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Response of water and nutrient fluxes to improvement fellings in a tropical montane forest in Ecuador

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Management of natural forests might be one option to reduce the high deforestation rate in Ecuador. We therefore evaluated the response of water and nutrient cycles in a natural tropical montane forest to improvement fellings with the aim of favoring economically valuable target trees which will later be harvested with additional ecosystem impacts not considered here.

The study was conducted at ca. 1900–2200 m above sea level in the south Ecuadorian Andes on the east-exposed slope of the east cordillera. In June 2004, one of two paired ca. 10-ha large catchments was thinned by felling 10.2% of the initial basal area (dbh $\geq$ 10 cm) on 30% of the catchment. The stems remained in situ. We measured ecosystem fluxes from rainfall via throughfall and stemflow to soil solution (litter leachate, soil solution at 15 and 30 cm depth) and stream flow between May 2004 and May 2005. After the fellings, soil solutions were extracted from the gaps created by the felled trees and the forest next to the gaps. We determined aboveground water fluxes by direct measurement and soil water fluxes with a budget approach. In the solutions, we measured concentrations of NH_4^+-N, NO_3^--N, total dissolved N, PO_4^{3-}-P, total dissolved P, Ca, Mg, K, Na, and Cl^-. Fluxes were calculated as volume-weighted mean (vwm) concentrations times water fluxes. Dry deposition was estimated using Cl^- as inert tracer.

The fellings increased concentrations of N, K, Ca, and Mg in the organic layer of the resulting gaps compared with the forest next to the gaps (vwm concentrations of N: 6.4 mg l^{-1} in the forest next to the gap/8.7 mg l^{-1} in the gaps, K: 9.8/11, Mg: 1.8/3.0, Ca: 3.4/5.8). Lower nutrient concentrations and fluxes in the mineral soil of the gaps than in forest next to the gaps suggested that these nutrients were taken up by ground vegetation and target trees. The paired modified and undisturbed catchments had similar water and nutrient budgets. The fellings did not have a significant impact on the water and nutrient budget at the catchment scale.

1. Introduction

Tropical montane rain forests fulfill important functions including the protection of downslope areas from flooding, prevention of soil erosion and landslides, and maintenance of a constant baseflow during dry periods (Hamilton et al., 1995; Bruijnzeel, 2000). Furthermore, particularly the north Andean montane forests are extremely rich in vascular plant species (Henderson et al., 1991). By 1991 more than 90% of the original forest cover in the north Andes had been lost (Henderson et al., 1991; Hamilton et al., 1995). The north Andean state of Ecuador suffers the highest deforestation rate in South America (FAO, 2006). Therefore, income-generating alternatives for the growing population in the mountain areas of Ecuador that maintain the functions and the biological richness of the natural forest are urgently needed (De Koning et al., 1998; Aguirre, 2007).

When attempting to maintain forest functions and biodiversity and nevertheless provide economic benefit to the local population, natural forest management is considered to be an option. Compared with complete transformation of a forest into agricultural land, single-tree extraction represents a low-intensity impact (Bawa and Seidler, 1998; Chazdon, 1998; Günter and Mosandl, 2003). The technical aspects of natural forest management were developed during the last century (Lamprecht, 1986; Brünig, 1996; Dawkins and Philip, 1998), but up to now it is difficult to prove the sustainability of silvicultural systems. According to FAO (2006)

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sustainability involves seven key components: extent of forest resources, biological diversity, forest health and vitality, productive functions, protective functions, socio-economic functions, legal, policy, and institutional framework. An important criterion which affects several of the key components of sustainability of land use is that nutrient losses and gains are nearly balanced and that the used ecosystem maintains its function as regulator of water and nutrient cycles. However, in many cases the current logging standards are not compatible with these ideals of sustainability and conservation (Günter, 2001; Huth and Ditzer, 2001; Hering, 2003). Thus, sustainable management of highly diverse tropical forest ecosystems requires case-specific silvicultural solutions (Hutchinson, 1993). So called "improvement fellings", i.e. the removal of competitors to foster growth and regeneration of trees with high timber value are among the oldest silvicultural instruments (Dawkins and Philip, 1998). Improvement fellings reduce the crown competition and stimulate nutrient uptake and thus favor target trees that can later be harvested. This harvest will likely cause further disturbances of the forest ecosystem.

