



Spatial and temporal variation of seepage water chemistry after femel and small scale clear-cutting in a N-saturated Norway spruce stand

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Received 18 November 2003. Accepted in revised form 5 March 2004

Key words: aluminium, clear-cutting, nitrate, nutrient leaching, potassium, selective-cutting

Abstract

The chemistry of seepage water was studied before and after small scale clear-cutting and femel cutting (removing 20% of the trees) between 1999 and 2002 at the Höglwald site in southern Bavaria. The interventions were performed in February 2000 on mature, N-saturated Norway spruce (*Picea abies* (L.) Karst.) stands with high NO_3^- concentrations before felling. Seepage water was collected with suction cups at 40 cm soil depth in the following treatments: (I) a mature stand (control), (II) a femel-cut, and (III) a clear-cut. In the femel cut subvariants were created with suction cups (plots) at varying distances from pre-selected spruce, which were later removed. The femel treatment was replanted with beech (*Fagus sylvatica* L.) saplings. On the clear-cut, subvariants of planted beech (close to the stem, interstem area), planted spruce (interstem), or natural spruce regeneration were investigated. Clear-cutting caused high NO_3^- peaks (average values up to 2750 μM) during 2000 and 2001 in all planted tree subvariants during times of comparatively low water fluxes. In contrast to peak concentrations, flux weighted yearly average concentrations showed different trends. In 2000, flux weighted yearly average NO_3^- concentrations were significantly elevated, but only on the subvariants of the interstem area, which covered in the clear-cut plot ca. 62% of the area. However, the subvariant close to the stem (31% of clear-cut area), or the natural spruce regeneration subvariant (6% of clear-cut area) exhibited no significant felling effect. With respect to the whole treatment area, this resulted in no significant felling effect as compared with the control. In the next year (2001), flux weighted yearly average NO_3^- concentrations were not significantly affected by clear-cutting, while the concentrations were even reduced for all of the clear-cut subvariants in 2002. On the subvariant natural spruce regeneration, NO_3^- concentrations remained below the European limit of drinking water (806 μM) during almost the whole investigation period. Selective cutting resulted in slightly reduced NO_3^- concentrations in 2000 and 2001 on the femel treatment. However, no significant effect could be detected for any subvariant in the femel-cut, even not for the subvariant with suction cups closest to the felled spruce. In contrast to many other investigations, clear-cutting did not increase the NO_3^- problem on the treatment to a relevant extend. Quite contrary, a decline in NO_3^- concentrations to values below the EU level for drinking water and levels below the control and femel treatment just 2 years after cutting were observed. Al^{3+} concentrations showed nearly the same trend as NO_3^- , while Ca^{2+} , Mg^{2+} , and K^+ concentrations were affected to a lesser degree. Only in 2002 was Ca^{2+} significantly lower on the clear-cut as compared to the femel treatment, but not compared to the control. Mg^{2+} increased in 2000 on the clear-cut subvariants in the interstem area, but decreased in the years 2001 and 2002. Changes could be observed for K^+ only periodically on some subvariants, but not for the whole treatment area. Concentrations of SO_4^{2-} , Na^+ , and Cl^- were reduced after clear-cutting and remained nearly unchanged after femel cutting.

Introduction

Since the 19th century managed, Norway spruce (*Picea abies* (L.) Karst.) forests replaced in wide areas

of Southern Bavaria the often degraded forests, which resulted from exploitation of the formerly naturally occurring beech forests. Today the spruce stands in this region are well known for their productivity (Pretzsch, 1996; Röhle, 1991), and their profitability is usually

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higher than that of beech (*Fagus sylvatica* L.) forests (Kenk and Guehne, 2001). However, in view of the fact that besides wood production, sustainability and non-timber values are becoming increasingly important (Lindenmayer et al., 2000; United Nations, 1992), concern is rising about negative ecological impacts of these uniform Norway spruce stands. These even-aged stands are highly susceptible to snow and wind, as well as insect outbreaks (Kenk and Guehne, 2001), and appear to have a low value for nature conservation. In a long-term study at Höglwald, Southern Bavaria, which is exposed to high loads of nitrogen (Huber and Kreutzer, 2002; Huber et al., 2002; Rothe et al., 2002), NO_3^- concentrations in seepage water exceeded the European level of drinking water at the spruce stand for most of the study (Kreutzer, 1995; Rothe et al., 2002). Additionally, unexpectedly high N_2O emissions from the soils were reported at the Höglwald site (Butterbach-Bahl et al., 1998, 2002). In contrast, a nearby European beech stand exhibited very low NO_3^- concentrations, mostly due to the lower N- input via throughfall (Rothe et al., 2002).

Having recognised these problems, one goal of the official forest management administration in Bavaria is to gradually transform many of the pure coniferous stands in Southern Bavaria into mixed forests. Two methods are widely used to regenerate these forests: (1) the Bavarian State Forest Administration is regenerating the stands with selective cutting of trees (femel cutting). (2) Small scale clear-cuts (around 0.5 to 1 ha) are made if there is a high risk of wind throw, or by farmers in private forests, because of lower costs and easy practicability. At the Höglwald site, we investigated the impacts of regeneration with different ecological foci: emissions of greenhouse gases, diversity of ground vegetation and fauna (Huber and Baumgarten, 2004), seepage water quality, and post harvest nutrient losses (first results in Weis et al., 2001). We investigated the effects of a femel-cut (regenerated with beech saplings) and a small scale clear-cut (regeneration with beech or spruce saplings, or naturally regenerated spruce). Pre-treatment samples were collected in 1999 and felling was conducted in February 2000.

After clear-cutting the removal of the canopy leads to abrupt changes in microclimate, throughfall input, evapo-transpiration, nutrient uptake, and litter input (Robertson et al., 2000). After the cutting, mineralisation 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). However, selective cutting is widely believed to reduce the negative effects during the regeneration, due to the shading effect of the remaining stand and to the successive nutrient-uptake of the newly established regeneration (natural or planted).

In this paper, we present the influence of clear-cutting and the first stage in the femel cutting system (removing 20% of the canopy with subsequent planting of beech saplings) on seepage water quality for the first four years of the study. Our hypotheses are:

(a) Clear-cutting causes drastically elevated NO_3^- concentrations for 2–3 years, with higher maximum, and elevated yearly average concentrations (flux weighted) compared to control, and femel-cutting.

(b) There is no difference in the elemental concentrations after clear-cutting whether the planted species is spruce, or beech. However, a dense natural regeneration (spruce trees from seedlings grown up under the old stand) before cutting may reduce the leaching of NO_3^- and cations.

(c) Femel-cutting will cause an increase in NO_3^- on subvariants close to felled spruce with lower intensity compared to the clear-cut. This effect will diminish on subplots with increasing mature stand character (subplots with increasing distance from the felled tree). This will result in no significant differences on a treatment area scale. (d) The higher NO_3^- concentrations after cutting are the driving force for an increased leaching of the accompanying charge balancing cations Al^{3+} , K^+ , Ca^{2+} , and Mg^{2+} following cation exchange processes in the mineral soil.

(e) After the felling the water fluxes are increased (lower interception) while the elemental input decreases with lower rates of dry deposition. This results in a decrease of the ion concentrations (SO_4^{2-} , Cl^- , and Na^+), which are less affected by mineralisation or cation exchange processes.

Materials and methods

Site description

The Höglwald site is a long-term ecological monitoring and experimentation site. The research at the site started in 1983, with the focus on biogeochemistry and ecosystem research. For more details about the stand and site characteristics see Kreutzer and Weiss (1998). The forest district of the Höglwald (370 ha) is situated in the hilly landscape of Southern Bavaria about 70 km

north of the Alps and 50 km west of Munich (center) at 11°11' E and 48°30' N. The forest is situated on a flat hilltop at an altitude of 540 m above sea level, surrounded by intensively managed farmland (cattle breeding, dairy, and maize). The proportion of agricultural land to forest land (predominantly small conifer forests with Norway spruce (*Picea abies* (L.) Karst) is approx. 2:1 (Kreutzer and Weiss, 1998). The climate is suboceanic. The region belongs to the temperate broad-leaf zone, originally dominated by beech (*Fagus sylvatica* L.). For the period 1984 to 2001 mean annual precipitation at the Höglwald forest was 933 mm, mean annual temperature 7.7 °C, mean difference between the warmest and the coldest month 17.5 °C, and the mean number of days exceeding 10 °C mean temperature was 155. During the observation period from 1999 to 2001, the climatic data differed from the long-term means. The mean temperature in this period was 8.2 °C and the mean bulk precipitation amounted to 1161 mm on average. The soil is a Parabraun Earth (Central European System) (USGS: Typic Hapludalf; FAO: Dystric Cambisol), strongly acidified in the topsoil and weakly aquic in the argillic horizon. The mineral soil is covered by an organic layer 6 to 8 cm thick. The humus form is moder. In the mineral soil, no coarse material (> 2 mm) is present. In the organic layer the base saturation of the cation exchange capacity is relatively high (40 to 80%), whereas the pH values are extremely low with a minimum in the Oh horizon of 2.75 (KCl). The base saturation in the A horizon (0–40 cm depth) is low (5–10%) and the Al-saturation is high (80–90%), with pH values of around 4.0 to 4.5 (KCl). In the B horizon (40–130 cm depth) the base saturation increases with depth as well as pH values. The C/N ratio varies in the forest floor between 23 and 27. The site is nitrogen saturated according to the definition of Ågren and Bosatta (1988). Throughfall input is 30 kg N ha⁻¹ y⁻¹ and more (Rothe et al., 2002), with additional input of dry deposition of N-gases like NO₂ (Butterbach-Bahl et al., 2002) and NH₃ (Huber et al., 2002) to the soil and via the canopy (Geßler et al., 2002). NO₃⁻ leaching via seepage with approx. 30 kg N ha⁻¹ y⁻¹ (Rothe et al., 2002), and the emissions of N-trace gases of approx. 8 kg N ha⁻¹ y⁻¹ in form of N₂O and NO_x (Butterbach-Bahl et al., 2002) is high. The investigated stands are mature Norway spruce plantations, the second generation after beech, cultivated in 1910/11 with three or four-year-old plants. The natural vegetation type is classified as submontane *Asperulo-Fagetum luzuletosum*, an acidophytic community transient to *Luzulo-Fagetum oxal-*

