The Effects of Leaching and Whole-tree Harvesting on Cation Budgets of Several Forests

D. W. JOHNSON,   J. M. KELLY,   W. T. SWANK,   D. W. COLE,   H. VAN MIEGROET,
J. W. HORNBECK,   R. S. PIERCE,   AND   D. VAN LEAR

The Effects of Leaching and Whole-tree Harvesting on Cation Budgets of Several Forests

D. W. Johnson,* J. M. Kelly, W. T. Swank, D. W. Cole, H. Van Miegroet,
J. W. Hornbeck, R. S. Pierce, and D. Van Lear

ABSTRACT

The effects of acid deposition, natural leaching, and harvesting on base cation export from forests in Maine, Tennessee, South Carolina, North Carolina, and Washington were compared. Based on previous literature, we hypothesized that base cation export via whole-tree harvesting (WTH) would be nearly independent of soil exchangeable base cation supplies; base cation export via leaching would be strongly dependent on the exchangeable cation supplies as well as the input, production, and mobility of anions. This hypothesis was supported by the data: mixed deciduous sites in Tennessee had among the highest base cation (principally Ca) exports via WTH (1.2 kmol$_c$ ha^{-1} yr^{-1}), yet the lowest soil exchangeable supplies (35 kmol$_c$ ha^{-1}), whereas the Washington sites had by far the highest base cation leaching (5–41 kmol$_c$ ha^{-1} yr^{-1}, due to very high levels of naturally produced HCO$_3^-$) and soil exchangeable supplies, yet only relatively moderate base cation exports via WTH (0.4 kmol$_c$ ha^{-1} yr^{-1}). Sulfate was either the dominant or a major anion in soil solutions from the eastern sites, suggesting that atmospheric deposition was a major factor in soil leaching from these sites (0.2–2.1 kmol$_c$ ha^{-1} yr^{-1}), but total leaching rates from these sites were much lower than at the Washington sites. Indeed, some of the eastern sites showed a net annual accumulation of one or more base cations from atmospheric deposition. Some of the southeastern sites with Ultisols showed the expected net retention of SO$_4^{2-}$, but the sites in the Tennessee Valley near Oak Ridge showed surprisingly little ecosystem SO$_4^{2-}$ retention, perhaps due to lower soil SO$_4^{2-}$ adsorption in these soils than in other Ultisols.

Early evaluations of the nutritional effects of intensive harvesting considered atmospheric inputs solely from the perspective of possible nutrient amendments (e.g., Boyle et al., 1973; Weetman and Webber, 1972). Conversely, early acid deposition research considered only the potential effects of atmospheric inputs upon cation nutrient loss. Since harvesting and atmospheric deposition occur together, it is essential to consider their combined effects on forest nutrient status, particularly as harvesting intensity increases, in terms of both shorter rotations and greater biomass utilization.

There are some fundamental differences between plant uptake and leaching processes that lead to some in-

teresting contrasts in how these two processes cause cation export from forest ecosystems. Plants selectively take up individual cations according to their nutritional needs, which vary considerably among species. Soil exchangeable supplies may have some effect on but by no means control the uptake rate of a given cation. The leaching of a given cation, on the other hand, is strongly controlled by the amount of that cation on soil exchange sites, the selectivity coefficients governing the exchange of that cation with other cations (Reuss, 1983), and the total cation (or anion) concentration in soil solution. If uptake of a particular cation is sufficient to cause depletion of that cation from soil exchange sites during a rotation (e.g., Alban, 1982), the leaching rate of that cation will decrease and the leaching rate of other cations will increase. When uptake depletes the soil exchange complex of all major nutrient cations, the soil may become acidified to the extent that Al^{3+} becomes the dominant cation in soil solution, and the ecosystem accumulates atmospherically deposited base cations (e.g., Ulrich et al., 1980). Uptake of sufficient quantities of one or more cations can effectively conserve them within the ecosystem by reducing the supply of these cations on exchange sites, thereby reducing leaching. With whole-tree harvesting (WTH), however, supplies of the cations conserved in the biomass are suddenly removed, and ecosystem capitals are significantly reduced.

