Research Articles

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Effects of Acidic Solutions on Element Dynamics in Monsoon Evergreen Broad-leaved Forest at Dinghushan, China

Part 1: Dynamics of K, Na, Ca, Mg and P

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Abstract

Background. Acid deposition has become a concern in south China in recent years. This phenomenon has increased to a dramatic extent with the large use of cars and coal-fueled power plants. As a consequence, soils are becoming acidified and their element dynamics will change. A decrease in the nutrient availability will lead to slower plant growth and maybe to a change in the forest type with current species being replaced by new ones with less nutrient requirements. Because of these reasons, it is important to understand how the dynamics of elements will change and what mechanism is part of the process. This knowledge is important for modeling the acidification process and either finding ways to counter it or to predict its consequences. The primary purpose of this study was to provide information about how the dynamics of K, Na, Ca, Mg and P are affected by acid deposition in a typical forest in southern China.

Methods. Experimental soils and saplings were collected directly from the monsoon evergreen broad-leaved forest in Dinghushan. All saplings were transplanted individually into ceramic pots in August 2000 and placed in an open area near their origin site. Pot soils were treated weekly from October 2000 to July 2002 with an acidic solution at pH 3.05, pH 3.52, pH 4.00 or pH 4.40, or with tap water as a control. The concentrations of SO_4^{2-}, NO_3^-, K^+, Na^+, Ca^{2+}, Mg^{2+} and available P and the pH were measured in soil and leachate samples taken at different times. The sapling leaves were collected and their element concentrations were measured at the end of the experiment.

Results and Discussion. Concentrations of soil exchangeable Ca and Mg decreased quickly over time, although only Ca showed changes with the acidic solution treatment and soil exchangeable K was stable because of soil weathering. Leaching of K, Mg and Ca was dependent upon the treatment acidity. Soil available P decreased slowly without any correlation with the acidity of the treatment. All the NO_3^- added by the treatment was taken up by the plants, but the SO_4^{2-} added accumulated in the soil. Amongst the plant species, *Schima superba* was little affected by the treatment, the leaf P content was affected in *Acmena acuminatissima* plants and *Cryptocarya concinna* was the most susceptible species to soil acidification, with a marked decrease of the leaf K, Ca and Mg concentrations when the treatment acidity increased.

Conclusions. Simulated acid deposition affected the dynamics of K, Ca and Mg in the monsoon evergreen broad-leaved forest. The dynamics of Ca in the soil and of K, Mg and Ca in the soil leachates were affected by the acidic solution treatment. If such a soil acidification occurs, *Cryptocarya concinna* will be amongst the first affected species, but *Schima superba* will be able to sustain a good growth and mineral nutrition.

Recommendations and Perspectives. Acid deposition will lead to imbalance the nutrient elements in the evergreen broad-leaved forest because of accelerated leaching losses of soil exchangeable Ca and Mg. Measures should be developed to slow down soil acidification or nutrient decrease.

Keywords: Acid deposition; *Acmena acuminatissima*; China; *Cryptocarya concinna;* monsoon forest; nutrient element dynamics; *Schima superba*; soil acidification; soil leachate

Introduction

Under naturally occurring conditions, the release of organic acids from leaf decomposition, the dissociation of the H_2CO_3 coming from the CO_2 dissolution and the cation uptake by plants will tend to cause an acidification of forest soils. However, this acidification will be balanced in the natural forest by other processes like the uptake of anion, the decomposition and decarboxylation of the organic matter and the weathering of soil minerals (Sato and Takahashi 1996). But this is likely to change. After comparing the soil chemical parameters in several locations from southern China with the same parameters measured in studies conducted 30 years ago, Pan (1992) and Dai et al. (1998) found that the soil pH had considerably decreased. This decrease, ranging from 0.1 to 1.0 pH units, suggests that a soil acidification had occurred. Several studies conducted on soil acidification showed that it originates mostly from acid deposition of sulfur and nitrogen oxides in south China (Liao et al. 1998, Larssen and Carmichael 2000). These oxides are mainly of anthropogenic origin as they are produced by the car traffic (NOx) and coal power plants (SO_2). In a fast developing country like China, the traffic is expected to increase to a great extent during the coming years and consequently also the emissions of gaseous nitrogen oxides. The same happens to the sulphur emissions from coal-fired power plants, although a cleaner technology is now in use and policies have been written to help restrict the SO_2 emissions. As a result, acid deposition in China is likely to become more serious and to extend to larger areas (Foell et al. 1995, Busch et al. 2001, Rodhe et al. 2002). This may result in further soil acidification in the future, as predicted by models (Larssen et al. 1999, 2000).