idetosum. The spruce stand is growing vigorously, is full-stocked, and has a closed canopy. According to inventories made in 1995 the basal area of the stand was 77.2 m² ha⁻¹ and growing stock was 1269 m³ ha⁻¹ with a mean current increment from 1992 to 1995 of approx. 20 m³ ha⁻¹ y⁻¹. The last thinning was performed in 1975. The ground vegetation is dominated by mosses (mostly *Thuidium tamariscinum*).

Experimental design

In 1999, the equipment was installed in an area with uniform stand and site conditions. Felling was performed with a harvester at the end of February 2000. The pre-treatment period before felling lasted from June 1999 to February 2000. Stems including bark were removed, and slash remained on the treatment sites. The small scale clear-cut (1.0 ha) was divided into one part regenerated with beech and into one part regenerated with spruce. 5-year old trees were planted in April 2000. The femel treatment (0.9 ha) was selectively cut by removing 20% of the basal area of the stand. The fellings were nearly equally distributed over the whole area to retain the stability of the stand and avoid instability by gap formation. The whole area was planted with five year old beech saplings.

The following treatment (variant) abbreviations are used throughout the paper:

- C control (mature Norway spruce)
- F femel (selective-cutting), regeneration with planted beech saplings
- CC clear-cut, with planted beech, planted spruce, or natural spruce regeneration

The experimental treatments are separated by game fences and small utility roads. The femel treatment is situated in the centre of the three treatments, 250 m NE is the control, and 150 m SE the clear-cut treatment.

Throughfall fluxes (ion, and water fluxes) of the different variants are presented in Table 1. The interventions cause an increase in the heterogeneity of the stand, considered in the experimental set-up with different subvariants with randomly distributed plots with suction cups of 1 m² area.

We investigated 10 plots at the control site with one suction cup each. The number of plots on the subvariants of the femel treatment and the clear-cut treatment are listed below. Subvariants with their abbreviations on the femel-cut treatment:

Table 1. Troughfall fluxes of water, and ions on the control (C), felled plot (F), and clear-cut (CC) treatment in the years 1999 (June to December), 2000, 2001, and 2002. *Note that fluxes in 1999 were from a seven month period

Year	Plot	Flux l m^{-2}	NH_4^+	Ca^{2+}	Mg^{2+}	K^+ $\text{mmol}_c \text{ m}^{-2}$	Na^+	NO_3^-	SO_4^{2-}	Cl^-
1999*	C	335	60	17	8	30	4	27	31	13
	F	315	52	13	6	25	6	21	28	11
	CC	314	53	13	6	27	5	23	30	11
2000	C	644	134	33	14	44	11	81	51	22
	F	714	102	30	12	39	9	71	41	19
	CC	1058	64	18	4	8	6	42	35	10
2001	C	659	99	29	13	56	14	50	48	28
	F	740	78	29	11	56	14	46	39	26
	CC	1170	48	17	3	10	13	34	24	13
2002	C	805	105	35	16	52	15	80	44	25
	F	868	95	36	16	54	17	80	44	25
	CC	1227	42	16	4	13	15	32	25	12

- 'f < 1.5 m': suction cups (SC) close to the stem, < 1.5 m from felled tree (5 plots)
- 'f 1.5–3 m': SC in the interstem area, 1.5–3 m from felled tree (5 plots)
- 'f 3–4.5 m': SC in the interstem area, 3–4.5 m from felled tree (6 plots)
- 'f 4.5–6 m': SC in the interstem area, 4.5–6 m from felled tree (6 plots)

Subvariants with their abbreviation on the clear-cut treatment:

- 'ccb > 1.5 m': SC in the interstem area, (> 1.5 m from the stem) planted beech (5 plots)
- 'ccb < 1.5 m': SC close to the stem, (< 1.5 m from the stem) planted beech (5 plots)
- 'ccs > 1.5 m': SC in the interstem area, planted spruce (5 plots)
- 'ccn': SC in the natural spruce area (5 plots)

Sampling and chemical analysis

Seepage water was sampled below the main rooting zone, with suction cups (tension ceramic lysimeters, SKF 100, Haldenwanger GmbH, Berlin). These soil water samplers were installed vertically, in hand-augered holes (40 cm depth from the top of the mineral soil). Disturbance was kept to a minimum during installation by use of mats, and when complete, access to the sampler areas was restricted. The samples were taken at intervals ranging from 10 to 30 days. We used a self-made portable 12 V vacuum pump to partially evacuate glass bottles attached to the samplers.

A vacuum suction of approximately 60 kPa is applied for seven days prior to sample collection. The glass bottles are protected against light by plastic buckets. The samples were collected within a week after tension was applied. pH was measured with an Ingold glass-electrode on unfiltered samples. Afterwards water samples were filtered using membrane filters with a pore size of 0.45 μm (Schleicher and Schuell, NC45) and stored at 4 °C until they were analysed. Aluminium (Al), calcium (Ca), iron (Fe), magnesium (Mg), sodium (Na), and potassium (K) were analysed with an ICP (Perkin Elmer, Optima 3000). Chloride (Cl^-), nitrate (NO_3^-), and sulphate (SO_4^{2-}) were determined with ion-chromatography (Dionex, IC2020I). Meteorological standard data were available from the German Weather Service from a nearby station (Augsburg-Mühlhausen). Supplementary climatic measurements were taken on each plot at hourly intervals for precipitation, air temperature, relative humidity, and photoactive radiation (PAR).

Determination of the water fluxes

Water fluxes were calculated for the estimation of the flux weighted yearly average concentrations of the corresponding variant or subvariant. The fluxes were estimated with a mechanistic model (Weis et al., 2001), which uses hourly values of bulk precipitation, air temperature, relative humidity, and wind speed as input data. It includes up to three different vegetation layers (mature stand, regeneration, and ground vegetation). Stand temperature, humidity and wind speed

is calculated for each vegetation layer. The evaporation rate is calculated from vapour pressure deficit (VPD) and multiplied by a specific evaporating surface to calculate interception loss, transpiration, and soil evaporation. Concerning interception loss, the evaporating surface is the wet leaf and bark surface. A leaf area index of 4.9 for the control and 4.5 for the femel variant was estimated for the site. Using a form factor of 1.25, we get a total needle surface of 12.25 and 11.25 m² per m² soil surface. The leaf surface of the moss layer was estimated with 2 m² per m² soil surface. The ground vegetation at the clear-cut was characterised using a leaf surface of 2, 3, and 4 m² per m² soil surface for the years 2000, 2001, and 2002 (see also Breuer et al., 2003; Asner et al., 2003). The evaporating surface for transpiration is calculated from leave surface, the species specific amount and distribution of stomata per leaf area, and the surface of the leaf mesophyll cells surrounding the sub-stomatal cavity as proposed by Kramer and Boyer (1995). Foliation and leaf senescence are modelled using a sigmoid function. The transpiration loss in the soil follows the fine root distribution, with 10% in the organic layer, 60% in the first 20 cm of the mineral soil, 20% between 20 and 40 cm, 5% between 40 and 60 cm, 3% between 60 and 80 cm, and 2% between 80 and 100 cm. Rooting of mosses was restricted in the model to the organic layer, and ground vegetation was assumed to have equal amounts of roots in the organic layer and the first 20 cm of the mineral soil. Soil water fluxes for the organic layer and the soil depth 20, 40, 60, 80, 100, and 120 cm were calculated using the Darcy-Richards equation (Richards, 1931). For every soil layer and each of the three variants the relationship between soil matrix potential, soil water content, and soil hydraulic conductivity were determined at undisturbed soil cores (\varnothing 15 cm, height 20 cm) with a set of one TDR-probe (Trime P2, IMKO-GmbH, Ettlingen, Germany) and two tensiometers (miniature pressure-transducer tensiometer T5, UMS GmbH, Munich, Germany) installed horizontally in the core. Modelled matrix potential and water content were compared to measured values. Hourly measurements have been carried out since 2000 with horizontally installed TDR-Probes (Trime-P2, rod length 110 mm, -P3 or -EZ, rod length 160 mm, IMKO GmbH, Ettlingen, Germany) and tensiometers (pressure-transducer tensiometer T4, UMS GmbH, Munich, Germany). At the control variant, one replicate each was located in the organic layer (TDR only), at 10, 30, 50, 70, and 90 cm soil depth. Four replicates each were positioned in the organic layer, at

10, and 30 cm at the femel and clear-cut treatment. For all three treatments (control, femel, and clear-cut), the time courses were simulated sufficiently well.