In small forest gaps, such as those created by improvement fellings, nutrient concentrations in soil and soil moisture increase (Silver et al., 1996; Denslow et al., 1998; Muscolo et al., 2007; Schrumpf et al., 2007). The latter improves the nutrient supply of regrowth on the clearings and of the plants next to the gaps. From strong anthropogenic or natural forest interventions e.g., by slash and burn agricultural practices (Williams et al., 1997) or hurricanes (Silver et al., 1996; Schaefer et al., 2000) it is known that nutrient export rates increase because of reduced nutrient uptake by the damaged forest and because of enhanced nutrient release by mineralization. However, Silver et al. (1996) found for a wet subtropical forest in Puerto Rico, that nutrient pools returned to pre-disturbance conditions within a three-week period after biomass harvesting or a three-month period after gap formation by a hurricane. Ostertag (1998) reported that canopy gaps in a lowland rain forest in Costa Rica had reduced fine root length and biomass compared to undisturbed forest and this effect was more pronounced in a less fertile than in a fertile soil. Only in the less fertile soil, root growth was stimulated by fertilizer application. Ostertag (1998) therefore concluded that belowground responses to canopy opening may depend on soil fertility.

A suitable method to monitor water and element cycles in forests and to assess the response of forests to stress, is the catchment approach, i.e. the measurement of input, output, and internal water and element fluxes within a well defined and usually not too large catchment (Bruijnzeel, 1990; Likens and Bormann, 1995; Matzner, 2004). Catchment budgets of water and elements indicate directions of ecosystem development in response to stress such as direct anthropogenic interferences or nutrient and pollutant inputs. Prerequisites for the catchment approach are that the catchment is water-tight and all input and output fluxes of water or any studied element are registered. However, it is difficult to determine the water-conducting properties of the subsoil and underlying rock and topographic catchment borders are not necessarily the same as hydrological ones. Catchment approaches were used at several locations in the temperate zone (e.g., Bäumler, 1995; Likens and Bormann, 1995; Matzner, 2004) but rarely in tropical rain forests (Bruijnzeel, 1990, 2000).

Our objective was to set up water and nutrient budgets of selected chemical species of paired undisturbed and modified catchments (by improvement fellings) under natural tropical montane forest in Ecuador to assess the response of the forest to moderate thinning with the purpose of favoring potentially valuable target trees. We hypothesized that the reduced nutrient use in gaps produced by the fellings improved the nutrient supply of the target trees while water and nutrient budgets at the scale of the ca. 10 ha-large catchments responded to the thinning with increased nutrient export.

2. Materials and methods

2.1. Study sites

Two forested microcatchments (MC2 and MC5 in Fig. 1) between the cities of Loja (4°00′S 79°12′W) and Zamora (4°05′S 78°58′W) in the province of Zamora-Chinchipe in south Ecuador were selected (Wilcke et al., 2001). Catchment MC2 was ca. 9.1 ha and MC5 ca. 12.1 ha large. The catchments are located at ca. 1900–2200 m above sea level (a.s.l). All catchments drain into the Rio San Francisco, which flows into the Amazon basin.

Weather data are recorded since April 1998 on a clear-cut area in ridge-top position at 1952 m a.s.l. between MC2 and MC5 (Fig. 1). The mean annual temperature at 1952 m a.s.l. is 15.2 °C. The coldest months are June and July, respectively, with a mean temperature of 14.4 °C; the warmest month is November with a mean temperature of 16.1 °C. The average temperature gradient between the station at 1952 m and another station at 2927 m a.s.l. is a decrease of 0.6 °C per 100 m increase in elevation (Bendix et al., 2008). The distribution of the annual precipitation is unimodal with a maximum between April and September and without a pronounced dry season (Fleischbein et al., 2005). Mean humidity was 86% (with 90% continuously from April to June 2001) and 79% in November 2000. The mean speed of the mainly easterly winds for the period between April 1998 and April 2001 was 1.5 m s^{-1} with a maximum of 7.9 m s^{-1}.