One of the consequences of the fast soil acidification following high input rates of gaseous oxides is an alteration of the biogeochemical cycles in the forest ecosystems. Generally, cumulative effects of acid deposition will lead to nutrient leaching (Oene 1992, Baba et al. 1995, Renner 1995) as the amount of H^+ cations in the soil increases and they replace the mineral cations in the argilo humic complex. A nutrient depletion will then follow in the long term for plants in which growth can be affected, especially for species with high nutrient demands. When the soil acidity increases, the concentration of free moving metallic ions (e.g. Al^{3+}, Mn^{2+}) will increase. Those metals show very high degrees of toxicity for many plants (Phillips and Yanai 2004, Joner et al. 2005). Most studies up to now have been conducted on cultivated soils. However, some attention has been paid to the dynamics of elements in forest ecosystems exposed to acid deposition. Nissinen and Hari (1998) showed that the accumulation of N in the soil of boreal Scots pine stands will lead to major changes in nutrient availability and N leaching. Driscoll et al. (2001) demonstrated that forest soil acidification leads to increase the leaching of cations such as K, Ca and Mg from the rhizosphere soil, which may cause nutrient imbalances for tree species. However, the acidification process varies greatly among different soils, as it is largely determined by soil properties and environmental conditions (Merino et al. 2000). Hence, it is necessary to study the consequences of acid deposition on every specific soil as they may differ with what was observed on other soils.

Information on changes in the element dynamics following soil acidification is important. It is useful for computer modeling and for a prediction of the future changes. With this knowledge, we will be able to offer the policy-makers good alternatives to remedy this problem, given that the origin of the problem will not disappear soon. Until now no investigation on this important environmental issue has been made in southern China forests. Although not exploited intensively for wood products, these forests represent a typical ecosystem of south China and a unique source of biodiversity for the native species. For those reasons, we decided to study the changes of element dynamics following soil acidification at Dinghushan Reserve, a typical forest environment of southern China. Our goal was to show (1) whether the acid deposition had an impact on the soil chemistry, (2) what the changes were and (3) how some of the typical tree species of these forests reacted to the changes.

1 Materials and Methods

1.1 Study area

About 80 km west of Guangzhou city, Dinghushan Reserve (Dinghushan) is the first natural reserve in China and is also part of the Network of the World's Biosphere Reserve organized by MAB, UNESCO. Located between 23°09'21" and 23°11'30" N and 112°30'39" and 112°33'41" E, Dinghushan is near the tropic of cancer. It is covered by tropical-subtropical forests, whereas 2/3 of the areas on this latitude are desert or semi-desert. This is due to the monsoon event that permits a high yearly water input, despite a dry season with very low rains. This humid subtropical monsoon climate is characterized at Dinghushan by the following environmental conditions: a total yearly solar radiation of 46×10^6 kJ/m^2 for the complete spectrum and of 23×10^3 kJ/m^2 for the plant active radiations (PAR); an annual mean temperature of 21°C with a maximum of 28°C in July and a minimum of 12.6°C in January; yearly precipitation comprised of between 1500 mm and 2300 mm with an average value of 1900 mm; a mean relative humidity of 80% and two main seasons, a wet/rain season from April to September and a dry season from October to March. As a consequence of this special climate, the forest climax in the area is the monsoon evergreen broad-leaved forest (MEBF). Some of its distinctive features are illustrated in Kong et al. (1997). In Dinghushan, the forest has been undisturbed for more than 400 years, which makes it one of the most valuable forest areas for scientific research in China.

1.2 Experimental design

We chose to study the effects of soil acidification on three tree species, *Schima superba*, *Cryptocarya concinna* and *Acmena acuminatissima,* that are three dominant species of the Chinese MEBF canopy. For each species, we collected thirty saplings that were two years old and 30 cm tall in August 2000 from the understory of the MBEF. They were then all transplanted individually into ceramic pots (diameter 30 cm, height 20 cm) where they were left to adapt for two months. Each pot was filled with around 6 kg of fresh lateritic soil collected from the upper layer of the MEBF soil in the same area as the saplings. This upper layer is acidified faster than the subsoil. Prior to the start of the experiment, the concentration of the major element was determined in the cultivation soil. The results are displayed in **Table 1**. After planting, all pots were placed in an open area near the MEBF in Dinghushan, where they were exposed to the natural rainfall in order to keep their environment as close as possible to the field conditions.

The pots were arranged as four random complete blocks. Each block was made of five different treatments with three plant species per treatment for a total of 15 pots per block (one sapling per pot). To simulate acid rain, the saplings

Table 1: Concentration of the major elements in the 0~10cm soil layer in MEBF at Dinghushan (mg/kg, except pH)

Parameters	pH	Exchangeable cations				Available P	NO_3^-	SO_4^{2-}
		K	Na	Ca	Mg			
Means SD	3.22 ±0.01	67.4 ±0.4	11.9 ±0.2	434.2 ±10.2	62.4 ±0.0	4.48 ±0.00	66.1 ±1.2	9.28 ±1.32

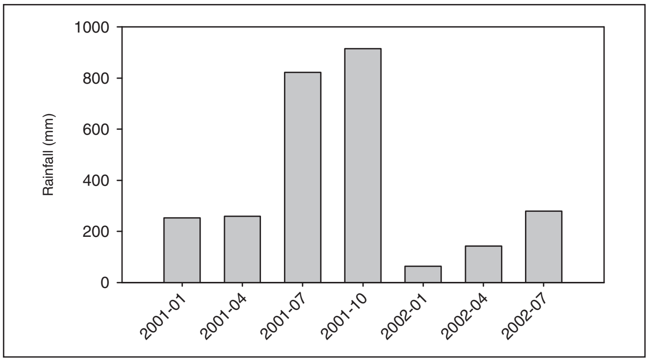

Fig. 1: Rainfall during the experiment (mm). Each value is the cumulated rain over the three months preceding the date