Calculations and statistical analysis

Flux weighted yearly elemental concentrations were calculated by division of the yearly elemental fluxes by yearly water fluxes.

Calculation of mean flux weighted yearly average concentrations on a plot scale were a compromise between the area covered de facto by each subvariant on the treatment, operating expense for the suction cups, and practicability of statistical tests. Dependent on the femel cutting strategy, the percentage of each of the four femel subvariants on the total femel treatment can differ considerably. In our case, where the felled trees were nearly equally distributed over the whole area to retain the stability of the stand (not nested to form a gap), a good approximation to reality was to calculate mean average concentrations including all suction cups (22 plots) of the four femel subvariants without weighting one subvariant. On the clear-cut treatment mean average concentrations were calculated including five plots each at the subvariants 'ccb>1.5 m', 'ccb<1.5 m', 'ccs>1.5 m', as well as the average concentration from 'ccn' (in total 16 values). The values of the subvariant 'ccb<1.5 m' which represents the area close to the stem in the beech part of the clear-cut were also used in the calculations for the area close to the stem in the spruce part of the clear-cut. The lower weight of the natural spruce regeneration area ('ccn') with 1 of 16 suction cups takes into account that ca. 6% of the whole clear-cut area is covered by natural spruce regeneration.

Statistical calculations were done using the statistic package SPSS 11.5 for Windows. In order to detect a possible felling effect, a covariant analysis was performed. Before the felling, we investigated each suction cup in a pre-treatment period from June 1999 to February 2000 to acquire information about the spatial heterogeneity of each treatment. Within the pre-treatment period all seasons could be included to also consider potential inter-annual variability. For each element and each suction-cup a mean flux weighted pre-treatment concentration was calculated. This concentration was used as covariate for the respective suction cup in covariant analysis of the (sub-) variants. In a first analysis, all subvariants were tested against the control; in a second, the variants control, femel, and clear-cut were tested among each other. A

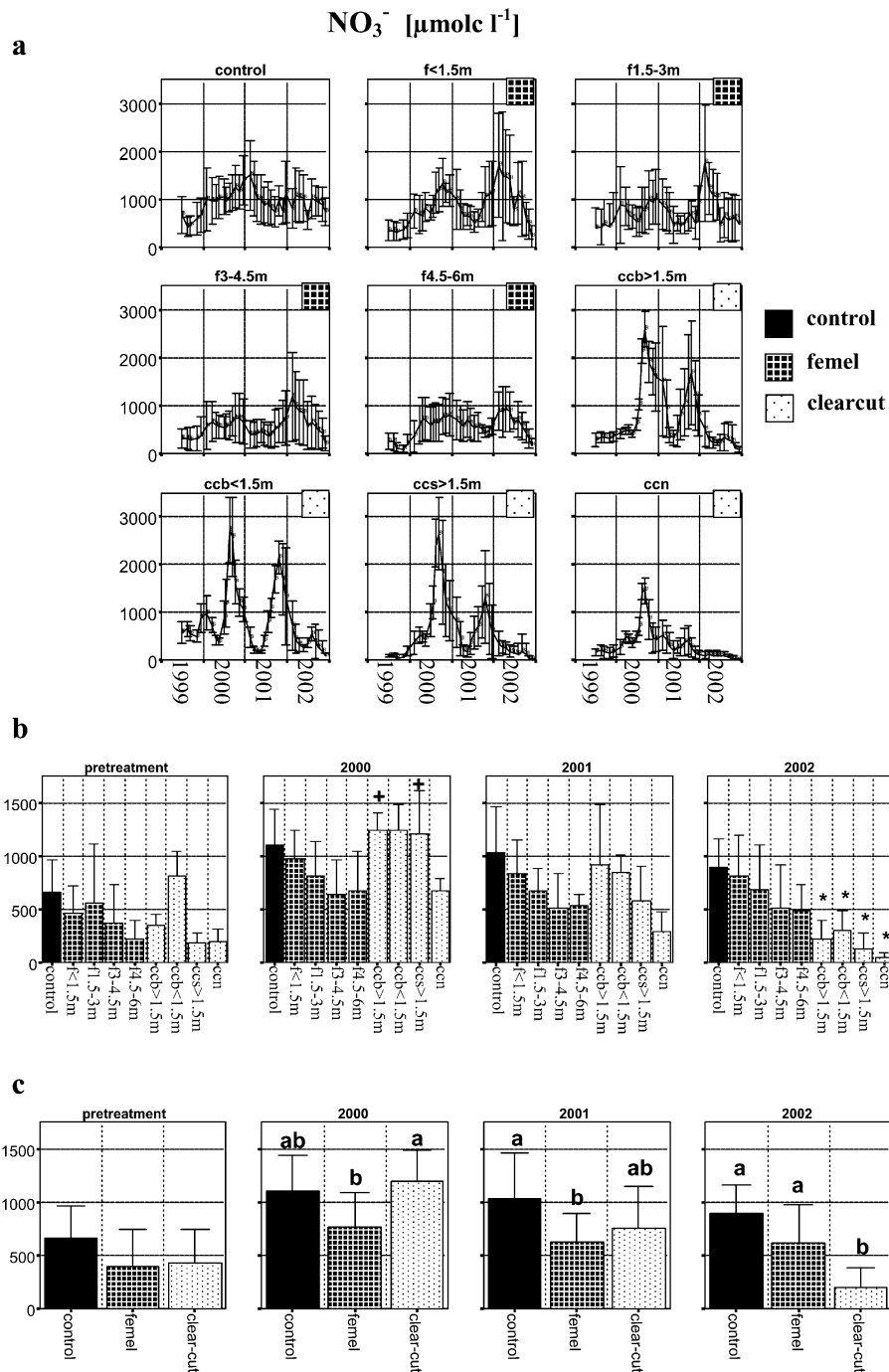


Figure 1. (a) Mean NO_3^- concentrations (\pm standard deviation) on control and for the different subvariants on the felm, and clear-cut treatment. Subvariants on the felm treatment differ in distance from a felled tree with: a) 'f<1.5 m': less than 1.5 m; b) 'f1.5–3 m': 1.5–3 m; c) 'f3–4.5 m': 3–4.5 m; d) 'f4.5–6 m': 4.5–6 distance. Subvariants on the clear-cut treatment: 'ccb>1.5 m': distant from the stem, planted with beech; 'ccb<1.5 m': close to the stem; 'ccs>1.5 m': distant from the stem, planted with spruce; 'ccn': natural spruce. (b) Mean flux weighted yearly NO_3^- concentrations on the control, and for different subvariants on the felm, and clear-cut treatment. '+' indicates a significant increase in the concentration due to felling; '*' indicates a significant decrease in the concentration due to felling (analysis of covariance, $P < 0.05$). (c) Mean flux weighted yearly NO_3^- concentrations on the control, felm, and clear-cut treatment. Significant differences due to felling ($P < 0.05$) between the different treatments tested are indicated by different indices (a, b, c).

