

Landscape and Watershed Processes

The Effects of Throughfall Manipulation on Soil Leaching in a Deciduous Forest

D. W. Johnson,* P. J. Hanson, and D. E. Todd, Jr.

ABSTRACT

The effects of changing precipitation on soil leaching in a deciduous forest were examined by experimentally manipulating throughfall fluxes in the field. In addition to an ambient treatment (AMB), throughfall fluxes were reduced by 33% (DRY treatment) and increased by 33% (WET treatment) using a system of rain gutters and sprinklers on Walker Branch Watershed, Tennessee. Soil leaching was measured with resin lysimeters in the O horizons and with ceramic cup lysimeters in the E (25 cm) and Bt (70 cm) horizons. Large and statistically significant treatment effects on N fluxes were found in the O horizons (lower N fluxes in the DRY and higher N fluxes in the WET treatment). Together with the greater O horizon N content observed in the DRY treatment, this suggested that N was being immobilized at a greater rate in the DRY treatment than in the AMB or WET treatments. No statistically significant treatment effects on soil solution were found in the E horizons with the exception of ($\text{Ca}^{2+} + \text{Mg}^{2+}$) to K^+ ratio. Statistically significant treatment effects on electrical conductivity (EC), pH, Ca^{2+} , Mg^{2+} , K^+ , Na^+ , SO_4^{2-} , and Cl^- were found in the Bt horizons due to differences between the DRY and other treatments. Despite this, calculated fluxes of Ca^{2+} , Mg^{2+} , K^+ , Na^+ , SO_4^{2-} , and Cl^- were lowest in the DRY treatment. These results suggest that lower precipitation will cause temporary N immobilization in litter and long-term enrichment in soil base cations whereas increased precipitation will cause long-term depletion of soil base cations.

CLIMATE CHANGE could affect the cycling of nutrients and productivity of forest ecosystems in a number of ways. To date, most studies have emphasized the effects of increased temperature or elevated carbon dioxide; but changes in precipitation may have equal or greater effects (Kirschbaum, 1995). Specifically, changes in precipitation could cause (i) changes in productivity due to changes in both water and soil nutrient availability, (ii) changes in water yield, and (iii) changes in water quality (Johnson et al., 1998, 2001; Ulrich, 1983; Lamersdorf et al., 1995). Changes in soil N availability can have short-term effects on plant productivity because forests are often N limited. Ulrich (1983) noted pulses of NO_3^- and Al during warm, dry years in the Solling forest ecosystem in Germany. He hypothesized that drought intensifies N mineralization and nitrification during summer, resulting in NO_3^- pulses during rewetting periods. In acidic soils, the NO_3^- pulse is accompanied by a pulse in soil solution Al concentrations. Results of a large field research project involving the

imposition of drought through roofs in the forest subcanopy to five forest ecosystems in Europe failed to support Ulrich's (1983) hypothesis, but did suggest that natural periods of drought could produce such pulses (Lamersdorf et al., 1995).

Watershed-level studies of hydrology in humid forest ecosystems often show that evapotranspiration (ET) remains relatively constant despite large variations in precipitation (Luxmoore and Huff, 1989; Likens et al., 1977). Thus, ET consumes a larger proportion of precipitation under dry conditions than under wet conditions, and soil water flux (SWF) is affected disproportionately by changes in precipitation. The magnitude of this effect varies with the amount of precipitation compared with ET in a nominal year. This is illustrated by the results of simulation modeling of the effects of changing precipitation in several forest ecosystems (Johnson et al., 2001). In a red spruce (*Picea rubens* Sarg.) forest in the Smoky Mountains of North Carolina where ET consumed only 33% of precipitation, simulated precipitation changes of -28% and $+28\%$ caused -36% to $+35\%$ changes (respectively) in simulated SWF. In a slash pine forest in Florida where ET consumed 88% of precipitation, -33% and $+33\%$ changes in simulated precipitation caused -85% and $+122\%$ (respectively) changes in simulated SWF. Thus, these simulations suggested that the effects of changing precipitation on SWF are greater in sites where ET consumes a greater proportion of precipitation. On Walker Branch Watershed, the site of the current study, simulated changes of -33% and $+33\%$ in precipitation caused -43% and $+47\%$ (respectively) changes in simulated SWF (Johnson et al., 1998, 2001).

Reduced SWF can cause increased ionic concentrations in soil solutions, especially for conservative ions such as Cl^- , whose concentrations are not significantly buffered by adsorption reactions (Johnson et al., 2001). For anions such as H_2PO_4^- and SO_4^{2-} , adsorption to Fe and Al hydrous oxides may minimize concentration increases due to drought. In the case of HCO_3^- , concentrations are controlled by pH and the partial pressure of CO_2 in soil and pH, and could be affected either positively or negatively by drought. The effects of drought on cations in soil solution should, in theory, be a function

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Abbreviations: AMB, no treatment, ambient throughfall; DO, dry, open (location in the DRY treatment between troughs); DRY, treatment where throughfall flux was decreased by 33%; DT, dry, treated (location in the DRY treatment under troughs); EC, electrical conductivity; NuCM, Nutrient Cycling Model; ortho-P, ortho-phosphate; SWF, soil water flux; TDE, Throughfall Displacement Experiment; WET, treatment where throughfall flux was increased by 33%; WO, wet, open (location in the WET treatment removed from drip points); WT, wet, treated (location in the WET treatment near drip points).

Table 1. Some chemical and physical properties of the Fullerton soil (after Johnson et al., 1981).

Horizon and depth (cm)	Bulk density	Percent > 2 mm	C	N	C to N ratio	pH [†]	Extractable P	CEC [‡]	Ca ²⁺	Mg ²⁺	K ⁺	Base saturation
	g cm ⁻³	%	— mg g ⁻¹ —		g g ⁻¹		mg kg ⁻¹		cmol. kg ⁻¹			%
A (0–7)	1.14	17	36.1	0.16	22.6	4.6	7	6.19	3.28	0.27	0.23	61
E (7–38)	1.14	29	0.78	0.04	19.5	4.9	2.3	2.03	0.19	0.03	0.07	14
BA (18–32)	1.57	27	0.21	0.02	10.5	4.6	0.4	5.37	0.22	0.11	0.07	7
Bt (50–80)	1.54	23	0.17	0.02	8.5	4.8	0.2	13.59	0.42	0.66	0.11	9

[†] In 1 M CaCl₂.

[‡] Cation exchange capacity; 1 M NH₄Cl extraction.

of changes in total ionic concentration. As noted by Reuss (1983), increased total ionic concentrations cause trivalent cations to increase disproportionately to di- and monovalent cations (to the 3/2 and third power, respectively) and divalent cations to increase as the square of monovalent cations.

To investigate the potential effects of changing precipitation on forest ecosystems, the Throughfall Displacement Experiment (TDE) was established on Walker Branch Watershed, Tennessee in 1993. Three different throughfall amounts were tested: -33% (DRY), ambient (no change, AMB), and +33% (WET). Throughfall amounts were controlled using a system of rain gutters and sprinklers. In a previous paper, we described the results of 13 months of data on soil solution chemistry from this site and included projections of longer-term trends using the Nutrient Cycling Model (NuCM; Johnson et al., 1998). In that paper, we found higher electrical conductivity (EC) and SO₄²⁻, Cl⁻, K⁺, Ca²⁺, Mg²⁺, and Na⁺ concentrations in Bt horizon soil solutions (70 cm) in the DRY than either the AMB or WET treatments. We found no treatment effects on pH, HCO₃⁻, NH₄⁺, NO₃⁻, or ortho-phosphate (ortho-P) concentrations. As is usually the case in N-limited ecosystems, soil solution concentrations of NH₄⁺, NO₃⁻, and ortho-P were at or near trace levels at most times, limiting our ability to evaluate treatment effects on these ions.

In this paper, we update the Bt horizon soil solution data set and add data on soil solution from surface (E, 25 cm) soil horizons and on N and P fluxes measured with resin lysimeters from O horizons. We use this data to test our hypotheses, based on the literature review above and previous research at this site. Our hypotheses were that changes in throughfall inputs will cause:

- (i) concomitant changes in N and P fluxes in O horizons;
- (ii) inverse changes in EC and ion concentrations in soil solutions;
- (iii) concomitant changes in SO₄²⁻ to Cl⁻ ratios (because of greater changes in conservative anions such as Cl⁻ than adsorbing anions like SO₄²⁻); and
- (iv) inverse changes in (Ca²⁺ + Mg²⁺) to K⁺ ratios (because of greater changes in divalent than in monovalent cations).

SITE AND METHODS

Site Description

The throughfall displacement system for the experiment is located on the Walker Branch Watershed (35°58' N, 84°17'

W), a part of the U.S. Department of Energy's (DOE) National Environmental Research Park near Oak Ridge, Tennessee (Johnson and Van Hook, 1989). Long-term mean annual precipitation is 1358 mm and mean temperature is 14.2°C. Soils are classified as Fullerton series (fine, kaolinitic, thermic Typic Paleudult). Some chemical and physical properties are given in Table 1. Depth to bedrock at this location is approximately 30 m. The site was chosen because of its uniform slope, consistent soils, and a reasonably uniform distribution of vegetation. White oak (*Quercus alba* L.), chestnut oak (*Quercus montana* Willd.), and red maple (*Acer rubrum* L.) dominate the site, but it contains 16 other tree species (Hanson et al., 1995, 1998, 2001). Stand basal area averages 20 to 25 m² ha⁻¹. The experimental area was located at the upper divide of the watershed so that lateral flow of water into the soils at the top of the plots would not confound attempts to create a reduced soil water treatment. The site was also chosen because of its southern aspect, which was intended to increase the effects of the reduced moisture.

