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Changes in nutrient distribution in forests and soils of Walker Branch watershed, Tennessee, over an eleven-year period

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Abstract. Changes in vegetation, litter, and soil nutrient content were measured in selected plots on Walker Branch watershed, Tennessee, from 1972–73 to 1982. The watershed has been allowed to revert to forest since 1942, before which it consisted of small subsistence farms and woodland pastures. Changes in Ca status were of particular interest because initial nutrient cycling characterizations indicated that net Ca accumulation in vegetation could have caused large decreases in soil exchangeable Ca^{2+} within 20 years.

Decreases in forest floor and subsoil (45–60 cm) N, exchangeable Ca^{2+}, and Mg^{2+} content were noted in several plots from 1972 to 1982. Surface soils (0–15 cm) showed either no change or, in some cases (e.g., N and exchangeable K^+ in certain plots), increases over the 11-year period. Reductions in forest floor and subsoil exchangeable Ca^{2+} and exchangeable Mg^{2+} on cherty, upper slope oak-hickory and chestnut oak forests were most striking. The changes in Ca^{2+} are thought to be due primarily to high rates of Ca incorporation into woody tissues of oak and hickory species. Reductions in forest floor and subsoil exchangeable Mg^{2+} could not be accounted for by woody increment; leaching may have played a major role in causing these decreases. Changes in P and exchangeable K^+ were variable, with both increases and decreases.

There were significant increases in exchangeable Al^{3+} in both subsoils and surface soils of certain plots, but these were not accompanied by decreases in exchangeable base cations or consistent decreases in pH. Dissolution of interlayer Al from 2:1 clays may be the cause of the exchangeable Al^{3+} increases.

These results suggest a general decline in fertility, especially with regard to Ca and Mg in those forests with low soil Ca and Mg supplies. Monitoring of further changes (if any) in these ecosystems will continue as the currently aggrading forests approach steady state.

Introduction

There have been numerous studies on the effects of afforestation and species conversion on soil chemical properties (e.g. Ovington 1958a, b; Nihlgard 1971; Rennie 1955; Alban 1982; Turner & Kelly 1977). Early work on this subject in the United Kingdom was initiated as a result of concerns that afforestation or site conversion to conifers would lead to soil deterioration through podzolization (Rennie 1955).

Recent concerns over acid deposition effects have rekindled interest in documenting long-term changes in soil acidity and nutrient status (Ulrich et al. 1980; Linzon & Temple 1980; Malmer 1976). In assessing the causes of changes in soil acidity, it is important to consider both leaching and vegetation uptake. Preferably, studies of long-term soil changes should be accompanied by detailed element cycling studies. This is seldom the case, but notable exceptions are the studies by Ulrich et al. (1980) in a beech forest in Germany, where acid deposition is thought to have produced marked changes in both soil and soil solution chemistry, and studies by Van Miegroet & Cole (1984), where nitrification coupled with nitrate leaching in a N-fixing red alder stand caused soil acidification.

Walker Branch Watershed (WBW) has been subjected to several intensive element cycling studies, including analyses of acid deposition effects on soil leaching in two forest ecosystems (chestnut oak and a yellow-poplar stands) (Richter et al. 1983; Johnson et al. 1985). These studies showed that although acid deposition likely had accelerated the rate of soil leaching by at least a factor of two (to current rates of 1.7 and 2.7 $\mathrm{kmol}(+)\cdot\mathrm{ha}^{-1}\cdot\mathrm{yr}^{-1}$ in the chestnut oak and yellow-poplar sites, respectively), soil exchangeable cation (SEC) reserves were sufficiently large (84 and 113 $\mathrm{kmol}(+)\cdot\mathrm{ha}^{-1}$) that no perceptible changes in soil base cation status or acidity were expected in less than many decades or even centuries (Richter et al. 1983; Johnson et al. 1985). On the other hand, both these and previous nutrient cycling studies on WBW found that soil exchangeable Ca^{2+} supplies were very low (20–30 $\mathrm{kmol}(+)\cdot\mathrm{ha}^{-1}$) relative to the rather large vegetation Ca uptake and accumulation rates (1.6–2.0 $\mathrm{kmol}(+)\cdot\mathrm{ha}^{-1}\mathrm{yr}^{-1}$), especially in forest types dominated by oaks (*Quercus* spp.) and hickories (*Carya* spp.) (Henderson et al. 1978; Cole & Rapp 1981; Johnson et al. 1985). The budgets suggested that sequestering of Ca in tree woody tissues could, in itself, cause a complete depletion of soil exchangeable Ca^{2+} in the oak-hickory and chestnut oak forest types in 10 to 20 years unless vegetation uptake was supplemented by weathering and/or deep rooting (beyond 60 cm). Clearly, weathering and deep rooting must occur in these forests or the trees would not survive. However, the combination of low soil pools and high vegetation uptake rates led us to hypothesize that decreases in soil exchangeable Ca^{2+} would occur.

Given the historical data base and, in many cases, the original samples from early nutrient cycling studies on WBW (Henderson & Harris 1975; Henderson et al. 1978), we were able to test the validity of some of the projections described above by a systematic resampling of original study plots. Soil leaching data are absent for these plots, but one plot was adjacent (20 m) to the intensively studied chestnut oak stand described above. Potential effects of vegetation increment on soil changes could be assessed from 1973 to 1982 biomass and nutrient content data in all plots.

Site

Walker Branch watershed is a 97.5-ha forested catchment located near Oak Ridge in eastern Tennessee. It is underlain by dolomite from which the residual soils were formed. Soils of the Fullerton and Bodine series occupy over 90% of the watershed. Fullerton soils belong to the clayey, kaolinitic, thermic family of Typic Paleudults. They occupy the ridge tops and upper slope positions. Bodine soils belong to the loamy-skeletal, silicaceous thermic family of Typic Paleudults. They occupy intermediate and lower slopes. Areas of Claiborne soils (fine-loamy, silicaceous Typic Paleudults) occupy minor areas in the major stream bottoms on alluvial or colluvial deposits.