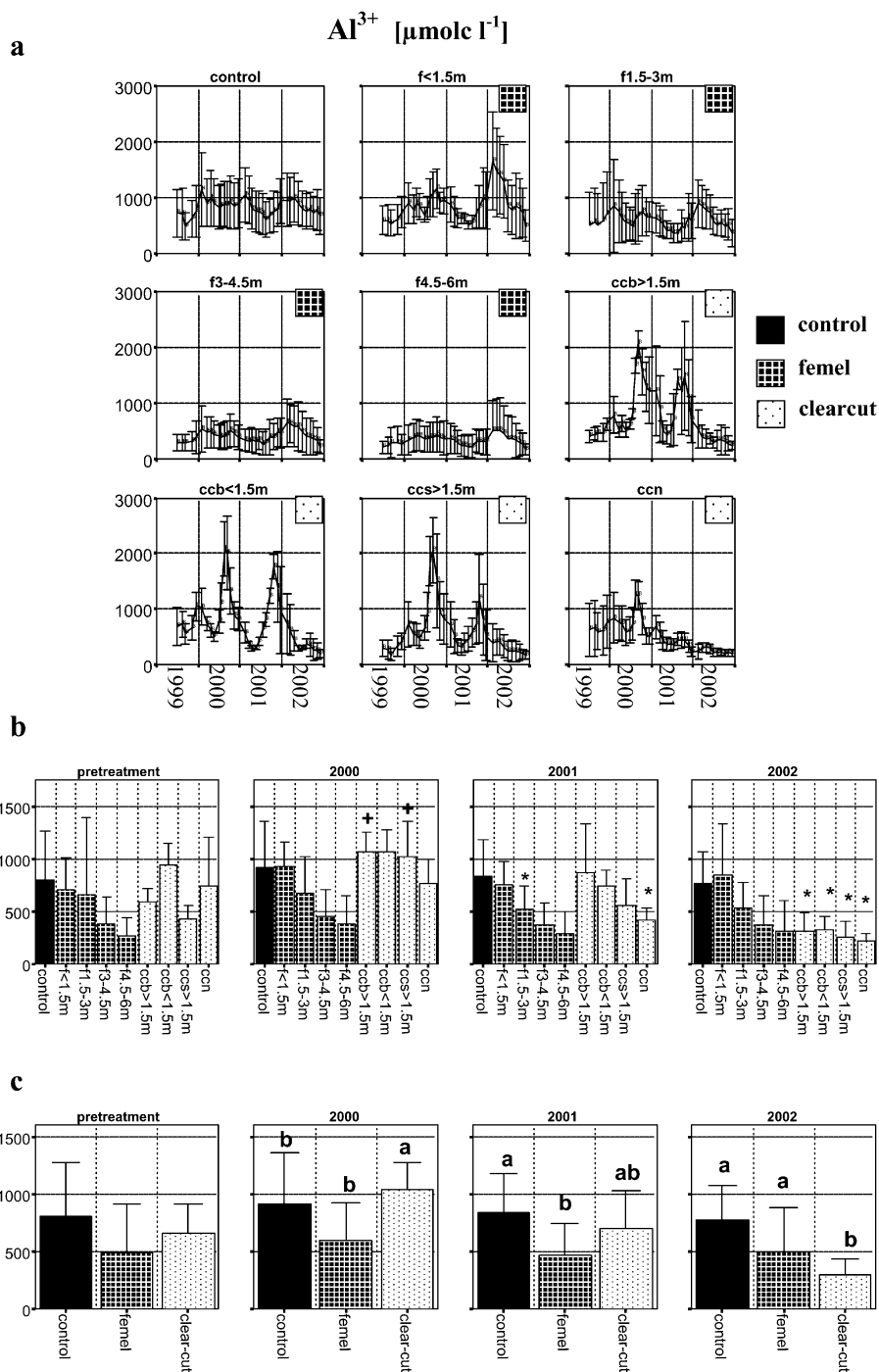


Figure 2. (a) Mean Al concentrations (\pm standard deviation) on the control and for the different subvariants on the felmed, and clear-cut treatment. Subvariants on the felmed treatment differ in distance from a felled tree with: a) 'f<1.5 m': less than 1.5 m; b) 'f 1.5–3 m': 1.5–3 m; c) 'f 3–4.5 m': 3–4.5 m; d) 'f 4.5–6 m': 4.5–6 distance. Subvariants on the clear-cut treatment: 'ccb>1.5 m': distant from the stem, planted with beech; 'ccb<1.5 m': close to the stem; 'ccs>1.5 m': distant from the stem, planted with spruce; 'ccn': natural spruce. (b) Mean flux weighted yearly Al concentrations on the control, and for different subvariants on the felmed, and clear-cut treatment. '+' indicates a significant increase in the concentration due to felling; '*' indicates a significant decrease in the concentration due to felling (analysis of covariance, $P < 0.05$). (c) Mean flux weighted yearly Al concentrations on the control, felmed, and clear-cut treatment. Significant differences due to felling ($P < 0.05$) between the different treatments tested are indicated by different indices (a, b, c).

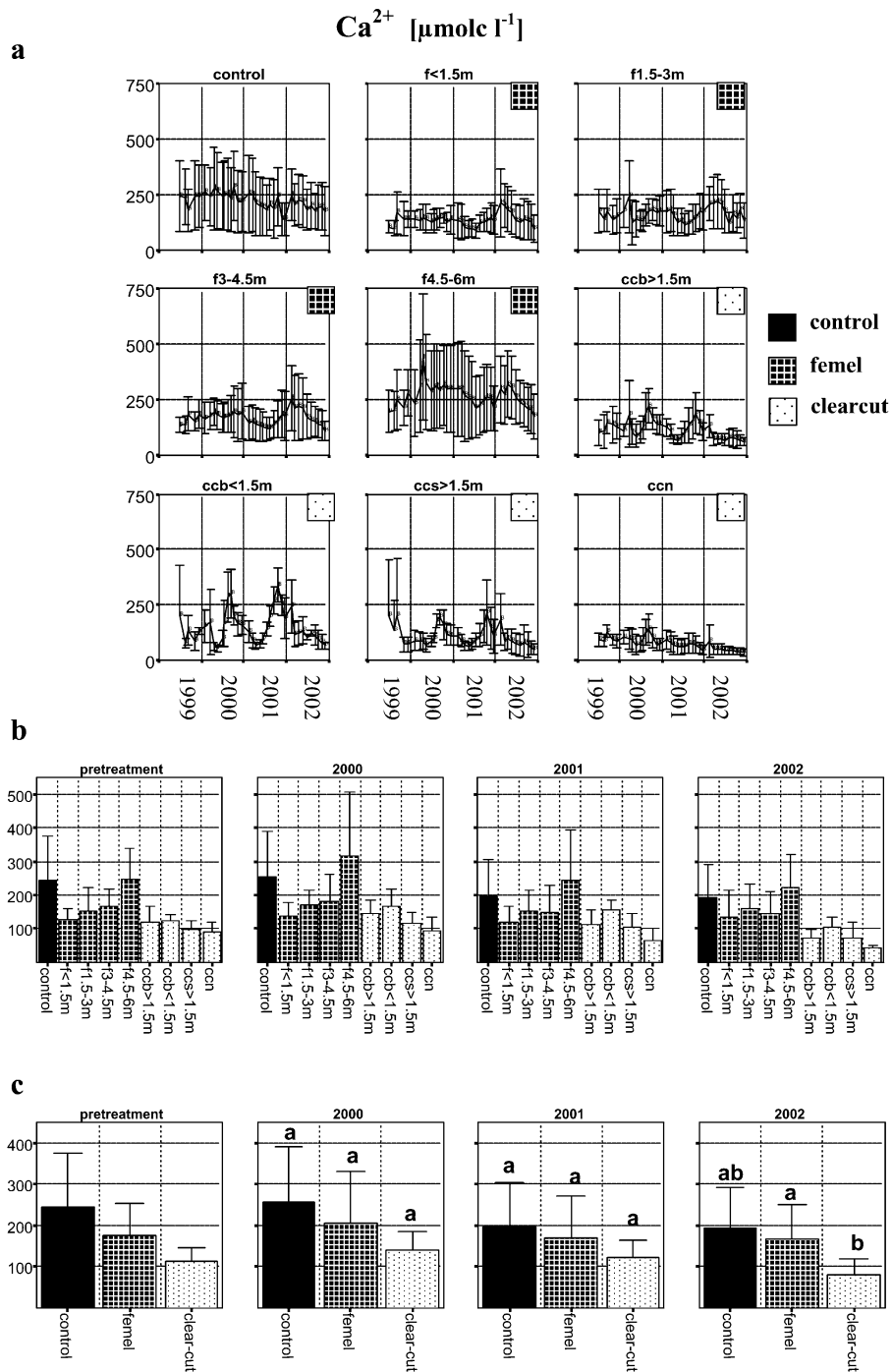


Figure 3. (a) Mean Ca^{2+} concentrations (\pm standard deviation) on the control and for the different subvariants on the femel, and clear-cut treatment. Subvariants on the femel treatment differ in distance from a felled tree with: a) 'f<1.5 m': less than 1.5 m; b) 'f 1.5–3 m': 1.5–3 m; c) 'f 3–4.5 m': 3–4.5 m; d) 'f 4.5–6 m': 4.5–6 distance. Subvariants on the clear-cut treatment: 'ccb>1.5 m': distant from the stem, planted with beech; 'ccb<1.5 m': close to the stem; 'ccs>1.5 m': distant from the stem, planted with spruce; 'ccn': natural spruce. (b) Mean flux weighted yearly Ca^{2+} concentrations on the control, and for different subvariants on the femel, and clear-cut treatment. '+' indicates a significant increase in the concentration due to felling; '*' indicates a significant decrease in the concentration due to felling (analysis of covariance, $P < 0.05$). (c) Mean flux weighted yearly Ca^{2+} concentrations on the control, femel, and clear-cut treatment. Significant differences due to felling ($P < 0.05$) between the different treatments tested are indicated by different indices (a, b, c).