Experimental Design

The experimental design and its performance have been described in detail by Hanson et al. (1995, 1998). Briefly, the manipulations of throughfall levels reaching the forest floor are made with a system designed to passively transfer precipitation from one experimental plot to another. There are three 80- × 80-m plots in the TDE: one WET, one DRY and one ambient (AMB). Each of these plots is divided into 100 eight- by eight-meter subplots that serve as the locations for repetitive, nondestructive measurements of soil and plant characteristics. Throughfall precipitation is intercepted in approximately 2000 subcanopy troughs (0.3 × 5 m) suspended above the forest floor of the dry plot (approximately 33% of the ground area is covered). The intercepted throughfall is then transferred by gravity flow across an ambient plot and distributed onto the wet treatment plot through paired drip holes spaced approximately 1 m apart. The troughs are arranged in 21 rows of 80 to 90 troughs each. Reductions in soil moisture anticipated from the experimental removal of 33% of the throughfall are comparable with the driest growing season of the 1980s drought (Cook et al., 1988), which resulted in sapling mortality and reduced growth of some vegetation (Jones et al., 1993).

Soil Solution and Soil Element Flux Measurements

O Horizon Nitrogen and Phosphorus Fluxes

Resin lysimeters (as described by Susfalk, 2000, and Johnson et al., 2001) were added in 1998 beneath the O horizons of all treatments to collect cumulative soil N and P leaching. The O horizon was too thin for installation of ceramic cup lysimeters, and we have had difficulties with plate tension lysimeters in O horizons at this site in the past because of frequent drying episodes during which lysimeters lose tension. Resin lysimeters are tension free and thus are not subject to problems with wetting and drying. Resin lysimeters are also

very cost effective in that only one analysis per year per sampling location is required. Resin lysimeters have been used successfully to obtain indices of leaching in several previous studies and microbial transformations have generally been found to be minimal (Kjønass, 1999; Schnabel, 1983; Schnabel et al., 1993; Sibbesen, 1978; Susfalk, 2000).

The resin lysimeters used initially (in 1998) in this study consisted of a 5.5-cm-long, 4-cm-i.d. PVC pipe within which a resin bag was sandwiched between layers of washed silica sand. Ten grams of oven-dried Rexyn I-300 (H-OH) resin (Fisher Scientific, Lari Lawn, NJ) were placed in a section of nylon pantyhose, using cable ties to secure each end. This resin bag was placed on a 20 g layer of moist, washed silica sand at the bottom of the tube and covered with another 20 g layer of silica sand. In 1999, nylon resin bags were replaced by resin trapped between two Nitex nylon sheets (ICN Biomedicals, Aurora, OH). Susfalk (2000) found that this change did not result in a significant difference in the N or P trapped by the resin lysimeters in a separate study. In order to keep the sand in the tube until installation, the bottom of each PVC tube was covered with cheesecloth held in place with a rubber band. The first set of resin lysimeters were installed in June 1998 by excavating a small hole and tunneling beneath the O horizons. The resin lysimeters were co-located within 2 m of the cluster of ceramic cup lysimeters ($n = 9$ per treatment and strata). For the DRY and WET treatments, the resin lysimeters were stratified according to position to account for the influence of trough collection (DRY) and drip redistribution (WET) systems. In the DRY treatment, one set of resin lysimeters was placed directly beneath a trough and one set between troughs at each location ($n = 9$ for each, for a total of 18 replicates). For the WET treatment, one set of resin lysimeters was placed beneath an area that was receiving water from a driphole and one that was not ($n = 9$ for each, for a total of 18 replicates). The resin lysimeters were removed in June 1999 and replaced with the modified (1999) design. The 1999 resin lysimeters were removed and replaced in June 2000.

After the collection, resins were removed from the lysimeters, placed into 250-mL Erlenmeyer flasks, and extracted with 100 mL of 1.0 M KCl with shaking for 1 h. The extract was filtered (Whatman [Maidstone, UK] No. 1) and stored at 4°C until analysis. The extracts were analyzed for NH_4^+ , NO_3^- , and ortho-P by automated colorimetric analysis at the Water Analysis Laboratory at the Desert Research Institute, Reno, NV. Three 10-g replicates of untreated resins were extracted in the same way and served as blanks. Fluxes were calculated from the amount of NH_4^+ , NO_3^- , and ortho-P extracted from the resins (minus blanks) divided by the surface area of the lysimeters (12.6 cm²). Due to a laboratory error, ortho-P was not measured in the 1998–1999 collection.

Mineral Horizon Soil Solution Collection

Soil solutions from the E (25 cm) and Bt (70 cm) horizons were collected with ceramic cup tension lysimeters (Soil Moisture Equipment Corp., Santa Barbara, CA). The lysimeters were installed prior to the initiation of treatment in a three by three array in each plot at a 7.9-m spacing (nine replicates per treatment and depth). Some pre-treatment collections were made in 1992 and reported earlier (Johnson et al., 1998). Post-treatment sampling was initiated in 1996. Prior to collection of samples, each lysimeter was set to a tension of -40 kPa. Soil solutions were analyzed for electrical conductivity (EC), pH and HCO_3^- (by titration to pH 5.0), Ca^{2+} , Mg^{2+} , K^+ , and Na^+ by atomic absorption; NH_4^+ , NO_3^- , and ortho-P by automated colorimetric analysis; and Cl^- and SO_4^{2-} by Dionex (Sunnyvale, CA) ion chromatography. In most cases

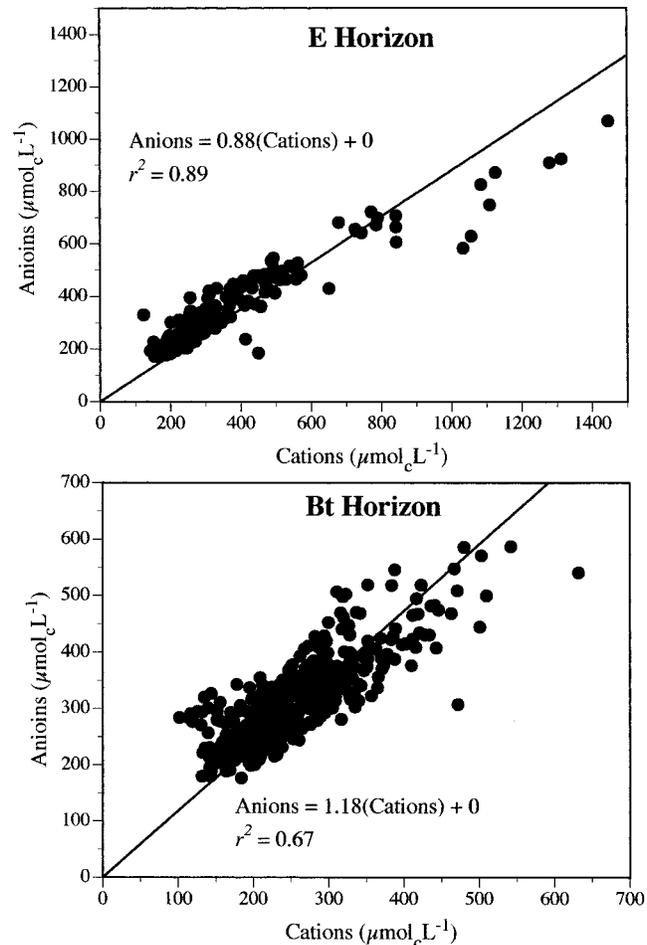


Fig. 1. Plot of total anions against total cations in soil solutions from the E and Bt horizons.

(64% of all samples), pH values were 5.0 or less and thus HCO_3^- concentrations were 0. Of the remaining cases where pH values were above 5.0, there was insufficient sample for titrations in 31% of the samples, so these analyses were omitted in favor of analyses for other ions (which take much less volume). Thus, HCO_3^- analyses are lacking in 11% of the samples overall and it was not possible to include HCO_3^- in all charge balance calculations. Within the data set containing HCO_3^- , the charge balance of $\text{Ca}^{2+} + \text{Mg}^{2+} + \text{K}^+ + \text{H}^+ + \text{Na}^+$ (major cations, neglecting NH_4^+ and Al) against $\text{HCO}_3^- + \text{Cl}^- + \text{SO}_4^{2-}$ (major anions, neglecting ortho-P and NO_3^-) indicated an anion deficit at higher total cation concentrations in the E horizon soil solutions and a slight cation deficit in the Bt horizon solutions (Fig. 1). The anion deficit in the E horizons could be due to unmeasured organic anions, and the cation deficit in the Bt horizon could be due to Al, which was not measured (pH values were often low enough for significant amounts of Al to be present in measurable concentrations).

