků lesních ekosystémů na ploše nebo v měřítku povodí je velmi užitečná pro integraci různých účinků dodatečného vstupu Mg cestou vápnění či hnojení.

Klíčová slova: kyselá depozice; okyselení půdy; bilance prvků; vápno; dolomit; hnojení; pH půdy; saturace bázemi; pufrační kapacita; historie stanoviště

Vápnění se provádělo a ještě se provádí v mnoha evropských zemích od začátku devadesátých let. Důvody a cíle vápnění v lesních ekosystémech mohou být rozmanité. Hlavním cílem kompenzačního vápnění je neutralizovat aktuální kyselou depozici za účelem zabránění další acidifikace půdy, zvýšení saturace bázemi a pufrační kapacity půdy a redukce možných toxických nebo antagonistických účinků zvýšené koncentrace hliníku v okyselených půdách. Další ošetření vápněním, často zvané meliorační vápnění, je zaměřené hlavně na zlepšení forem humusu, na doplňování zásob živin ztracených buď vyluhováním, nebo těžbou biomasy, na prohloubení kořenového systému nebo se provádí jako meliorace stanoviště pro usnadnění přirozené nebo umělé obnovy či transformace lesních porostů.

Rozdíly v reakci lesních ekosystémů na vápnění mohou být způsobeny množstvím, formou a četností aplikace a dále také rozpustností, reakční kinetikou, obsahem a složením použitého materiálu. Kromě toho vliv stejného ošetření na různých stanovištích nebo různého ošetření na stejném stanovišti může vést ke zcela rozdílným výsledkům vlivem mnoha dalších faktorů, jako je klima, historie stanoviště, hospodaření, podmínky prostředí (např. aktuální a historické míry depozice), formy humusu, poměr C/N, zásoba dusíku a chemické vlastnosti půdy (např. pH, saturace bázemi, výměnná kapacita kationtů). Bilance prvků v lesních ekosystémech v měřítku plochy nebo povodí je velice užitečná pro integraci různých vlivů dodatečného vstupu Mg cestou vápnění či hnojení. To nakonec vede k závěru, že je sotva možné předpovědět vliv aplikace vápna na stanovištích, kde tyto faktory nejsou známy, a to vysvětluje rozmanitost různých poznatků z mnoha experimentů a studií.

Pro vyhodnocení účinků ošetření je důležité mít na mysli faktor „čas". Většina studií byla prováděna pouze několik málo let po aplikaci, a proto zachycuje pouze počáteční vlivy vápnění na lesní ekosystémy. Jakékoliv hodnocení musí brát v úvahu cíle ošetření, změny podmínek prostředí, alternativy hospodaření, společensko-ekonomické faktory a konečně rovnováhu mezi možnými riziky vlivem ošetření a možnými vlivy absence působení.

Corresponding author:

PD Dr. rer. nat. habil. Wolfgang Schaaf, Chair of Soil Protection and Recultivation, Brandenburg University of Technology, Konrad-Wachsmann-Allee 6, D-03044 Cottbus, Germany
tel.: + 49 355 694 240, fax: + 49 355 692 323, e-mail: schaaf@tu-cottbus.de