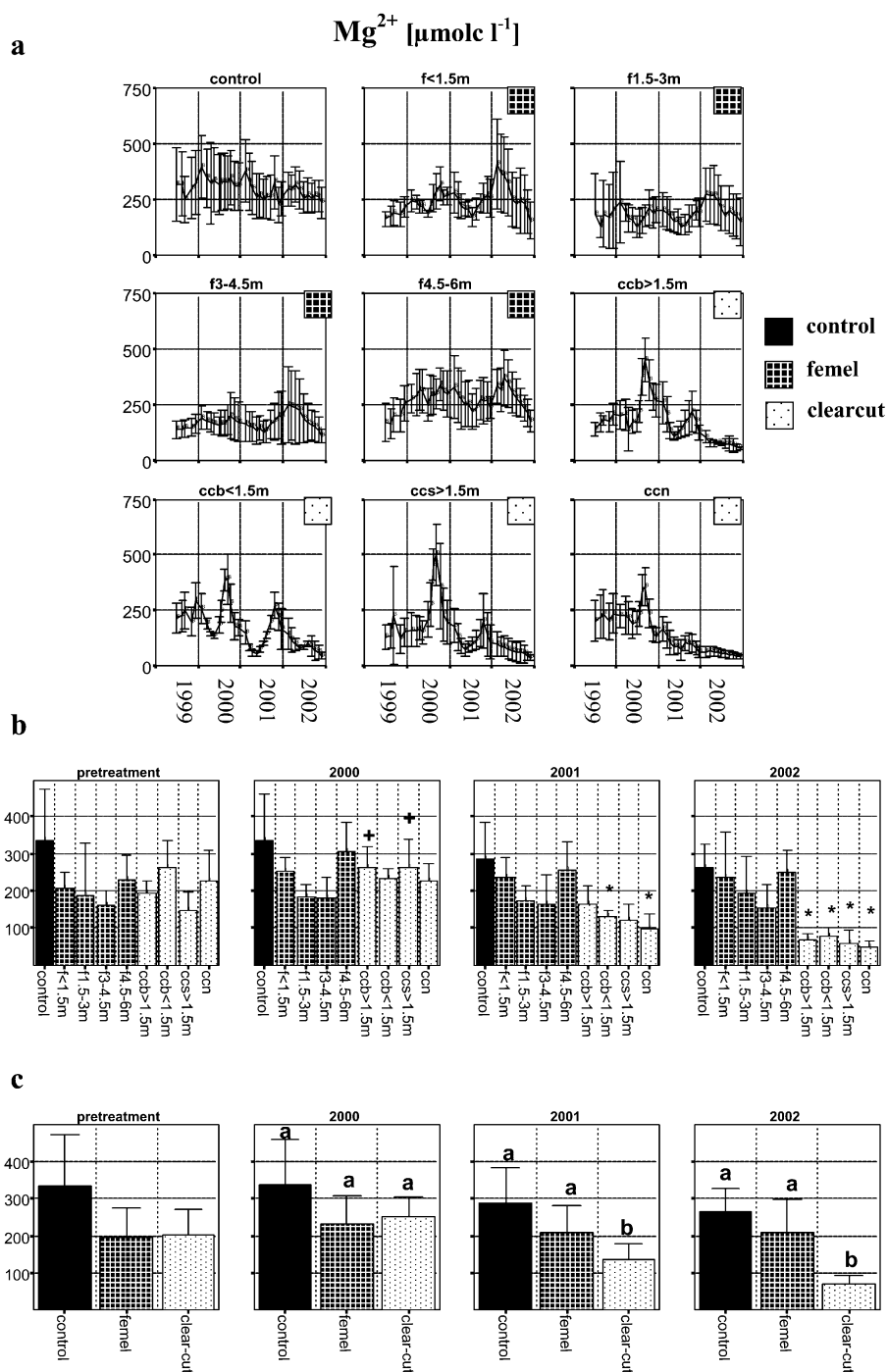


Figure 4. (a) Mean Mg^{2+} concentrations (\pm standard deviation) on the control and for the different subvariants on the felmel, and clear-cut treatment. Subvariants on the felmel treatment differ in distance from a felled tree with: a) 'f<1.5 m': less than 1.5 m; b) 'f 1.5–3 m': 1.5–3 m; c) 'f 3–4.5 m': 3–4.5 m; d) 'f 4.5–6 m': 4.5–6 distance. Subvariants on the clear-cut treatment: 'ccb>1.5 m': distant from the stem, planted with beech; 'ccb<1.5 m': close to the stem; 'ccs>1.5 m': distant from the stem, planted with spruce; 'ccn': natural spruce. (b) Mean flux weighted yearly Mg^{2+} concentrations on the control, and for different subvariants on the felmel, and clear-cut treatment. '+' indicates a significant increase in the concentration due to felling; '*' indicates a significant decrease in the concentration due to felling (analysis of covariance, $P < 0.05$). (c) Mean flux weighted yearly Mg^{2+} concentrations on the control, felmel, and clear-cut treatment. Significant differences due to felling ($P < 0.05$) between the different treatments tested are indicated by different indices (a, b, c).

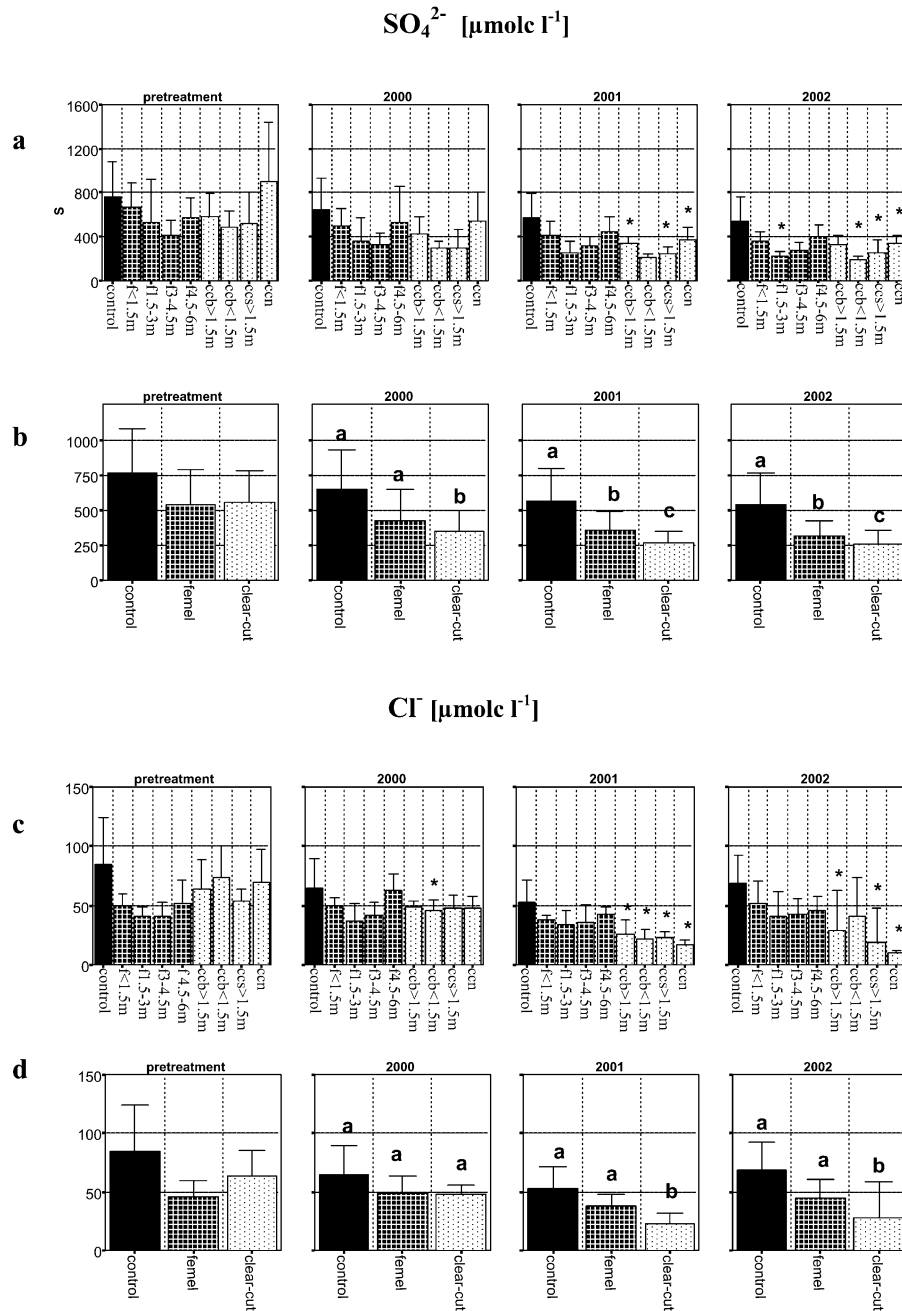


Figure 5. (a) Mean flux weighted yearly SO_4^{2-} concentrations in the pre treatment period, and in the years 2000, 2001, and 2002 on control, and for the different subvariants on the felmel, and clear-cut treatment. '+' indicates a significant increase in the concentration due to felling; '*' indicates a significant decrease in the concentration due to felling (analysis of covariance, $P < 0.05$). Subvariants on the felmel treatment differ in distance from a felled tree with: a) 'f<1.5 m': less than 1.5 m; b) 'f 1.5–3 m': 1.5–3 m; c) 'f 3–4.5 m': 3–4.5 m; d) 'f 4.5–6 m': 4.5–6 m distance. Subvariants on the clear-cut treatment: 'ccb>1.5 m': distant from the stem, planted with beech; 'ccb<1.5 m': close to the stem; 'ccs>1.5 m': distant from the stem, planted with spruce; 'ccn': natural spruce. (b) Mean flux weighted yearly SO_4^{2-} concentrations on the control, felmel, and clear-cut treatment. Significant differences due to felling ($P < 0.05$) between the different treatments tested are indicated by different indices (a, b, c). (c) Mean flux weighted yearly Cl^- concentrations in the pre-treatment period, and in the years 2000, 2001 and 2002 on control, and felmel, and clear-cut treatment. (d) Mean flux weighted yearly Cl^- concentrations on control, felmel, and clear-cut treatment.