Some of the interactions between uptake, leaching, and harvesting described above were illustrated by the results of a previous study in which we compared the combined effects of atmospheric deposition, natural leaching, and WTH on the Ca, K, and Mg budgets of mixed oak with those of a nearby loblolly pine forests near Oak Ridge, TN (Johnson and Todd, 1987). Both sites were subject to rather high atmospheric S inputs (1.1–1.3 kmol$_c$ ha^{-1} yr^{-1}), and SO$_4^{2-}$ was the dominant anion in all solutions. Large accumulations of Ca in woody tissues had reduced soil exchangeable Ca^{2+} and HCO$_3^-$ leaching in the mixed oak site relative to the loblolly pine site. Vegetation and soil exchangeable Mg^{2+} levels were similar in the two sites, but because soil exchangeable Ca^{2+} to Mg^{2+} ratios differed (greater in the loblolly pine than in the mixed oak soil), and because both sites were subjected to nearly the same leaching pressure from SO$_4^{2-}$, Mg^{2+} leaching in the mixed oak site was greater than in the loblolly pine site.

In this paper, we report the results of a cooperative study of the combined effects of WTH, acid deposition, and natural leaching processes on the cation budgets of selected forest ecosystems in the southeastern, northeastern, and northwestern USA. Detailed discussions of the results from each of the individual sites are given in other publications (Johnson et al., 1982, 1985a, b; Johnson and Todd, 1987; Bigger and Cole, 1983; Smith et al., 1986; Van Lear et al., 1983; Boring et al., 1981; Kelly, 1984; Swank, 1984). Our objective here is to sum-

marize and compare the results of these individual studies in order to compare the relative importance of acid deposition, natural leaching, and WTH among these sites and to determine if there are any systematic patterns that relate to soil, vegetation type, or atmospheric input. Based on the above discussion and previous literature, we hypothesized that base cation export via WTH would be nearly independent of soil exchangeable base cation supplies, whereas base cation export via leaching would be strongly dependent on the exchangeable cation supplies as well as the input, production, and mobility of anions. We also hypothesized that the potential effects of atmospheric S deposition on soil leaching would be mitigated by soil SO_4^{2-} adsorption in the southeastern sites (e.g., Johnson et al., 1980; Johnson and Todd, 1983), and that sites with oak-hickory vegetation would have large base cation exports associated with WTH (e.g., Marion, 1979; Cole and Rapp, 1981; Johnson et al., 1982).

SITES AND METHODS

This study (funded by the USEPA), combined and added to on going effects of sawlog vs. WTH [funded by the Dep. of Energy (DOE); Mann et al., 1988] and effects of acid deposition on forest nutrient status [funded by the Electric Power Res. Inst. (EPRI); Johnson et al., 1985a]. All three studies were coodinated at Oak Ridge National Laboratory. The institutions, principal investigators, and sites involved in these studies are listed in Table 1.

The Department of Energy's WTH study called for estimates of major nutrient (N, P, K, Ca, and Mg) distribution in vegetation, litter and soils, fluxes via precipitation and soil leaching, and biomass and nutrient export by harvesting. Some sites (Coweeta, NC; Chesuncook, ME; Clemson, SC) also measured nutrient export via streamwater. The specific methods used at each site have been described in detail in previous papers (Johnson et al., 1982, 1985a; Johnson and Todd, 1987; Bigger and Cole, 1983; Smith et al., 1986; Van Lear et al., 1983; Boring et al., 1983; Swank, 1984), and only a few of the more pertinent details will be repeated here. Ceramic cup lysimeters (of varying size) and, in most cases, bulk precipitation and

throughfall collectors had been in place for 2 to 3 yr at each site. [The depth of lysimeter placement varied from site to site from a minimum of 45 to a maximum of 100 cm, depending on soil conditions (e.g., presence of fragipan, bedrock, etc).] The current comparative study required additional analyses (i.e., SO_4^{2-} and HCO_3^-) on routinely collected water samples and, in some cases, the installation of precipitation and throughfall collectors (open-bucket type).

The EPRI-funded acid deposition study required estimates of major ion fluxes in five undisturbed forests. Precipitation, throughfall (open-bucket), stemflow, and soil solutions were collected monthly and analyzed for Ca^{2+}, K^+, Mg^{2+}, Na^+, NH_4^+, pH, HCO_3^-, SO_4^{2-}, NO_3^-, Cl^-, and orthophosphate. Ceramic plate lysimeters set to a constant tension of 10 kPa were used in these studies. Detailed descriptions of methods are reported in previous papers (Johnson et al., 1985b; Kelly, 1984). At three of these sites, the USEPA project required an analysis of N, P, K, Ca, and Mg removal should the sites be subjected to whole-tree harvesting. The data necessary to make these calculations (assuming complete removal of trees) was available from other studies at these sites (Johnson et al., 1985b; Kelly, 1984).