were watered once a week with 800 mL of an acidic solution at pH 3.05, pH 3.52, pH 4.00 or pH 4.40, or with tap water for the control. The tap water came from a natural lake at Dinghushan with an average pH value of 5.26. From January to October 2000, the pH value of atmospheric rainfall ranged from 4.36 during the dry period to 5.61 during the wet one. As we suppose that the rain acidity will probably increase in the future, we chose our values for the acidic solution pH to start around the lowest pH values observed in the natural rain and then decrease by steps of about 0.5 units. Acidic solutions were prepared by adding an acid mix made of H_2SO_4 and HNO_3 in a 1:1 mole ratio to the tap water. No fertilizer was supplied to the pots. The treatments were carried out from October 2000 to July 2002. The rain intensity was measured directly on site and is presented in **Fig. 1** as the cumulated amount of rain between one measure and the following one. Over one year, we watered the plants with 42 mm of water (or 42 L) so we consider this had no effect on the element dynamics given the ratio to the natural rains.

1.3 Sample collection and analysis

Soil samples were collected in each pot in October 2000, January, July and October 2001 and January and July 2002. They were collected as three cores of 1 cm of diameter from the upper soil layer (0 to 10 cm) for each time. Soil samples were air dried and sieved to pass a 2 mm mesh. Dead roots and plant residues were picked out. Soil leachate samples were collected from the bottom of the pots after the soils had been leached with 800 ml of tap water in April, July and October 2001 and January, April and July 2002. The pH values as well as the concentrations of K, Na, Ca, Mg and available P in leachates were measured immediately after collection. This sampling method was not meant to follow the natural dynamics of the element in leachates over time, but to provide data about the effect of the treatment acidity on leachates. Depending on the sapling sizes, about 20 to 60 mature leaves were collected in both January and July 2002. Approximately 1/8 of the tree leaves were sampled for the first measurement, which had no effect on their growth. Leaf samples were dried for three days at 70°C in an oven and then ground prior to chemical analysis.

Soil pH was determined with a glass electrode in the supernatant after shaking for 2h and sedimentation in a beaker for 24 h in 0.1M KCl. The soil to extractant ratio was 1:2.5. Soil leachate pH was directly determined using the glass electrode method. Exchangeable base cations in the soil (K, Na, Ca and Mg) were extracted with a 1M ammonium acetate (NH_4AC) solution adjusted to pH 7.0 and then measured by atomic absorption spectrometry (GBC932AA, GBC Scientific Equipment, Australia). Leachate K^+, Na^+, Ca^{2+} and Mg^{2+} were directly analyzed with atomic absorption spectrometry. The soil NO_3^- was extracted using a 10 mM $CaSO_4$ solution with a 1:5 soil to extractant ratio. The leachate NO_3^- and the soil extracted NO_3^- were determined using the phenol disulfonic acid spectrophotometric method (Nicholas and Nason 1957). As for SO_4^{2-}, the soil was dissolved in water (soil water ratio of 1:5) and then filtered. The SO_4^{2-} in this solution as well as the leachate SO_4^{2-} were measured using the barium sulfate gravimetric method (APHA Standard Methods, 20th ed., Method 4500-SO4 D, 1998). The soil available P was extracted with a solution containing 25 mM HCl and 30 mM NH_4F (the soil to extractant ratio was 1:7). The extracted P and the soluble P in the soil leachates were measured by the stannous chloride method (APHA Standard Methods, 20th ed., pp. 4–145, Method 4500-P D, 1998). Foliar N, S and P concentrations were determined photometrically after leaves were ground in a ball mill and digested using K_2SO_4-H_2O_2 (N and P) or HNO_3-$HClO_4$ (S). Foliar concentrations of Ca and Mg after leaf incineration and K after leaf digestion using K_2SO_4-H_2O_2 were measured by atomic absorption spectrometry.

1.4 Data analyses

Data analyses were carried out using the SAS software. For all statistical tests, we chose the probability level to reject the null hypothesis to be inferior or equal to 0.05. In order to obtain a global picture of the effect of acidic solution treatment on the soil and soil leachates over the whole experiment, we compared the treatments using a mixed model where the different soil parameters where expressed as linear combination of the treatment pH. We used ANOVA followed by Tukey multiple comparison test to study the effect of pH on: (1) the soil elements (for each time separated), (2) the leaf element content between treatments. We used the T-test for dependent samples to analyze: (1) the time dynamics for soil elements, where the T-test was performed on each pair of times, (2) the difference between two seasons for one leaf element.

2 Results

2.1 Dynamics of acidity

Fig. 2 shows the evolution of the soil acidity. Through the 2 years during which the experiment lasted, there was a significant increase of the soil pH in the control pots as well as a significant decrease in the pH 3.52 and pH 3.05 treatments ($p<0.05$). In the pH 3.05 treatment, this decrease was higher than 0.6 pH units. The effect of treatments was significant ($p<0.01$) with an increasing acidity of the soil as the watering solution acidity increased. The soil pH at the end of the experiment ranged from 3.35 in the control to 2.8 in the pH 3.05 treatment. The same effect appeared in the leachates that showed a significant decrease of the pH ($p<0.01$) correlated with the decrease of the soil pH across the treatments (**Table 2**).