The forest at the study site is described as "bosque siempreverde montano", evergreen montane forest (Balslev and Øllgaard, 2002). According to the classification of Bruijnzeel and Hamilton (2000), it is a Lower Montane Forest. Lauraceae, Rubiaceae, Melastomataceae and Euphorbiaceae are the most important tree families of the area. The tallest forest is found on lower slopes and in ravines where the canopy reaches 25 m with some emergents reaching up to 35 m. The ground flora is dominated by ferns and large herbs (Homeier et al., 2002; Paulsch, 2002; Homeier, 2004). Most of the trees have some vascular epiphytes (Paulsch, 2002). The forest has a stem density of 500–1250 stems ha^{-1} with dbh $\geq$ 0.1 m (diameter at breast height,) and of 1100–3100 stems ha^{-1} with dbh $\geq$ 0.05 m (Homeier, 2004).

The bedrock consists of interbedding of palaeozoic phyllites, quartzites and metasandstones (the "Chiguinda unit" of the "Zamora series" in Hungerbühler, 1997). Most soils are developed in surface sediments caused by landslides and possibly periglacial drift. The soils are mainly Humic Dystrudepts (USDA-NRCS, 1998) in both microcatchments. Most soils are shallow, loamy-skeletal with high mica (a layer silicate) content (Yasin, 2001). The thickness of the organic layer ranges 2–43 cm (mean: 16 cm, Wilcke et al., 2002).

2.2. Improvement fellings

The improvement fellings were realized in MC5. On the experimental area of about 3.4 ha (horizontal projection) in MC5, we registered all individuals with dbh > 20 cm of the target species *Tabebuia chrysantha* Nichols., *Cedrela* sp., *Podocarpus oleifolia* D.Don, *Nectandra membranaceae* Griseb, *Hyeronima asperifolia* Pax & K. Hoffmann, *Hyeronima moritziana* Pax & K. Hoffmann, *Inga acreana* Harms, *Clusia ducuoides* Engl., and *Ficus subandina* Dugand. The two species *T. chrysantha* and *Cedrela* sp. can be considered as economically most valuable. We selected 137 trees with straight trunks and regular crowns as target trees to be favored by improvement fellings of 40.3 (standard error, s.e. 6.4)