To ensure comparability of analytical results, 6 to 8 subsamples of plant tissue and water from all sites were cross-checked by analysis at ORNL. These cross-checks revealed no major differences (i.e., regressions of the two analyses had slopes of 0.87–1.08) in the analytical results reported here.

Soil solution fluxes at the various sites were estimated by multiplying volume-weighted-average concentrations by water flux estimates from ancillary watershed studies (Coweeta, NC; Camp Branch, TN; Walker Branch, TN); or evapotranspiration estimates (Pack Forest, WA; ORNL, TN; Clemson, SC; Maine). For details concerning water budget estimates, see Johnson et al. (1985b), Johnson and Todd (1987), Bigger and Cole (1983), Smith et al. (1986), Van Lear et al. (1983), Boring et al. (1981), Kelly (1984), and Swank (1984).

To make a combined assessment of effects of atmospheric deposition (both positive and negative), natural leaching, and WTH on the base cation status of these ecosystems, we used a modification of the nutrient budget approach first proposed by Boyle et al. (1973) for evaluating the effects of WTH. Boyle's approach involves comparisons of nutrient inputs via precipitation, mineralization, and weathering and outputs in biomass via WTH with soil reserves. The approach also involves making very general and suitably cautious observations as to the potential for depletion of various nutrients and taking into ac-

Table 1. Sites, principal investigators, forest types, ages, biomass, and soil classifications for the study sites. Ages in parenthesis are assumed rotation ages.

Research group (Task leader)	Site location	Forest type	Age	Biomass, Mg ha⁻¹	Soil classification
Oak Ridge Natl. Lab. (D.W. Johnson)	Oak Ridge Natl. Environ. Res. Park (NERP), TN	Mixed deciduous (primarily *Quercus* and *Carya*)	30–150† (50)	134	Paleudults
		Loblolly pine (*Pinus taeda*)	30	145	Paleudults
Oak Ridge Natl. Lab. (D.W. Johnson)	Walker Branch Watershed, TN	Chestnut oak (*Quercus prinus*)	30–150‡ (50)	234	Paleudults
		Yellow poplar (*Liriodendron tulipifera*)	40‡	128	Fragiudults
Southeastern Forest Exp. Stn. (W.T. Swank)	Coweeta Hydrologic Lab., NC	Mixed deciduous (primarily *Quercus* and *Carya*)	Mixed; Avg = 80 (80)	183	Hapludults
Northeastern Forest Exp. Stn. (R.S. Pierce & J.M. Hornbeck)	Chesuncook Lake, ME	Spruce-fir (*Picea rubens–Abies balsamea*)	60	230	Haplorothods Haplaquepts
Clemson Univ. (D. Van Lear)	Clemson, SC	Loblolly pine	41	134	Hapludults
Univ. of Washington (D.W. Cole)	Charles Pack Forest, WA	Douglas fir (*Pesudotsuga menziesii*)	50	318	Dystrochrepts
		Red alder (*Alnus rubra*)	50	147	Dystrochrepts
Tennessee Valley Authority (J.M. Kelly)	Camp Branch, TN	Mixed deciduous (primarily *Quercus*)	Mixed; Avg = 90 (90)	118	Hapludults

† Woodland pasture abandoned and allowed to reforest in 1942.   ‡ Cleared land abandoned and allowed to reforest in 1942.

count the sometimes major uncertainties in these estimates. The most significant uncertainties are the weathering and mineralization estimates and the potential errors in extrapolating precipitation and soil leaching data from a short sampling period (usually 3–5 yr) to a full rotation (30–150 yr). In our case, we had the additional uncertainty as to the actual rotation age to be used in the uneven-aged mixed deciduous stands in the southeastern USA. We modified Boyle's approach by adding leaching data, including an analysis of the potential role of atmospheric deposition in causing leaching, and omitting estimates of mineralization and weathering, for which we had little data. We opted to present estimates of leaching and WTH export on an annualized as well as on a per-rotation basis to better facilitate intersite comparisons by normalizing for the rotation age variable. Finally, we used the leaching fluxes in the control sites as estimates of base cation fluxes over a rotation. Most sites showed increased leaching following harvesting (Mann et al.,

1988), except for one site (Red alder—Pack Forest, WA) showed decreased leaching following harvesting (Bigger and Cole, 1983). These changes were temporary, however (Mann et al., 1988), and thus, the control plot fluxes are taken to be the best (though probably slightly low) estimates of leaching rates for the full rotation.

