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Effects of Acid Rain on Forest Nutrient Status

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Environmental Sciences Division
Publication No. 2498

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ENVIRONMENTAL SCIENCES DIVISION

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SUMMARY

The overall purpose of this study was to evaluate the effects of acid deposition on the nutrient status of five diverse forest ecosystems: two (chestnut oak and yellow-poplar) on Walker Branch Watershed, Tennessee; one (scarlet oak) on Camp Branch Watershed, Tennessee; and two (red alder and Douglas-fir) at the Thompson site, Washington. These sites represent both a diversity in soils, climate, and vegetation and a gradient in pollutant sulfur inputs, from a high value of near 30 $kg \cdot ha^{-1} \cdot year^{-1}$ in Walker Branch to a low value of about 9 $kg \cdot ha^{-1} \cdot year^{-1}$ at the Thompson site. A special study of the effects of acid and sulfur on decomposer invertebrates was included in the Walker Branch study.

Atmospheric acid inputs exceeded natural H^+ production by carbonic acid in the Tennessee sites and roughly equaled natural H^+ production at the Douglas-fir site. The cumulative effects of 40 years of nitrogen fixation in the red alder site had resulted in an internal H^+ production by nitrification and nitrate leaching that far exceeded atmospheric H^+ inputs at that or any other site. The very high natural leaching rates in the red alder stand have significantly acidified, and continue to acidify, that soil.

Although atmospheric acid input caused increased leaching from the other soils, base cation loss rates were not large enough to rapidly deplete soil exchangeable cation reserves, and no long-term significant soil acidification by acid deposition is foreseen. Furthermore, scarce or potentially limiting cations (such as Ca^{2+} in one of the Tennessee sites) are being conserved despite accelerated leaching rates.

Atmospheric sulfur inputs exceed forest sulfur requirement in all of the five sites. Excess sulfur accumulates to a minor extent as SO_4^{2-} in vegetation, but soils are the major repository for excess SO_4^{2-} in these ecosystems. The greatly enhanced nitrogen status of the red alder stand appears to have resulted in lowered SO_4^{2-} in both vegetation and soils at that site as compared to the adjacent nitrogen-poor Douglas-fir site.

Decomposer invertebrates appeared to be affected negatively by large applications of SO_4^{2-}, either as $KHSO_4$ or K_2SO_4. Forest floor buffering prevented large changes in pH with acid SO_4^{2-} treatments. Results indicate that effects of acid deposition on decomposer invertebrates are unlikely except at input levels much higher than ambient.

The results of this study apply only to the five sites investigated in terms of the effects of acid deposition. However, the processes studied in these sites--nitrification, carbonic acid formation, sulfate absorption--may be examined in a variety of forest ecosystems to assess acid deposition.

ABSTRACT

JOHNSON, D. W., D. W. COLE, J. M. KELLY, J. W. WEBB, D. D. RICHTER, and H. VAN MIEGROET. 1985. Effects of acid rain on forest nutrient status. ORNL/TM-9729. Oak Ridge National Laboratory, Oak Ridge, Tennessee. 54 pp.

In five forest sites (three in eastern Tennessee and two in western Washington) the effect of natural carbonic acid production on soil leaching was equaled or exceeded by that of atmospheric acid inputs. In a nitrogen-fixing red alder site in Washington, however, internal leaching by nitrification and nitric acid formation far exceeded atmospheric H^+ inputs at any site. All other sites retained NO_3^-, and soil SO_4^{2-} adsorption reduced the effectiveness of atmospheric H_2SO_4 inputs on soil leaching in two of the Tennessee sites and in the Washington red alder site.

The very high natural leaching rates in the red alder stand have significantly acidified, and continue to acidify, that soil. At all other sites, base cation loss rates were not large enough to rapidly deplete soil exchangeable cation reserves, and no long-term significant soil acidification by acid deposition is foreseen. Furthermore, scarce or potentially limiting cations (such as Ca^{2+} in one of the Tennessee sites) are being conserved despite accelerated leaching rates.

Atmospheric sulfur inputs exceeded forest sulfur requirement in all five sites. Excess sulfur accumulated to a minor extent as SO_4^{2-} in vegetation, but soils were the major repository for excess SO_4^{2-}. The greatly enhanced nitrogen status of the red alder stand appeared to have lowered SO_4^{2-} in both vegetation and soils at that site as compared to the adjacent nitrogen-poor Douglas-fir site.

Decomposer invertebrates appeared to be affected negatively by unrealistically large applications of SO_4^{2-}, either as $KHSO_4$ or K_2SO_4. Forest floor buffering prevented large changes in pH with acid SO_4^{2-} treatments. Results indicate that effects of acid deposition on decomposer invertebrates are unlikely except at input levels much higher than ambient.

Section 1

INTRODUCTION

The overall purpose of this study was to determine the effects of acid precipitation on the nutrient status of five diverse forest ecosystems: two on Walker Branch Watershed, Tennessee; one on Camp Branch Watershed, Tennessee; and two at the Thompson Research Site, Washington. Specifically, our objectives were as follows:

Objective 1: Determine the direct effects of H^+ and SO_4^{2-} deposition on nutrient status in five forest ecosystems.

Objective 1a: Determine the input of H^+ in acid rain relative to the input of H^+ by natural leaching processes in the soil.

Objective 1b: Determine the fate of incoming SO_4^{2-} and its effects on sulfur cycling and nutrient leaching from the ecosystem.

Objective 2: Determine the effects of H^+ and SO_4^{2-} deposition on the decomposer invertebrate community in a deciduous forest ecosystem (Walker Branch Watershed) and the consequences of this relative to phosphorus availability and cycling.

The first major task under Objective 1a was to evaluate the relative contributions of atmospheric H^+ input vs internal H^+ generation (primarily by carbonic acid formation but also by natural formation of H_2SO_4, HNO_3, and organic acids) in the leaching of nutrient cations from soils at the five study sites. This provides a means of assessing the effects of acid rain on cation losses from the ecosystem within the context of natural rates of cation loss. To evaluate the total base cation removal from the soil, it is also necessary to estimate cation uptake by forest vegetation (Sollins et al. 1980). Cation removal from soils by plant uptake must be distinguished from leaching, however, because the cations are removed from the soil but not from the ecosystem. In this overall study, only leaching effects were considered, but effects of plant uptake on base cation removal were estimated for some sites (Johnson et al., in press).

The second major task (Objective 1b) was to determine the fate of incoming SO_4^{2-} and its effects on sulfur cycling in these ecosystems. The emphasis in most acid rain studies has been on the effects of H^+, but there are several reasons to carefully consider the effects of elevated SO_4^{2-} inputs as well. First, SO_4^{2-} contains an essential plant nutrient, sulfur (S), and information is available concerning the effects of sulfur deficiency on forests (Turner et al. 1980); but little is known about the effects of excess sulfur. Furthermore, it has been repeatedly shown that atmospheric sulfuric acid inputs will have no effect on cation leaching from forest soils unless the sulfate anion is mobile in the ecosystem (Johnson and Cole 1977, Johnson et al. 1979, Singh et al. 1980, Lee and Weber 1982).

The invertebrate study was conducted to determine the effects of neutral and acidic SO_4^{2-} deposition on soil and litter arthropods and the consequent effects on available phosphorus (P). Soil invertebrates are an essential component in decomposition and elemental cycling processes in forest ecosystems (Crossley 1977). These invertebrates play an integral role in the fragmentation of litter and may also be important in regulating rates of nutrient release in forest ecosystems (Satchell 1974, McBrayer 1977). Availability of phosphorus to plants appears to be strongly controlled by decomposer invertebrates (McBrayer 1977).

Section 2

SITES AND METHODS

The EPRI work was coupled with a National Science Foundation (NSF)-funded basic study of soil-leaching processes at Walker Branch Watershed, Tennessee, and Thompson Site, Washington. The central hypothesis of the NSF study was that elemental leaching losses from forest ecosystems can be predicted by understanding and quantifying processes regulating anion production and mobility within the soil. More specifically, the following hypotheses were addressed:

- The mobility of sulfate and phosphate in soil is related to the free iron (Fe) and aluminum (Al) content of the soil. This, in turn, is a function of the original chemical composition of the soil parent material and the degree to which it has been weathered.
- The ratio of nitrate to bicarbonate in the soil solution is regulated in part by the ratio of available nitrogen (N) to available carbon (C) in the soil.
- Any addition (natural or otherwise) that raises the soil solution pH will mobilize HCO_3^- and absorbed anions and thus shift the anion composition of the soil solution.

These hypotheses were tested in a series of manipulative experiments in two plots--one nitrogen-rich and one nitrogen-poor--at each site. Treatments included (1) sufficient urea-nitrogen to double litter nitrogen content, (2) sufficient sawdust to double litter weight, and (3) 0.5 eq/m^2 NaH_2PO_4, Na_2SO_4, NaCl, and $NaNO_3$ salts. Effects of these treatments on soil solution composition will be reported elsewhere.

EPRI funding allowed the expansion of this study to address the acid deposition issue in several respects. It allowed the addition of instrumentation and analyses to the existing plots (precipitation, throughfall, litterfall, additional lysimeters, as well as additional analyses for sulfur and nitrogen in soils and vegetation), the addition of a new site (Camp Branch), and the addition of an invertebrate study.

Two of the three treatments in the NSF study, urea-nitrogen and sawdust application experiments, were expanded with EPRI funding to examine

sulfur-nitrogen-carbon interactions at all sites. Site descriptions and details of methods are given below.

SITES

Walker Branch Watershed, Tennessee

Walker Branch Watershed is situated in the ridge-and-valley physiographic province near Oak Ridge, Tennessee (Grigal and Goldstein 1971). The climate type is humid mesothermal with moderate summer and winter temperatures. Mean annual temperature is 14.3°C and precipitation averages 130 cm/year, mostly as rain (Henderson et al. 1977). Underlying the watershed is Knox dolomite, a cherty, dense to coarsely crystalline rock of late Cambrian to early Ordovician age.

The chestnut oak stand on Walker Branch is located on a ridgetop on the Fullerton series soil, a Typic Paleudult. Vegetation type is chestnut oak, described by Grigal and Goldstein (1971). Species consist primarily of chestnut oak (*Quercus prinus* L.), hickory (*Carya* spp.), red maple (*Acer rubrum* L.), white oak (*Quercus alba* L.), black oak (*Q. velutina* Lam.), blackgum (*Nyssa sylvatica* Marsh.), sourwood (*Oxydendrum arboreum* L.), and yellow-poplar (*Liriodendron tulipifera* L.), with occasional dogwood (*Cornus florida* L.) and northern red oak (*Q. rubra* L.). Understory is absent.