Fullerton soils are characterized by a very thin (5 cm) dark grayish brown cherty silt loam A horizon; a thicker (20–40 cm) brown cherty silt loam E horizon; discontinuous, thin (strong brown) AB (10–15 cm) and yellowish-red Btl (10–20 cm) horizons, and thick red cherty clay (50–100 cm) Bt2, Bt3, and Bt4 horizons. Depth of soil to bedrock ranges from 20 to 30 m on ridgetops. Bodine soils have less clay and less reddish color in the Bt horizons, and a greater percent of coarse fragments, than do Fullerton soils. Claiborne soils are characterized by a dark brown silt loam Ap horizon (15–25 cm thick), by a reddish brown silt loam BA horizon (10–20 cm thick), a reddish brown silty clay loam Bt horizon (20–45 cm thick), a yellowish red silt clay loam Bt2 horizon (38–76 cm thick), and a red clay Bt3 horizon (38–76 cm thick).

Vegetation is primarily oak-hickory (*Quercus* spp.-*Carva* spp.), with scattered pine (*Pinus echinata* Mill. and *P. virginiana* Mill.) on the ridgetops and mesophytic hardwoods (predominantly *Liriodendron tulipifera* L. and *Fagus grandifolia* Ehrh.) in protected coves and stream bottoms. Mean basal area is approximately 21 $m^2 \cdot ha^{-1}$. Precipitation averages 151 $cm \cdot year^{-1}$, with approximately 57% draining to streams (Henderson et al. 1978). The four major forest types as identified by Grigal & Goldstein (1971) include oak-hickory (*Quercus* spp. – *Carya* spp.), chestnut oak (*Q. prinus*), yellow-poplar (*Liriodendron tulipifera*), and pine (*Pinus echinata* and *P. virginiana*).

Methods

Two-hundred and ninety-eight permanent 0.1-ha, nested inventory plots were established on WBW in 1967 as described by Harris et al. (1973). Results from eight selected plots (representing a range of soil and vegetation types) are reported here; a full discussion of the vegetation changes on WBW will be presented elsewhere. Measurements of diameter at breast height (dbh) taken in 1973 and in 1982 were used in this comparison. Biomass was estimated from the pine, hardwood, and yellow-poplar equations given by Harris et al. (1973). Net increments of biomass and nutrients in vegetation were estimated from the differences between biomass and nutrient contents (in live and dead trees) between 1973 and 1982. Nutrient content in vegetation was obtained by multiplying the biomass of each component (foliage, branch, bole) in each of 20 major species (constituting over 95% of total biomass on the watershed) by nutrient concentrations in that component and species. Nutrient concentrations were derived by destructive sampling on both WBW (Henderson et al. 1978) and a nearby whole-tree harvesting site (Johnson et al. 1982). Results of these two samplings were pooled because values for the species analyzed were not significantly different. Pie-shaped sections of woody tissues and samples of July 1982 foliage were analyzed for N, P, K, Ca, and Mg. Plant tissues were dry ashed overnight at 520 °C. The ashed samples were digested with 1 *N* HCl and diluted to volume with deionized distilled water. Potassium, Ca, and Mg were analyzed by flame atomic absorption, and P was determined by an automated colorimetric procedure. Nitrogen was determined by Kjeldahl digestion in a block digestor followed by ammonium analysis by an automated colorimetric procedure. National Bureau of Standards orchard and pine leaves were run as blind samples and analyses were accepted only if results were within 10% of published values.

In addition to a vegetation survey, 24 core plots were intensively sampled for forest floor weight and nutrient content and soil N at 15-cm intervals to a 60 cm depth on a seasonal (quarterly) basis during 1972. Samples were taken randomly from a 12- × 12-m grid subplot (1 sample per m^2 square, 3–7 samples per 12- × 12-m subplot) within each inventory plot. Samples were dried, and soils were sieved and stored. Unfortunately the 15- to 30- and the 30- to 45-cm-depth soil samples were lost. Quarterly samples were also taken from intensively studied Fullerton, Bodine, and Claiborne series soil sites in 1972 and analyzed to determine seasonal variations in extractable P and exchangeable Ca^{2+}, Mg^{2+}, and Al^{3+}.

In the spring of 1982, eight of the 24 original core plots were resampled using the same 12- × 12-m grid system originally established. Brief descriptions of these plots are given in Table 1. All samples were taken as in the original collection [0.25-m^2 circular ring for forest floor, bucket auger at 15-cm intervals to a depth of 60cm for soils, 4–6 samples per subplot]. All soils from both periods were analyzed for pH (1:1 soil:solution in H_2O and in 0.01 M $CaCl_2$), total C (LECO Analyzer), total N (Kjeldahl, using block digestor and autoanalyzer analysis for NH_4^+), CEC, and exchangeable Ca^{2+}, Mg^{2+}, K^+, Na^+, and Al^{3+} (by unbuffed 1 M NH_4Cl extraction followed by an ethanol wash and 1 M KCl to displace NH_4^+), and extractable P (0.5 M HCl plus 1 M NH_4Cl; Olsen & Dean 1965).

Soils were bulked by horizon and plot for total P, K, Ca, and Mg analyses. Soil samples for total analyses were dissolved by treating the samples with HF and HNO_3 and evaporating to dryness. Then $HCLO_4$ and HNO_3 were added to the samples which again were evaporated to dryness. The residue was dissolved in 1:1 HCl and diluted to volume with deionized distilled water. Potassium, Ca, and Mg were analyzed by flame atomic absorption and P was determined by autoanalyzer. Standard soils from the Canada Centre for Mineral and Energy Technology (Soils SO-1, SO-2, and SO-4) were run as blind samples and results were within 10% of published values.

T-tests were used to ascertain significant differences (95% confidence level between old (1972) and new (1982) nutrient concentrations. Soil nutrient contents (on an aerial basis) were calculated using bulk density and perient gravel data for each soil series from Peters et al. (1970).