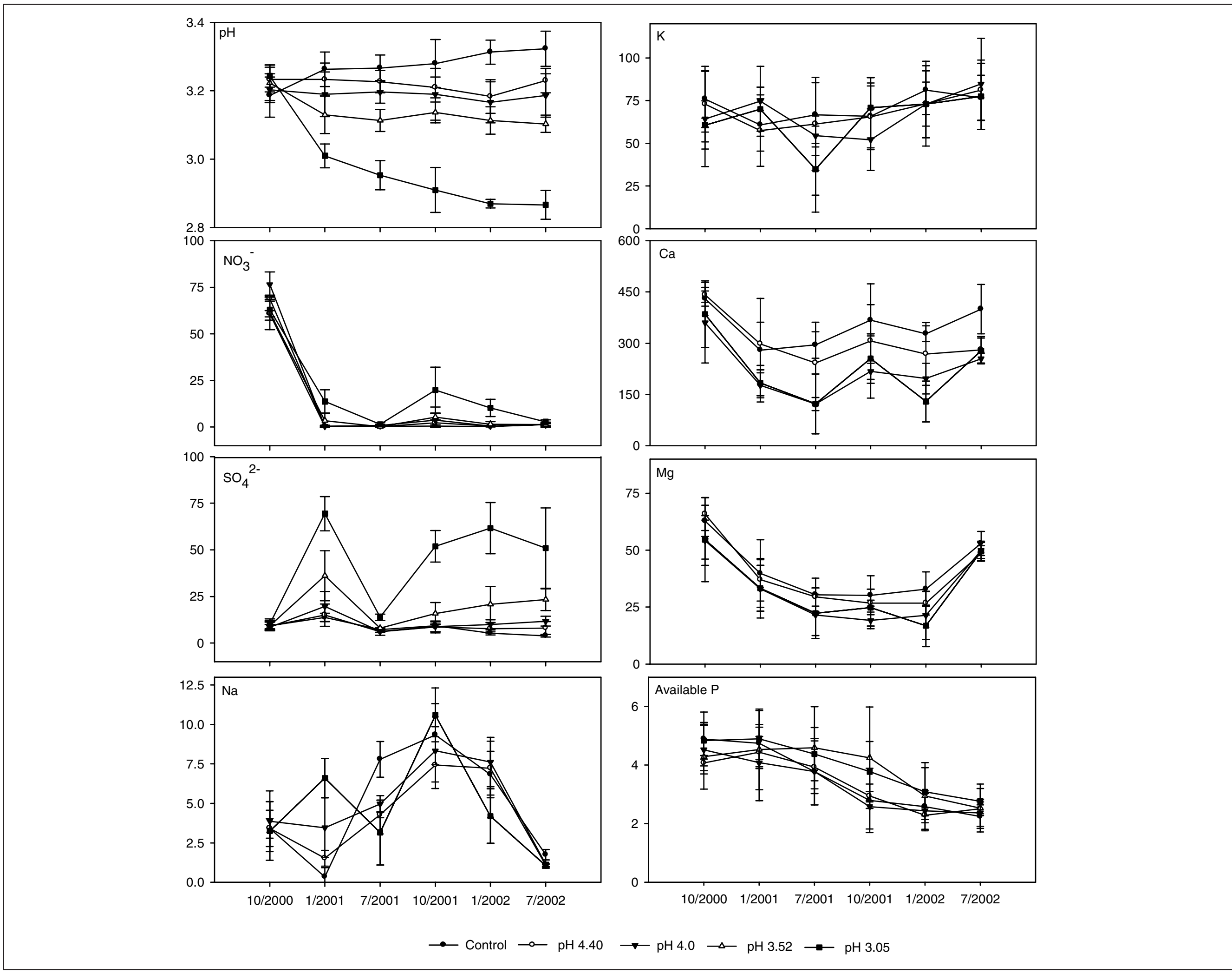

Fig. 2: Effect of acidic solution treatments on soil pH and elemental concentration changes during the experimental period (mg/kg, except pH)

2.2 Dynamics of SO_4^{2-} and NO_3^-

The NO_3^- concentrations in all the treatment soils decreased very fast and significantly, and reached a near zero value between the time saplings were planted and the first measure (see Fig. 2). After this, it did not show any significant variation except in the pH 3.05 treatment where it increased in October 2001 ($p<0.05$). According to our measures, only the pH 3.05 treatment leached significantly more NO_3^- than the other treatments (see Table 2).

Between treatments, SO_4^{2-} showed a greater variation than NO_3^- (see Fig. 2). The control, the pH 4.40 and the pH 4.00 treatments were not different, whereas the pH 3.52 treatment soil SO_4^{2-} concentration was two times higher ($p<0.05$) and the pH 3.05 treatment four times higher ($p<0.01$). Over time, the soil SO_4^{2-} increased in January 2001 in correlation to the soil pH and returned to the initial value in July 2007. After this, it increased again starting after July 2001 in only the pH 3.52 and pH 3.05 treatments. The SO_4^{2-} leaching was greatly influenced by the treatment (see Table 2) and the different treatments showed increasing SO_4^{2-} leaching rates in negative correlation to the pH of the treatments ($p<0.001$).