Mineral Horizon Ion Flux Calculations

Ion fluxes were calculated from average ion concentrations for each water year and simulated soil water fluxes. Daily estimates of soil water content, water potential, and rates of percolation were estimated from a revised version of the TEHM/PROSPER model (Huff et al., 1977; Luxmoore, 1983) coded using Ithink modeling software (High Performance Systems, 2000) for Macintosh computer. A comparison of measured and

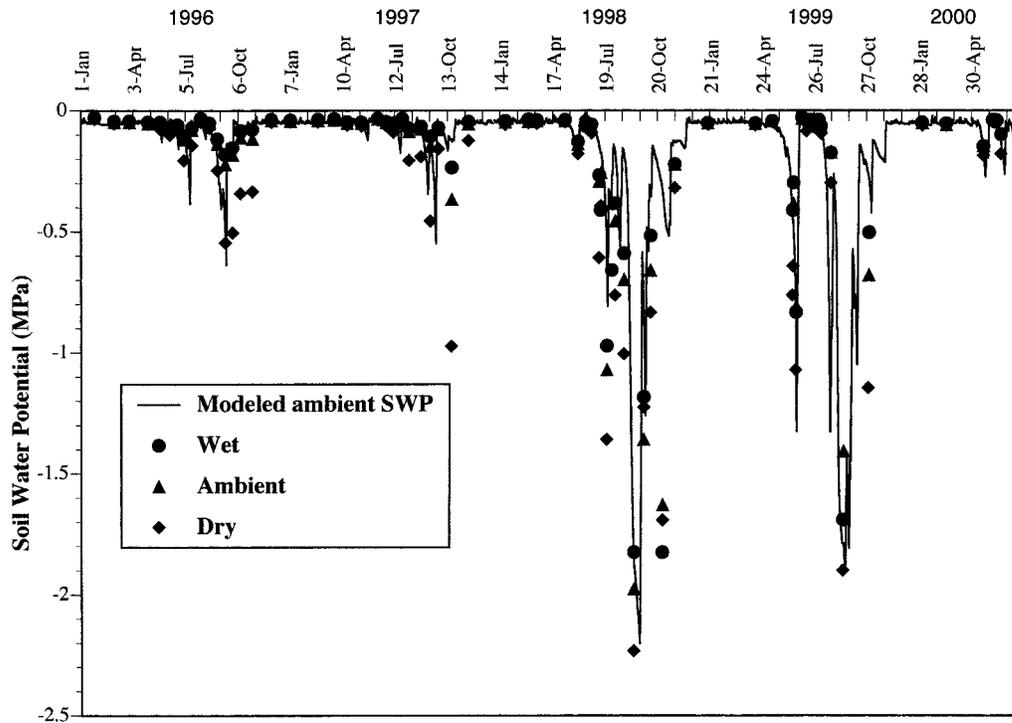


Fig. 2. Soil water potential at 0 to 30 cm in the treatment plots from 1 Jan. 1996 through 31 Dec. 2000.

modeled soil water potentials for the period from 1993 through 2000 is provided in Fig. 2, showing good agreement for a range of wet and dry years.

Soil water content (v/v) was measured in each subwatershed with a time domain reflectometer (TDR, Soil Moisture Equipment Corp.) following the procedure of Topp and Davis (1985), which has been developed for soils with a high coarse fraction (Drungil et al., 1987). Sample locations for TDR measurements (310) were distributed along an 8- × 8-m grid across the TDE site, resulting in more than 100 soil water monitoring locations per treatment area. At each location, two pairs of TDR waveguides were installed in a vertical orientation (0–0.35 m and 0–0.7 m). The TDR soil water content measurements were adjusted to reflect the coarse fraction of these soils (mean coarse fraction of 14%) and converted to soil water potentials using soil moisture retention curves for the A, A/E, and E/B horizons (Hanson et al., 1998). This data is available on the website data archive (<http://www.esd.ornl.gov/programs/WBW/Datadte.htm>)

Forest Floor Sampling

Organic (O) horizons were sampled in February 1999. Sample locations were designated in each treatment plot along upper, middle, and lower slope transects. In each treatment area five locations were designated per transect for a total of 15 sampling locations in each treatment area. To minimize any influence of trough placements on O horizon accumulation or decomposition, post-treatment sampling within the dry treatment plot included twice the area (i.e., one under and one between troughs) per location. Identical sampling locations were avoided between years.

O horizons were sampled within 0.25 m² circular templates. Coarse twigs and large pieces of nonfoliar litter were removed first followed by the collection of the foliar component of the Oi horizon. The Oe and Oa horizons were sampled together because Oa horizons were often too thin to sample separately. All samples were oven-dried for 48 h (100°C) and dry mass was determined. Complete samples were subsequently ground

in a large Wiley mill to a 20 mesh and a portion of the homogenized ground material was combusted at 500°C to obtain ash content. Total dry mass per unit ground area (m²) for each organic horizon was corrected for ash content. Other subsamples were ground to a fine powder and analyzed for C and N on a PerkinElmer (Wellesley, MA) 2400 CHNS analyzer at the Desert Research Institute, Reno, NV.

Statistical Analyses

Statistical analyses were performed using the General Linear Model (GLM) in DataDesk software (Data Description, 1988; Velleman, 1997). Within GLM, analysis of variance was used for overall treatment effects and Bonferonni post-hoc tests for differences between AMB, DRY, and WET treatments. Because of high variability, the data for the resin lysimeter fluxes was log-transformed for statistical analysis.

RESULTS

Hydrologic Fluxes

Precipitation amounts and simulated soil water fluxes for water years 1996 through 2000 are shown in Table 2. Precipitation values for the AMB treatment represent field measurements and those for the DRY and WET treatments represent 0.67 and 1.33 times those values, respectively. As noted in previous papers (Johnson et al., 1998, 2001), these simulations suggest that soil water flux (SWF) is affected disproportionately by treatments: reducing precipitation by –33% in the DRY treatments caused decreases in simulated SWF of –46 to –59%, and increasing precipitation by +33% in the WET treatment increases simulated SWF by +44 to +57% (Table 2). It is also noteworthy that annual and seasonal precipitation input to the TDE site varied widely during the four-year sampling period (Table 2).

Table 2. Measured water year (1 October through 30 September) precipitation, and simulated cumulative drought days soil water flux using the PROSPER model.

Water year	Ambient	Dry	Wet
Precipitation (mm)			
1996	1618	1077	2156
1997	1612	1074	2149
1998	1205	802	1606
1999	1257	837	1676
2000	1187	791	1582
Soil percolation (cm)			
1996	85.3	34.7	134.0
1997	94.7	45.9	142.2
1998	76.3	39.0	111.3
1999	65.4	30.3	101.7
2000	52.9	28.4	81.2
Cumulative drought (Mpa d⁻¹)			
1996	-35	-52	-29
1997	-28	-46	-25
1998	-108	-171	-62
1999	-128	-215	-98
2000	-52	-140	-32

In 1996 and 1997, annual precipitation inputs were above average, mostly because of higher inputs during the January to July period. Conversely, annual precipitation in water years 1998, 1999, and 2000 was below average; and the growing seasons in 1998 and 1999 were characterized by late-season droughts. A brief early spring drought also occurred in June 1999 (Fig. 2).

O Horizon Fluxes

The fluxes of NH₄⁺, NO₃⁻, NH₄⁺ plus NO₃⁻, and ortho-P measured by the resin lysimeters are given in Fig. 3 and the statistical analyses are presented in Table 3.

Table 3. Probability values for repeated measures analysis of variance (ANOVA) and Bonferonni post-hoc tests on log-transformed values for resin lysimeter N and P fluxes. A = ambient; DO = DRY, open; DT = DRY, under troughs; WO = WET, open; WT = WET, near drip point; ND = not determined.

Parameter	NO ₃ ⁻	NH ₄ ⁺	NO ₃ ⁻ + NH ₄ ⁺	Ortho-P
Treatment	<0.01	<0.01	<0.01	<0.01
Date	0.14	0.87	0.31	ND
Treatment × date	0.65	0.76	0.70	NS
DO-A	0.91	1.00	1.00	1.00
DT-A	<0.01	<0.01	<0.01	0.22
DT-DO	0.01	<0.01	<0.01	0.36
WO-A	1.00	1.00	1.00	1.00
WO-DO	1.00	1.00	1.00	0.97
WO-DT	<0.01	<0.01	<0.01	0.04
WT-A	<0.01	<0.01	<0.01	0.16
WT-DO	<0.01	<0.01	<0.01	0.06
WT-DT	<0.01	<0.01	<0.01	<0.01
WT-WO	<0.01	0.01	<0.01	0.60

(Note that ortho-P values were not measured in the 1998–1999 collection.) As expected, results for the resin lysimeters were sensitive to their placement with regard to the troughs in the DRY treatment and with regard to the drip locations on the WET treatment. Between troughs of the DRY treatment (DRY–open, DO) and between the drip holes of the WET treatment (WET–open, WO) resin-derived fluxes from the O layer were not significantly different from the fluxes in the ambient (A) treatment. Fluxes were, however, strongly affected when resin lysimeters were placed underneath troughs or near drip holes. Fluxes of NH₄⁺, NO₃⁻, and NH₄⁺ + NO₃⁻ beneath the troughs (DRY–treatment, DT) were significantly (*p* < 0.05) lower than in the lysimeters located in the open (A, DO, and WO). Fluxes of NH₄⁺, NO₃⁻, and NH₄⁺ + NO₃⁻ near drip holes (WET–treatment,