The yellow-poplar stand is located in a sinkhole depression on the Tarklin series soil, a Typic Fragiudult. Vegetation type is yellow-poplar, described by Grigal and Goldstein (1971). Species consist primarily of yellow-poplar with occasional red maple, white oak, black locust (*Robinia pseudoacacia* L.), persimmon (*Diospyros virginiana* L.), black cherry (*Prunus serotina* Ehrh.), and Carolina buckthorn (*Rhamnus caroliniana* D.C.). Understory consists of dogwood, occasional eastern red cedar (*Juniperus virginiana* L.), and Japanese honeysuckle (*Lonicera japonica*).

Thompson Site, Washington

The Thompson site is located in the western foothills of the Cascade Mountains near Landsburg, Washington. The climate type is maritime, with cool, dry summers and wet, moderate winters. Mean annual temperature is 9.8°C and mean annual precipitation is 136 cm, most of which falls as rain between October and March. The soil at the study site belongs to the Alderwood series, a Dystric Entic

Durochrept, and is composed of loosely consolidated ablation till of varying depth over a compacted basal till.

The research plots are located in a naturally regenerated 48-year-old stand of red alder (Alnus rubra Bong.) and in a 48-year-old plantation of Douglas-fir (Pseudotsuga menziesii Bong. Carr.) immediately adjacent to it. The understory in the red alder stand is dominated by sword fern (Polystichum munitum Kaulf. Presl.) intermixed with occasional western hemlock (Tsuga heterophylla Raf. Sarg.), huckleberry (Vaccinium parviflorum Smith), vine maple (Acer circinatum Pursh.), Oregon grape (Berberis nervosa Pursh. Nutt.), and elderberry (Sambucus callicarpa Pursh.). Understory in the Douglas-fir stand is much less predominant and consists of salal (Gualtheria shallon Pursh.), Oregon grape (Berberis nervosa Pursh. Nutt.), huckleberry (Vaccinium spp.), and mosses.

Camp Branch Watershed, Tennessee

The Camp Branch Experimental Watershed is located on the Cumberland Plateau within the boundaries of Fall Creek Falls State Park in Bledsoe County, Tennessee. The study area encompasses the major portion of the area drained by the south fork of Camp Branch; the weir is located about 175 m upstream from the confluence of the north and south forks. Camp Branch is a tributary of Cane Creek, which is a tributary of Caney Fork River, which forms the main body of the U.S. Corps of Engineers Center Hill Reservoir.

The Camp Branch watershed occupies a total of 94 ha and ranges in elevation from 597.5 m at its highest point to 518.3 m at the weir.

The climate of the Cumberland Plateau is temperate and continental. The winters are moderate, with short cold periods, and the summers are mild to hot. Precipitation is well distributed over the year. The average annual precipitation for the Camp Branch watershed is about 144.3 cm, mostly in the form of rain. Mean annual temperature is 12 °C.

The study site was located on an upper slope plateau on the Lily series soil, a Typic Hapludult derived from acid sandstone. Vegetation consisted of second growth forest in which oak species constitute approximately 70% of the basal area. Quercus coccinea, Q. stellata, Q. alba, and Q. velutina are dominant (Kelly 1979).

METHODS

Site Characterization and Monitoring

At each plot, bulk precipitation and throughfall were collected by means of plastic funnels fitted to polyethylene bottles; litterfall was collected in litter traps; and soil solutions were collected beneath litter, A horizons, and B horizons (at rooting depth) by means of tension lysimeters (Cole 1968). Collections were made on approximately a monthly basis. Vegetation (foliage, branch, bole, roots), litterfall (leaf and nonleaf), and forest floor samples were analyzed for total nitrogen by Kjeldahl digestion and total sulfur by a nitric-perchloric acid digestion procedure at A and L Laboratories, Memphis, Tennessee.

Soil samples were analyzed for percent gravel; bulk density (by determining pit volume with backfilling and weighing); pH (1:1 soil:water ratio); percent carbon (LECO furnace); percent nitrogen (Kjeldahl digestion followed by Autoanalyzer analysis for NH_4^+); cation exchange capacity and exchangeable cations (1 N NH_4Cl followed by alcohol rinse and NaCl extraction; atomic absorption analysis on the NH_4Cl extract for Ca^{2+}, K^+, and Mg^{2+}; and Autoanalyzer analysis on the NaCl extract for NH_4^+); iron and aluminum (by citrate-dithionite and by oxalate methods; USDA 1972); extractable SO_4^{2-} (by shaking with distilled water followed by Autoanalyzer analysis); soluble absorbed SO_4^{2-} (by shaking with 0.032 M NaH_2PO_4 in a 1:5 soil:solution ratio for 1 h, followed by barium chloranilate analysis; Bertolacini and Barney 1957); and total sulfur (by a nitric-perchloric acid digestion followed by turbidometric analysis of SO_4^{2-} at A and L Laboratories, Memphis, Tennessee).

Water samples from Camp Branch and Walker Branch were analyzed by the Analytical Chemistry Division of ORNL, and water samples from the Thompson site were analyzed at the College of Forest Resources, University of Washington. Cross-comparisons between laboratories were made and results were comparable. All waters were analyzed for pH, conductivity, Ca^{2+}, Mg^{2+}, K^+, and Na^+ (the latter four by atomic absorption), NH_4^+, NO_3^-, PO_4^{3-}, Cl^-, SO_4^{2-} (by Technicon Autoanlyzer), and alkalinity by titration. Further checks of analyses were made by charge balance between cations and anions (except where organic anions were suspected) and by comparing calculated with measured conductivity (Golterman and Clymo 1969).

Soil Invertebrate Studies

Study plots were established on a ridgetop site on Fullerton series soil on Walker

Branch Watershed. Vegetation was of the chestnut oak type described by Grigal and Goldstein (1971).

Two replicates were established, each consisting of five 8- x 8-m plots. Plots were randomly selected, with each plot exposed to one of five treatments based on the estimated annual input of SO_4^{2-}-S (18 kg ha^{-1} $year^{-1}$) to Walker Branch Watershed (Shriner and Henderson 1978). Treatments consisted of two-times annual SO_4^{2-} (3.6 g SO_4^{2-}-S m^{2-} $year^{-1}$) (2S), ten-times annual SO_4^{2-} (18 g SO_4^{2-}-S m^{-2} $year^{-1}$) (10S), two-times SO_4^{2-} and H^+ (3.6 g SO_4^{2-}-S m^{-2} $year^{-1}$ plus 0.112 g H^+ m^{-2} $year^{-1}$) (2H), ten-times SO_4^{2-} and H^+ (18 g SO_4^{2-}-S m^{-2} $year^{-1}$ plus 0.562 g H^+ m^{-2} $year^{-1}$) (10H), and a control (distilled water only) (C). Recent measurements of H^+ inputs to Walker Branch (Richter et al. 1983) indicate that H^+ contributes 0.6 keq ha^{-1} annually. The 2H and 10H applications supply an additional 1.12 and 5.62 keq ha^{-1} $year^{-1}$, respectively. The H^+ treatments, though not exact, adequately represent two and ten times the annual H^+ inputs to Walker Branch.

Surface soil and litter samples were taken at approximately monthly intervals between November 1980 and March 1982. Because the number of monthly invertebrate samples greatly exceeded the number of Tullgren funnels, all plots were not sampled on the same dates. Usually, one replicate was sampled on a given date and the second replicate was sampled one week later. This procedure allowed inferences to be made concerning interactions between treatment, replicate, and month since replicates were sampled at varying time periods after treatment applications.

Soil and litter samples were extracted for invertebrates using modified Tullgren funnels (Tullgren 1918).

Soils were analyzed for pH, available phosphorus (Bray No. 2 extraction), and sulfate (SO_4^{2-}) quarterly, while aluminum (Al^{3+}) and potassium (K^+) levels (by 1 N NH_4Cl extraction) were measured at 6-month intervals.

Microbial and microfloral activity in the forest floor was estimated by measuring CO_2 evolution according to the method of Edwards (1982). Ten litter samples per plot were taken from the 10S and control treatments on two occasions in August 1982. Samples were collected from the O_e horizon, weighed to 1 g (moist wt), incubated for at least 4 d, and flushed with CO_2-free air. Carbon dioxide evolution from each sample was measured three times over a 72-h period.

Section 3

RESULTS

INTERNAL VS EXTERNAL H^+ PRODUCTION AND EFFECTS ON BASE CATION LOSSES.

At each of the five sites, precipitation pH increased upon passing through the forest canopy (Figure 3-1). In all but the red alder site, pH continued to rise in soil solution, allowing the net production of HCO_3^-. In the red alder stand, pH is reduced in the A-horizon solution because of soil acidification by a natural internal process: nitrification (Van Miegroet and Cole 1984). Soil solutions in the red alder stand are dominated by nitrate, indicating that internally produced HNO_3 by nitrification is the major cation leaching agent. The predominance of NO_3^- in this stand is due to overabundance of nitrogen due to nitrogen-fixation by the red alder. All other sites are nitrogen-deficient, and thus atmospheric NO_3^- is rapidly immobilized in the ecosystem as noted in Figure 3-1.

Sulfate was the major anion in solutions at the Fullerton, Tarklin, and Camp Branch sites, implying that atmospheric inputs play a major role in cation leaching at these sites (Figure 3-1). As discussed below, however, not all atmospherically deposited SO_4^{2-} is associated with H^+, and thus the large leaching rates of SO_4^{2-} at these sites do not necessarily result in equivalent cation loss from the ecosystem (Richter et al. 1983). Furthermore, as is noted later, some individual base cations may be conserved even when the sum of base cations shows a net loss (Johnson and Richter 1984).

Both the Fullerton and Camp Branch soils adsorb SO_4^{2-} in the B horizon (Johnson et al. 1982, Richter et al. 1983), and because of charge balance considerations (i.e., total cations must equal total anions), this results in a reduction in cation concentration and leaching (Johnson and Cole 1980). In the Tarklin soil, which does not adsorb SO_4^{2-}, cation concentrations are not reduced in B-horizon solutions (Figure 3-1). At the Washington sites, which are receiving lower atmospheric deposition rates, SO_4^{2-} is less important in regulating cation loss, accounting for less than half of the anion flux in the Douglas-fir soil and as little as 7% of total anion leaching in the red alder soil.