2.3 Dynamics of the elements in the soils

Between treatments, only the Ca^{2+} ion amounts in the soils changed significantly ($p<0.001$). None of the other cations or the available P were affected by the acidic solution treat-

Table 2: Average values of the soil leachate pH and elemental concentrations (mg/L, except pH; n=72). Data within columns that do not have common indices are significantly different (p<0.05)

Acidic solutions	pH	K	Na	Ca	Mg	P	NO_3^-	SO_4^{2-}
Control	4.80^a	1.14^a	0.89^a	3.30^a	0.359^a	0.0940^a	0.79^a	11.6^a
pH 4.40	4.16^b	1.57^a	0.95^a	3.70^a	0.448^b	0.0845^a	1.33^a	16.5^b
pH 4.00	3.71^c	1.27^a	0.83^a	3.20^a	0.482^b	0.0813^a	1.72^a	21.3^c
pH 3.52	3.64^c	1.33^a	0.87^a	5.19^{ab}	0.532^b	0.0862^a	1.40^a	33.4^d
pH 3.05	3.23^d	3.19^b	1.39^a	5.53^b	0.842^c	0.0908^a	8.55^b	86.2^e

ment. The Ca^{2+} concentration decreased by around 1/3 of the initial value in the pH 4.00, pH 3.52 and pH 3.05 treatments, with a little more in the pH 3.05 treatment soil, although this was not statistically significant. The results were different for the leachates (see Table 2). The leaching of Ca^{2+} was higher in the pH 3.05 treatment ($p<0.05$), the K^+ leaching was 2.5 times significantly higher in the pH 3.05 treatment ($p<0.01$) and the leachate Mg^{2+} concentrations were also affected as they increased while the pH of the treatment decreased ($p<0.01$). The Mg^{2+} leaching was 2 times higher in the pH 3.05 treatment than in the control.

Over time, it appears in Fig. 2 that the amount of K^+ in the soil increased a little, although the changes were not significant. The concentration in Na^+ first increased slowly but significantly ($p<0.05$) with a peak around 2 times the initial value in October 2001, after what it decreased further to reach values below the initial concentration ($p<0.001$). Between the planting and the first measure, the amount of the soil Ca^{2+} ions decreased by half in all treatments ($p<0.05$), then started to increase slowly. This increase was not significant for any treatments. As for the soil Mg^{2+}, it decreased by around 4 times in all treatments between October 2000 and January 2001 ($p<0.01$) and increased at the time of the last sampling in all treatments too ($p<0.001$). Finally, the amount of available P decreased slowly during the experiment for all treatments with a significant difference between the initial and last values ($p<0.05$).

2.4 Dynamics of elements in plants

Fig. 3 shows the element content in the sapling leaves at the end of the experiment during the dry (January) and the wet (July) seasons. For the three species, the amount of NO_3^- increased by 50% in both seasons when the pH of the treatment decreased ($p<0.01$). The values were around 14 mg/g for *Schima superba* (*Schima s.*) and *Acmena acuminatissima* (*Acmena a.*) and 22 mg/g for *Cryptocarya concinna* (*Cryptocarya c.*). Only the *Cryptocarya c.* saplings grown in the pH 3.05 treatment showed changes in the leaf K concentrations that increased significantly ($p<0.01$). The season had no effect on the leaf K. The S content was only affected by the treatment in *Schima s.* in July ($p<0.001$): the leaf concentrations increased by two times in the pH 3.52 and pH 3.05 treatments. In January, the leaf Ca content was not affected by the treatment, but it decreased in the leaves of the *Cryptocarya c.* and *Acmena a.* saplings grown on the pH 3.05 treatment in July ($p<0.05$). Only *Cryptocarya c.* showed a sensibility of the Mg uptake to the acidity of the treatment ($p<0.001$) as its leaf Mg content decreased along with the pH of the treatment (except under the pH 3.52 treatment) in both seasons. *Acmena a.* exhibited a non-significant decrease of leaf Mg. Finally, *Acmena a.* was the only species to show a sensibility of the leaf P to the treatment with a concentration in the pH 3.05 treatment that was half of the other treatments in both seasons ($p<0.01$).