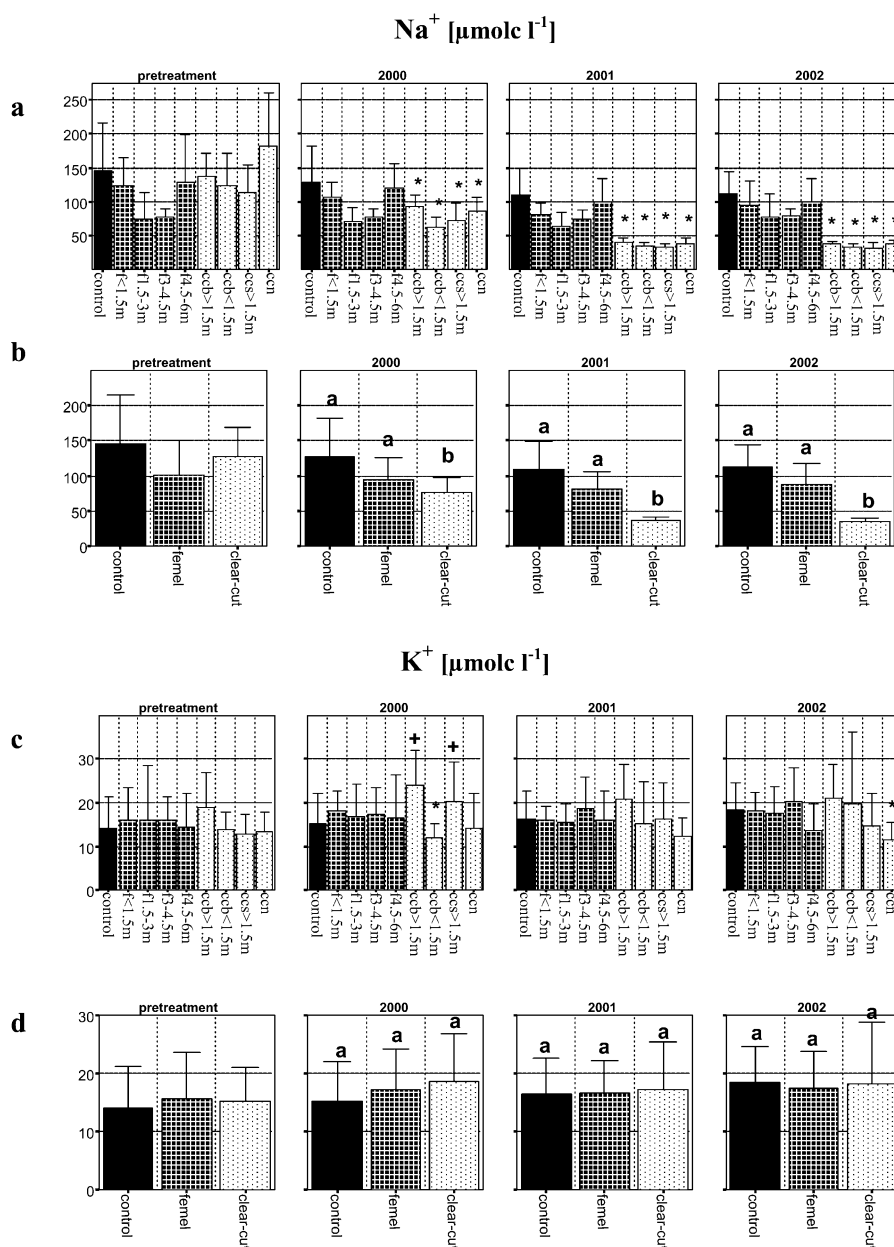


Figure 6. (a) Mean flux weighted yearly Na^+ concentrations in the pre-treatment period, and in the years 2000, 2001, and 2002 on control, and for the different subvariants on the felmed, and clear-cut treatment. '+' indicates a significant increase in the concentration due to felling (analysis of covariance, $P < 0.05$); '*' indicates a significant decrease in the concentration due to felling (analysis of covariance, $P < 0.05$). Subvariants on the felmed treatment differ in distance from a felled tree with: a) 'f<1.5 m': less than 1.5 m; b) 'f 1.5–3 m': 1.5–3 m; c) 'f 3–4.5 m': 3–4.5 m; d) 'f 4.5–6 m': 4.5–6 distance. Subvariants on the clear-cut treatment: 'ccb>1.5 m': distant from the stem, planted with beech; 'ccb<1.5 m': close to the stem; 'ccs>1.5 m': distant from the stem, planted with spruce; 'ccn': natural spruce. (b) Mean flux weighted yearly Na^+ concentrations on the control, felmed, and clear-cut treatment. Significant differences due to felling ($P < 0.05$) between the different treatments tested are indicated by different indices (a, b, c). (c) Mean flux weighted yearly K^+ concentrations in the pre-treatment period, and in the years 2000, 2001 and 2002 on control, and felmed, and clear-cut treatment. (d) Mean flux weighted yearly K^+ concentrations on control, felmed, and clear-cut treatment.

Table 2. Estimated water fluxes in the years 1999 (June to December), 2000, 2001, 2002, and for the pre-treatment period (June 1999 to February 2000). *Note that fluxes in 1999 were from a seven month period

Plot	Calculated fluxes in 40 cm depth (mm)				
	1999*	Pre-treatment	2000	2001	2002
C	214	263	574	563	711
F	214	259	645	630	782
CC	213	261	1008	1003	1101

critical probability level of 0.05 was used to indicate significant differences.

Results

The impacts of cutting on the different (sub-)variants are presented for NO_3^- , Al, Ca^{2+} , and Mg^{2+} in figures (Figures 1–4), each separated into three parts. The first part (a) of the graph shows the time series of the nine subvariants with standard deviation. The second part (b) shows the flux weighted yearly concentration of the respective ion for each year and the nine subvariants. The effect of felling is marked with ‘+’ for enhanced, and ‘*’ for reduced concentrations. The third part (c) shows the calculated average concentration of the respective ion for the variants control, femel, and clear-cut. Statistical differences due to cutting of the trees are indicated by different letters (a, b, c). For Cl^- , SO_4^{2-} , K^+ , and Na^+ flux weighted yearly average values are shown for the different variants, and subvariants (Figures 5 and 6). Table 2 presents the estimates of the water fluxes in different periods.

NO_3^-

On the control NO_3^- concentrations were mostly above the European limit for drinking water ($50 \text{ mg L}^{-1} = 806 \mu\text{M}$, Figure 1a–c). The highest NO_3^- concentrations (Figure 1a) were found at the clear cut in the year of felling (September, or October 2000), within the subvariants with planted saplings ($\text{ccb} > 1.5 \text{ m}$, $\text{ccb} < 1.5 \text{ m}$, $\text{ccs} > 1.5 \text{ m}$). In addition, in 2001 distinct peak concentrations occurred for the clear-cut subvariants; however, with lower concentrations than in 2000. The clear cut with subvariant natural spruce (ccn) showed a pronounced peak in 2000, but smaller than for the other clear-cut subvariants, and a hardly

recognisable peak in 2001. All peaks were restricted to a few sampling periods during summer to autumn. In the first months of 2002, higher concentrations could be detected on the femel variant for the subvariants close to the felled trees ($f < 1.5 \text{ m}$, and $f1.5\text{--}3 \text{ m}$). However, the effect was restricted to a few suction cups, indicated by the high standard deviation.

Flux weighted concentrations on the control were lower in the pre-treatment period than in 2000, 2001, and 2002 respectively (Figure 1b). Clear-cutting caused significantly higher concentrations in 2000 (compared to the femel variant, but not to the control), and significantly reduced concentrations in 2002 (femel, and control; Figure 1c). Femel-cutting reduced NO_3^- significantly only in 2001 (compared to control), indicating a small reduction of NO_3^- distributed equally for all subvariants where the felling effect was not significant (compare Figure 1c with 1b). Statistically significant felling effects were restricted to subvariants on the clear cut (Figure 1b). In the year 2000 in the interstem area of the clear-cut ($\text{ccb} > 1.5 \text{ m}$, and $\text{ccs} > 1.5 \text{ m}$) flux weighted yearly NO_3^- concentrations were significantly elevated (Figure 1b). No significant felling effect was detected for the subvariants close to the stem ($\text{ccb} < 1.5 \text{ m}$), the natural spruce regeneration subvariant (ccn), or for all subvariants in 2001. In 2002, felling reduced NO_3^- concentrations significantly for all clear-cut subvariants.

Al

Al showed nearly the same trend as NO_3^- (compare Figure 1a with Figure 2a). Clear-cutting caused significantly higher Al concentrations in 2000, on femel and control, and for the subvariants of the interstem area ($\text{ccb} > 1.5 \text{ m}$, and $\text{ccs} > 1.5 \text{ m}$, Figure 2b and 2c), but reduced concentrations for all subvariants in 2002 (Figure 2b). Femel-cutting did reduce Al concentrations in 2001 for the femel subvariant $f1.5\text{--}3 \text{ m}$. Cutting also reduced concentrations on the clear-cut subvariant with natural spruce regeneration (ccn) in 2001 (Figure 2b).