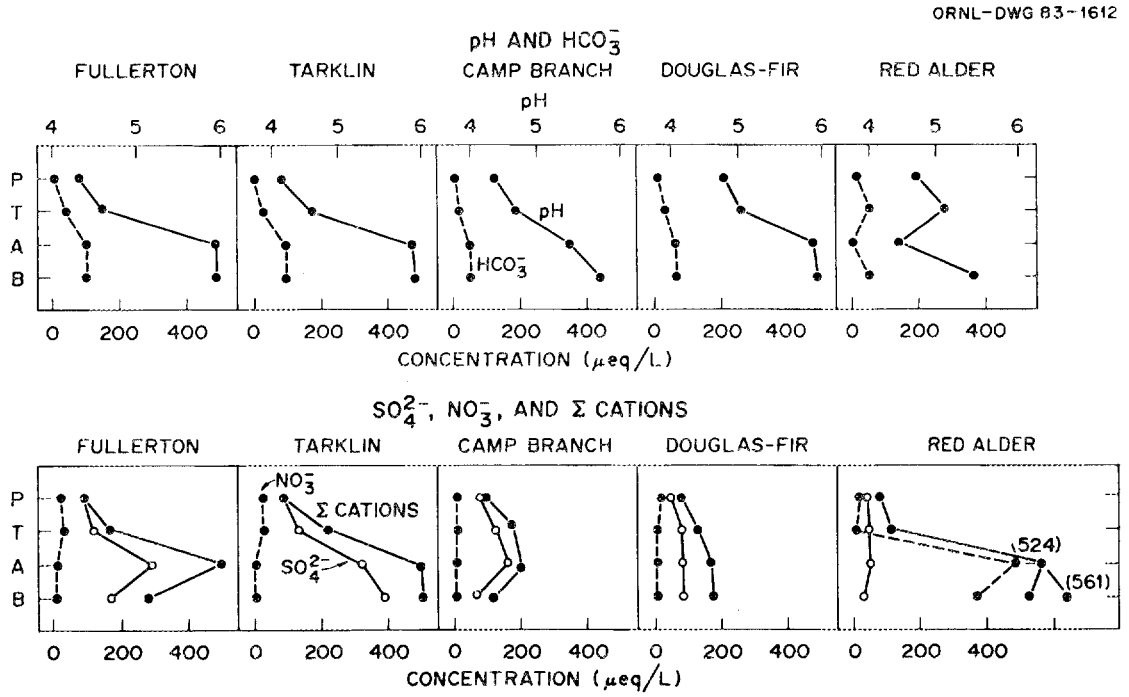

Figure 3-1. pH, HCO_3^-, SO_4^{2-}, NO_3^-, and Σ cations in soil solutions from five forested research sites. P = precipitation, T = throughfall, A = A horizon soil solution, B = B horizon soil solution.

Annual fluxes for the five sites illustrate the interactive effects of internal H_2CO_3 and HNO_3 generation, atmospheric H^+ and SO_4^{2-} deposition, and SO_4^{2-} adsorption on total H^+ input to these ecosystems and net cation leaching from them. All sites accumulate incoming H^+, which is either absorbed along with SO_4^{2-} and NO_3^-, neutralized along with adsorption of SO_4^{2-} or exchanged for base cations that leach from soils with SO_4^{2-} or NO_3^-. Bulk precipitation inputs of SO_4^{2-} to the Camp Branch and Washington red alder sites exceed leaching losses, whereas the Tarklin site shows a large net SO_4^{2-} loss. The Fullerton and Douglas-fir sites appear to be approximately in balance with respect to SO_4^{2-} (Table 3-1). However, according to complementary studies by Lindberg et al. (1979) on Walker Branch, dry deposition of sulfur adds 50 to 100% to the sulfur inputs via bulk precipitation at the Fullerton and Tarklin sites. Thus, it appears that the Fullerton soil is still accumulating SO_4^{2-}, although the rate of accumulation seems to be decreasing (see Temporal Trends in Solution Concentration and Flux, this section).

Table 3-1

INPUT AND OUTPUT OF H^+, SO_4^{2-}, NO_3^-, AND HCO_3^- FROM EPRI INTENSIVE STUDY SITES

	H^+	SO_4^{2-}	NO_3^-	HCO_3^-	Cations[a]	H_2O
	(keq•ha^{-1}•year^{-1})					(cm)
Walker Branch, Tennessee[b]						
Bulk Precipitation	0.69	0.92	0.14	0	1.31	131
Fullerton						
Leaching (80 cm)	<0.01	1.22	<0.01	0.58	1.78	61
Balance	+.69	-0.30	+0.14	-0.58	-0.47	+70
Internal H^+[c]	0.58					
Tarklin						
Leaching (50 cm)	<0.01	2.75	<0.01	0.50	2.75	61
Balance	+.69	-2.83	+0.14	-0.50	-1.44	+70
Internal H^+[c]	0.50					
Camp Branch, Tennessee[d]						
Bulk Precipitation	0.55	0.62	0.22	0	1.07	140
Leaching (100 cm)	0.01	0.44	<0.01	0.29	0.66	70
Balance	+0.55	+0.18	+0.22	-0.29	+0.41	+70
Internal H^+[c]	0.16					
Thompson, Washington[e]						
Bulk Precipitation	0.33	0.52	0.16	0.16	1.19	166
Douglas-Fir						
Leaching (40 cm)	0.05	0.76	0.01	0.56	1.61	114
Balance	+0.28	-0.24	+0.15	-0.40	-0.42	-52
Internal H^+[c]	0.40					
Red Alder						
Leaching (40 cm)	0.06	0.36	3.79	0.42	4.97	128
Balance	+0.27	+0.17	-3.64	-0.26	-3.77	-38
Internal H^+[2]	3.90					

[a] $Ca^{2+} + Mg^{2+} + K^+ + Na^+ + H^+$.

[b] From May 1981 to April 1984.

[c] From internal H_2CO_3 and HNO_3 production.

[d] From May 1981 to April 1983.

[e] From January 1981 to May 1984.

The sulfate absorption isotherms for soils from the five sites are shown in Figure 3-2. (Plots of absorbed SO_4^{2-} do not include current SO_4^{2-} content, only the addition SO_4^{2-} adsorption with increasing solution SO_4^{2-} concentration.) Surprisingly, the red alder soil had the highest sulfate adsorption capacity, followed by the Lily and Douglas-fir soils. Fullerton and Tarklin soils showed the lowest adsorption. Reasons for the high sulfate adsorption capacity in the red alder soil are not fully understood. Amorphous plus iron and aluminum oxide contents of the Douglas-fir and red alder soils are nearly identical (Table 3-2). The red alder soil has lower pH and percent base saturation (%BS) in surface horizons but not in the B horizon. Further aspects of the properties providing the red alder soil its high adsorption capacity will be explored in the new EPRI-funded project "Integrated Forest Study on Effects of Atmospheric Deposition."

The extent to which a site accumulates SO_4^{2-} (by soil adsorption) from atmospheric deposition depends on both the SO_4^{2-} adsorption capacity of the soil and the soil solution SO_4^{2-} concentration. The steeper the slope of the adsorption isotherm, the greater the adsorption capacity of a soil at a given solution concentration. Thus, the differences in SO_4^{2-} accumulation between the red alder (accumulating SO_4^{2-}) and the Douglas-fir (not accumulating) sites, both of which receive the same atmospheric sulfur inputs, are consistent with differences in soil SO_4^{2-} adsorption properties (Table 3-1; Figure 3-2). Similarly, the differences between the Fullerton (accumulating SO_4^{2-}) and the Tarklin (not accumulating) sites are consistent with differences in soil SO_4^{2-} adsorption properties, at least at the low concentration ranges (Table 3-1; Figure 3-2). The amount of SO_4^{2-} retention in the field cannot be directly compared to soil adsorption properties of the Fullerton and Douglas-fir sites, for example, because they receive very different atmospheric inputs.

Those sites which accumulate SO_4^{2-} and NO_3^- are not fully susceptible to leaching caused by atmospheric H_2SO_4 and HNO_3 inputs. Adsorption of SO_4^{2-} causes OH^- release and the creation of a cation exchange site (Rajan 1978); thus, both incoming SO_4^{2-} and H^+ are retained (or H^+ is neutralized by OH^- release) without concomitant base cation loss. Similarly, biological NO_3^- uptake results in OH^- release; thus, HNO_3 can be retained in the ecosystem without concomitant base cation loss.

The net production of HCO_3^- and NO_3^- must be accompanied by an equivalent net production of H^+ within the soil because of carbonic and nitric acid formation, respectively. Figure 3-3 illustrates carbonic acid formation and leaching. As incoming rain enters the CO_2-enriched soil atmosphere, H_2CO_3 is formed and partly dissociates into H^+ and HCO_3^-. The H^+ displaces base