RESULTS AND DISCUSSION

Exports of sum of base cations (SBC), Ca, K, and Mg via solution and WTH on a per-rotation basis along with soil exchangeable and total contents (where available) of these cations are given in Table 2. Annual fluxes via solution (bulk precipitation, throughfall, and soil solution) combined with average annual exports of these cations

Table 2. Fluxes per rotation, soil exchangeable and total contents of sum of base cations (SBC), Ca, K, and Mg on a per-rotation basis in the various study sites. Numbers in parentheses indicate net ecosystem balances.

Site	Vegetation	Fluxes per rotation			Soil capital†	
		Bulk precip.	Leaching	WTH	Exchange-able	Total
				kmol$_c$ ha^{-1}		
			Sum of base cations			
Clemson, SC	Loblolly pine	19	23 (−4)	12 (−16)	71	ND‡
Oak Ridge, TN	Loblolly pine	15	48 (−33)	13 (−46)	83	1370
Oak Ridge, TN	Mixed oak	34	84 (−50)	61 (−111)	35	1450
Walker Branch, TN	Chestnut oak	22	83 (−61)	82 (−143)	84	3000
Walker Branch, TN	Yellow poplar	17	106 (−89)	72 (−161)	113	1700
Camp Branch, TN	Mixed oak	106	17 (+89)	33 (+56)	95	ND
Coweeta, NC	Mixed oak	54	47 (+7)	66 (−59)	68	5540
Pack Forest, WA	Red alder	50	2076 (−2026)	20 (−2046)	415	ND
Pack Forest, WA	Douglas fir	33	256 (−223)	28 (−251)	154	ND
Chesuncook, ME	Spruce-fir	5	107 (−102)	30 (−132)	85	3800
			Calcium			
Clemson, SC	Loblolly pine	6	8 (−2)	7 (−9)	46	756
Oak Ridge, TN	Loblolly pine	10	33 (−23)	8 (−31)	64	255
Oak Ridge, TN	Mixed oak	19	36 (−17)	55 (−72)	18	245
Walker Branch, TN	Chestnut oak	14	12 (+2)	70 (−68)	50	200
Walker Branch, TN	Yellow poplar	11	56 (−45)	57 (−102)	80	200
Camp Branch, TN	Mixed oak	44	3 (+41)	27 (+14)	40	ND
Coweeta, NC	Mixed oak	18	8 (+10)	57 (−47)	47	125
Pack Forest, WA	Red alder	11	1464 (−1453)	15 (−1468)	103	2360
Pack Forest, WA	Douglas fir	7	149 (−142)	12 (−154)	48	3360
Chesuncook, ME	Spruce-fir	2	59 (−57)	21 (−78)	67	540
			Potassium			
Clemson, SC	Loblolly pine	3	2 (+1)	2 (−1)	4	392
Oak Ridge, TN	Loblolly pine	1	4 (−3)	3 (−6)	6	596
Oak Ridge, TN	Mixed oak	3	8 (−5)	3 (−8)	5	593
Walker Branch, TN	Chestnut oak	2	3 (−1)	6 (−7)	10	500
Walker Branch, TN	Yellow poplar	2	4 (−2)	6 (−8)	9	500
Camp Branch, TN	Mixed oak	6	<0.1 (+6)	3 (+3)	15	ND
Coweeta, NC	Mixed oak	6	6 (0)	6 (−6)	13	3170
Pack forest, WA	Red alder	8	35 (−27)	3 (−30)	6	1230
Pack Forest, WA	Douglas fir	5	18 (−13)	10 (−23)	7	700
Chesuncook, ME	Spruce-fir	0.6	3 (−24)	5 (−7.4)	3	260
			Magnesium			
Clemson, SC	Loblolly pine	4	6 (−2)	3 (−5)	21	ND
Oak Ridge, TN	Loblolly pine	2	10 (−8)	2 (−10)	13	520
Oak Ridge, TN	Mixed oak	4	31 (−27)	3 (−30)	12	610
Walker Branch, TN	Chestnut oak	3	11 (−8)	6 (−14)	27	1040
Walker Branch, TN	Yellow poplar	3	18 (−15)	9 (−24)	17	560
Camp Branch, TN	Mixed oak	10	6 (+4)	3 (+1)	40	ND
Coweeta, NC	Mixed oak	8	21 (−13)	3 (−16)	8	2240
Pack Forest, WA	Red alder	6	487 (−481)	2 (−483)	306	1160
Pack Forest, WA	Douglas fir	4	52 (−48)	6 (−54)	99	1780
Chesuncook, ME	Spruce-fir	1	32 (−31)	4 (−35)	15	3000