Old (1972) and new (1982) soil samples were analyzed simultaneously

Table 1. Brief descriptions of core plots on Walker Branch watershed, Tennessee, sampled in 1972–73 and 1982.

Forest type[a]	Plot	Slope position	Dominant vegetation	Soil series
Pine	26	Ridgetop, slight depression	Shortleaf pine yellow-poplar	Fullerton silt loam
Yellow-poplar	98	Cove	Yellow-poplar, white oak	Claiborne cherty silt loam
Yellow-poplar	281	Lower slope near cove	Yellow-poplar	Claiborne silt loam
Chestnut oak	42	Ridgetop	Chestnut oak	Fullerton silt loam
Chestnut oak	179	Upper slope near ridgetop	Chestnut oak, red maple	Fullerton cherty silt loam
Oak-hickory	91	Secondary narrow ridgetop	Hickory, chestnut oak	Bodine cherty silt loam
Oak-hickory	107	Secondary ridgetop	Black oak, hickory	Fullerton cherty silt loam
Chestnut oak	237	Midslope, slight depression	Chestnut oak, white oak	Fullerton silt loam

[a] According to original classification by Grigal & Goldstein (1971).

Table 2. Nutrient distributions (in kg·ha^{-1}) in eight intensively sampled plots on Walker Branch watershed in 1982.

	Pine	Yellow-poplar		Chestnut oak		Oak-hickory		
Ecosystem component	Plot 26	Plot 98	Plot 281	Plot 42	Plot 179	Plot 91	Plot 107	Plot 237
Vegetation	Nitrogen							
Live	320	424	475	441	426	365	387	308
Standing dead	59	14	41	25	27	58	67	32
Net increment[a]	103	74	82	62	69	63	98	33
Forest floor	140	168	163	277	309	206	445	293
Soil	3,190	5,300	6,160	2,780	2,150	2,700	2,360	3,560
Total ecosystem	3,709	5,906	6,839	3,523	2,912	3,329	3,529	4,193
Vegetation	Phosphorus							
Live	26	32	37	29	32	26	28	22
Standing dead	5	1	3	2	2	4	5	2
Net increment[a]	5	6	7	5	6	4	7	2
Forest floor	9	3	9	15	21	17	26	18
Soil-extractable[b]	20	23	17	20	37	25	28	48
Soil-total	670	1,180	1,600	800	580	990	660	880
Total ecosystem	710	1,216	1,649	846	623	1,037	719	922
Vegetation	Potassium							
Live	187	230	241	224	213	230	286	162
Standing dead	38	7	20	24	14	50	55	19
Net increment[a]	20	38	37	35	30	35	58	18
Forest floor	10	14	12	22	25	14	31	23
Soil-exchangeable[c]	270	315	207	268	193	220	310	267
Soil-total	17,400	21,800	19,000	19,600	18,400	20,400	18,400	12,500
Total ecosystem	17,635	22,051	19,273	19,870	18,627	20,694	18,772	12,704

Vegetation	Calcium							
Live	495	1,020	824	1,240	1,244	1,290	1,630	930
Standing dead	100	40	76	129	57	296	334	103
Net increment[a]	145	188	160	183	238	235	400	169
Forest floor	168	232	187	562	150	355	353	340
Soil-exchangeable[c]	820	1,390	1,500	990	110	282	142	1,580
Soil-total	2,500	2,540	2,770	2,230	1,180	1,410	1,800	2,480
Total ecosystem	3,263	3,832	3,857	4,161	2,631	3,351	4,117	3,853
Vegetation	Magnesium							
Live	53	70	83	54	51	64	83	40
Standing dead	10	1	7	7	4	15	15	3
Net increment[a]	16	11	15	8	9	9	26	4
Forest floor	15	18	14	23	13	16	34	16
Soil-exchangeable[c]	120	172	174	170	29	49	43	141
Soil-total	4,700	8,240	6,860	5,200	4,870	7,000	5,330	5,530
Total ecosystem	4,778	8,329	6,964	5,284	4,938	7,095	5,462	5,589

[a] Net accumulation in live and dead tree biomass from 1973 to 1982; [b] NH_4F/HCl extraction; [c] NH_4Cl extraction.

during 1983–1984 to avoid laboratory bias. Potential changes in soil exchangeable Ca^{2+}, K^+, and Mg^{2+} during storage were checked by re-analyzing the 1972 samples from the seasonally sampled soil site. Regression analyses gave (in $cmol(+) \cdot kg^{-1}$) (old analysis) = (0.82) (new analysis) + 0.02, $r^2 = 0.97$, for Ca^{2+}; (old analyses) = (0.81) (new analyses) + 0.01, $r^2 = 0.95$, for Mg^{2+}; and (old analysis) = (1.03) (new analyses) + 0.002, $r^2 = 0.94$ for K^+. Laboratory bias as well as changes during soil storage could have contributed to the somewhat higher exchangeable Ca^{2+} and Mg^{2+} in the 1983–4 analyses, but it is clear that no *major* changes in exchangeable Ca^{2+}, Mg^{2+}, or K^+ occurred during soil storage.

Forest floor samples had originally been analyzed for N, P, K, Ca, and Mg in 1972, so only a selected number of samples (n = 24) were re-analyzed during 1983–1984. Results showed good correlations except in the case of K: for N [(old analysis) = (0.87) (new analysis) + 0.25, $r^2 = 0.82$], for P [(old analysis) = (0.87) (new analysis) − 0.0001, $r^2 = 0.90$], for K^+ [(old analysis) = 0.93 (new analysis) + 0.009, r^2 0.35], for Ca [(old analysis) = 0.89 (new analysis) + 0.14, $r^2 = 0.95$], and for Mg [(old analysis) = (1.10) (new analysis) − 0.002, $r^2 = 0.76$] (all values in $mg \cdot g^{-1}$). It is noteworthy that old analyses tend to be slightly lower than new analyses except in the case of Mg.