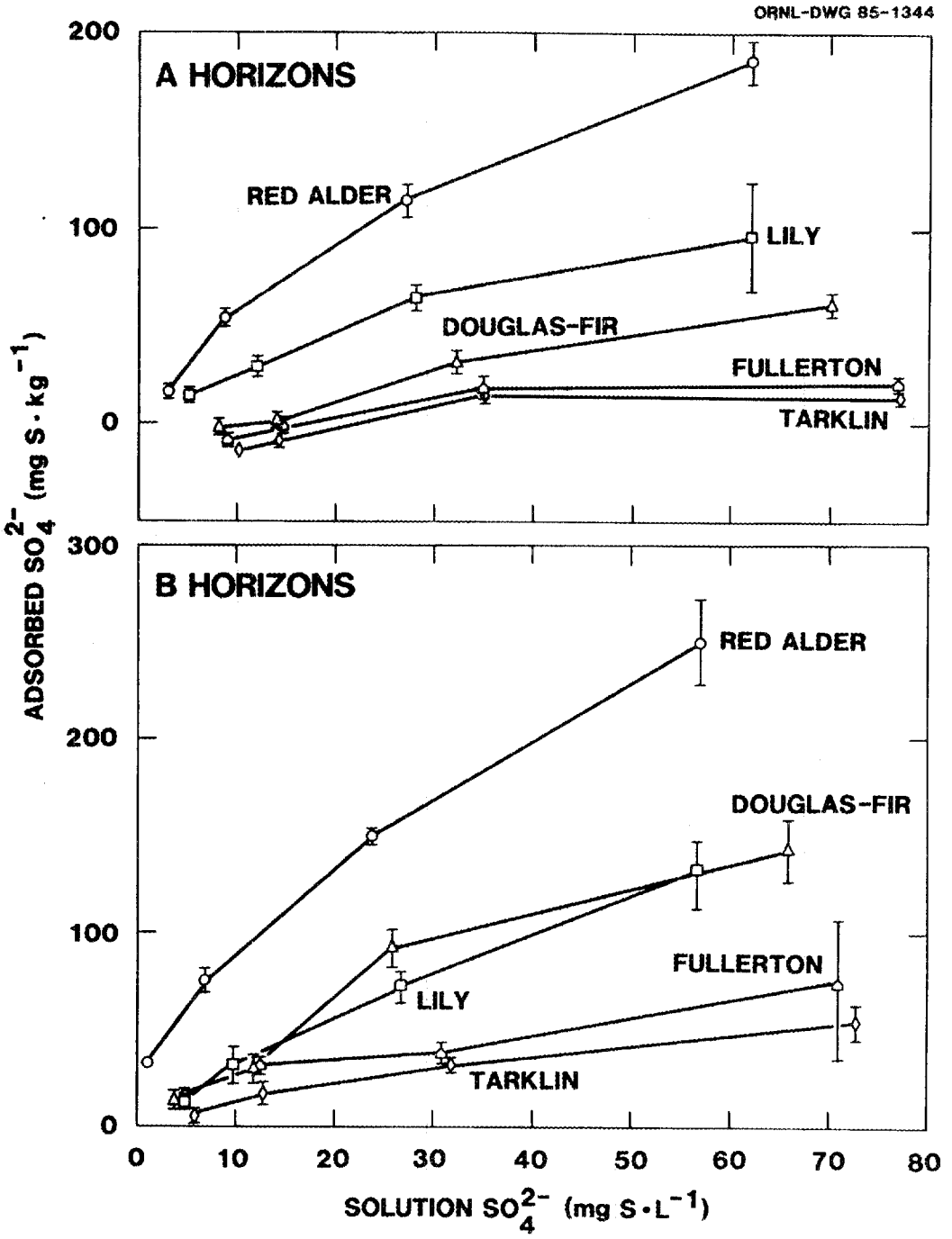

Figure 3-2. Sulfate adsorption isotherms for the soils from the five EPRI study sites (Fullerton and Tarklin from Walker Branch, Tennessee; Lily from Camp Branch, Tennessee; and Alderwood series with red alder and Douglas-fir vegetation at the Thompson Site, Washington). Absorbed SO_4^{2-} does not include SO_4^{2-} initially present in the soil but only that accumulated during SO_4^{2-} additions.

Table 3-2

INPUT AND OUTPUT OF BASE CATIONS FROM EPRI INTENSIVE STUDY SITES

	H^+	Ca^{2+}	Mg^{2+}	K^+	Na^+	Base Cations
	(keq ha^{-1} or keq ha^{-1} $year^{-1}$)					
	Walker Branch, Tennessee[a]					
Bulk Precipitation	0.69	0.35	0.08	0.05	0.05	0.53
Fullerton						
Leaching (80 cm)	<0.01	0.27	0.28	0.06	1.16	1.78
Balance	+0.69	+0.08	-0.20	-0.01	-1.11	-1.25
Soil Exchange	557	45	26	8.4	-	79.4
Tarklin						
Leaching (50 cm)	<0.01	1.49	0.47	0.10	0.68	2.74
Balance	+0.69	-1.14	-0.39	-0.05	-0.63	-2.21
Soil exchange	126	84	17	9	-	110
	Camp Branch, Tennessee[b]					
Bulk Precipitation	0.55	0.26	0.12	0.06	0.08	0.52
Leaching (100 cm)	0.02	0.09	0.28	0.03	0.24	0.64
Balance	+0.53	+0.15	-0.16	0.03	-0.16	-0.12
Soil exchange	274	13.4	21.1	6.6	1.2	42.3
	Thompson, Washington[c]					
Bulk Precipitation	0.33	0.18	0.08	0.05	0.55	0.86
Douglas-fir						
Leaching	0.05	0.50	0.28	0.06	0.73	1.57
Balance	+0.28	-0.31	-0.20	-0.01	-0.18	-0.71
Soil exchange	214	21	3.6	4.9	-	29.5
Red alder						
Leaching	0.06	2.77	0.80	0.39	0.95	4.91
Balance	+0.27	-2.58	-0.72	-0.34	-0.40	-4.05
Soil exchange	159	19	2.1	4.5	-	25.7

[a]May 1981 - April 1984.

[b]May 1981 - April 1983.

[c]January 1981 - May 1984.

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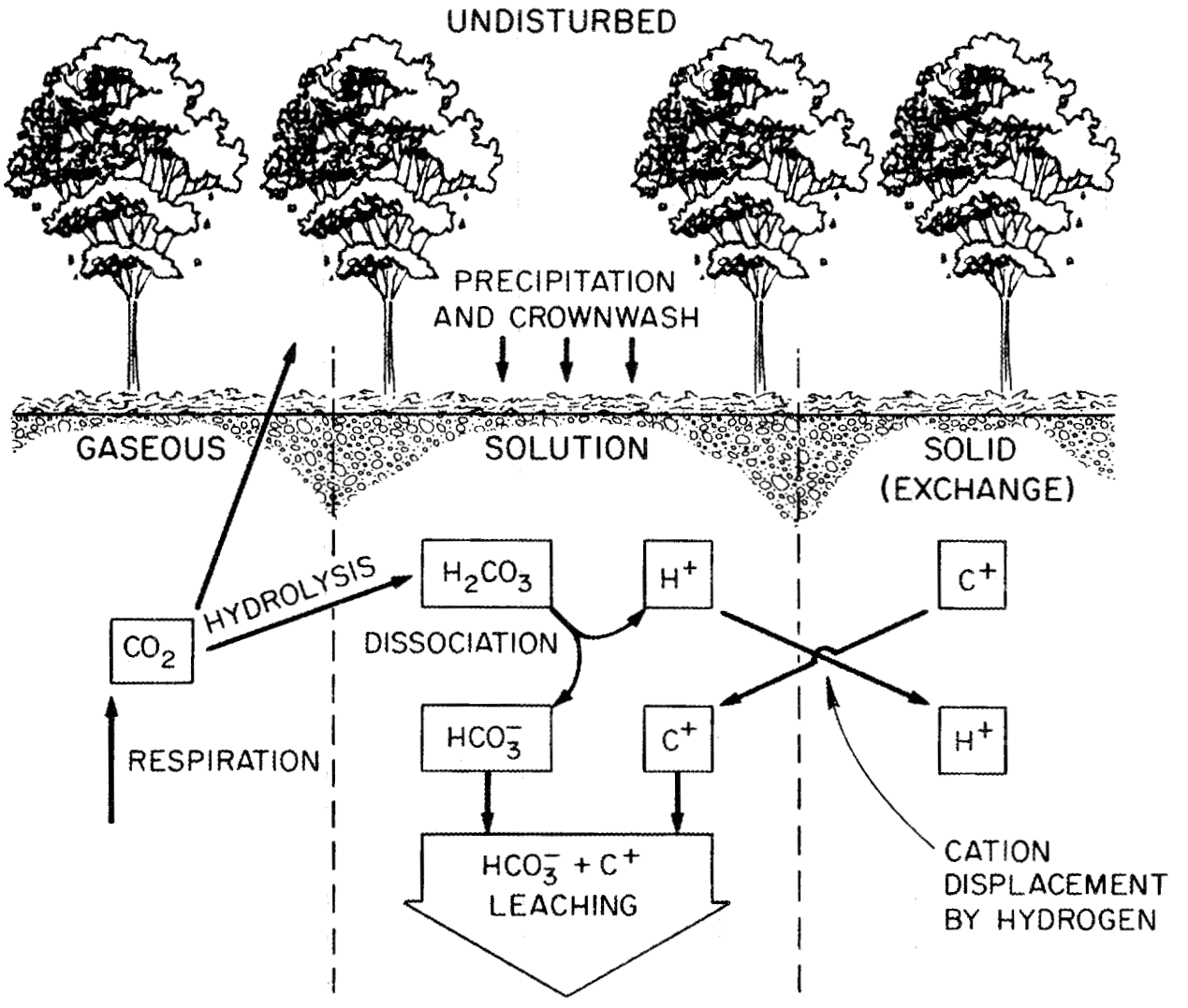

Figure 3-3. Schematic diagram of carbonic acid leaching in soils (C^+ = base cation).

cations and a cation-HCO_3^- solution leaches from the system. Similarly, H^+ is produced during nitrification, as illustrated in Figure 3-4. In both cases the H^+ does not appear in solution but displaces base cations, just as atmospherically deposited H^+ does. Thus, we can calculate internal H^+ production by internal H_2CO_3 and HNO_3 formation from the net output of HCO_3^- and NO_3^-, respectively, and compare this to measured atmospheric H^+ input (Table 3-1). At all but the red alder site, atmospheric H^+ input and internal production by HCO_3^- formation are within 50% of one another, but internal NO_3^- production at the red alder site greatly exceeds H^+ inputs from atmospheric input or H_2CO_3 production at any site, even the ones most heavily impacted by acid deposition. Clearly, nitrification in a nitrogen-saturated ecosystem such as the red alder site

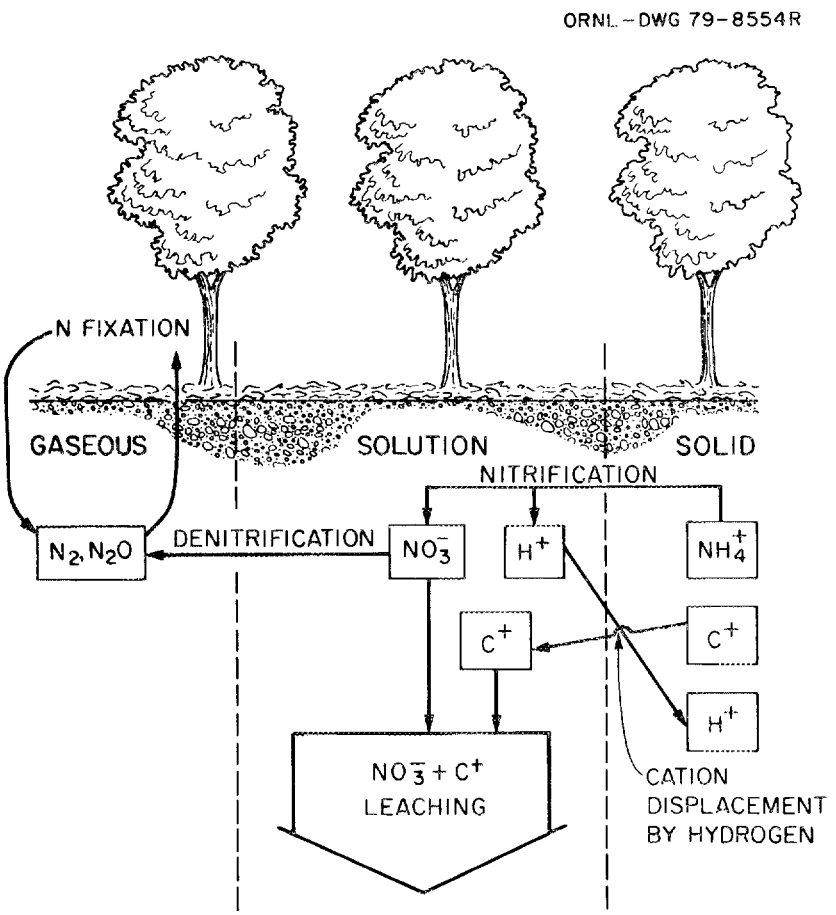

Figure 3-4. Schematic diagram of nitrification and nitrate leaching in soils (C^+ = base cation).

is a powerful acidifying agent that must be accounted for in acid deposition assessments.