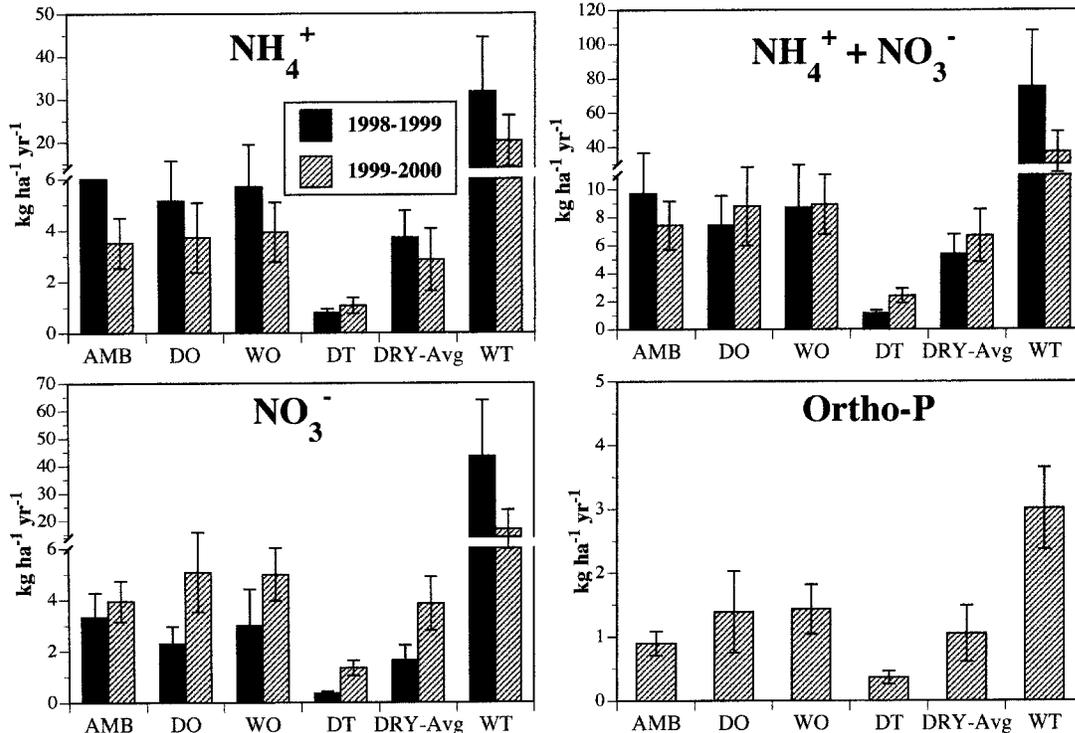


Fig. 3. Measured fluxes of NH₄⁺, NO₃⁻, NH₄⁺ + NO₃⁻, and ortho-phosphate in resin lysimeters. Note that ortho-P was not measured in 1998–1999. Standard errors are shown. See Table 3 for statistical analyses.

WT) were significantly ($p < 0.05$) greater than in the other treatments. Differences in NH_4^+ , NO_3^- , and $\text{NH}_4^+ + \text{NO}_3^-$ fluxes between the WT and other treatments were statistically significant ($p < 0.01$) in all cases. Fluxes of NH_4^+ , NO_3^- , and $\text{NH}_4^+ + \text{NO}_3^-$ in the DRY, undertrough (DT) treatment were significantly lower than in any other treatment. Differences in ortho-P fluxes were significant only in a few cases (WT-DT, WO-DT, and, accepting the 0.06 probability level, WT-DO). There were no statistically significant effects of date on NH_4^+ , NO_3^- , and $\text{NH}_4^+ + \text{NO}_3^-$ fluxes. (The effects of date could not be tested on ortho-P fluxes because of the lack of data for 1998–1999).

The area-weighted average fluxes of NH_4^+ , NO_3^- , and $\text{NH}_4^+ + \text{NO}_3^-$ in the DRY treatment (i.e., 0.67 times the between-trough flux plus 0.33 times the beneath-trough flux) gave an approximation of the effects of the treatment on the plot as a whole. These fluxes were 45 to 52% lower than those in the AMB treatment in 1998–1999, but only 10 to 20% lower than the AMB treatment

in 1999–2000 (Fig. 3). It was not possible to accurately calculate aerially weighted average fluxes for the WET treatment because the areas affected by drip holes were both uncertain and variable over time, depending upon rainfall intensity and duration.

Soil Solution Concentrations

There were no statistically significant treatment effects on E horizon (25 cm) soil solution concentrations of any measured ion (Fig. 4 and 5; Table 4). There were significant effects on EC, where the values for the DRY treatment were significantly greater than those for the WET treatment, and for the $(\text{Ca}^{2+} + \text{Mg}^{2+})$ to K^+ ratio, where the values for the AMB treatment were significantly greater than for the WET treatment (Fig. 4 and 5; Table 4). Date was a significant factor for pH, EC, Mg^{2+} , Na^+ , SO_4^{2-} , Cl^- , NO_3^- , and SO_4^{2-} to Cl^- ratio. Treatment \times date was a significant factor for pH and EC, indicating that treatment differences changed over time.

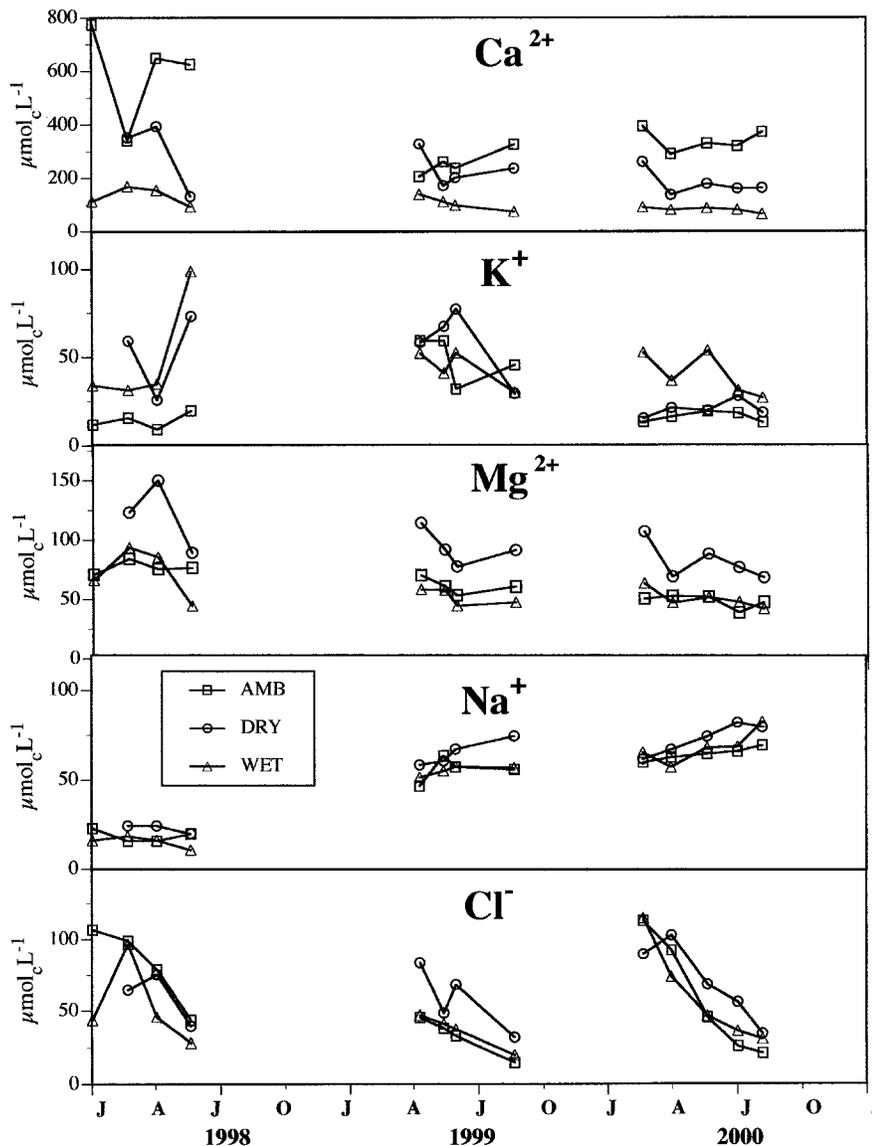


Fig. 4. Concentrations of Ca^{2+} , K^+ , Mg^{2+} , Na^+ , and Cl^- in E horizon soil solutions (25 cm). See Table 4 for statistical analyses.