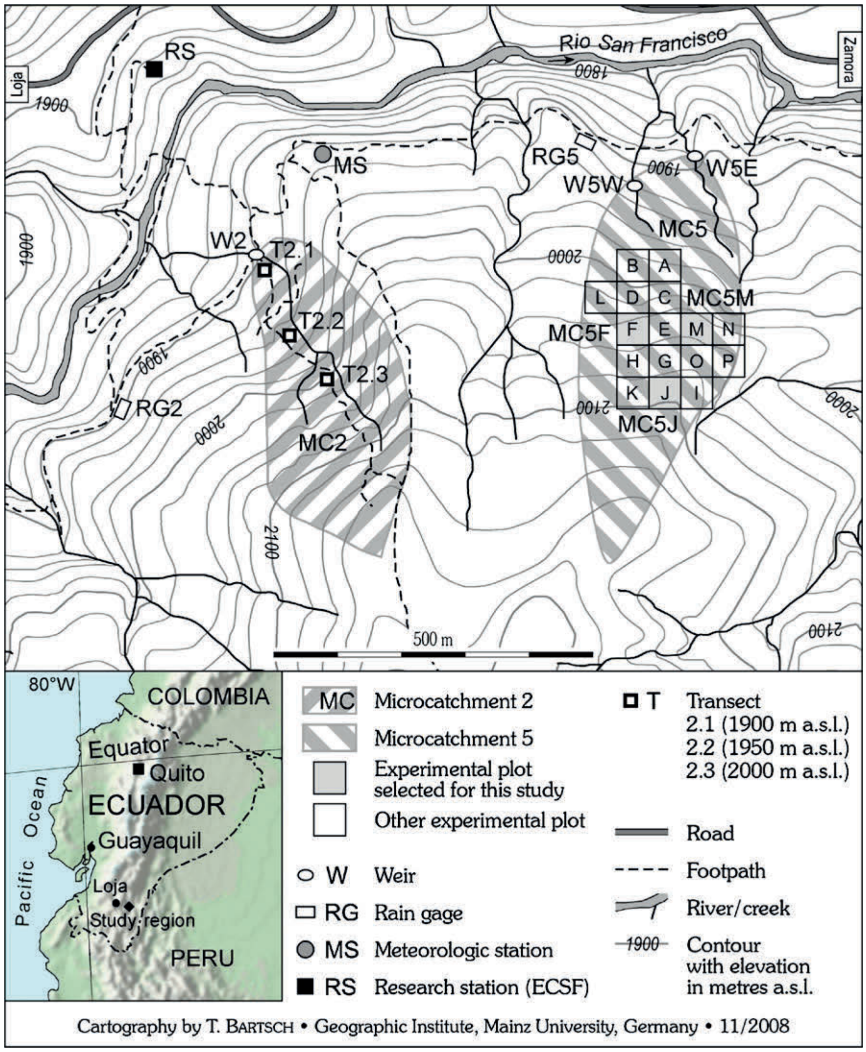

Fig. 1. Location of the study sites.

trees per ha. The most effective crown competitors of the target trees were removed in June 2004 in order to stimulate growth and natural regeneration of the target trees. The felled trees had mostly dominant in some cases codominant social positions. The average dbh of the felled trees was 29.2 cm (s.e. 1.0). In some cases only one tree was removed to favor two target trees and only in cases when both competitors were codominant we removed two competitors for one target tree. All felled stems, branches and leaves were left on the forest floor at the place where accumulated after the fellings.

Before the silvicultural treatment, the stands in MC2 and MC5 had different basal areas and stem densities (Table 1). The stand at MC5 had a slightly higher canopy openness already before the experiment than MC2, which was further increased after the thinning (Table 1). The logging intensity in MC5 corresponded to the removal of 3.6 (s.e. 0.55) $m^2\ ha^{-1}$ or 10.2% of the basal area (dbh $\geq$ 10 cm). Due to the fellings 10.8% (s.e. 1.9) of the surface suffered collision damages by felled crowns. This area was calculated immediately after felling as product from crown length and width for all felled trees. The structural parameters for the undisturbed catchment MC2 serving as reference area were sampled on a permanent plot of 5.2 ha for trees with dbh $\geq$ 20 cm cm and 20 subplots with a sampling area of 0.25 ha for trees with dbh $\geq$ 10 cm. Structural data for the modifed catchment MC5 originate from a permanent plot with 3.4 ha for trees with dbh $\geq$ 20 cm and 16 subplots with a sampling area of 0.22 ha for trees with dbh $\geq$ 10 cm.

Table 1
Structural properties (means and standard errors in brackets) of the stands (trees with dbh $\geq$ 10 cm) in the undisturbed catchment (MC2) serving as reference area and the modified catchment (MC5) in which the improvement fellings took place before and after the improvement fellings. All data refer to the slope-corrected area.

Property	Undisturbed forest	Modified forest	
		Before the fellings	After the fellings
Canopy openness (%)	5.0 (0.22)	6.7 (0.14)	11.9 (1.29)
Stem density (n ha^{-1})	642 (47)	743 (62)	703 (59)
Basal area (m^2 ha^{-1})	23.9 (1.6)	35.3 (1.8)	31.7 (1.7)

2.3. Solution sampling

To collect incident precipitation, two gaging stations each consisting of five fixed funnel gages were installed on deforested sites between 1900 and 1950 m a.s.l. adjacent to each catchment. In the undisturbed natural forest (definition according to FAO, 2004) in catchment MC2, three transects (at 1900–1910, 1950–1960, and at 2000–2010 m a.s.l.) were located upslope on the lower part of a 38–70° slope (Fig. 1). In the modified catchment MC5, three of sixteen 50 m $\times$ 50 m plots of the whole experimental design (Günter et al., 2008) were placed randomly for detailed study (Fig. 1). Along each transect (MC2) and on each experimental plot (MC5), twenty fixed throughfall collectors were installed. Along the transects in MC2, the throughfall collectors were placed randomly along a ca. 20 m-long line. On the plots in the modified natural forest (definition according to FAO, 2004) the collectors were placed evenly spaced along the two diagonals of the experimental plot because we aimed to collect the area-representative throughfall of a given experimental plot. Throughfall and rainfall collectors were 2-l polyethylene sampling bottles and circular funnels with a diameter of 115 mm. Throughfall collectors were installed at a height of 0.3 m above the soil surface.