All sites show a net loss of base cations by leaching, but the rates of loss and the contributions of atmospheric H^+ inputs vs internal H^+ production to this loss vary quite considerably among the sites (Table 3-2). The Fullerton and Tarklin sites have the same atmospheric H^+ inputs and similar internal HCO_3^- generation rates, yet net cation losses differ by a factor of 3 due primarily to the differences in SO_4^{2-} mobility between these soils. The Douglas-fir and red alder stands also have the same atmospheric inputs and very similar internal HCO_3^- generation rates, but the rates of base cation loss differ by sixfold due to internal NO_3^- production in the red alder soil. The Camp Branch site has relatively low atmospheric inputs, internal H^+ generation, and cation loss rates (the latter in part due to considerable SO_4^{2-} adsorption also).

CATION LOSSES AND SOIL CATION RESERVES

Of what significance are net cation losses to these ecosystems? The simplest way to assess this is to compare net losses with exchangeable cation capital. Thus, the net leaching losses of base cations for Douglas-fir, red alder, Camp Branch, Fullerton, and Tarklin sites equal 2.4, 15.8, 0.3, 1.6, and 2.0%, respectively, of exchangeable cation capital (Table 3-3). Because soil exchangeable cations are continually replaced by weathering, most of these loss rates are judged to be

Table 3-3

NET CATION LEACHING LOSSES AS A PERCENT OF EXCHANGEABLE CATION RESERVES IN EPRI INTENSIVE STUDY SITES

Site	Soil Depth (cm)	Ca^{2+} (%)	Mg^{2+} (%)	K^+ (%)	Base Cations (%)
Walker Branch					
Fullerton	80	0	0.8	0.01	1.6
Tarklin	50	1.4	2.3	0.6	2.0
Camp Branch	60	0	0.7	0	0.3
Thompson Site					
Douglas-fir	50	1.5	5.6	0.2	2.4
Red alder	50	13.5	34.3	7.6	15.8

insignificant. The exceptionally large rate of cation loss in the red alder stand (due to nitrification and nitrate leaching) has apparently caused a significant reduction in pH and exchangeable cation capital, however. As compared to the adjacent Douglas-fir stand, surface soil pH is 0.4 units lower (4.6 vs 5.0) and base saturation is 50% lower (15 vs 27%) in the red alder stand (Table 3-4). (The similarities in cation content on a soil volume basis in Table 3-3 are due to estimated differences in percent gravel content.)

Although all sites show a net loss of total base cations, some individual cations are conserved at certain sites. The Fullerton site shows a slight gain of Ca^{2+} and a slight net loss of K^+, but large losses of Mg^{2+} and especially Na^+ (Table 3-2). The Tarklin site shows net losses of all base cations larger than those at the Fullerton site, except for Na^+. We do not know the source of Na^+ in subsoils at these sites at this time. Contamination from nearby Na^+-treated plots is possible, but spatial and temporal trends in Na^+ concentration do not support this, nor do dye application tests indicate the existence of subsurface horizontal flow between treated and untreated plots. Furthermore, exchangeable Na^+ in control plots has not changed since Na^+ salt application in nearby treated plots (D. W. Johnson, ORNL, unpublished data). Total soil analyses indicate large nonexchangeable Na^+ reserves in these soils (D. W. Johnson, ORNL, unpublished data), and the availability of this Na^+ for leaching will be explored in our new EPRI proposal.

The Camp Branch site has net losses of Mg^{2+} and Na^+, a similar pattern to that at the Fullerton site (Table 3-2). Small accretions of Ca^{2+} and K^+ are found at this site. The relatively low leaching losses of base cations from this soil are attributed to relatively low SO_4^{2-} inputs compared to the other Tennessee sites, SO_4^{2-} adsorption by subsoils, and relatively low rates of HCO_3^- leaching.

The Douglas-fir and red alder sites show net losses of all base cations, but losses from the red alder soil are much larger in every case. A significant net loss of Na^+ is noted in the red alder soil as was the case in the Tennessee soils.

In terms of the percent of exchangeable reserves, the net losses of Ca^{2+}, Mg^{2+}, and K^+ are much higher in the red alder soil than in any other case (Table 3-3). This is especially striking in the case of Mg^{2+}, where only a 3-year turnover time is indicated for exchangeable reserves. Weathering rates must be fairly high in the red alder soil to keep exchangeable cation reserves from being very rapidly (3-10 years) depleted. It is clear, however, that weathering processes have not

Table 3-4

CATION EXCHANGE CAPACITY (CEC), pH, EXCHANGEABLE CATIONS, AND BASE SATURATION AND AMORPHOUS IRON AND ALUMINUM OF SOILS (<2 mm FRACTION) FROM EPRI INTENSIVE STUDY SITES (FROM JOHNSON et al. 1981)

Horizon	pH	CEC[a]	Ca^{2+}	Mg^{2+}	K^+	BS[b]	Fe[c]	Al[c]
		-----(meq 100 g^{-1} <2 mm fraction)-----				(%)	(%)	(%)
				Fullerton - Chestnut Oak				
A1	4.6	6.2	3.28	0.27	0.23	61	0.94	0.39
A2	4.9	2.0	0.19	0.03	0.07	15	0.49	0.17
B1	4.6	5.4	0.22	0.11	0.07	7	1.10	0.20
B2	4.8	13.6	0.42	0.66	0.11	9	2.49	0.33
				Tarklin - Yellow-Poplar				
A1	4.6	7.6	4.52	0.68	0.41	74	0.51	0.15
A2	5.0	3.8	1.85	0.33	0.21	63	0.47	0.18
B1	4.7	3.7	0.69	0.19	0.09	26	0.63	0.18
B2	4.7	3.9	0.95	0.28	0.09	34	0.75	0.17
				Lily - Scarlet Oak				
A	4.2	3.5	0.20	0.11	0.09	12	0.72	0.24
B1	4.4	3.5	0.13	0.11	0.06	9	1.19	0.30
B2	4.6	3.9	0.15	0.37	0.08	16	1.60	0.30
				Alderwood - Douglas-Fir				
A1	5.0	12.3	2.66	0.32	0.33	27	1.30	0.80
A2	5.2	9.1	1.01	0.16	0.19	15	1.52	0.91
B21	5.1	8.1	0.47	0.12	0.14	9	1.37	0.55
B22	5.2	5.2	0.27	0.10	0.14	10	1.18	0.43
C	5.1	7.6	0.26	0.17	0.26	8	0.94	0.41
				Alderwood - Red Alder				
A1	4.6	10.0	1.03	0.16	0.30	15	1.31	0.84
A2	4.8	9.8	1.18	0.13	0.23	16	1.52	0.91
B21	5.2	7.4	1.14	0.10	0.21	20	1.22	0.65
B22	5.2	6.2	0.81	0.11	0.17	18	1.03	0.45
C	5.1	9.4	0.82	0.11	0.20	12	0.96	0.39

[a] NH_4Cl extraction.

[b] % base saturation.

[c] Citrate-dithionite extraction (amorphous plus oxide forms).

kept pace with leaching, because base saturation and pH of surface horizons have been reduced relative to the Douglas-fir soil (Table 3-4). There is evidence that the red alder soil has higher concentrations of exchangeable Ca^{2+} and K^{+} in B horizons than the Douglas-fir soil in the <2-mm soil fraction (Table 3-4) (Van Miegroet and Cole 1984). This indicates either (1) translocation of Ca^{2+} and K^{+} from surface to subsurface horizons, (2) high rates of weathering, or (3) both in the red alder soil.

TEMPORAL TRENDS IN SOLUTION CONCENTRATION AND FLUX

There are significant year-to-year variations in concentrations and fluxes of some ions. Since the installation of the lysimeter system, we have seen a steady rise in SO_4^{2-} concentration in the B2 horizon of the Fullerton soil (Figure 3-5). No trend was noted in A-horizon solutions (although seasonal variations were pronounced) or in the Tarklin B2-horizon solution (Figure 3-5). Seasonal variations are not reflected in either B2-horizon solution because of buffering by SO_4^{2-} adsorption.

The trend in Fullerton B2-horizon SO_4^{2-} may represent the breakthrough of SO_4^{2-} in the Fullerton soil, and we may have witnessed the transition from a sulfate-adsorbing system to a nonadsorbing (steady-state) system with all the attendant implications for increased cation leaching by atmospheric H_2SO_4

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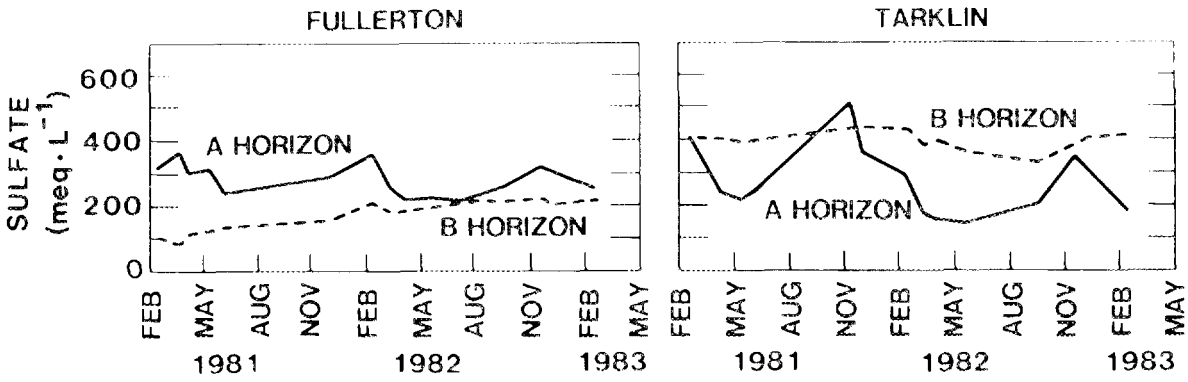

Figure 3-5. Sulfate concentrations in soil solution from the Fullerton and Tarklin sites with time.