† Soil depths included are: Clemson, 50 cm; Oak Ridge loblolly, 45 cm; Oak Ridge mixed oak, 45 cm; Walker Branch chestnut oak, 80 cm; Walker Branch yellow poplar, 50 cm; Camp Branch, 100 cm; Coweeta, 100 cm; Park Forest (both sites), 50 cm; Chesuncook, 60 cm.
‡ ND = not determined.

via WTH are depicted in Fig. 1. Solution fluxes of total anion, SO_4^{2-}, HCO_3^-, and NO_3^- are depicted in Fig. 2. The variations associated with these estimates are unknown, as is usually the case in studies of this kind, because we have no way of estimating the errors associated with water flux or tree biomass estimates. Uncertainties with respect to rotation age in the mixed deciduous sites are also significant, and differences in assumed rotation age (from 50 to 90 yr; Table 2) have a major effect on calculated variations in annualized cation exports by WTH. To give some perspective to the latter, we have plotted "error bars" next to the annualized cation exports by WTH for the sites in question indicating the range values that could be obtained from rotation ages of 50 to 90 yr. We are also able to present standard deviations of weighted average annual solution concentrations in most instances, and these are given in Fig. 3 and 4.

When expressed on a per-rotation basis, the expected pattern of greater SBC removal in southeastern mixed deciduous forests than in the coniferous forests did materialize to some extent: the removals of Ca and SBC by WTH were greatest in the Oak Ridge, Walker Branch, and Coweeta sites and least in the Clemson and Oak Ridge loblolly pine forests. When expressed on an annualized basis, however, no particularly striking pattern of SBC[1] or Ca export with respect to forest type emerges, especially given the uncertainty associated with rotation

ages in the southeastern mixed deciduous sites. The reasons for this are clearly associated with the generally greater rotation ages in the southeastern deciduous sites than in the other sites (Table 1). The Camp Branch mixed oak site had a rather low estimated export via WTH due to both a relatively long rotation age and a low biomass estimate (Table 1). The Pack Forest sites (both red alder and Douglas fir) had higher annual K exports via WTH than the other sites, even when accounting for the uncertainties of rotation age in the southeastern deciduous sites, but this is not reflected in the SBC patterns, since K is a relatively minor component of SBC.

As expected, the magnitudes of SBC, Ca, K, and Mg exports via WTH at these sites were unrelated to the exchangeable supplies of these cations in the soil. For instance, the Pack Forest red alder site had the largest pools of exchangeable base cations, Ca^{2+}, and especially Mg^{2+}, yet exports of these cations via WTH at this site were relatively low. In contrast, the Oak Ridge Mixed Oak site had the lowest exchangeable Ca^{2+} and base cation pools of any site, yet Ca and base cation exports via WTH at this site were relatively high (Table 2).

There was tremendous variability in leaching rates: total cation and anion leaching ranged from highs of over 30 $kmol_c$ ha^{-1} yr^{-1} at the Pack Forest red alder site to a lows of less than 0.2 $kmol_c$ ha^{-1} yr^{-1} at the Camp Branch site (Fig. 1 and 2). These variations were due almost entirely to variations in soil solution concentrations, which were in turn due to differences in SO_4^{2-}, NO_3^-, and HCO_3^-