To collect litter leachate, three zero-tension lysimeters were installed along each transect in the undisturbed forest and near the target tree (distance ca. 1–2 m) and in the gap created by the improvement fellings in the modified forest. The lysimeters, consisting of plastic boxes covered with a polyethylene net (0.5 mm mesh), had a surface area of 0.20 m $\times$ 0.14 m and were 0.15 m high. The boxes were connected with a 1-l polyethylene sampling bottle with plastic tubes. The lysimeters were installed from a soil pit below the organic layer parallel to the surface. The organic layer was not disturbed, most roots in the organic layer remained intact.

Mineral soil solution was sampled by three suction lysimeters (mullite suction cups, 1 μm $\pm$ 0.1 μm pore size) at each of the 0.15 and 0.3 m mineral soil depths at each sampling station with a vacuum pump. The vacuum was held permanently and adjusted to the matric potential. The solutions of all three lysimeters per depth were combined to one sample. We equipped all three transects in the undisturbed forest and the three selected experimental plots of the modified forest with such sampling stations. While in the undisturbed forest one sampling station was placed at each transect, in the modified forest we installed paired stations near the target trees (at a distance of 1–2 m) favored by improvement fellings and in the gap created by the fellings. Both, zero-tension lysimeters and suction cup lysimeters did not collect the soil solution quantitatively. Stream water samples were collected from the center of the stream above our weirs to avoid contamination by the weir material.

All samples were collected weekly. Sampling in the undisturbed forest started in April 1997 (Wilcke et al., 2001) and in the modified forest in April 2004. For this study, data of a one-year period between 5 May 2004 and 4 May 2005 were considered. Between 5 May and 16 June 2004 – the phase preliminary to the improvement fellings – rainfall, throughfall, litter leachate, and mineral soil solution at the 0.15 and 0.3 m soil depths were collected near the target trees, while the instrumentation in the gaps created by the fellings was only operational on 1st July 2004 after the improvement fellings which was completed on 22 June 2004.

Immediately after sampling, pH was measured with a glass electrode in unfiltered aliquots which were discarded after measurement. Another aliquot of 100 ml was filtered (ashless white ribbon paper filters, pore size, 4–7 μm, Schleicher and Schuell, Dassel, Germany), frozen at the day of sampling, and transported to Germany in frozen state.

2.4. Water and element fluxes

Water fluxes by rainfall and throughfall were calculated as means of the fluxes of all individual collectors for each gaging station. For the budgets, we used the gaging stations for rainfall adjacent to each of the undisturbed and modified catchments (Fig. 1) to determine the water input into the catchments. Horizontal rain contributed $< 6\%$ to the total water input (Bendix et al., 2008) and was neglected. The altitudinal change in rainfall in the altitudinal belt covered by the study catchments which were entirely below the condensation point (Bendix et al., 2008) was considered small and also neglected. Further rainfall data were obtained from the automatic meteorologic station at 1952 m a.s.l. (Fig. 1) in high resolution (Bendix et al., 2008). Catchment-wide throughfall was derived by averaging the values of the three throughfall gaging stations in each catchment. Stemflow was neglected because it accounted for less than 2% of the total water input to the soil as measured in the undisturbed forest (Fleischbein et al., 2006).

Water fluxes in soil were modeled by modifying the Soil Water Balance (SWB) model (DVWK, 1996). According to the SWB model, we determined the water fluxes out of the organic layer, the 0–0.15 m mineral soil layer, and the 0.15–0.3 m mineral soil layer. For the organic layer fluxes were calculated as throughfall (input) minus independently determined transpiration (output) minus (or plus) change in stored water in the soil layers as calculated by the difference in water contents of the respective soil layer between two soil water content measurements with frequency domain reflectometry (FDR). For deeper soil layers, the input is the output of the overlying soil layer. We assumed direct evaporation from the soil as negligible and derived weekly transpiration rates by partitioning the annual difference between throughfall and stream flow of each catchment proportionally to the individual weeks according to the measured interception losses (i.e. rainfall-throughfall). Weekly transpiration rates were furthermore distributed among the soil layers according to the root length densities of the respective soil layer taken from Soethe et al. (2006). We assumed a linear relationship between water uptake of the vegetation and fine root abundance. We used soil water-content measurements logged by FDR probes in the lower part of the undisturbed forest at 0.1, 0.2, 0.3, and 0.4 m depths for all transects since differences in soil water content were little pronounced because of the overall wet environment of the study site (Fleischbein et al., 2006). Data gaps of soil water fluxes (because of lacking soil water contents) were substituted with the help of a regression of weekly soil water fluxes on weekly throughfall volumes. The SWB approach does not consider lateral flow occurring as response to the rare rain storm events (Goller et al., 2005).