Source: Adapted from Johnson et al. 1984.

deposition. If this is the case, it presents a unique opportunity to directly observe the effects of SO_4^{2-} mobility on cation leaching rates.

The difference in sulfur saturation is also reflected by the SO_4^{2-} concentration of solutions collected at the Washington sites, which are subject to relatively low atmospheric sulfur deposition rates. At the red alder site, SO_4^{2-} concentrations and annual sulfur flux significantly decrease from the A to the B horizon, which is indicative of net sulfur retention within that part of the ecosystem. The same pattern was observed under Douglas-fir in 1980 only, but later observation years generally show a sulfur enrichment of the B-horizon solution, suggesting desorption of SO_4^{2-}. These patterns will be further followed to see whether and to what extent they persist.

SULFUR DISTRIBUTION AND CYCLING

Atmospheric sulfur inputs exceeded forest sulfur requirements in all cases (Table 3-5). Requirement is here defined as that sulfur used to form new foliar and aboveground woody tissues. Adding belowground tissues in the Tennessee sites may increase requirement estimates by about one-third (Johnson et al. 1982), but this still remains well below even bulk precipitation sulfur inputs in all cases. No estimates of belowground sulfur requirements are available for the Washington sites at this time, but it is unlikely that this information would cause a major change in the relationship between sulfur input and annual sulfur requirement.

Two additional considerations lend support to the assertion that atmospheric sulfur inputs exceed forest sulfur requirements in these ecosystems. First, bulk precipitation inputs underestimate true sulfur inputs via dry deposition. This is evident when comparing foliage sulfur content with net sulfur fluxes from the crown (i.e., leaf litterfall + crownwash + stemflow) (Table 3-6). Dividing foliage content by net sulfur flux from the crown gives the number of times the foliar sulfur pool would have to turn over each year to account for net sulfur fluxes from the crown if net sulfur flux from the crown was assumed to be part of the internal sulfur cycle. The figures for the three Tennessee sites range from 2.1 to 3.1 turnovers per year. No other nutrient shows such an unrealistically high cycling rate, and thus most net SO_4^{2-} flux from the crown can be assumed to be due to dry depositions at these sites. At the Washington sites the situation is far less clear in that the Douglas-fir stand retains 3 to 4 years' needles, confounding the foliar turnover calculation somewhat. It is highly

Table 3-5

FLUXES OF SULFUR IN FIVE FOREST ECOSYSTEMS (kg ha^{-1} $year^{-1}$)

Component	Fullerton	Tarklin	Camp Branch	Douglas-Fir	Red Alder
Bulk precipitation	16.0	16.0	9.9	8.4	8.4
Crownwash[a]	9.7	7.0	3.5	0.5	0
Stemflow	6.2	3.1	2.2	1.4	0
Litterfall					
Leaf[b]	6.9	5.6	3.3	3.1	5.4
Other	0.1	0.2	0.04	0.6	4.9
Total	7.0	5.8	3.34	3.7	10.3
Wood increment	0.3	0.5	0.4	1.2	0.5
Requirement[c]	8.7	7.5	3.3	4.8	4.6
Leaching	18.7	43.5	7.0	12.2	5.7

[a]Crownwash = throughfall - bulk precipitation.

[b]Collected in littertraps.

[c]Requirement = increment in new foliage + wood.

Table 3-6

FOLIAGE SULFUR CONTENT AND NET SULFUR RETURN FROM CANOPIES AT FOREST SITES

Component	Fullerton	Tarklin	Camp Branch	Douglas-Fir	Red Alder
Foliage sulfur content (kg ha^{-1})	8.4	7.5	2.9	25.9	4.1
Net Sulfur Removal from canopy (kg ha^{-1} $year^{-1}$)					
Leaf litterfall	6.9	5.6	3.3	3.1	5.4
Crownwash	9.7	7.0	3.5	0.5	0
Stemflow	6.2	3.1	2.2	1.4	0
Total	22.8	15.7	9.0	5.0	5.4
Foliage Turnover[a] ($year^{-1}$)	2.7	2.1	3.1	0.2	1.3

[a]Net sulfur removal ÷ foliage sulfur content.

noteworthy that in the red alder stand the foliar sulfur turns over close to once a year, while no net foliar leaching by crownwash can be observed. This implies a lack of both dry deposition and internal SO_4^{2-} leaching. The latter is consistent with the observation that the red alder stand has virtually no foliar SO_4^{2-}-S pool (Tables 3-7 and 3-8). This is no doubt due to the enriched nitrogen status at the red alder site. By contrast, the adjacent Douglas-fir stand has fully 42% of its foliar sulfur in SO_4^{2-} form. The Tennessee sites have only 24 to 33% of their foliar sulfur as SO_4^{2-}, a somewhat surprising result since sulfur inputs are higher than those at the Washington sites.

The total sulfur distributions at these five sites exhibited some basic similarities as well as some specific differences. The soil was the major pool in all cases, constituting 88 to 97% of total ecosystem sulfur content. For all sites it is clear that wood, not foliage, is the major aboveground sulfur pool in trees. The understory constitutes a large sulfur pool in the red alder stand, perhaps due in part to its advanced stage of stand development (i.e., the alder overstory is beginning to break up). The understory is less important at the Douglas-fir site and virtually absent at the Tennessee sites.

Forest floor sulfur content (in midsummer) was roughly equal to aboveground tree sulfur content in the Fullerton and Tarklin sites but more than tree content at the Camp Branch and the red alder sites. Forest floor sulfur content was very large at the red alder site, primarily due to sword fern litter.

The SO_4^{2-} distributions in these five sites paralleled the total sulfur distributions in many ways, but differed in specific ways related to soil SO_4^{2-} adsorption properties and nitrogen-sulfur interactions. The differences in SO_4^{2-} content between the Fullerton and Tarklin soils are apparently due to differences in SO_4^{2-} adsorption capacity, which in turn are due to differences in Fe + Al oxide content (Richter et al. 1983) (Table 3-4). The differences in SO_4^{2-} content between the Douglas-fir and red alder soils are more likely due to differences in nitrogen status, because soil mineralogy is similar and the red alder soil has a greater SO_4^{2-} adsorption capacity (Table 3-2; Figure 3-2). One of the factors that may contribute to the greater SO_4^{2-} adsorption capacity of the red alder soil is its lower initial SO_4^{2-} content than the Douglas-fir soil (Table 3-7).

It has long been known that nitrogen status affects SO_4^{2-} content of vegetation (Kelly and Lambert 1972, Turner et al. 1980). Our urea fertilization vegetation verified the accepted pattern of reduced foliar SO_4^{2-} with increasing nitrogen,

Table 3-7

DISTRIBUTION (in kg ha^{-1}) OF TOTAL SULFUR AND SO_4^{2-}-S IN FIVE FOREST ECOSYSTEMS.

Component	Fullerton	Tarklin	Camp Branch	Douglas-Fir[a,b]	Red Alder[a]
			Total S		
Foliage	8.4	7.5	2.9	25.9	4.2
Woody	26.6	18.5	9.2	62.1	45.2
Tree	35.0	26.0	12.1	88.0	44.4
Understory	nil	nil	nil	3.9	15.7
Aboveground	35.0	26.0	12.1	91.9	65.1
Forest Floor	34.9	24.1	22.8	15.9	80.5
Soil	1306	1378	1545	1412	1099
Σ Ecosystem	1376	1428	1570	1520	1245
			SO_4^{2-}		
Foliage	2.5	2.5	0.7	10.9	0.1
Woody	6.1	5.3	2.3	8.3	4.1
Tree	8.6	7.8	3.0	19.3	4.2
Understory	nil	nil	nil	1.4	3.6
Aboveground	8.6	7.8	3.0	20.6	7.8
Forest Floor	1.0	0.8	0.9	2.4	3.9
Soil	226	142	460	162	67
Σ Ecosystem	236	151	464	185	78.7

[a]Woody biomass calculated using allometric equations developed by Gholz et al. (1979).

[b]Including few alder trees in the Douglas-fir stand.

Table 3-8

EFFECTS OF NITROGEN FERTILIZATION OF FOLIAR SULFUR CONCENTRATION (in %)

	Total S		Organic S		SO_4^{2-}	
	Control	Urea	Control	Urea	Control	Urea
Fullerton	0.18	0.16	0.12	0.13	0.054	0.032
Tarklin	0.20	0.14	0.14	0.12	0.065	0.026
Camp Branch	0.12	0.12	0.09	0.10	0.030	0.026
Red Alder	0.103	0.131	0.10	0.128	<0.003	<0.003
Douglas-fir						
Current	0.153	0.132	0.101	0.112	0.051	0.020
Older	0.175	0.182	0.095	0.105	0.079	0.076

except in the nitrogen-rich red alder stand where foliar SO_4^{2-} levels were trace to begin with. We were not, however, able to tie up all foliar SO_4^{2-}-S in the other sites, even with repeated fertilization on the Fullerton and Tarklin sites (D. D. Richter, ORNL, unpublished data). No doubt this is due to the relatively large soil SO_4^{2-} pools at these sites. It is also interesting to note that, according to the literature on coniferous species (Kelly and Lambert 1972, Turner et al. 1977), foliar SO_4^{2-}-S in the red alder stand is at deficiency levels. Yet soil SO_4^{2-} pools are still relatively large and urea fertilization seems to result in increased organic sulfur accumulation in foliage (Table 3-8). In short, foliar SO_4^{2-}-S in the absence of soil SO_4^{2-}-S information may not be as good an index of ecosystem sulfur status as previously thought (Kelly and Lambert 1972, Turner et al. 1977).

EFFECTS OF H^+ AND SO_4^{2-} DEPOSITION ON DECOMPOSER INVERTEBRATE COMMUNITIES.