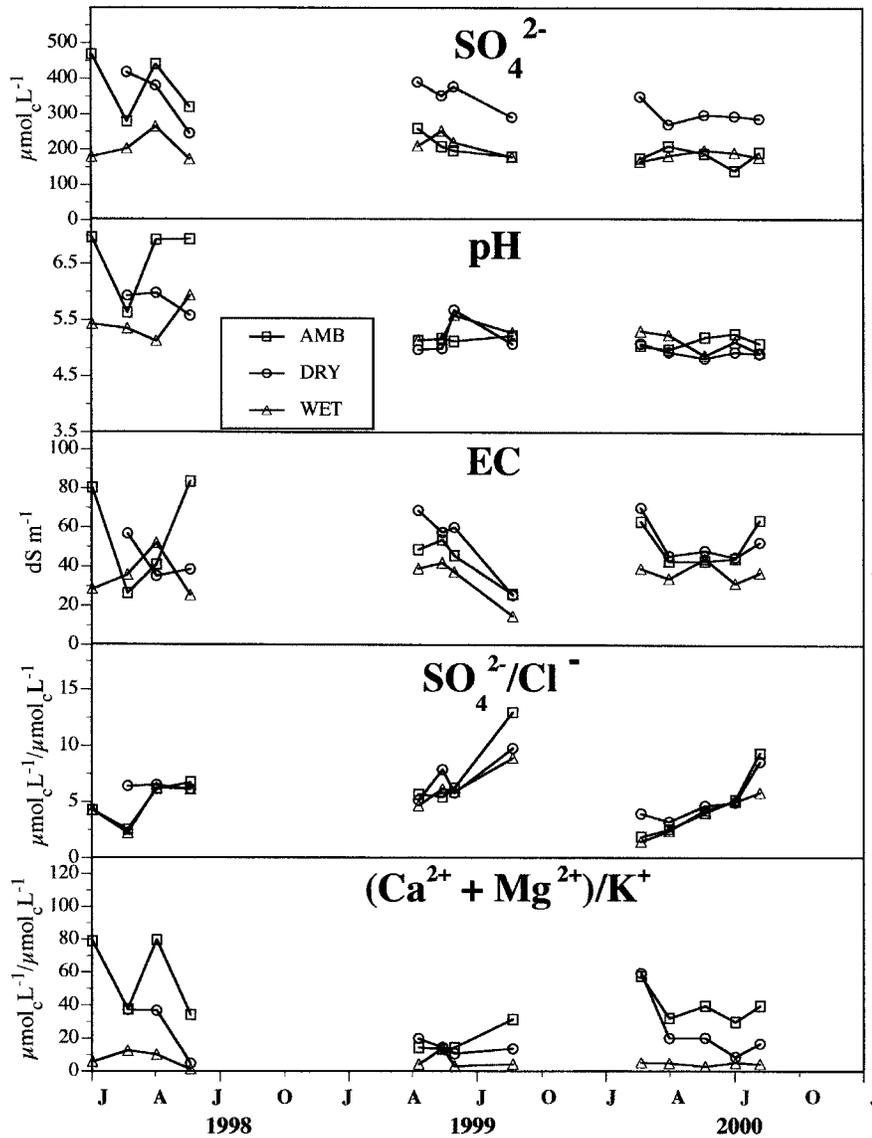


Fig. 5. Sulfate, pH, electrical conductivity (EC), SO_4^{2-} to Cl^- ratios, and $(Ca^{2+} + Mg^{2+})$ to K^+ ratios in E horizon soil solutions (25 cm). See Table 4 for statistical analyses.

In the Bt horizon (70 cm) soil solutions, treatment effects were statistically significant ($p < 0.01$) for pH, EC, SO_4^{2-} to Cl^- ratio, $(Ca^{2+} + Mg^{2+})$ to K^+ ratio, and all measured ions except for HCO_3^- , ortho-P, NH_4^+ ,

and NO_3^- (Fig. 6 and 7; Table 5). Insufficient data may have been a factor in the lack of treatment effects for HCO_3^- , ortho-P, NH_4^+ , and NO_3^- . Ortho-P concentrations were below detection limits (2 mmol L⁻¹) in 87%

Table 4. Probability values for repeated measures analysis of variance (ANOVA) and Bonferonni post-hoc tests on the effects of treatment on soil solution concentrations from the E horizons (25 cm).

Ion	Treatment	Date	Treatment × date	Bonferonni post-hoc tests		
				Ambient-dry	Ambient-wet	Wet-dry
pH	0.79	<0.01	0.06	1.00	0.90	0.96
Electrical conductivity	0.02	<0.01	<0.01	0.95	0.10	0.06
Ca^{2+}	0.16	0.47	0.51	0.90	0.18	0.74
K^+	0.23	0.17	0.54	1.00	0.37	0.54
Mg^{2+}	0.19	0.07	0.98	0.22	0.93	0.32
Na^+	0.77	<0.01	0.63	1.00	0.87	0.97
SO_4^{2-}	0.13	0.04	0.21	0.26	1.00	0.14
Cl^-	0.57	<0.01	0.31	0.79	1.00	0.66
HCO_3^-	0.21	0.36	0.69	0.48	0.28	0.99
NO_3^-	0.16	<0.01	0.91	0.17	0.83	0.35
NH_4^+	0.99	0.94	0.93	1.00	1.00	1.00
SO_4^{2-} to Cl^- ratio	0.46	<0.01	0.20	0.76	0.99	0.52
$(Ca^{2+} + Mg^{2+})$ to K^+ ratio	0.02	0.24	0.57	0.99	0.04	0.09

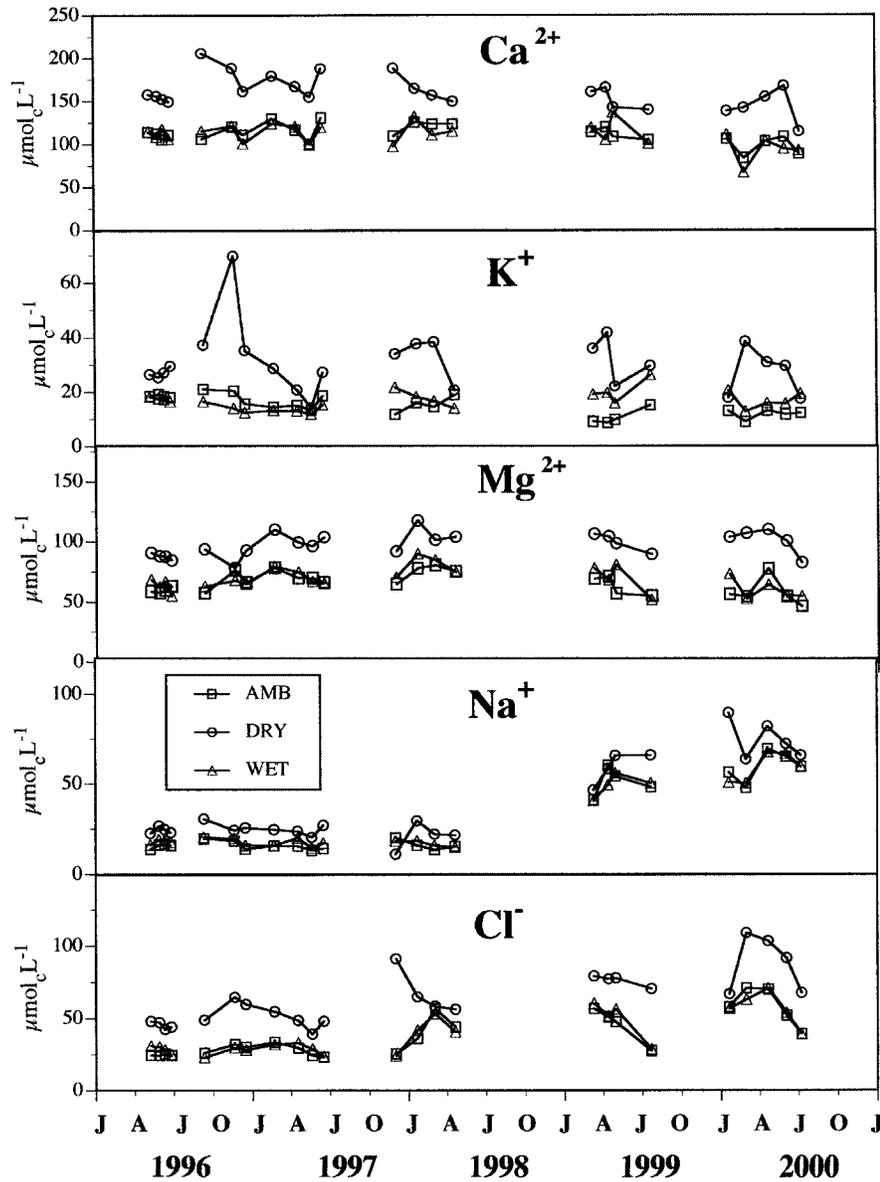


Fig. 6. Concentrations of Ca²⁺, K⁺, Mg²⁺, Na⁺, and Cl⁻ in Bt horizon soil solutions (70 cm). See Table 5 for statistical analyses.

of the samples, NH₄⁺ concentrations below detection limits (1 mmol_c L⁻¹) in 58% of the samples, and NO₃⁻ concentrations were below detection limits (4 mmol_c L⁻¹) in 43% of the samples. Date was a significant fac-

tor for all measured soil solution properties, but the interaction term (treatment × date) was significant (*p* < 0.05) for K⁺ and (Ca²⁺ + Mg²⁺) to K⁺ ratio, suggesting that the effects of treatment changed over time in these

Table 5. Probability values for repeated measures ANOVA and Bonferonni post-hoc tests on the effects of treatment on soil solution concentrations from the Bt horizons (70 cm).

Ion	Treatment	Date	Treatment × date	Bonferonni post-hoc tests		
				Ambient-dry	Ambient-wet	Wet-dry
pH	<0.01	<0.01	0.99	0.05	0.99	0.07
Electrical conductivity	<0.01	<0.01	0.19	<0.01	0.15	<0.01
Ca ²⁺	<0.01	<0.01	0.99	<0.01	0.95	<0.01
K ⁺	<0.01	<0.01	<0.01	<0.01	0.76	<0.01
Mg ²⁺	<0.01	<0.01	0.92	<0.01	0.67	<0.01
Na ⁺	<0.01	<0.01	0.62	<0.01	0.91	<0.01
SO ₄ ²⁻	<0.01	<0.01	0.97	<0.01	0.57	<0.01
Cl ⁻	<0.01	<0.01	0.20	<0.01	0.94	<0.01
HCO ₃ ⁻	0.73	<0.01	0.05	0.00	0.88	0.88
NO ₃ ⁻	0.34	<0.01	0.93	0.43	0.93	0.47
NH ₄ ⁺	0.43	<0.01	0.22	0.99	0.48	0.99
SO ₄ ²⁻ to Cl ⁻ ratio	<0.01	<0.01	0.14	<0.01	0.15	<0.01
(Ca ²⁺ + Mg ²⁺) to K ⁺ ratio	0.02	<0.01	<0.01	0.63	0.02	0.50

cases. It appeared that the differences in K^+ between the DRY and other treatments decreased over time, while the differences in $(Ca^{2+} + Mg^{2+})$ to K^+ ratio in the WET and the other treatments increased over time (Fig. 6 and 7).