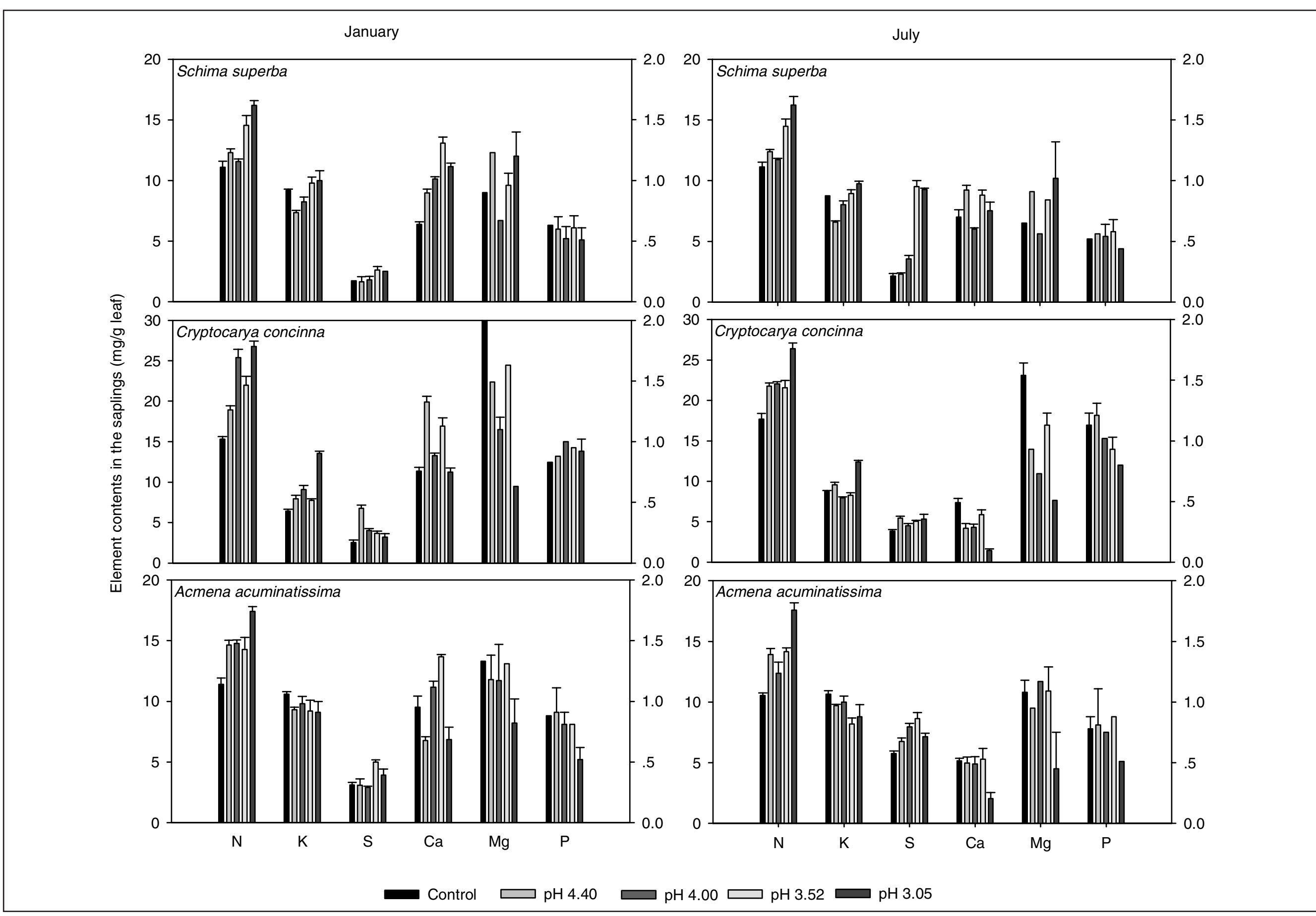

Fig. 3: Effect of acidic solution treatments on mean foliar elemental concentrations of *Schima superba*, *Cryptocarya concinna* and *Acmena acuminatissima* saplings during the experimental period (mg/g; n=4) ± SD. The Mg and P follow the right scale; other elements follow the left scale

3 Discussion

The soil pH during our experiment showed great differences between acidic solution treatments. It decreased obviously in the pH 3.52 and the pH 3.05 treatments. Although this result is not new, it is interesting to see that a very acid soil like the lateritic soil we studied can be acidified even more when exposed to a high input of H^+. Our results, however, show that the soil acidification will remain a slow process as long as the rain pH is higher than the soil pH. As acid rains with pH such as 3.05 do not occur often, this means for the MEBF that, although its soil will be sensible to the acidity of rains, it will acidify only slowly.

The soil NO_3^- concentrations decreased soon after the start and remained low during all the experiment. This is probably because we planted the young saplings in a lateritic soil that had little organic matter which decomposition could release NO_3^- in the soil for their use. As a consequence, they completely used the NO_3^- brought by the initial soil and the treatment. Tropical soils are often low in NO_3^- and the acid deposition may act as a fertilizer for this element. Fabian et al. (2005) showed that NO_3^- originating from forest fires and transported by the wind and the rain had a fertilizing effect on the acid soils of Ecuador tropical forests. In the pH 3.05 treatment, the NO_3^- increased later as the amount brought by the treatment was high enough to fulfill the needs of the saplings. This treatment also had a very high NO_3^- leaching. It shows that if NO_3^- was to accumulate in the MEBF soil, it would leach with possible consequences on the water usability.

The behavior of the SO_4^{2-} anions was different. The amount provided by the treatments increased with their acidity and directly affected the soil concentrations, which steadily increased in the pH 3.52 and pH 3.05 treatments. The SO_4^{2-} leaching rate was high in all treatments compared to the soil concentrations: 10 to 20 mg/L in the control up to 80 mg/L in the pH 3.05 treatment when the SO_4^{2-} input increased. This shows a low ability of Dinghushan MEBF lateritic soil to retain the SO_4^{2-} and the results provided by the pH 3.05 treatment let us suppose that the SO_4^{2-} saturated the soil with concentrations around 60–70 mg/kg of soil. It could thus be a key element in the leaching of exchangeable cations (Bergholm et al. 2003) as they have a high ability to attract them and make them leach along (Ulrich 1983).

In our experiment, the treatment had no effect on the Na^+ leaching and the changes that occurred in the soil Na^+ were of a small scale. This was due to the low total Na concentration in the soil. Na is not a plant nutrient so that small Na^+ concentrations have no consequences.