Two and ten times the annual ambient inputs of both SO_4^{2-} alone and H^+ plus SO_4^{2-} (acid-sulfate) were applied at 5-week intervals to the forest floor of Walker Branch Watershed. Over a 14-month period, densities of litter macroarthropods averaged 19% lower in the high SO_4^{2-} treatment compared to the control (Figure 3-6). Soil arthropod populations increased significantly in response to the low SO_4^{2-} and to both acid-SO_4^{2-} treatments (Table 3-9). Fungivores and predators in the soil and litter exhibited responses similar to those observed in the total populations. The treatments produced chronic and apparent acute effects on forest floor arthropods. Chronic effects on litter macroarthropod populations were observed in the high SO_4^{2-} treatments (Figure 3-6). Treatment effects on litter populations were especially severe during late summer and early autumn, normally a time of environmental stress caused by low soil and litter moisture.

Graphical inspection revealed apparent acute treatment effects in June 1981 and January 1982 when arthropod sampling took place 3 to 7 d after treatments were applied. In both months, the low SO_4^{2-} treatment apparently increased soil arthropod densities, while both ten-times treatments decreased densities compared to the control. The effect is illustrated for June in replicate 1 of Figure 3-7.

The chronic effects of the high SO_4^{2-} treatment probably resulted from the large K^+ additions to the forest floor (438 kg ha^{-1} $year^{-1}$). Potassium sulfate was used in the SO_4^{2-} applications. The apparent acute effects of both ten-times treatments also probably resulted from concentrated additions of

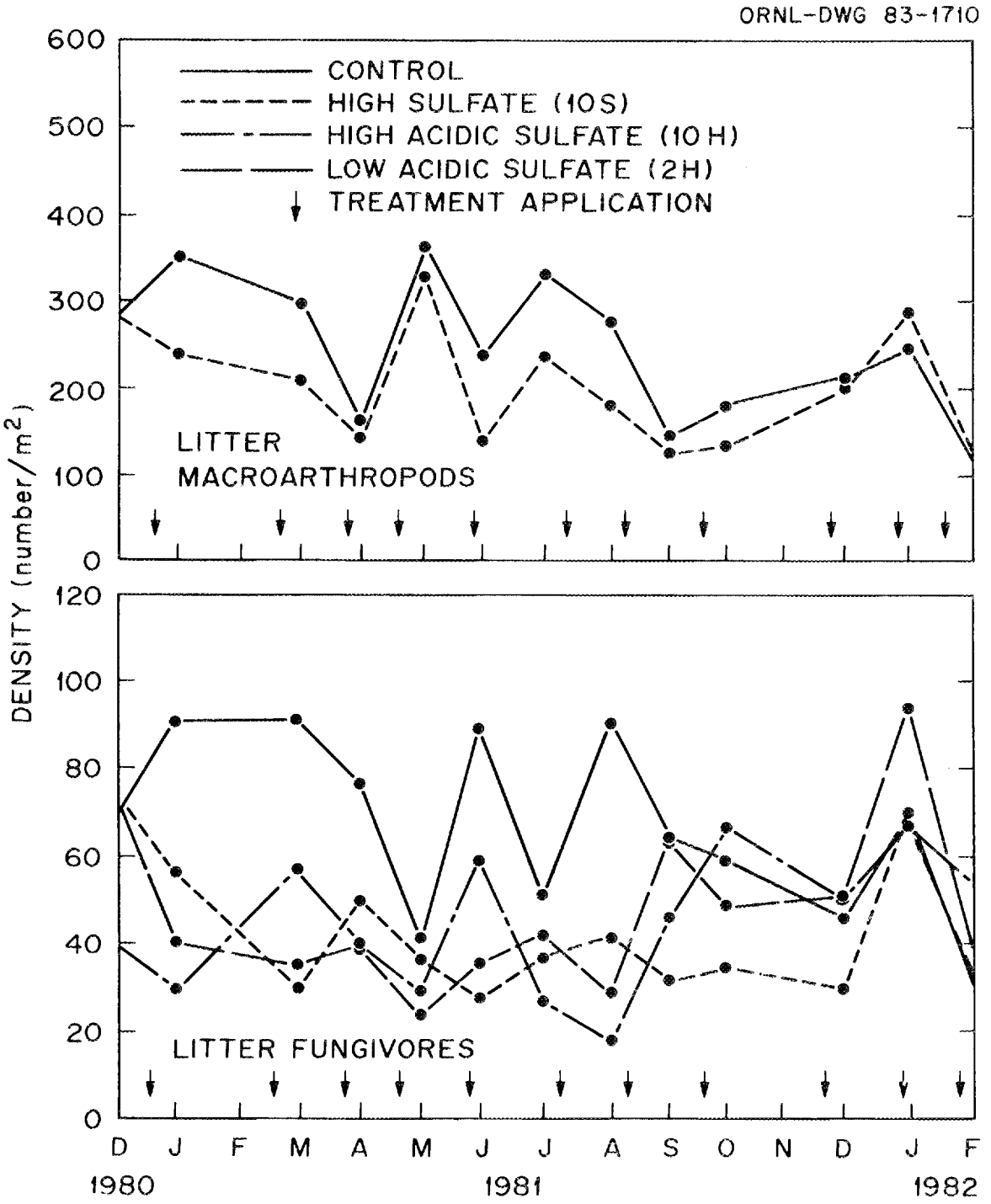

Figure 3-6. Monthly litter macroarthropod and litter fungivore densities in the high sulfate and control, high sulfate, low and high acidic sulfate, and control treatments, respectively.

Table 3-9

MEANS (NUMBER/m^2) AND RESULTS OF DUNCAN'S MULTIPLE RANGE TEST FOR SOIL MESOFAUNA (C=CONTROL, 2S= LOW SULFATE, 10S = HIGH SULFATE, 2H = LOW ACID SULFATE, AND 10H = HIGH ACIDIC SULFATE INPUT)

Treatment	Mean	Duncan Grouping
2S	42,900	A
2H	41,500	A
10H	40,000	A
C	35,900	B
10S	35,100	B

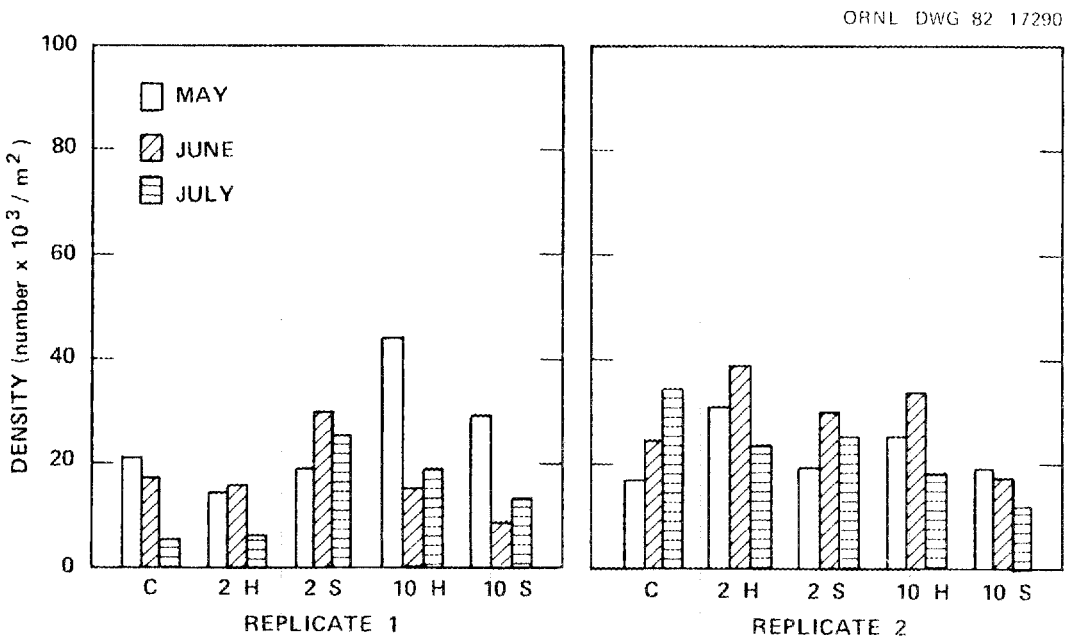

Figure 3-7. Mean soil mesofauna densities in the five treatments in May, June, and July. (C = control, 2S = low sulfate, 10S = high sulfate, 2H = low acidic sulfate, 10H = high acidic sulfate.)

K^+ to the forest floor. The overall positive response of soil mesofauna to the high acid-SO_4^{2-} and to both two-times treatments may be due to the fact that small additions of K^+ fertilizers have a beneficial effect on forest floor arthropods (e.g., Cary et al. 1971). Soil arthropods were not adversely affected by the hydrogen ion additions, probably because most of the H^+ was buffered by the litter.

The treatments may have indirectly affected forest floor arthropods by influencing microfloral activity. Respiration measurements taken in the litter showed that evolution of carbon dioxide was 26% less in the high SO_4^{2-} treatment than in the control. Analysis of soil nutrients showed increased levels of K^+ and SO_4^{2-} corresponding to the 2- and 10-times treatments. Soil pH, Al^{3+}, and available phosphorus appeared unaffected by the treatments. Soil pH was unaffected by the acid-SO_4^{2-} treatments because most of the additional H^+ was neutralized by the forest floor (Figure 3-8). Analysis of litter leachate during a single treatment interval (Figure 3-9) indicated that only 31% of the H^+ applied in the high acid-SO_4^{2-} treatment reached the mineral soil. Thus, even after 18 months of treatments, the forest floor litter still served as an excellent buffer against large increases in H^+ inputs to the soil environment. Beginning in April 1982, we increased the H^+ and SO_4^{2-} additions in the two-times treatments by a factor of ten. In December 1982 and again in June 1983, we noted a small (about 5%) but statistically significant decrease in numbers of soil microfauna in these new treatments. Because we have not observed decreased pH or phosphorus or increased Al^{3+} concentrations in soil samples from these plots, we cannot be certain that the effect is directly related to treatments. We noted an apparent reduction of litter on the 20-times plots compared to other plots, which could be a treatment effect and could lead directly (e.g., lower solution pH) or indirectly (e.g., increased drying of soil) to decreases in soil fauna. In any case, such a small decrease would be unlikely to exert significant effect on nutrient cycling and apparently only occurs under conditions of unrealistically high acid/sulfate input.