To quantify stream flow, Thompson (V-notch) weirs (90°) with sediment basins were installed in the lower part of each catchment and water levels were recorded manually twice per week. At the modified catchment, the stream flow of two branches of the stream was monitored with two weirs because these two branches only merged below the experimental plots at a location which was difficult to access. The calculated stream flow of the two weirs was combined to calculate the total stream flow of the modified catchment. For the undisturbed MC2, the weir was manually calibrated with ruler, bucket, and stop watch ($n = 29$ manually measured streamflow-water level pairs, calibration function derived by regression of ln(stream flow) on ln(water level), $r^2 = 0.84$), while for the modified catchment we used the water level-stream flow function published for notches with ideal 90° angles (Dyck and Peschke, 1995). Annual stream flow fluxes were calculated by assuming that the water level measured twice per week was representative for the time between one and the next measurement. This might underestimate the total stream flow because storms are likely underrepresented in our water level measurements. The rare strong storms (usually less than 5 per year, Fleischbein et al., 2006) cause overflow of our weir and cannot be registered. To assess the plausibility of our annual stream flow estimates we compared it to previously modeled five years of stream flow in MC2 with a hydrological model that was calibrated with directly measured water levels (Wilcke et al., 2008).

Missing values of rainfall, throughfall, and stream flow were substituted with the help of regression functions of these fluxes on the rainfall recorded at the meteorologic station at 1952 m a.s.l. (Fig. 1).

2.5. Chemical analyses

After export of the filtered 100-ml aliquots from Ecuador to Germany in frozen state, Ca, Mg, K, and Na concentrations were determined with flame atomic absorption spectroscopy (AAS). Furthermore, water samples were analyzed colorimetrically with a continuous flow analyzer (CFA) for concentrations of dissolved inorganic N (NH_4^+-N and NO_3^--N + NO_2^--N, hereafter referred to as NO_3-N), total dissolved N (TDN, after UV oxidation to NO_3), dissolved inorganic P (PO_4-P), total dissolved P (TDP), and total dissolved Cl^-. Some samples had concentrations below the detection limit of the analytical methods (0.075 mg l^{-1} for N, 0.2 mg l^{-1} for P, 0.001 mg l^{-1} for Ca, Mg, K, and Na). For calculation purposes, values below the detection limits were set to zero (for Ca, Mg, Na, K: <0.01%, N <1%, and P <45% of the values were below detection limit). Thus, our annual means underestimate the real concentration of chemical constituents and mean concentrations can be smaller than the detection limit.

2.6. Calculations and statistics

Annual element fluxes were calculated for rainfall, throughfall, and stemflow by multiplying the respective annual volume-weighted mean (VWM) concentrations with the annual water fluxes. Element fluxes with stream flow were calculated by multiplying flow-weighted mean (FWM) concentrations with the measured annual stream flow and relating the annual flux to the surface area of the catchments (undisturbed: 9.1 ha, modified: 12.1 ha).

To estimate the dry deposition and quantify canopy leaching we used the model of Ulrich (1983). The total deposition (TD) of an element i was calculated with Eq. (1).

$$\mathrm{TD_i} = \mathrm{RFD_i} + \mathrm{DD_i} \tag{1}$$

Here, RFD is bulk rainfall deposition measured at the three gaging stations adjacent to the study forest (including water-soluble coarse particulate deposition) and DD is dry deposition estimated with Eq. (2). This estimate of DD includes water-soluble dry particulate and gaseous deposition.

$$\mathrm{DD_i} = (\mathrm{TFD_{Cl}}/\mathrm{RFD_{Cl}})\mathrm{RFD_i} \quad \mathrm{RFD_i} \tag{2}$$

where $\mathrm{TFD_{Cl}}$ represents the throughfall deposition of Cl^-. The quotient $\mathrm{TFD_{Cl}}/\mathrm{RFD_{Cl}}$ is called deposition ratio. The canopy budget (LEA) was calculated with Eq. (3). Positive values of LEA indicate leaching, negative ones uptake of an element i by the canopy.

$$\mathrm{LEA_i} = \mathrm{TFD_i} \quad \mathrm{RFD_i} \quad \mathrm{DD_i} \tag{3}$$