Pronounced seasonal variations in exchangeable Ca^{2+}, K^+, and extractable P were noted in samples from the 0- to 15-cm depth of the seasonally sampled soil site (see results); therefore, only the 0- to 15-cm-depth samples taken in March and April of 1972 were analyzed and compared with those taken in March of 1982. Seasonal variations at lower soil depths were minimal.

Results

Aboveground nutrient distribution

The distribution of nutrients in 1982 and the net nutrient increments in vegetation from 1973 to 1982 are given in Table 2. As noted by Henderson et al. (1978) for the WBW as a whole, vegetation contained a higher proportion of total ecosystem Ca (18–49%) than of any of the other major nutrients (8–16% of total N, 2–5% of total P, 1.1–1.4% of total K, and 0.9–1.8% of total Mg). Vegetation Ca was a particularly important component of ecosystem Ca content in the cherty, upland chestnut oak and oak-hickory plots (nos. 91, 107, and 179), where it exceeded soil exchangeable Ca^{2+} content by 5 to 10 times and accounted for 40–70% of the total

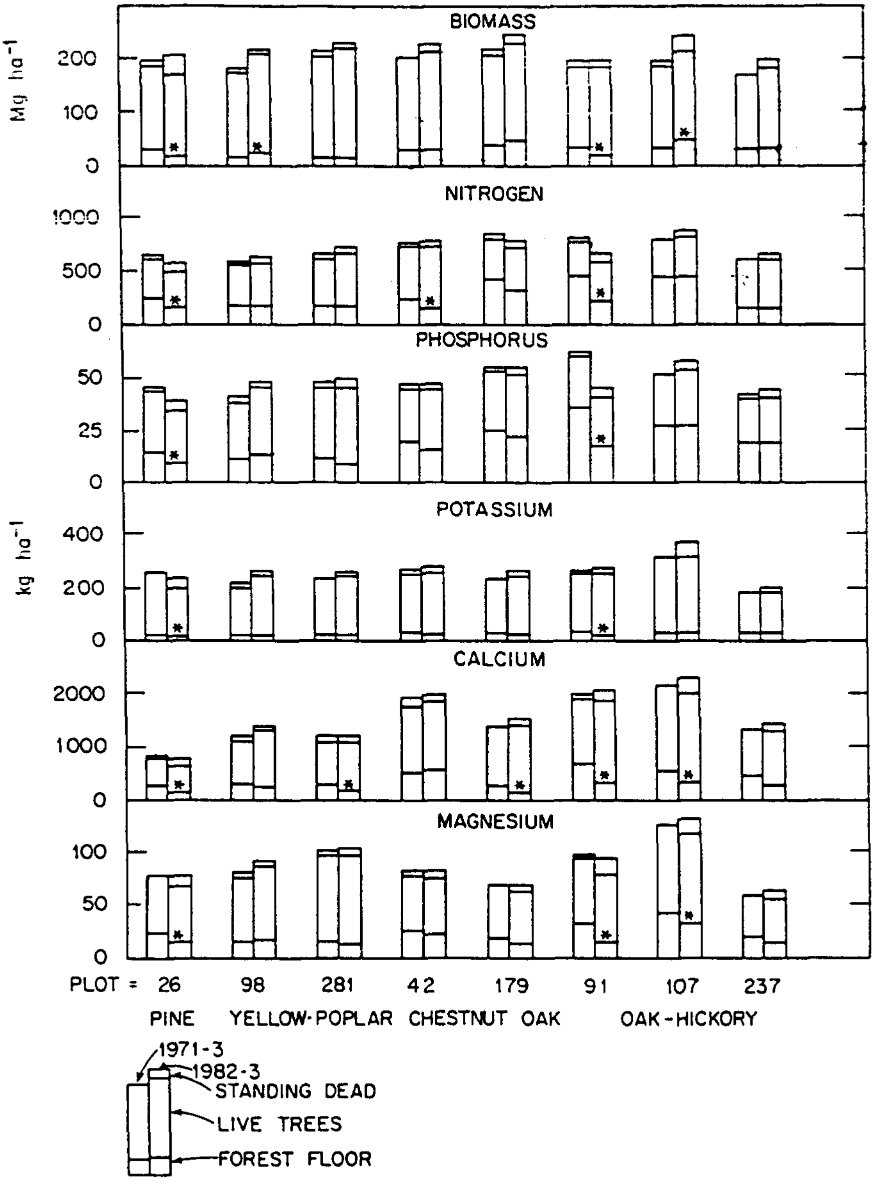

Fig. 1. Changes in biomass and nutrient content in vegetation and forest floor in selected plots from Walker Branch watershed from 1972–73 to 1982. (* denotes statistically significant difference, 95% confidence level, t-test.)

ecosystem Ca content (to a soil depth of 60 cm). Furthermore, the vegetation Ca increment from 1973 to 1982 in these plots equalled 80 to 280% of the soil exchangeable Ca^{2+} content, implying that the vegetation Ca increment alone could have caused a significant reduction in soil exchangeable Ca^{2+} over this time period unless soil exchangeable Ca^{2+} supplies were rapidly replenished by weathering or by deep rooting. In contrast, the vegetation increment of N, P, K, and Mg from 1973 to 1982 equalled only 3–20% of soil exchangeable or extractable contents of these elements (Table 1).

There were some differences in vegetation, forest floor, and total soil Ca among the eight resampled plots, but the major differences were in soil exchangeable Ca^{2+}. In the three cherty, upland chestnut oak and oak-hickory plots (91, 107, 179), soil exchangeable Ca^{2+} constituted only 8 to 20% of total soil Ca whereas soil exchangeable Ca^{2+} constituted 33 to 64% of total soil Ca in the other five plots (to a soil depth of 60 cm).

The changes in aboveground tree (live and standing dead) and forest floor biomass and nutrient content from 1972–73 to 1982 are given in Fig. 1. Because the intermediate (15–45 cm) soil depths were not sampled in 1972, a full accounting of changes in soil nutrient cannot be made, but comparisons between the top (0–15 cm) and the deepest (45–60 cm) soil sampling depths are given later.