[1] Sodium is not considered in this analysis because it is not an essential nutrient, and its concentration in the tissues of temperate forest trees is normally very low ($<5\%$) relative to concentrations of Ca, K, and Mg. Sodium analysis in the Walker Branch trees confirms that Na can be neglected in that case.
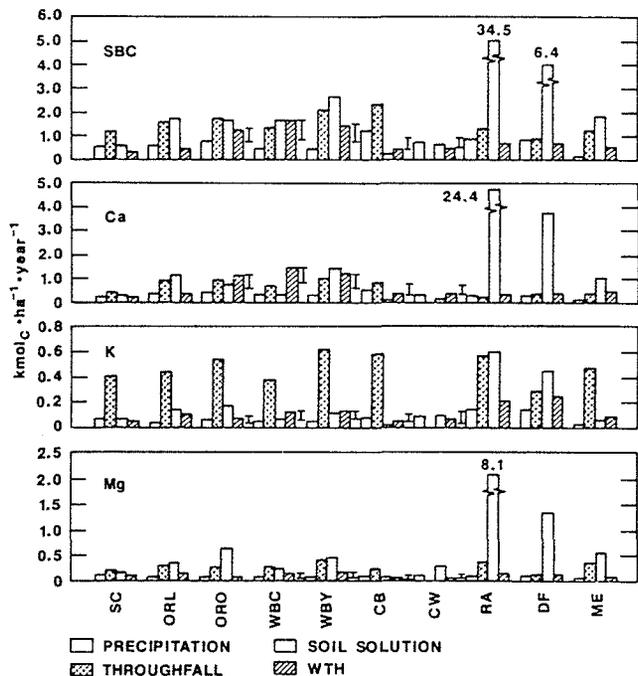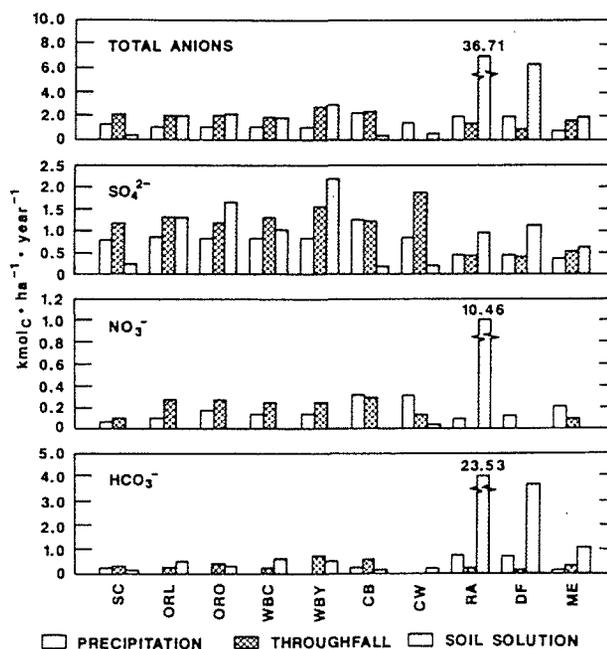

Fig. 1. Fluxes of the sum of base cations (SBC), Ca, K, and Mg by bulk precipitation, throughfall, soil solution, and whole-tree harvesting on an annualized basis in the various study sites. *SC* = Clemson, SC; *ORL* = loblolly pine, Oak Ridge, TN; *ORO* = mixed deciduous, Oak Ridge, TN; *WBC* = chestnut oak, Walker Branch Watershed, TN; *WBY* = yellow polar, Walker Branch Watershed, TN; *CW* = mixed deciduous, Coweeta, NC, *CB* = mixed deciduous, Camp Branch, TN; *DF* = Douglas fir, Pack Forest, WA; *RA* = red alder, Pack Forest, WA; *ME* = spruce-fir, Chesuncook, ME. Error bars represent uncertainties due to rotation age in uneven-aged stands.

Fig. 2. Fluxes of total anions, SO_4^{2-}, NO_3^-, and HCO_3^- in bulk precipitation, throughfall, and soil solution in the various study sites. See Fig. 1 for abbreviations.