If deep water leakage and changes in soil moisture and groundwater storages are negligible, the catchment water budget can be described with Eq. (4).

$$\mathrm{RF} = \mathrm{SF} + \mathrm{ET} \tag{4}$$

where RF is incident precipitation, SF is stream flow, and ET is evapotranspiration (i.e. the sum of Ei, Es and Et, see below). The changes in soil moisture and groundwater storages can be neglected in our study because of the budgeting interval of a whole year. The rainfall interception losses (Ei) were derived from Eq. (5).

$$\mathrm{Ei} = \mathrm{RF} \quad \mathrm{TF} \tag{5}$$

where TF is throughfall. Soil evaporation (Es) from the forest floor was neglected and transpiration losses (Et) were calculated with Eq. (6).

$$\mathrm{Et} = \mathrm{TF} \quad \mathrm{SF} \tag{6}$$

Complete time series of the various measurement stations were compared with the non-parametric Wilcoxon test for dependent data sets. Mean properties of the treatments (undisturbed vs. modified, and for soil solutions also gaps vs. forest next to gaps in the modified forest) before and after the improvement fellings were compared with one-way ANOVA followed by Student-Newman-Keuls test (if variances were homogeneous) or Games-Howell test (if variances were not homogeneous). The same tests were used to compare dissolved ion concentrations between the two catchments or three treatments (undisturbed forest, modified forst gap and modified forest next to gap) for individual months considering the individual sampling weeks as replicate concentration measures of the given month. If only two means of two groups of variables were compared, the results of the ANOVA were identical with those of a *t*-test. Statistical tests of differences in pH were run with H^+ concentrations. Statistical analyses were conducted with SPSS 15.0 (SPSS Inc., Chicago, IL, USA). Significance was set at $p < 0.05$.

3. Results and discussion

3.1. Nutrient supply of the target trees

One aim of the improvement fellings was to reduce nutrient uptake of competing trees and thereby increase the nutrient supply of the target trees. To test if this aim was reached we compared the chemical composition of the soil solution below the organic layer and at the 0.15 and 0.30 m mineral soil depths in the gaps created by the improvement fellings and in the forest next to the gaps in the modified catchment.

The pH of litter leachate (undisturbed forest: 4.6 and modified forest: 4.7) was not significantly different and did not respond to the fellings as indicated by almost unchanged pH after the fellings (undisturbed forest: 4.5, modified forest next to gaps: 4.4, gaps: 4.6). The mineral soil solution at 0.15 m depth had the same pH (4.4) in undisturbed and modified forest and was slightly more acid in the modified forest (4.5) than in the undisturbed forest (4.7) at 0.30 m depth before the fellings. After the fellings, the pH remained unchanged in the undisturbed forest (0.15 m: 4.4/0.30 m: 4.7) and became slightly more acid in the forest next to the gaps of the modified catchment (4.2/4.2) but increased significantly in the gaps (4.8/4.9) possibly indicating chemical reduction processes caused by increased soil moisture. This is supported by the finding that at 0.15 m soil depth the mean matric potential of our observation period was 6.8 MPa in the gap, 7.9 MPa in the forest next to the gap, and 10 MPa in the undisturbed forest reflecting a gradient of soil moisture along this line (own unpublished results). Increased soil moisture in gaps compared to undisturbed forest was also reported by Muscolo et al. (2007).

There was no temporal trend in the course of the concentrations of all studied nutrients in litter leachate (Fig. 2). The few extreme values are responses to dry periods particularly at the end of September 2004, in mid November 2004, and in the second half of January 2005 where weekly rainfall was 3.5–6.8 mm which is far below the mean weekly rainfall of 57 mm. Increases in ion concentrations are partly attributable to concentration effects which is supported by the simultaneous increase in the concentrations of Cl^- which can be considered as an inert tracer indicating concentration/dilution effects in the organic layer (Fig. 2). Concentration increases, however, can also be explained by enhanced mineralization because it is well known that C and N mineralization rates increase for a few days following the rewetting of a dry soil (Birch, 1958; Bloem et al., 1992; Cui and Caldwell, 1997; Franzluebbers et al., 1994)

During the preliminary phase (05/05–16/06/2004), the litter leachates of the experimental stations in the modified forest had