As noted in a previous paper (Johnson et al., 1998), observed treatment effects in the Bt horizon were most often due to differences between the DRY treatment and the other treatments; differences between the WET and AMB treatments were significant only in the case of $(Ca^{2+} + Mg^{2+})$ to K^+ ratio (where WET was lower than AMB) (Fig. 7 and Table 5). The Bt horizon soil solutions in the DRY treatment had consistently higher pH, EC, and all measured ions except HCO_3^- , ortho-P, NH_4^+ , and NO_3^- throughout the four-year sampling period. Soil solution Na^+ concentrations in the Bt horizon increased significantly over time ($r^2 = 0.67, p < 0.0001$)

during the sampling period. The Bt horizon Na^+ concentrations increased threefold in the spring of 1999 and remained elevated through the spring of 2000 (Fig. 6). Concentrations of Cl^- and EC also increased significantly ($r^2 = 0.28, p < 0.0001$ and $r^2 = 0.21, p < 0.0001$; respectively) in the spring of 1999 but to a lesser degree than for Na^+ (Fig. 6). Other statistically significant trends included a very slight decrease in Ca^{2+} ($r^2 = 0.01, p < 0.0128$) and pH ($r^2 = 0.01, p < 0.0185$). There were no significant trends in the concentrations of other ions, although seasonal variations were often substantial and the effects of date were significant in all cases.

Calculated Ion Fluxes in Mineral Soil

Calculated fluxes of Ca^{2+} , K^+ , Mg^{2+} , Na^+ , Cl^- , and SO_4^{2-} for the Bt horizon are shown in Fig. 8. Error bars

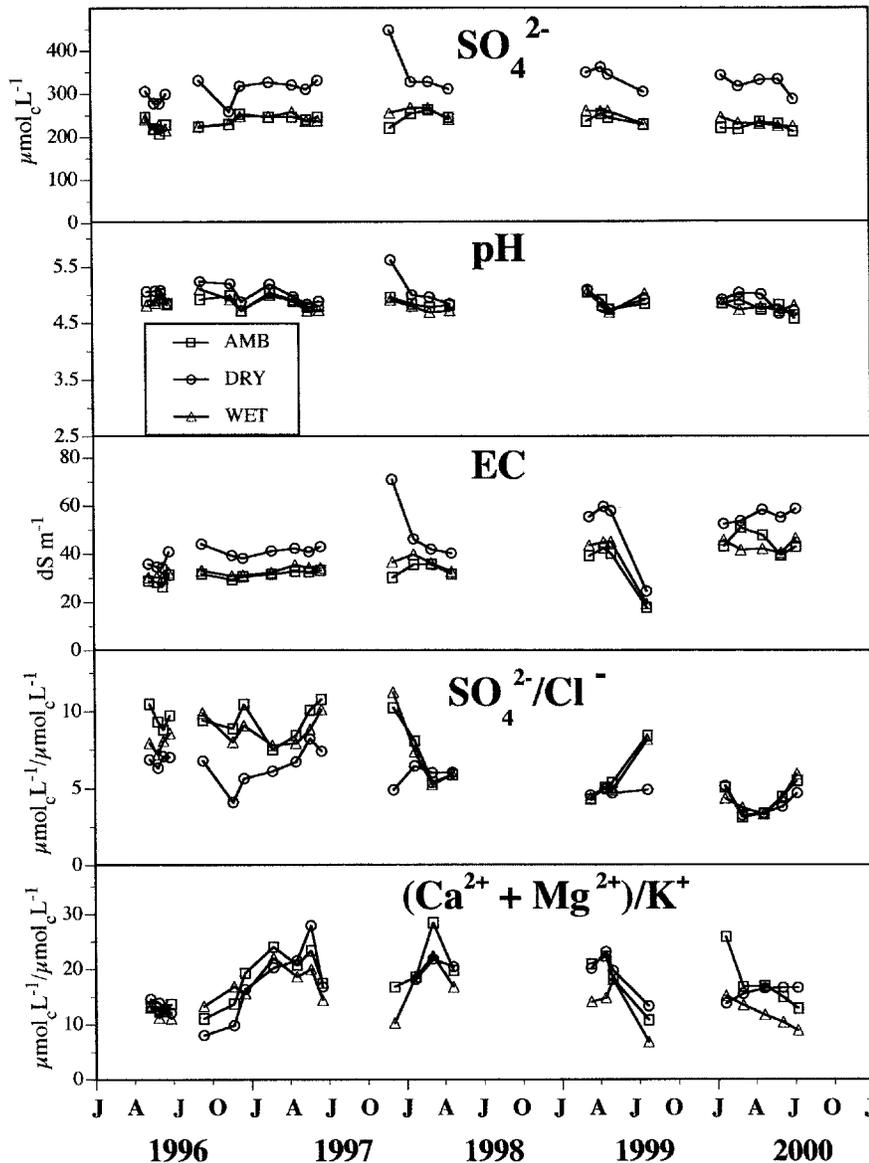


Fig. 7. Sulfate, pH, electrical conductivity (EC), SO_4^{2-} to Cl^- ratios, and $(Ca^{2+} + Mg^{2+})$ to K^+ ratios in Bt horizon soil solutions (70 cm). See Table 5 for statistical analyses.

are not shown because the flux values are calculated from modeled soil water flux data, and thus it is not possible to conduct statistical analyses of the flux data.

The drought of 1998–2000 caused reductions in calculated Ca^{2+} , K^+ , Mg^{2+} , and SO_4^{2-} fluxes over the sampling period. This was mostly because of the reduction in soil water flux; only Ca^{2+} concentration showed a slight downward trend over time (Fig. 6 and 7). Calculated fluxes of Na^+ increased sharply in 1999 and 2000, caused by the increase in soil solution Na^+ concentration (Fig. 6).

Calculated Cl^- flux also increased over the sampling period as a result of the increase in Cl^- concentration.

Litter Mass and Nitrogen Content

The mass and N content of standing litter in February 1999 is presented in Table 6. There were no statistically significant differences in mass or N content in Oi horizons, but significant differences in mass and N content in the Oe + Oa, nonleaf component, and in total O horizon were noted (Table 6). The differences were

Table 6. Mass and N content of litter sampled in February 1999 (five years after treatment). Treatment *P* values for analysis of variance (ANOVA) given and letters denote significant differences ($p < 0.05$) using Bonferonni post-hoc tests on log-transformed values. Standard errors are given.

Horizon	Treatment			<i>P</i>
	Ambient	Dry	Wet	
		Mass (kg ha⁻¹)		
Oi	4 144 ± 241a	4 056 ± 215a	4 035 ± 344a	0.96
Oe + Oa	5 086 ± 587a	8 425 ± 582b	5 603 ± 555a	<0.01
Nonleaf	1 591 ± 247a	2 829 ± 302b	1 630 ± 131a	<0.01
Total	10 821 ± 746a	15 310 ± 820a,b	11 628 ± 817a,c	<0.01
		N concentration (mg g⁻¹)		
Oi	11.2 ± 0.4a	10.6 ± 0.2b	10.1 ± 0.3a,b	0.05
Oe + Oa	11.1 ± 0.5a	13.3 ± 0.3b	14.0 ± 0.7b	<0.01
Nonleaf	5.1 ± 0.6a	4.9 ± 0.6a	5.2 ± 0.2a	0.87
		N content (kg ha⁻¹)		
Oi	55 ± 5a	47 ± 2a	44 ± 4a	0.13
Oe + Oa	102 ± 10a	181 ± 11b	117 ± 12a	<0.01
Nonleaf	8 ± 1a	14 ± 2b	8 ± 1a	<0.01
Total	165 ± 10a	242 ± 13b	169 ± 14a	<0.01