All plots had a net increase in aboveground (vegetation plus forest floor) biomass from 1973 to 1982, although in some cases (plots 26, 107), this was due in large measure to an increase in standing dead biomass. Obviously, in these cases mortality exceeded the growth of live trees. Outbreaks of the southern pine beetle in the early 1970s and the hickory borer in the late 1970s are largely responsible for the mortality in the pine and oak-hickory forest types.

Despite the overall increase in aboveground biomass, several plots showed either no change or a net decrease in aboveground nutrient content, primarily due to decreases in forest floor nutrient content (Fig. 1). In some cases (plots 26 and 91), the decreases in forest floor nutrient content were due largely to decreases in forest floor biomass, but in other cases the decrease in nutrient content was clearly due to a decrease in nutrient concentration: several plots showed a decrease in forest floor nutrient content with no change in forest floor biomass (N in plot 42, Ca in plots 281 and 179), and some plots showed either no change or a decrease in forest floor nutrient content with increased forest floor biomass (plots 98 and 107). Indeed, there was a marked trend toward decreased Oe horizon nutrient concentration from 1971 to 1982 in most plots (Fig. 3).

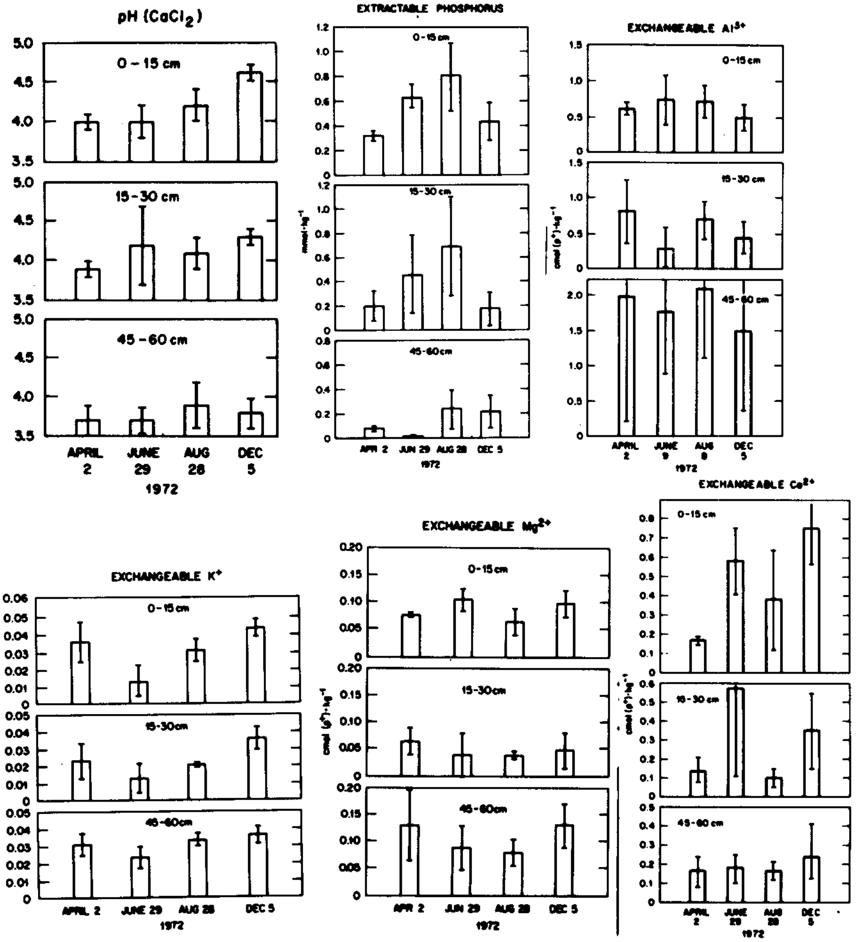

Fig. 2. Seasonal variations in pH (1:1 ratio in 0.01 *M* $CaCl_2$), extractable P (NH_4F/HCl), exchangeable Ca^{2+}, Mg^{2+}, K^+, and Al^{3+} (1 *M* NH_4Cl) in a soil profile on Walker Branch watershed. (Ca^{2+}, Mg^{2+}, and pH data after Johnson 1986.)

Seasonal variations in soils

Oi (recognizable leaf litter) horizon weights and nutrient concentrations are known to vary seasonally (weight by 40%, nitrogen by 20%) on WBW as a result of litterfall pulses (G.S. Henderson, unpubl. data 1972). Forest floor sampling was done in the spring of 1971 and 1982 to avoid the confounding effects of seasonal variation; however, Oi weights and concentrations might well vary from year to year as well as with season. Oe (decomposed litter) horizons show much less seasonal variation (weight by $\leqslant 21\%$, nitrogen by