Ca^{2+}

Ca^{2+} peaks are less pronounced than Al peaks after clear-cutting (Figure 3a). Ca-concentrations are usually lower than Al concentrations. Under the simplified assumption that all Al is Al^{3+} , the median Ca/Al ratio was 0.22, in 90% of all cases the ratio is < 1.0 , and 8% of the values are even below 0.1. Clear-cutting, and femel cutting had low effects on Ca^{2+}

concentrations (Figure 3b, c). Only in 2002, Ca^{2+} was significantly reduced on the clear-cut (compared to *femel*, not to control, Figure 3a). No significant effect on flux weighted yearly Ca^{2+} concentrations could be detected for any of the subvariants (Figure 3b).

Mg²⁺

Clear-cutting caused distinct Mg^{2+} peak concentrations for the clear-cut subvariants in 2000, which were less pronounced in 2001 (Figure 4a). Statistically significant was the cutting effect in 2000 for *ccb*>1.5 m, and *ccs*>1.5 m with elevated concentrations, and in 2001 for *ccb*>1.5 m, and *ccn*, and in 2002 for all clear-cut subvariants with reduced concentrations (Figure 4b). Clear-cutting reduced Mg^{2+} in 2001 and 2002 compared to *femel* and control (Figure 4c).

Other anions (SO₄²⁻, and Cl⁻)

Clear-cutting significantly reduced SO_4^{2-} concentrations for three of four subvariants in 2001, and 2002 (Figure 5a). In 2000, 2001, and 2002 clear-cutting reduced SO_4^{2-} concentrations (Figure 5b). In 2001 and 2002 cutting also reduced SO_4^{2-} concentrations on the *femel* compared to the control (Figure 5b). Cl^- was only reduced after clear-cutting in 2001 and 2002 (Figure 5d). The effect was significant for nearly all subvariants in these years (Figure 5c). SO_4^{2-} and Cl^- did not show such distinct peaks (time series not shown) as observed for NO_3^- , Al, and Mg^{2+} . However, single Cl^- peaks were detected on the clear-cut treatment with high standard deviations.

Other cations (K⁺, Na⁺)

The felling decreased Na^+ concentrations on all clear-cut subvariants (Figure 6a, b). K^+ concentrations were very low in seepage water. K^+ was elevated on *ccb*>1.5 m, and *ccs*>1.5 m in 2000 (Figure 6c). Cutting of the trees caused no statistically significant differences for the different treatments (Figure 6d).

Discussion

NO₃⁻ leaching after clear-cutting compared with other investigations

In the present study at the N-oversaturated Höglwald site, with inputs in throughfall of approx. 30 kg ha⁻¹ y⁻¹ and more, distinct average NO_3^- peaks up to

2750 μM were detected after clear-cutting. An increase in NO_3^- concentrations in seepage water after clear-cutting is often described in the literature (for example Borman and Likens, 1979; Vitousek et al., 1979); however, the magnitude varies widely between different sites (Mellert et al., 1996; Weis et al., 2001) and seems to be determined by the N-status, fertility, and productivity of the ecosystem (Berden et al., 1997; Katzensteiner, 2003; Ring, 1995). High and distinct NO_3^- peak concentrations were also found after natural catastrophic events. Mellert et al. (1996) detected maximum NO_3^- concentrations of more than 4500 μM NO_3^- and average concentrations up to 3500 μM in most of 13 investigated wind-thrown spruce stands in Bavaria. After bark beetle attack in the Bavarian Forest National Park maximum concentrations were up to 2000 μM NO_3^- in seepage water, despite a dilution effect due to the relatively high precipitation on the site (Huber et al., 2004).

Moderately elevated NO_3^- concentrations were reported from case studies in the Austrian Alps (Katzensteiner, 2003) and in Northern Bavaria (Weis et al., 2001), though in both cases peak NO_3^- concentrations remained below the European limit for drinking water quality. Dahlgren and Driscoll (1994) reported NO_3^- concentrations of < 500 μM in the streamwater after clear-cutting Hubbard Brook Experimental Forest.

In contrast to the above mentioned works, also much lower NO_3^- concentrations were observed, especially on N-limited or non N-saturated sites (Ring, 1995; Berden et al., 1997). For example, in the study of Johnson and Todd (1998) maximum NO_3^- concentrations of approx. 25 μM were measured. After clear-cutting a N-limited mixed coniferous stand in Finland, NO_3^- remained below 20 μM (Piirainen et al., 2002). In an Irish catchment study, maximum NO_3^- concentrations of approx. 40 μM were observed (Cummins and Farrell, 2003). Relatively low NO_3^- concentrations occurred also in catchment studies on a majority of sites, which have been felled in Great Britain (Reynolds and Edwards, 1995; Neal et al., 1998). Attention and concern about the effects of clear-cutting on drinking water quality and site sustainability are often highlighted by distinct peak NO_3^- concentrations. However, most of the peaks in the present study appeared during times of relatively low water fluxes. Such peak concentrations may be overrated in respect to groundwater contamination, when in addition water fluxes are not considered. To assess the potential load for ground water resources, data with flux

weighted yearly average NO_3^- concentration are more vital, and also the spatial heterogeneity should be considered. Despite the high peak concentrations in our study, flux weighted NO_3^- concentrations were only significantly elevated due to the felling in the year directly after the clear-cutting in the area distant from the stem, which represents approx. 62% of the whole clear-cut area. In the area close to the stem (31% of the clear-cut) or in the natural regeneration area (6% of the clear-cut) no significant differences could be detected. The concentrations were not significantly different between clear-cut and control treatment. In the following year, no significant differences occurred for any of the subvariants. One year later, in the third year after clear-cutting, for all subvariants even significantly lower flux weighted NO_3^- concentrations were detected compared to femel-cutting, and control. This was mainly due to the removed canopy causing lower N-inputs and higher water fluxes via throughfall. Further, the subsequent emission of the N-trace gases and N_2 can diminish the pool of N-compounds.

NO_3^- leaching is a long-lasting problem at the Höglwald site, with nearly permanently elevated concentrations above the EU-level for drinking water, since the site was first monitored after starting the investigations in 1984 (Kreutzer et al., 1991). In regard to these long time spans of elevated NO_3^- concentrations, short periods of dramatically increased concentrations in the clear-cut does not increase NO_3^- leaching for this site to a relevant extend. In addition, one must take into consideration that almost two years after the cutting NO_3^- concentrations decreased to levels far below the control and below the EU-level of drinking water quality. This is the most striking difference, as most other investigations were made on sites, where NO_3^- was not present prior to cutting in critically elevated concentrations.

NO_3^- leaching after thinning or femel cutting

Femel-cutting in the present investigation could even reduce significantly NO_3^- concentrations in the first years. It was a slight effect, distributed over the whole femel treatment, and only recognisable in statistics because of the high number of plots on the treatment. On sites in the Alps with low NO_3^- concentrations in the old stand, forest thinning (removing 20–40% of the trees) increased NO_3^- concentrations slightly (Bäumler and Zech, 1999). Bauhus (1994) showed, that if a nested group of trees is removed the risk of NO_3^- leaching increased. However, after

the initial step of femel-cutting, which was investigated in the present study, further cuttings will be required as defined by the forest management plan, with additional risks of NO_3^- leaching.

Impact of regeneration and ground vegetation on NO_3^- leaching

Indications of a beneficial impact by pre-existing dense regeneration are given from our natural regeneration variant on the clear-cut. On this subvariant, peak and average NO_3^- concentrations remained relatively low after the felling. Also, Mellert et al. (1996) found lower NO_3^- concentrations in wind thrown stands with dense birch regeneration. In contrast to the regeneration, ground vegetation did not reduce NO_3^- concentrations effectively at our site. In the densely forested Höglwald site approximately 80% of the forest floor is covered by mosses. The coverage of the mosses decreased after felling, and herbs could not occupy the area rapidly enough to reduce NO_3^- concentrations as was the case in many other investigations (Bauhus, 1994; Emmet et al., 1991; De Keersmaeker, 2000; Mellert et al., 1996; Parfitt et al., 2002; Stevens and Hornung, 1990; Weis et al., 2001).

Al leaching and stress parameters

In the present investigation, Al showed nearly the same trend as NO_3^- with high peak concentrations in the first year after clear-cut. The leaching of cations is dependent on the concentrations of NO_3^- or other mobile anions (Dahlgren and Driscoll, 1994; Van Miegroet and Cole, 1985). During nitrification H^+ is produced and exchanged by other cations in the soil matrix. The composition of cations in the soil solution strongly reflects soil properties of the horizon where the suction cup is installed (Mellert et al., 1996). A high correlation of Al and NO_3^- was also determined in other investigations at the Höglwald site (Kreutzer et al., 1991), where the cation exchange sites in 40 cm depth are to 80–90% saturated with Al (Kreutzer and Weiss, 1998).

Root damages caused by toxic levels of Al may be problematic for the development of young beech or spruce saplings on the clear-cut, in addition to other unfavourable conditions (rapid change in light intensity, risk of frost events, mice, and others). The extreme Al pulses, and the low Ca:Al ratios during the growing season on the clear cut, but also on the femel cut, may be a risk according to Al-stress parameters (Rost-Siebert, 1983; Ulrich, 1988).