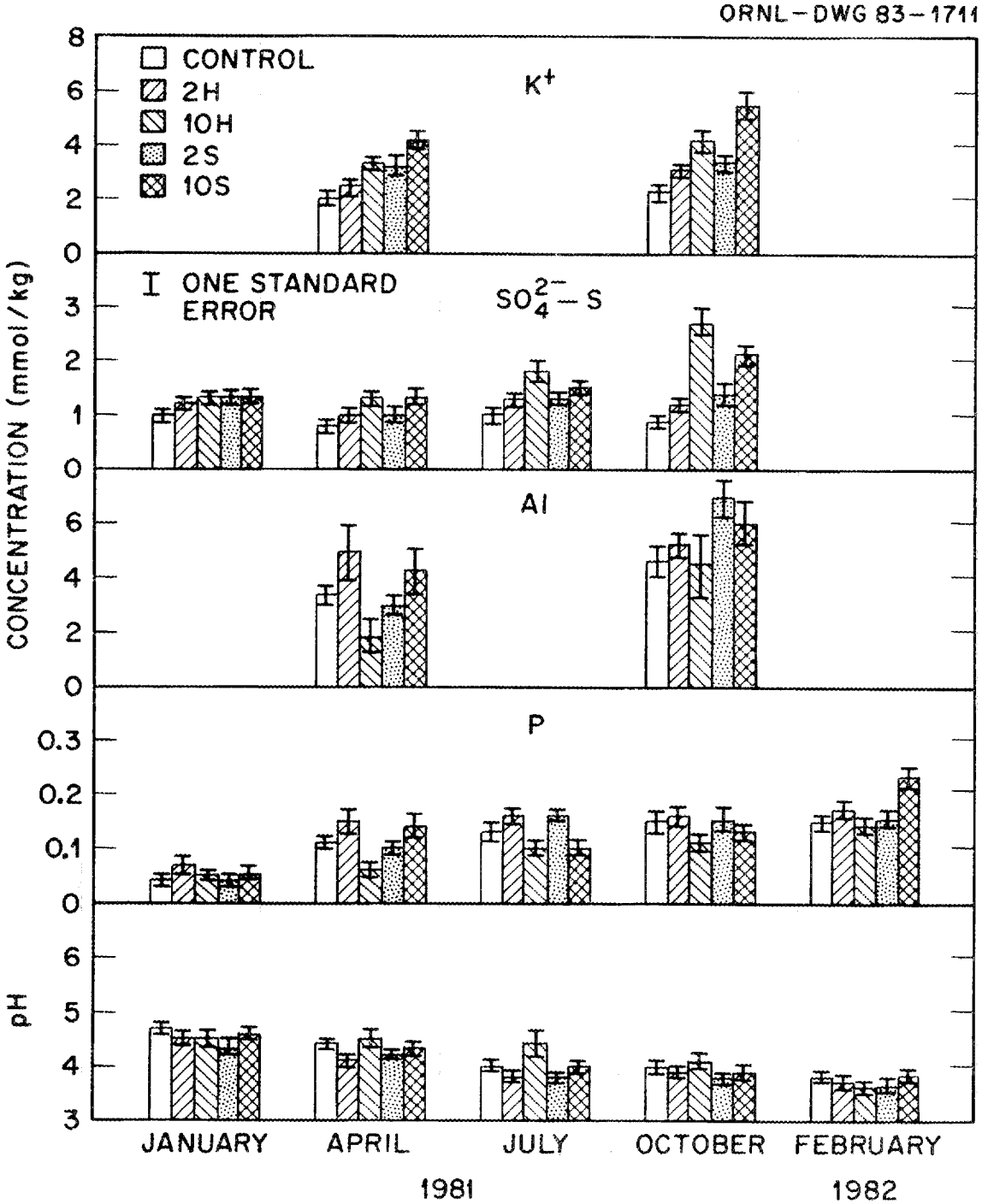

Figure 3-8. Soil exchangeable K^+, water-soluble SO_4^{2-}, exchangeable aluminum, Bray P (No. 2), and pH in the control, low acidic sulfate (2H), high acidic sulfate (10H), low sulfate (2S), and high sulfate (10S) treatments.

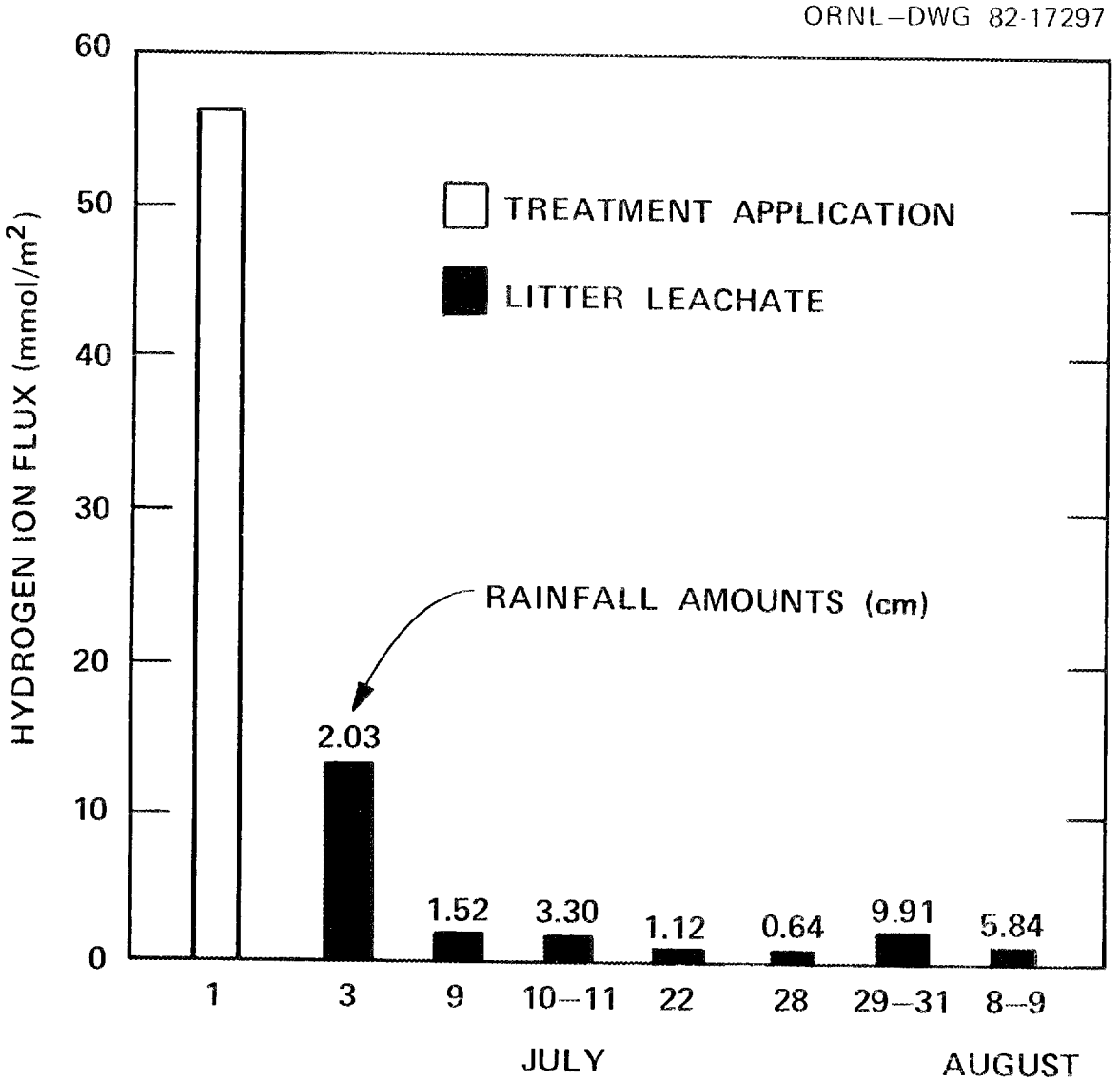

Figure 3-9. Hydrogen ion concentration in the treatment application and the litter leachate in the high acidic sulfate (10H) plots during the July-August 1982 treatment interval.

Section 4

SUMMARY AND CONCLUSIONS

Studies of acid deposition effects on soil leaching in the five forest sites (three in eastern Tennessee and two in western Washington) revealed that internal leaching processes by carbonic acid production approximately equaled atmospheric acid inputs. In a nitrogen-fixing red alder site in Washington, internal leaching by nitrification and nitric acid formation far exceeded atmospheric H^+ inputs at any site. Soil SO_4^{2-} adsorption reduced the effectiveness of atmospheric H_2SO_4 inputs on soil leaching in two of the Tennessee sites (Fullerton site and Camp Branch) and in the Washington red alder site.

In no case did atmospheric acid input cause sufficiently large rates of leaching to rapidly deplete soil exchangeable cation reserves. In most cases, annual leaching rates were $\leq$5% of soil exchangeable reserves and, with replenishment by weathering, no long-term significant soil acidification by acid deposition is foreseen. Furthermore, scarce or potentially limiting cations (such as Ca^{2+} in the Fullerton site) are being conserved despite accelerated leaching rates, whereas more abundant cations (Mg^{2+} at the Fullerton site) are preferentially leached. However, the very high natural leaching rates in the red alder stand have significantly acidified, and continue to acidify, that soil.

Atmospheric sulfur inputs exceed forest sulfur requirement in all of the five sites. Excess sulfur accumulated to a minor extent as SO_4^{2-} in vegetation, but soils were the major repository for excess SO_4^{2-} in these ecosystems. The greatly enhanced nitrogen status of the red alder stand appeared to have resulted in lowered SO_4^{2-} (and higher organic sulfur) in both vegetation and soils at that site as compared to the amounts in the adjacent nitrogen-poor Douglas-fir site. Urea-nitrogen fertilization reduced foliar SO_4^{2-} levels in all but the red alder site (where foliar SO_4^{2-} levels were trace to begin with), but nitrogen fertilization did not reduce foliar SO_4^{2-} to levels thought to be sulfur-deficient. Large soil SO_4^{2-}-S reserves may have been drawn upon to retain foliar SO_4^{2-}-S levels in fertilized plots.

Decomposer invertebrates appeared to be affected negatively by large applications of SO_4^{2-}, either as $KHSO_4$ or K_2SO_4, but were enhanced by lower applications. Forest floor buffering prevented large changes in pH with acid SO_4^{2-} treatments. Results indicate that effects of acid deposition on decomposer invertebrates are unlikely except under much higher input levels than ambient.

The assessment of acid deposition effects on these sites can be applied only to these five sites and does not necessarily represent regional patterns. However, we have identified some key processes--nitrification, carbonic acid formation, and sulfate adsorption--that mediate and determine the effects of acid deposition on these sites, and the applicability of process-level research may be tested and applied to acid deposition evaluations in a variety of forest ecosystems.

Section 5

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