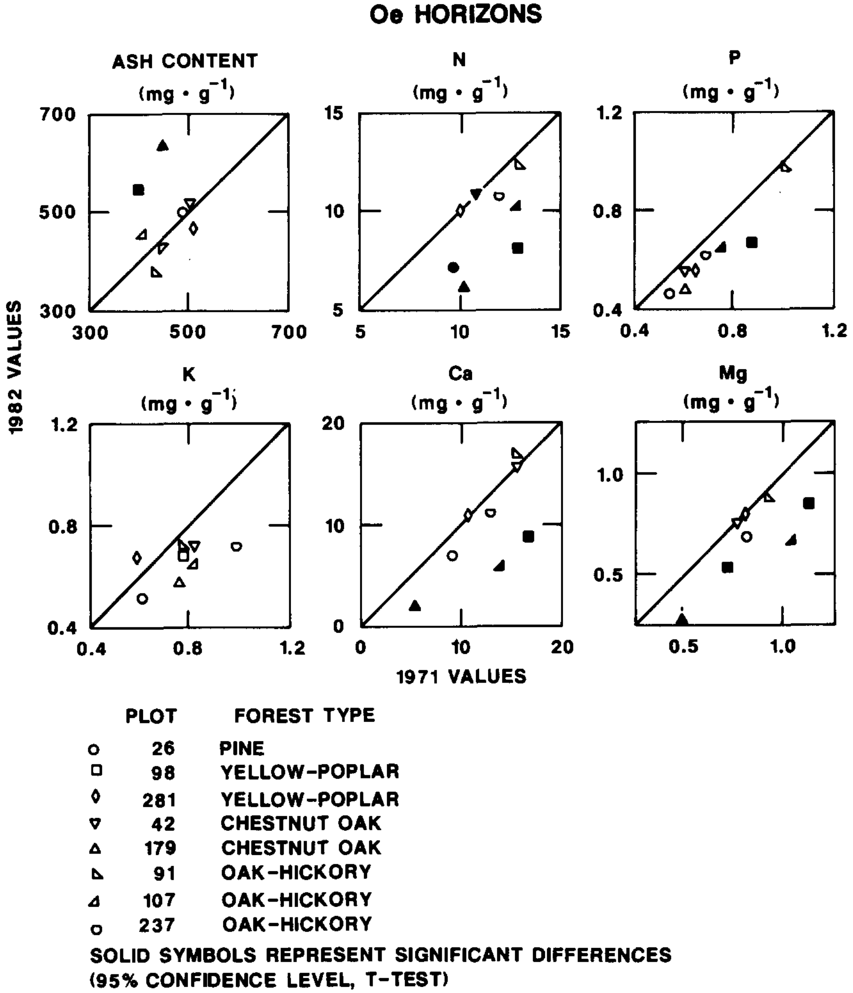

Fig. 3. 1982 vs 1971 average values for ash content, total N, P, K, Ca, and Mg in Oe horizons from selected plots on Walker Branch watershed. Solid symbols represent significant differences (95% confidence level, t-test) between 1982 and 1971 values.

≤ 7%), however (G.S. Henderson, unpubl. data 1972). Seasonal variations in P, K, Ca, and Mg in Oe horizons have not been determined, but given the slight variations in N, one would expect variations in these nutrients to be minimal.

Haines & Cleveland (1981) noted seasonal variations in several soil chemical and physical properties and point out the need to account for seasonal variations in sampling for long-term changes in soils. Seasonal variations in pH, extractable P, exchangeable K^+, and exchangeable Ca^{2+} were pronounced in the 0–15 cm depth of the seasonally-sampled Fullerton soil site adjacent to plot 179 (Fig. 2). Seasonal variations in exchangeable Al^{3+} and Mg^{2+}, if they occurred were masked by spatial variability. However, we have previously noted seasonal variations in exchangeable Al^{3+} in a Tarklin soil (Typic fragiudult) supporting a yellow-poplar forest site on Walker Branch (Johnson & Todd 1984), so the possibility cannot be excluded. The seasonal variations in all but extractable P become minimal deeper in the soil. The seasonal variations in extractable P, exchangeable Ca, and exchangeable K^+ in the 0–15 cm depth convert to approximately 20, 120, and 10 $kg \cdot ha^{-1}$, which are of the same magnitude as those estimated for litterfall and root turnover of these elements (Henderson et al. 1978).

Long-term changes in litter and soils

As noted above, there was a consistent trend toward lower nutrient concentrations in Oe horizons from 1971 to 1982 (Fig. 3). This trend was particularly marked, and most statistically significant for N, Ca, and Mg.

One significant potential source of error in the above comparisons is the depth of forest floor sampling. Variations in depth of forest floor sampling cause variations in the amount of surface mineral soil included in forest floor samples which can affect both weight estimates and nutrient analyses. A check of consistency in forest floor sampling is provided by analyses for ash content. The ash contents of Oe horizons sampled in 1971 and 1982 were comparable in all but two of the plots sampled (plots 98 and 179; Fig. 3). Thus, the apparent increase in forest floor weight from 1971 to 1982 in plot 98 and the apparent decrease in forest floor nutrient concentrations in plots 98 and 179 (Figs. 2, 3) are suspect. When expressed on an ash-free basis, the differences in 1971 vs 1982 Oe horizon P and Mg in plot 98 and Mg in plot 179 were no longer statistically significant. However, the other statistically significant differences noted in Figs. 1 and 3 remained statistically significant when expressed on an ash-free basis. Thus, the overall declines in nutrient concentrations between the two sampling dates apparently represent actual changes with time rather than differences in sampling technique.

There were some statistically significant differences in total N, extractable P, exchangeable K^+, Ca^{2+}, Al^{3+}, pH and CEC in the surface (0–15 cm) soils of certain plots between the spring 1972 and spring 1982 samplings, but no overall (i.e. across-plot) trends were evident (Fig. 4). Furthermore, differences in surface soils (where they existed) must be regarded with caution given the seasonal variations noted above. There were several statistically significant changes in subsoil (45–60 cm) chemical properties within certain plots from 1972 to 1982 which were presumably not confounded by seasonal