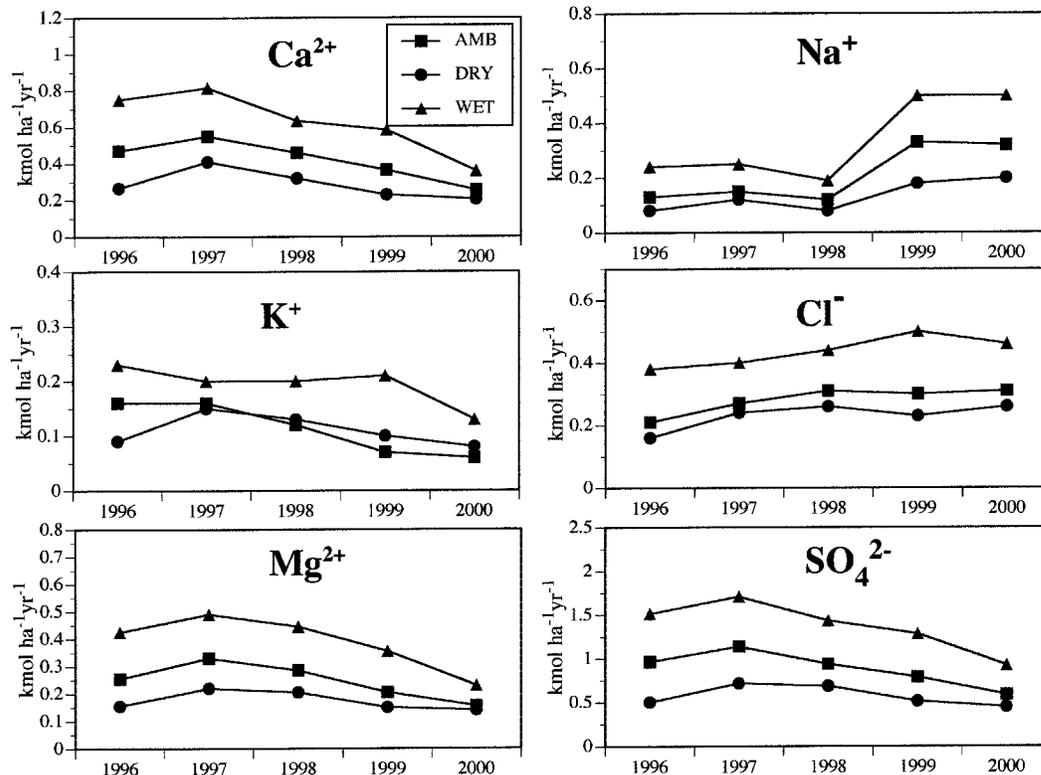


Fig. 8. Calculated fluxes of Ca^{2+} , K^+ , Na^+ , Mg^{2+} , Cl^- , and SO_4^{2-} in the Bt horizons. Values are based on average concentrations and modeled soil water flux. Error bars represent standard errors associated with average concentrations only. AMB = ambient, no treatment; WO = WET treatment, placed in the open away from drip points; DO = DRY treatment, placed between troughs; WT = WET treatment, placed near drip points; DT = DRY treatment, placed beneath troughs; DRY-Avg = area-weighted average fluxes in the DRY treatment.

due primarily to greater Oe + Oa mass in the DRY treatment; greater nonleaf mass in the DRY treatment also contributed to the overall difference. The DRY and WET treatments had lower Oi horizon N concentrations and greater Oe + Oa horizon N concentrations than the AMB treatments. Differences in standing litter layer mass were not attributable to differences in litter inputs, which were not significantly different among treatment plots from 1994 through 1998 (Joslin et al., 2000).

DISCUSSION

Potential Effects of Changes in Ion Inputs in Throughfall

The treatment effects in the O and Bt horizons could have been affected by differences in ion inputs in throughfall. Unfortunately, throughfall fluxes were not routinely measured. Based on previous studies at a nearby chestnut oak site on Walker Branch (Johnson et al., 1985; Johnson and Todd, 1990), the fluxes of NH_4^+ , NO_3^- , total N, and ortho-P in throughfall among the treatments would have been approximately 0.06, 0.08, 0.12, and 0.07 kmol ha^{-1} , respectively (Table 7). These values are within the range of the differences observed in the resin lysimeters between the AMB and DRY weighted average values (0.05 to 0.12 kmol for NH_4^+ , 0.01 to 0.12 kmol for NO_3^- , 0.06 to 0.31 $\text{kmol for NH}_4^+ + \text{NO}_3^-$, and 0.15 kmol for ortho-P) (Fig. 3). The differences in resin fluxes between the AMB- and DRY-under trough (DT) locations (2.4 to 5.6 kmol for NH_4^+ , 2.5 to 3.0 kmol for NO_3^- , and 5.0 to 8.6 $\text{kmol for NH}_4^+ + \text{NO}_3^-$, and 0.5 kmol for ortho-P) were far larger than could be accounted for by throughfall inputs. Similarly, the differences in resin fluxes of NH_4^+ , NO_3^- , total N, and ortho-P between the AMB and WET treatments were substantially greater than could be accounted for by throughfall inputs. As noted above, the resin lysimeters in the WET treatment were located near drip points and therefore measurements may have been elevated by larger than average water flux and also by inputs of N in irrigation water.

The changes in throughfall fluxes of K^+ could have had a major effect on calculated K^+ fluxes in Bt soil solution. Values for throughfall K^+ from the previous studies were greater than those calculated for soil solution flux in this study (Table 7 and Fig. 8). Changes in throughfall fluxes of Cl^- , and to a lesser extent Ca^{2+} , Mg^{2+} , and SO_4^{2-} , could also have caused changes in soil solution flux: previously measured throughfall fluxes

equaled 30 to 100% of those calculated for soil solution fluxes in this study (Table 7 and Fig. 6). In the case of Na^+ , however, throughfall fluxes were considerably lower than those calculated for soil solution, especially in water years 1999 and 2000.

Differences in throughfall fluxes alone could not account for all the observed differences in soil solution concentration; indeed, if throughfall fluxes were the only factor involved, there would be no differences in soil solution concentration (i.e., both ion and water fluxes would change by $\pm 33\%$ and concentrations would stay the same). It is clear from the treatment effects on soil solution concentration that soil water flux must have changed disproportionately in response to changes in throughfall in the DRY treatment, at least. It is not clear why soil solution concentrations in the WET treatment were not more substantially affected than they were. The lack of response in soil solution concentrations in the WET treatment caused soil leaching flux estimates to increase substantially above those in the AMB treatment. As noted above, some of this was certainly due to increased throughfall fluxes. Regardless of the contributions of throughfall fluxes to soil leaching fluxes, however, any change in soil leaching flux represents a net loss from the system, whereas throughfall fluxes include a proportion of atmospheric dry deposition and ions that have been recycled by vegetation (especially in the case of K^+). Thus, changes in soil leaching fluxes can be expected to have long-term consequences for soil change.

Evaluation of Hypotheses

The first hypothesis (i.e., changes in throughfall input will cause concomitant changes in N and P fluxes in O horizons) was supported for N but not P. Nitrogen fluxes were significantly greater in the WET treatment and lower in the DRY treatment than in the AMB treatment. This effect was exclusively due to those lysimeters placed either under troughs (DRY) or near drip points (WET); the lack of difference between those lysimeters placed in the open (AMB, WO, DO) is reassuring that the plots were comparable in other ways.

The second hypothesis (i.e., changes in throughfall input will cause inverse changes in EC and ion concentrations in soil solutions) was supported only in part. The DRY treatment did continue to cause increases in ion concentrations in Bt horizon soil solutions, but there were no statistically significant differences in E horizon solutions nor any differences between the WET and AMB treatments in the Bt horizons. The reasons for the greater treatment effect on the Bt than on E horizon solutions are not clear, but may relate to increased differences in soil water flux with increasing depth. Although most roots and therefore most evapotranspiration effects are concentrated in the surface horizons, water removal by roots does occur at depths greater than 25 cm, and, as discussed above, this could magnify treatment effects on soil water flux with depth. This hypothesis remains to be tested, however. The reasons for the lack of soil solution responses to the WET treatment are not known and merit further investigation.

Table 7. Ion fluxes in throughfall in a chestnut oak forest on Walker Branch Watershed (data from Johnson et al., 1985 and Johnson and Todd, 1990).

Ion	Amount
Ca^{2+}	0.33 ± 0.02
K^+	0.36 ± 0.04
Mg^{2+}	0.13 ± 0.02
Na^+	0.05 ± 0.01
Cl^-	0.19 ± 0.01
SO_4^{2-}	0.64 ± 0.04
Ortho-P	0.02 ± 0.002
Total P	0.03 ± 0.01
NH_4^+	0.18 ± 0.01
NO_3^-	0.23 ± 0.01
Total N	0.36 ± 0.07

The third hypothesis (i.e., changes in throughfall input will cause concomitant changes in SO_4^{2-} to Cl^- ratios) was supported for Bt horizons: SO_4^{2-} to Cl^- ratios were lower in the DRY treatment, suggesting that there was some buffering of soil solution SO_4^{2-} concentrations by adsorption. The Bt horizon solution SO_4^{2-} to Cl^- ratios declined and converged by water year 2000 when soil solution concentrations of Cl^- (and Na^+) increased in an apparent response to drought (Fig. 6). The Bt horizon SO_4^{2-} concentrations showed no change during water year 2000, however, again suggesting buffering and supporting the third hypothesis. Soil solution SO_4^{2-} concentrations were not completely controlled by buffering, however: SO_4^{2-} concentrations were consistently greater in the DRY than in the other treatments.

The fourth hypothesis (i.e., changes in throughfall input will cause inverse changes in $[\text{Ca}^{2+} + \text{Mg}^{2+}]$ to K^+ ratios) was supported in part: $(\text{Ca}^{2+} + \text{Mg}^{2+})$ to K^+ ratios were significantly lower in WET than in the DRY treatments in the E horizons and significantly lower in the WET than in the AMB treatment in both horizons. In a previous paper (Johnson et al., 1998), we noted no significant treatment effects on $(\text{Ca}^{2+} + \text{Mg}^{2+})$ to K^+ ratio.

General Observations

The resin lysimeters allowed us to measure treatment effects on N fluxes for the first time. Treatment effects on N fluxes could not previously be measured because soil solution NH_4^+ and NO_3^- concentrations were at or near trace levels at all times. Measurement of forest floor fluxes with tension lysimeters is difficult because the forest floor is thin and subject to extreme wetting and drying cycles. Also, resin lysimeters avoid the problems of exchange or adsorption that may be associated with ceramic lysimeters (Lajtha et al., 1999). Susfalk (2000) noted that resin lysimeters collected substantially greater amounts of NH_4^+ , NO_3^- , and ortho-P than ceramic cup lysimeters in a comparative study in the Sierra Nevada mountains of Nevada. The differences were thought to be due to adsorption by the ceramic and perhaps also because of spoilage of water samples between collections.