However, in an important part of the main rooting zone (humus layer), the base saturation is rather high (Kreutzer and Weiss, 1998) and Al appears mostly in complexed form and thus is not toxic (Schierl et al., 1986; Göttlein and Matzner, 1997). From acid irrigation experiments on mature trees at the Höglwald site indications are given for stress tolerance and defence mechanisms which prevent damage to the trees and permit the trees to achieve the high yield at this site (Hahn and Marschner, 1998; Kreutzer, 1995). Kreutzer et al. (1991) found also no damage on spruce seedlings after an enormous Al-mobilisation due to artificial acid irrigation on the Höglwald site. In the study of Ljungström and Stjernquist (1995) no correlation between growth of beech seedlings and Ca:Al ratio in soil solution was found. However, until now very little is known about the mechanisms of Al resistance of trees. Complexation of Al with phenolic substances present at the root surface and precipitation as phosphate may play an important role for immobilisation of Al (Heim et al., 2000). However, the authors mention, that high turnover rates of the fine root system would be necessary for this mechanism.

Nutrient cations (Ca, Mg, and K) in seepage water

K⁺ concentrations in the present study remained relatively unchanged after cutting. Possible influences may have compensated one another. The decrease in coverage of ground vegetation (mosses), was a source for K⁺, together with needle litter, dying roots, decaying wood, and other slash components (Arthur et al., 1993; Fahey et al., 1988; Katzensteiner, 2003). Additionally, the decrease in nutrient uptake can enhance K⁺ concentrations in seepage water. The elevated water-fluxes on the clear-cut (Table 2) together with lower K⁺ fluxes in throughfall (Table 1) dilutes K⁺ concentrations in the soil solution. However, on the clear-cut in the interstem area an increase in K⁺ concentrations could be observed for the year 2000. At the Hubbard Brook Experimental Forest clear-cutting resulted in large and long persisting losses of K⁺ in streamwater (Likens et al., 1994), which were highly correlated with NO₃⁻ (Dahlgren and Driscoll, 1994). A rapid increase in K⁺ concentrations was also found in drainage water after felling in Wales (Reynolds et al., 1995), in the Northern Limestone Alps (Katzensteiner, 2003), and in seepage water after bark beetle attack in the Bavarian Forest National Park (Huber et al., 2004). This K⁺ losses compromised a substantial de-

pletion of K for ecosystems with low K supply (see Katzensteiner, 2003; Huber et al., 2004).

Ca²⁺ was also slightly affected in the present study by the cuttings with nearly no significant effect on concentrations in seepage water. As for K⁺ the internal cycle of Ca²⁺ is interrupted on the clear-cut. As mentioned already for K⁺, different sources and sinks balanced one another and resulted in nearly unchanged Ca²⁺ concentrations after the fellings. Also, in a clear-cut at the Hubbard Brook Experimental Forest Ca²⁺ and Mg²⁺ showed much weaker correlation than K⁺ and Al to NO₃⁻ (Dahlgren and Driscoll, 1994). However, on windthrown sites, where Ca²⁺ was dominant on the cation exchange sites, it was also dominant in the soil solution (Mellert et al., 1996).

Compared to Ca²⁺, more pronounced peaks were found for Mg²⁺ in the first year after clear-cutting at our site. Mineralised Mg²⁺ from slash or dead plant material may be transported much faster than Ca²⁺. A lower binding capacity of Mg²⁺ to exchange sites is well known, which results in a higher mobility of Mg²⁺ compared to Ca²⁺ (Kreutzer, 1995). Later (2001, 2002) Mg²⁺ was reduced on clear-cut subvariants. The pool of Mg in bark, branches, twigs, needles, and roots is usually much smaller than of Ca, and K (see for example Katzensteiner, 2003). This will result in a much lower mineralisation rate of Mg from residues on the clear-cut than of Ca and K. A low mineralisation rate, reduced Mg fluxes in throughfall, higher water fluxes, and lower NO₃⁻ concentrations on the clear-cut may explain the reduction in Mg²⁺ concentrations.

Cl⁻, Na⁺, and SO₄²⁻ in seepage water

Decreasing Cl⁻ and Na⁺ concentrations occurred due to higher water fluxes on the clear-cut and the lower input via throughfall. Na⁺ was reduced more rapidly as Cl⁻, most probably because of a relatively higher proportion of Cl⁻ in slash, which may have been mineralised. Clear-cutting reduced atmospheric deposition of SO₄²⁻ at our site and doubled water fluxes in the mineral soil. This caused a significant reduction in SO₄²⁻ concentrations. A decline in this concentrations following felling has been reported also in other studies (Adamson and Hornung, 1990; Neal et al., 1992). However, effects may be more distinct closer to the sea, or on more SO₄²⁻ polluted sites. For example, clear-cutting on sites close to the sea caused a large reduction in streamwater Cl⁻, Na⁺, and SO₄²⁻ concentrations, due to a reduced dry deposition of sea salts

via removal by interception by the canopy (Reynolds and Edwards, 1995). Mineralisation processes play a minor role for SO_4^{2-} leaching after clear-cutting. Compared to the amounts of soil bound, and throughfall SO_4^{2-} , only small quantities of SO_4^{2-} can be mineralised since spruce slash contains relatively low amounts of S (N:S ratio is 10–15, Ulrich 1989). Dahlgren and Driscoll (1994) explained the decrease of SO_4^{2-} after clear-cutting, besides other processes, by an increased adsorption of the ion due to protonation of mineral soil surfaces by the acidification that follows clear-cutting. Kölling and Prietzel (1996) found a strong impact of nitrification on adsorption of sulphur compounds on oxides and hydroxides, and on solution of Al-hydroxysulphates. However, from our data we could not find any indication for such processes, whereas Mellert et al. (1996) found multiple correlation between NO_3^- and SO_4^{2-} resulting in elevated or decreased SO_4^{2-} concentrations depending on site conditions.

Conclusion and outlook

The clear-cut treatment at the N-saturated Höglwald site exhibited a pronounced but punctuated increase in NO_3^- and Al concentrations. In contrast to many other studies, the average NO_3^- concentrations are equally high or even higher on the mature N-saturated control as after the clear-cut. NO_3^- leaching has been and is currently a long-term problem of the stand, causing an increase in soil acidity, leaching of cations, and a risk for contamination of groundwater resources. Two years after the clear-cut, NO_3^- concentrations remained markedly below the EU level for drinking water. In future years we expect to get higher seepage water quality and quantity on the clear-cut. This is supported by the (1) experience, that nitrification pulses decreased mostly after two to three years, (2) lower N-input in throughfall, and (3) higher water fluxes due to the lowered interception. The present results demonstrate, that small clear-cuts may be carried out on N-saturated sites, without a risk of further NO_3^- contamination. Regardless, as shown by other investigations, clear-cuts have a high risk of enormous unwanted losses of nutrients. However, these losses are difficult to predict by foresters, as there are no practicable and accurate computer based model predictions available. With femel-cutting a dense natural or planted regeneration is usually established, in order to reduce the risks of unwanted side effects. The

first step of femel cutting is usually done to increase the light intensity under the canopy to establish a new tree generation. Following cuttings and the final felling may then be conducted with a lower risk of NO_3^- leaching. Among foresters it is a 'widely believed fact' that this strategy, and the high demand of the newly established regeneration reduces the risk for NO_3^- leaching for consecutive fellings. However, until now the advantages of this strategy are not supported by a sufficient number of investigations presented in refereed journals including all critical stages of regeneration.

Because of the low N-storage capacity of old mature stands, foresters should consider beginning the femel cutting much earlier than is the present forest practice in Germany (actually with stand age of 90 to 110 years in spruce stands), preferably before NO_3^- leaching starts. Unfortunately very little is known about a correlation between stand age and NO_3^- leaching on N-saturated sites.

With our experimental set-up, we tried to get a better handle on the spatial variations in the femel or clear-cut treatment. However, despite use of abundant number of suction cups, these calculations are a rough estimation for 'real' average concentrations. For example, there is still a lack of knowledge about the effects on the extraction line and on the ruts made by the harvester, which comprise significant portions of the total area.

All forest strategies do not eliminate the main problem for N-saturated forest ecosystems, which is the high anthropogenic N deposition. Only a decrease of the N-emissions can effectively reduce the risk of NO_3^- leaching. Agricultural policy should promote a sustainable agriculture with closed nutrient cycles and low NH_3 emissions to protect natural, or semi-natural ecosystems in rural, livestock dominated areas from high N-inputs.

Acknowledgements

The work was funded by the Bundesministerium für Bildung, Forschung und Technologie (BMBF), under contract number 0339733A. The Bayerische Staatsministerium für Landwirtschaft und Forsten helped with funding the projects B60 and St125. The authors would like to thank Angela Alkofer, Verena Rotter, Christian Mass, Michael Kohlpaintner, Rita Heibl, Daniel Glaser, and Wolfgang Petrik for excellent field work and laboratory measurements. Our resident mas-

ter student Jared David May from U.S. is gratefully thanked for improving the English and his valuable editorial inputs, two anonymous reviewers for critically reading and comments to improve the manuscript.

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Section editor: H. Lambers