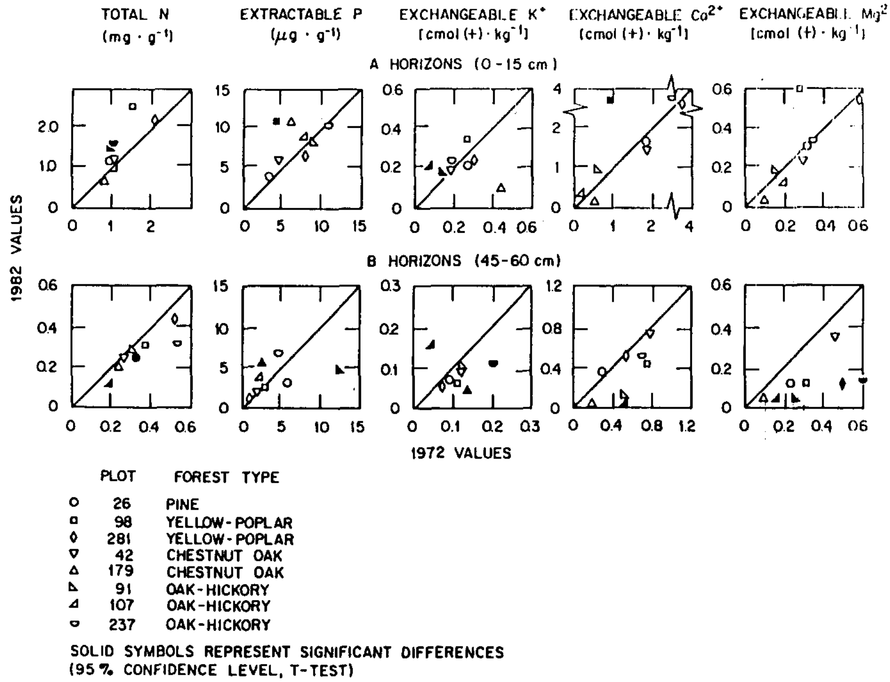

Fig. 4. 1982 v 1972 average values for total N, extractable P, exchangeable Ca^{2+}, K^+, Mg^{2+} in surface (0–15 cm) and subsurface (45–60 cm) soils from selected plots on Walker Branch watershed. Solid symbols represent significant differences (95% confidence level, t-test) between 1972 and 1982 values.

variations, however. The most notable overall (i.e. across-plot) trend was a decrease in subsoil exchangeable base cations (Figs. 4, 5). There was also a slight but consistent trend toward lower subsoil N concentrations (Fig. 4). The decrease in subsoil exchangeable base cations might be expected to be accompanied by decreases in pH, increases in exchangeable Al^{3+}, and perhaps a decrease in CEC (the latter due to reduction in the pH-dependant component of CEC). While there were statistically significant changes in these soil properties in some plots, these changes were not consistent with what would normally be expected from the observed changes in exchangeable base cations. For instance, both horizons of plot 26 and the surface horizon of plot 42 had a lower pH and a higher exchangeable Al^{3+} level in 1982 than in 1972, yet there were little or no differences in exchangeable base cations (Fig. 5). Also, the decreases in exchangeable base cations in subsoils from the cherty, upper slope plots (91, 107, and 179) were not accompanied by significant increases in levels of exchangeable Al^{3+} or decreases in pH; indeed, plots 91 and 107 showed significant *increases* in $CaCl_2$ pH from 1972

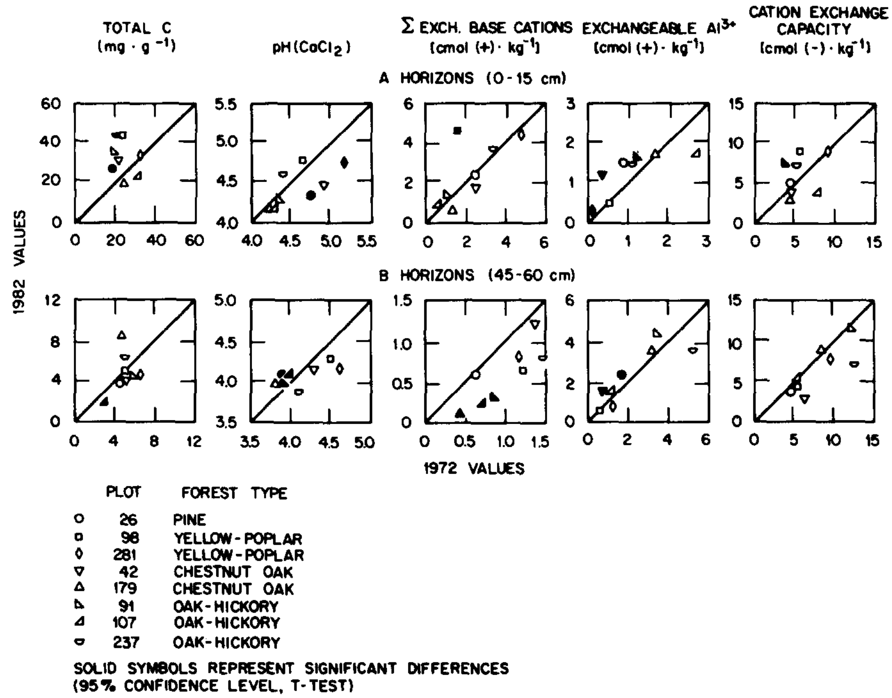

Fig. 5. 1982 vs 1972 average values for total carbon, pH (1:1 ratio in 0.01 M $CaCl_2$), exchangeable base cations ($\Sigma BC = Ca^{2+} + Mg^{2+} + K^+ + Na$), exchangeable Al^{3+}, and cation exchange capacity (1 M NH_4Cl extraction) in surface (0–15 cm) and subsurface (45–60 cm) soils from selected plots on Walker Branch watershed. Solid symbols represent significant differences between 1972 and 1982 values.

to 1982 (Fig. 5). On the other hand, the decreases in pH corresponded with increases in Al^{3+} (but not with decreases in base cations) in plots 281 and 42 at the 0- to 15-cm depths (Fig. 5).

Many of these apparent anomolies may be due to the very large proportions of Al^{3+} on the cation exchange complex. That is, a replacement of base cations by Al^{3+} in these very acid soils might well result in a relatively large decrease in base cations but only a small, perhaps undetectable, increase in Al^{3+}. Another possibility is the dissolution of interlayer polyhydroxyl Al^{3+} as suggested by Ulrich et al. (1980) to explain similar changes (increased Al^{3+} but no change in base cations) in soils at the Solling site in West Germany. Lee et al. (1984) note the presence of polyhydroxyl Al^{3+} in vermiculite clays of soils of the same series and adjacent to Walker Branch. It is possible that the seemingly anomolous releases of Al^{3+} from soils of plots 26 and 42 originated from this source.