The exchangeable K was not affected by the treatments and it increased a little over the time, although it is a macronutrient of plants. Haun et al. (1988) found similar results when they were studying the effects of simulated acid rain on the growth of seedlings in an orchard. At the same time, the K^+ leaching was higher in the pH 3.05 treatment. This great stability is then probably related to the soil weathering. The MEBF soil naturally has a relatively high concentration of K in the core soil that can be released after weathering. Moreover, K^+ is one of the cations that are replaced most easily by H^+ in their exchange sites (Binkley and Richter 1987) so that the weathering process for them is the fastest. The lack of difference between different acidic solution treatments showed the ability of the MEBF lateritic soil to compensate for the losses of this element at first, when the soil has important reserves of K^+.

Both soil Ca^{2+} and Mg^{2+} concentrations decreased rapidly after the start of the experiment. This is due to the young saplings that showed a very impressive growth during the first three months as they benefited from the full light and a high NO_3^- supply, and because our cultivation soil had not enough organic matter in decomposition to provide an input of elements like in the natural forest soil systems. After this initial period, the growth of plants as well as their use of other element decreased as the available NO_3^- was low. From this time, the concentration of both cations remained stable as the soil weathering compensated for the ions that leached. The heavy monsoon rains that occurred at this time prevented the element concentrations to increase as they were creating a higher leaching. At the end of the experiment, the increasing concentration of the Mg^{2+} must be put in relation with the very dry year, hence a low leaching. The soil acidification had a strong impact on both Ca and Mg as shown by the leaching rates that were higher in the treatments which were more acidic. However, only the soil Ca^{2+} concentration was significantly affected by the pH of the treatment. Although the Mg leaching rate was correlated with the treatment acidity, its value was small compared to the amount of exchangeable Mg that consequently did not show any change in relation with the acidity. This is because the soil Ca was available in very high concentrations (6 times more) so it was the first element affected by the higher leaching caused by the acidification. At the end of the experiment, the increase of both cation soil concentrations was faster in the pH 3.05 and pH 3.52 treatments as the H^+ accelerated their weathering.

In the soil, the available P concentrations decreased continuously. This is logical as P is a major nutrient for the plants. The decrease was quite important given the very small amount of P at Dinghushan. However, one can expect a high recycling rate for the P in the natural forest as well as plant species with lower P requirements than on other soils. The phosphor should not be a concern as it was not affected by the acidic solution treatment.

In the three sapling species, the treatment improved the amount of N in the leaves. This is a consequence of the input of NO_3^- with the treatment and the low availability of this nutrient in our cultivation soil as we discussed before. *Schima s.* was the species that showed the lowest changes in its leaf content across all the treatments and seasons. This species will probably be the less sensitive of the three here to the soil acidification as its deeper root system allows it to get mineral elements in the lower layers of the soil where they are less affected by the acid deposition. In our experiment, the young saplings of *Schima s.* had a root system 2 times deeper than the two other species at the end of the two years. It is quite clear that *Cryptocarya c.* will be the most sensible species here as this plant physiology requires a higher concentration of nutrients in the leaves (see Fig. 3). It was obvious in our experiment that its leaf Ca and Mg contents decreased a lot in July (around 2/3 of the January content). Although the rain season was not really wetter in our experiment than the dry one, changes occurred between the two seasons because the rain season is also the season of the growth. If the trees cannot uptake enough elements from the soil, they will distribute the elements from the older leaves to the new ones, leading to a decrease of the element concentration per leaf mass. At the end of the experiment, the growth of the three species was comparable in the control pots and *Cryptocarya c.* was around 25 cm smaller than the two others in the pH 3.05 treatment. *Acmena a.* results were less dramatic than those of *Cryptocarya c.*, although it is obvious that a high acid deposition could affect young trees of this species in a short amount of time. *Acmena a.* was also the only species to show a consistent sensibility to the soil available P.

Although the pH changes between each of our treatments was around 0.5 units, changes in the soil element dynamics were rather limited until the pH of the treatment decreased from 3.52 to 3.05. This result suggests that a threshold exists for the soil acidification in the MEBF lateritic soil and that the threshold value is around the soil pH. As long as the acid deposition pH is higher than the one of the soil, the acidification process is slow or inexistent. But if the pH of the acid deposition becomes lower than the soil pH, the soils like the one we studied at Dinghushan will acidify quickly (0.5 pH unit in two years) with the consequences we have shown on the dynamics of elements.

4 Conclusions and Recommendations

The acid deposition on the MEBF lateritic soil will have variable effects depending on its acidity. The soil acidification will be high unless the acid deposition is less acidic than the soil itself. For the moment, the most acid rains measured around Dinghushan did not meet the acidity threshold so the soil should not acidify strongly until the air pollution by N and S oxides increases. If such acidification occurs, the Ca will be the first nutrient affected followed shortly by the Mg, whereas the K will not be a concern in the initial period. The SO_4^{2-} concentration will increase greatly in the soil as well as the leachates. On the other hand, the acid deposition will have a benefit on plant growth as it brings NO_3^- in a soil that lacks this nutrient, but this effect will not last long under high acid deposition, as the other nutrients will become limiting factors. The P was not affected by the acidification in our experiment, although it should be given a special care regarding its very low concentration. *Schima superba* will resist to the soil acidification to a certain extent thanks to its deep root system, but *Cryptocarya concinna* will be very susceptible to the soil nutrient decrease as its nutrient demand is high. This will translate as reduced growth and possibly a greater mortality. If acid deposition becomes stronger in the future as is predicted, it could prove necessary to devise special protection policies for this species. As for *Acmena acuminatissima*, its sensibility to soil acidification is half that of the other species, but it will probably be strongly affected in cases of changes in the P dynamics.