With the resin lysimeters, we were able to detect large and statistically significant treatment effects on N fluxes from the forest floor (O horizons). The magnitudes of the N fluxes measured were within values that would be expected in this ecosystem. Nitrogen fluxes via litterfall and crownwash in this forest type are 34 and 3 $\text{kg N ha}^{-1} \text{ yr}^{-1}$, respectively (Johnson and Van Hook, 1989); thus, the resin lysimeter values suggested that there was a net uptake of N (by soil organisms, tree roots, or both) from the forest floor in the AMB and DRY treatments, as would be expected (with average fluxes of approximately 10 and 5 $\text{kg N ha}^{-1} \text{ yr}^{-1}$, respectively).

The differences in forest floor N content between treatments provide corroborating evidence that decomposition and N mineralization from litter are being reduced by the DRY treatment. Nitrogen content of litter

in the DRY treatment was about 70 kg N ha^{-1} greater than the AMB treatment (Table 6), suggesting a mean annual rate of N immobilization in the dry plot organic horizons of 12 $\text{Kg N ha}^{-1} \text{ yr}^{-1}$. This rate of immobilization is equal to about three times the difference in aerically weighted mineral N flux between the AMB and DRY plots in 1998–1999 (4.3 $\text{kg ha}^{-1} \text{ yr}^{-1}$; Fig. 3). In the 1999–2000 resin lysimeter collection, the aerically weighted fluxes in the DRY and AMB treatments were virtually identical (7.4 and 6.7 $\text{kg ha}^{-1} \text{ yr}^{-1}$, respectively), even though the fluxes beneath the troughs remained substantially and significantly lower in the DRY treatment. These differences could reflect error in estimation or they could reflect a leveling off of forest floor mass and N in the DRY treatment. Both the forest floor sampling in 1999 and the resin lysimeter data suggest greater sequestration of N in the forest floor in the DRY treatment.

Implications for Long-Term Soil Change and Plant Productivity

In a previous paper, we explored the implications of the treatments for long-term soil change using the Nutrient Cycling Model (NuCM; Johnson et al., 1998). The model predictions at that time indicated that, compared with the AMB treatment, soils in the DRY treatment would become enriched in base cations over a 30-year period, whereas those in the WET treatment would become depleted. The magnitudes of these changes did not suggest any measurable effect on plant productivity. These predictions remain to be tested. The NuCM also predicted that, because of slightly higher litter production in the WET treatment, forest floor mass and nutrient contents would increase relative to the AMB treatment, and the reverse would occur in the DRY treatment (lower forest floor mass and nutrient contents). This prediction was clearly not supported by the results of this study, where a significant increase in mass and N content in the DRY treatment and no effects in the WET treatment were found.

Perhaps the most significant effects relative to plant productivity noted in this study were the changes in O horizon leaching rates and the corresponding increases in forest floor N content in the DRY treatment. These effects are statistically significant, and excessive immobilization (or reduced mineralization) of N in the forest floor might reduce N availability for growth if sustained over long time periods. However, the rates of N immobilization observed here were relatively small and we hypothesize that they will eventually decline as a new steady state in the forest floor mass is reached.

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REFERENCES

- Cook, E.R., M.A. Kablack, and G.C. Jacoby. 1988. The 1986 drought in the southeastern United States: How rare an event was it? *J. Geophys. Res.* 93(D11):14257-14260.
- Data Description. 1988. DataDesk. Data Description, Ithaca, NY.
- Drungil, C.E.C., T.J. Gish, and K. Abt. 1987. Soil moisture determination in gravelly soils with time domain reflectometry. *Trans. ASAE* 32:177-180.
- Hanson, P.J., D.E. Todd, and J.S. Amthor. 2001. A six year study of sapling and large-tree growth and mortality responses to natural and induced variability in precipitation and throughfall. *Tree Physiol.* 21:345-358.
- Hanson, P.J., D.E. Todd, Jr., N.T. Edwards, and M.A. Huston. 1995. Field performance of the Walker Branch Throughfall Displacement Experiment. p. 307-313. *In* A. Jenkins et al. (ed.) Ecosystem manipulation experiments: Scientific approaches, experimental design, and relevant results. Ecosyst. Res. Rep. #20. Commission of the European Communities, Brussels.
- Hanson, P.J., D.E. Todd, M.A. Huston, J.D. Joslin, J. Croker, and R.M. Augé. 1998. Description and field performance of the Walker Branch Throughfall Displacement Experiment: 1993-1996. ORNL/TM-13586. Oak Ridge Natl. Lab., Oak Ridge, TN.
- High Performance Systems. 2000. Ithink. High Performance Systems, Hanover, NH.
- Huff, D.D., R.J. Luxmoore, J.B. Mankin, and C.L. Begovich. 1977. TEHM: A terrestrial ecosystem hydrology model. ORNL/NSF/EATC-27. Oak Ridge Natl. Lab., Oak Ridge, TN.
- Johnson, D.W., D.W. Cole, F.W. Horng, H. Van Miegroet, and D.E. Todd. 1981. Chemical characteristics of two forested Ultisols and two forested Inceptisols relevant to anion production and mobility. ORNL/TM-7646. Oak Ridge Natl. Lab., Oak Ridge, TN.
- Johnson, D.W., P.J. Hanson, D.E. Todd, R.B. Susfalk, and C.F. Trettin. 1998. Precipitation change and soil leaching: Field results and simulations from Walker Branch Watershed, Tennessee. *Water Air Soil Pollut.* 105:251-262.
- Johnson, D.W., D.D. Richter, G.M. Lovett, and S.E. Lindberg. 1985. The effects of atmospheric deposition on potassium, calcium, and magnesium cycling in two deciduous forests. *Can. J. For. Res.* 15: 773-782.
- Johnson, D.W., R.B. Susfalk, H.L. Gholz, and P.J. Hanson. 2001. Simulated effects of temperature and precipitation change in several forest ecosystems. *J. Hydrol.* 235:183-204.
- Johnson, D.W., and D.E. Todd. 1990. Nutrient cycling in forests of Walker Branch Watershed: Roles of uptake and leaching in causing soil change. *J. Environ. Qual.* 19:97-104.
- Johnson, D.W., and R.I. Van Hook (ed.) 1989. Analysis of biogeochemical cycling processes in Walker Branch Watershed. Springer-Verlag, New York.
- Jones, E.A., D.D. Reed, G.D. Mroz, H.O. Liechty, and P.J. Cattellino. 1993. Climate stress as a precursor to forest decline: Paper birch in northern Michigan, 1985-1990. *Can. J. For. Res.* 23:229-233.
- Joslin, J.D., M.H. Wolfe, and P.J. Hanson. 2000. Effects of altered water regimes on forest root systems. *New Phytol.* 147:117-129.
- Kirschbaum, M.U. 1995. The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biol. Biochem.* 27:753-760.
- Kjønass, O.A. 1999. In situ efficiency of ion exchange resins in studies of nitrogen transformation. *Soil Sci. Soc. Am. J.* 63:399-409.
- Lajtha, K., W.M. Jarrell, D.W. Johnson, and P. Sollins. 1999. Collection of soil solution. p. 166-182. *In* G.P. Robertson et al. (ed.) Standard soil methods for long-term ecological research. Oxford Univ. Press, New York.
- Lamersdorf, N.P., C. Bier, K. Blanck, M. Bredemeier, T. Cummins, E.P. Farrel, L. Rasmusson, and M. Ryan. 1995. Reactions of soil solution chemistry to drought: Results of the EXMAN project. p. 86-95. *In* A. Jenkins (ed.) Ecosystem manipulation experiments: Scientific approaches, experimental design, and relevant results. Ecosyst. Res. Rep. #20. Commission of the European Communities, Brussels.
- Likens, G.E., F.H. Bormann, R.S. Pierce, J.S. Eaton, and N.M. Johnson. 1977. Biogeochemistry of a forest ecosystem. Springer-Verlag, New York.
- Luxmoore, R.J. 1983. Water budget of an eastern deciduous forest stand. *Soil Sci. Soc. Am. J.* 47:785-791.
- Luxmoore, R.J., and D.D. Huff. 1989. Water. p. 197-232. *In* D.W. Johnson and R.I. Van Hook (ed.) Analysis of biogeochemical cycling processes in Walker Branch Watershed. Springer-Verlag, New York.
- Reuss, J.O. 1983. Implications of the Ca-Al exchange system for the effect of acid precipitation on soils. *J. Environ. Qual.* 12:591-595.
- Schnabel, R.R. 1983. Measuring nitrogen leaching with ion exchange resin: A laboratory assessment. *Soil Sci. Soc. Am. J.* 47:1041-1042.
- Schnabel, R.R., S.R. Messier, and R.F. Purnell. 1993. An evaluation of anion exchange resin used to measure nitrate movement through soil. *Commun. Soil Sci. Plant Anal.* 24:863-879.
- Sibbesen, E. 1978. An investigation of the anion-exchange resin method for soil phosphate extraction. *Plant Soil.* 50:305-321.
- Susfalk, R.B. 2000. Relationships of soil-extractable and plant-available phosphorus in forest soils of the eastern Sierra Nevada. Ph.D. diss. Univ. of Nevada, Reno.
- Topp, G.C., and J.L. Davis. 1985. Measurement of soil water content using time domain reflectometry (TDR): A field evaluation. *Soil Sci. Soc. Am. J.* 49:19-24.
- Ulrich, B. 1983. Soil acidity and its relation to acid deposition. p. 127-146. *In* B. Ulrich and J. Pankrath (ed.) Effects of accumulation of air pollutants in ecosystems. D. Reidel Co.
- Velleman, P.F. 1997. DataDesk Version 6.0 statistics guide. Data Description, Ithaca, NY.