Discussion

Seasonal variations in exchangeable Ca^{2+}, Al^{3+}, K^+, and Mg^{2+}, appear to be more significant than any long-term trends in the surface (0–15 cm) horizons of these soils. Similarly, seasonal variations in extractable P appear to be more significant than any long-term trends in either surface or subsurface horizons (Figs. 2 and 4). Seasonal variations in all but extractable P were minor and statistically insignificant in Oe and subsoil (45–60 cm) horizons, however, and the marked changes in the nutrient contents of these horizons (especially the decreases in Ca^{2+} and Mg^{2+}), deserve further scrutiny. Vegetation increment could have caused the subsoil exchangeable Ca^{2+} decreases in the three cherty, upland chestnut oak and oak-hickory forests (plots 91, 107, and 179). Vegetation increment per se could not have accounted for the subsoil exchangeable Mg^{2+} decreases in these three plots or in the other two plots (281 and 237) in which they occurred. However, it is possible that total vegetation uptake (which exceeds the net vegetation increment by 5- to 10-fold) and subsequent return via litterfall to the soil surface has resulted in a redistribution of nutrients from subsoils to surface soils, as suggested by Thomas (1967) for Ca "pumping" by dogwood trees on WBW. Without a full accounting of changes in all soil horizons, it is not possible to either confirm or reject this hypothesis.

The role of leaching in causing these changes is unknown because leaching has not been measured in these plots. It is safe to assume that soil leaching, which is thought to have been increased approximately two-fold by acid deposition on WBW (Johnson et al. 1985), has played some role in the observed decline in subsoil exchangeable base cations. This would apply especially to the reductions in subsoil exchangeable Mg^{2+} that cannot be accounted for by net vegetation increment. The changes observed in plot 42 are of particular interest in light of the forecasts made from element budgets in a nearby intensive acid deposition study plot [the chestnut oak site described by Richter et al. (1983) and Johnson et al. (1985)]. These studies indicated that soil exchangeable acidity and base cation reserves were sufficiently large in comparison to flux rates that little change in exchangeable acidity or base cations would be expected in any time frame less than several decades. In part, these forecasts are supported by the data from plot 42 in that little change in exchangeable base cations was noted (Fig. 5). Despite the constancy of exchangeable base cations, however, there were decreases in pH and marked increases in exchangeable Al^{3+}, as noted above.

In the Walker Branch studies cited above, total H^+ loading to the ecosystem was approximately $1.6\,\mathrm{kmol\,ha^{-1}\,yr^{-1}}$ from atmospheric H_2SO_4 and deposition and 0.5–$0.6\,\mathrm{kmol\,ha^{-1}\,yr^{-1}}$ from internal carbonic acid generation, creating a base cation leaching potential of about 2 kmol(+) $ha^{-1} yr^{-1}$ (Johnson, et al. 1985). Since several of the soils on Walker Branch absorb SO_4^{2-}, the actual rate of leaching is on the order of 1.0–2.0 kmol(+) $ha^{-1} yr^{-1}$, which would equal 10–20 kmol(+) ha^{-1} over the 10-year sampling period described here. Net vegetation increment over the 1972–1982 period ranged from 9–24 kmol(+)·ha^{-1}·yr^{-1} (Table 2). Thus, the effects of leaching and vegetation increment on base cation removal from soils were roughly of the same order of magnitude in these forests. Clearly, however, vegetation increment had a greater effect on Ca removal than Mg removal from soils (Table 2) and therefore the subsoil Mg^{2+} declines were less likely to be due to vegetation increment than the subsoil Ca^{2+} decline are. In the studies cited above, Mg^{2+} leaching (0.22–0.44 kmol(+)·$ha^{-1} - 1$) was more important than vegetation Mg^{2+} increment (0.1 kmol·ha^{-1}·yr^{-1}; Johnson, et al. 1975), and we hypothesize that the same would be true for the plots studied here.

The decreases in forest floor nutrient content may have resulted from either a decrease in litterfall nutrient concentration or an increase in forest floor nutrient turnover rate. Given the fact that litter (Oe) concentrations often decreased whereas litter weights increased, the former (decrease in litterfall nutrient concentration) appears more likely than the latter (increased forest floor nutrient turnover). In short, it appears that a general decrease in site fertility is manifested in lower subsoil and litter nutrient concentrations.

Conclusions

The general declines in forest floor and subsoil nutrient content in these mixed deciduous plots on WBW suggest a general decline in the nutrient status of those forest ecosystems. This decline is most marked and consistent with respect of Ca^{2+} and Mg^{2+} in those cherty, upland chestnut oak and oak-hickory forests with the lowest exchangeable Ca^{2+} and Mg^{2+} supplies. The decreases in subsoil exchangeable Ca^{2+} in these cases can be accounted for by the net vegetation Ca^{2+} increment (i.e. uptake minus return). Vegetation increment cannot account for subsoil exchangeable Mg^{2+} decreases, but vegetation uptake from subsoils followed by a return to surface soils may also have contributed to the subsoil exchangeable Mg^{2+} decreases in several plots. Soil leaching, as augmented by acid deposition, must have had some effect on the subsoil exchangeable Ca^{2+} and Mg^{2+} decreases, but its magnitude is unknown in these particular plots. Intensive leaching studies adjacent to one site indicated that total leaching (natural carbonic acid plus acid-deposition-induced sulfate leaching) had about the same magnitude of impact on total base cation export from soils as net vegetation increment did (Johnson 1985). However, net leaching of Ca^{2+} was much less than the net vegetation increment of Ca^{2+}, whereas the reverse was true for Mg^{2+}. Thus, we hypothesize that subsoil exchangeable Mg^{2+} declines are due primarily to leaching, whereas exchangeable Ca^{2+} declines are due primarily to vegetation uptake.

Finally, it should be repeated that Walker Branch is not an undisturbed ecosystem, but a patchwork of former farms and woodland pastures that has been allowed to revert to forest since 1942. Thus, there is no reason to expect that either the vegetation or the soils should be in a steady-state condition, and there is every reason to expect that changes will occur as the new forest develops and ages. It is our intention to maintain these long-term monitoring plots for future resampling and research to ascertain the continuing changes in this forest ecosystem.

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