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References

Baba M, Okazaki M, Hashitani T (1995): Effect of acidic deposition on forested andisols in the Tama hill region of Japan. Environ Pollut 89, 97–106

Bergholm J, Berggren D, Alavi G (2003): Soil Acidification Induced by Ammonium Sulphate Addition in a Norway Spruce Forest in Southwest Sweden. Water Air Soil Pollut 148, 87–109

Binkley D, Richter D (1987): Nutrient cycles and H budgets in a forest ecosystems. Adv Ecol Res 16, 1–51

Busch G, Lammel G, Beese FO, Feichter J, Dentener FJ, Roelofs G-J (2001): Forest Ecosystems and the Changing Patterns of Nitrogen Input and Acid Deposition Today and in the Future Based on a Scenario. Environ Sci Pollut Res 2, 95–102

Dai Z, LiuY, Wang X, Zhao DW (1998): Changes in pH, CEC and exchangeable acidity of some forest soils in southern China during the last 32–35 years. Water Air Soil Pollut 108, 377–390

Driscoll CT, Lawrence GB, Bulger AJ, Butler TJ, Cronan CS, Eagar C, Lambert KF, Likens GE, Stoddard JL, Weathers KC (2001): Acidic deposition in the Northeastern United States: sources and inputs, ecosystem effects, and management strategies. BioScience 51, 180–198

Fabian P, Kohlpaintner M, Rollenbeck R (2005): Biomass Burning in the Amazon-Fertilizer for the Mountaineous Rain Forest in Ecuador. Environ Sci Pollut Res 12 (5) 290–296

Foell W, Green C, Amann M, Bhattacharya S, Carmichael G, Chadwick M, Cinderby S, Haugland T, Hettelingh JP, Hordijk L, Kuylenstierna J, Shah J, Shrestha R, Streets D, Zhao D (1995): Energy use, emissions and air pollution reduction strategies in Asia. Water Air Soil Pollut 85, 2277–2282

Haun WG, Wolt J, Reyndlds JH (1988): Effects of acidic solution on soil solution composition and orchard grass seeding viability. Soil Sci Soc Am J 52, 1037–1043

Joner EJ, Eldhuset TD, Lange H, Frostegard AS (2005): Changes in the microbial community in a forest soil amended with aluminium in situ. Plant Soil 275, 295–304

Kong GH, Huang ZL, Zhang QM, Liu SZ, Mo JM, He DQ (1997): Type, structure, dynamics and management of the lower subtropical evergreen broad-leaved forest in the Dinghushan Biosphere Reserve of China. Tropics 6, 335–350

Larssen T, Carmichael GR (2000): Acid rain and acidification in China: the importance of base cation deposition. Environ Pollut 110, 89–102

Larssen T, Seip HM, Semb AJM, Ivar PM, Rolf DV, Espen L, Valter A, Tang D, Odd E (1999): Acid deposition and its effects in China: An overview. Environmental Science & Policy 2, 9–24

Larssen TU, Schnoor JL, Seip HM, Zhao DW (2000): Evaluation of different approaches for modeling effects of acid rain on soils in China. Sci Total Environ 246, 175–193

Liao B, Larssen T, Seip HM (1998): Response of five Chinese forest soils to acidic inputs: batch experiment. Geoderma 86, 295–316

Merino A, Macías F, García-Rodeja E (2000): Element Fluxes and Buffer Reactions in Acidified Soils from a Humid-Temperate Region of Southern Europe. Water Air Soil Pollut 120, 217–228

Nicholas DJD, Nason A (1957): Determination nitrate and nitrite. Methods Enzymol 3, 981

Nissinen A, Hari P (1998): Effect of nitrogen deposition on tree growth and soil nutrients in boreal Scots pine stands. Environ Pollut 102 (s1), 61–68

Oene VH (1992): Acid deposition and forest nutrient imbalance: A modeling approach. Water, Air and Soil Pollut 63, 33–50

Pan GX (1992): Acidification of soils in Mount Lushan over the last 35 years. Pedosphere 2, 179–182

Phillips RP, Yanai RD (2004): The Effects of $AlCl_3$ Additions on Rhizosphere Soil and Fine Root Chemistry of Sugar Maple (Acer Saccharum). Water Air Soil Pollut 159, 339–356

Renner R (1995): 'Scientific Uncertainty' Scuttles New Acid Rain Standard. Environ Sci & Tech 29, 464–466

Rodhe H, Dentener F, Schulz M (2002): The global distribution of acidifying wet deposition. Environ Sci Tech 36, 4382–4388

Sato K, Takahashi A (1996): Acidity neutralization mechanism in a forested watershed in Central Japan. Water Air Soil Pollut 88, 313–329

Ulrich B (1983): Soil acidity and its relations to acid deposition. In: Ulrich B, Pankrath J (eds), Effect of accumulation of air pollutants in forest ecosystems. D. Reidel Publishing Company, Dordrecht, pp 127–146

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