production and mobility and the exchangeable base cation levels in the soil. There was no relationship between precipitation H⁺ input and soil cation leaching whatsoever (Fig. 1–4). The effects of differing exchangeable Ca^{2+} contents on the leaching of Ca^{2+} and Mg^{2+} at the Oak Ridge loblolly pine and mixed oak sites discussed previously (see the introduction of this article and Johnson and Todd, 1987) are evident from the data in Table 2 and Fig. 1 and 3. The extremely high Ca^{2+}, Mg^{2+}, and HCO_3^- concentrations and fluxes in the soils at the Pack Forest red alder and Douglas fir sites clearly reflect the large exchangeable base cation capitals of these soils (Table 2). Base saturations ranged from 42 to 62% in the Douglas fir soil and from 62 to 88% in the red alder soil (Biggar and Cole, 1983). Drever and Hurcomb (1986) suggested that calcite weathering, in spite of the low abundance of this mineral in the substrate of various drainage basins in the Washington Cascades, acted as the principal source of solutes, particularly Ca^{2+}. Soils at the Pack Forest sites consist of glacial till derived from various parent material sources within the region, and may well contain calcareous material. The red alder site also has high soil solution NO_3^- concentrations and fluxes, as noted by Van Miegroet and Cole (1984) for another red alder site in Washington State. All other sites had very low soil solution NO_3^- concentrations and clearly showed a net annual accumulation of NO_3^- from atmospheric deposition (i.e., soil solution flux was much less than either bulk deposition or throughfall flux; Fig. 2). Bicarbonate was clearly the dominant anion in soil solutions from the red alder site; however, and soil solution HCO_3^- concentrations at this site greatly exceeded HCO_3^- and total anion concentrations of any site other site (Fig. 2).

Sulfate was the dominant anion (in $\mu mol_c\ L^{-1}$) in soil solutions from all but the Pack Forest and Maine sites, where HCO_3^- was dominant (Fig. 3). Soil solution SO_4^{2-}
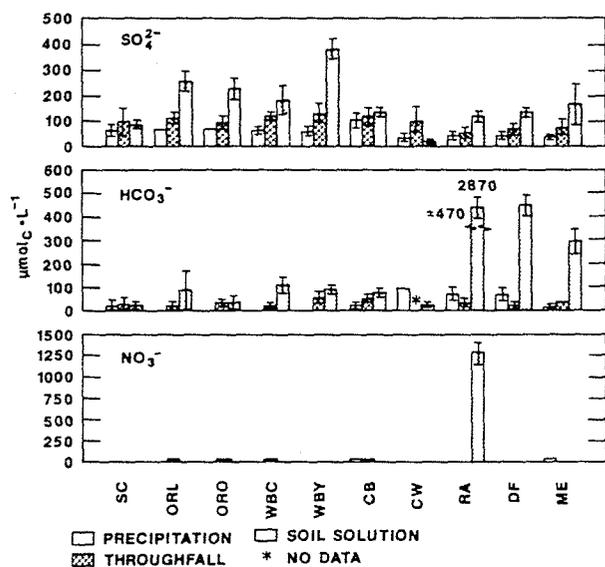
concentrations were highest in the Oak Ridge and Walker Branch sites and lowest in the Coweeta site (Fig. 3). Both ecosystem budgets and solution concentration profiles indicate that only the Clemson, Coweeta, and Camp Branch Sites were accumulating atmospherically deposited S. It is well known that bulk deposition underestimates total S deposition, however (Lindberg et al., 1986). Detailed measurements of wet and dry S deposition have shown that the chestnut oak and yellow poplar sites on Walker Branch receive total S inputs of approximately 1.6 $kmol_c\ ha^{-1}\ yr^{-1}$, or twice the flux estimated from bulk deposition (Lovett et al., 1986; Lindberg et al., 1986). Therefore the chestnut oak site was accumulating S at the rate of 0.6 $kmol_c\ ha^{-1}\ yr^{-1}$, or approximately 40% of total input. The yellow-poplar stand on Walker Branch does not accumulate SO_4^{2-} because of the low SO_4^{2-} adsorption capacity of that soil (Richter et al., 1983; Johnson et al., 1985b). Even with this higher estimate of total S deposition, the Oak Ridge mixed oak and loblolly pine sites (which are on the same soil series and within 5 km of the Walker Branch chestnut oak site) would not show any net S accumulation. We suspect that the shallower placement of lysimeters in the latter sites (45 cm vs. 80 cm in the Walker Branch chestnut oak site) may be a factor in these differences, since subsoils typically adsorb more SO_4^{2-} than surface soils. In any event, it is clear that the Ultisols from the Camp Branch, Clemson, and Coweeta sites retain more SO_4^{2-} than those from the Oak Ridge and Walker Branch sites.

The relative importance of leaching vs. WTH export varied considerably among sites and among individual cations. Leaching clearly exceeded WTH export of SBC at the Pack Forest (red alder and Douglas fir), Maine,
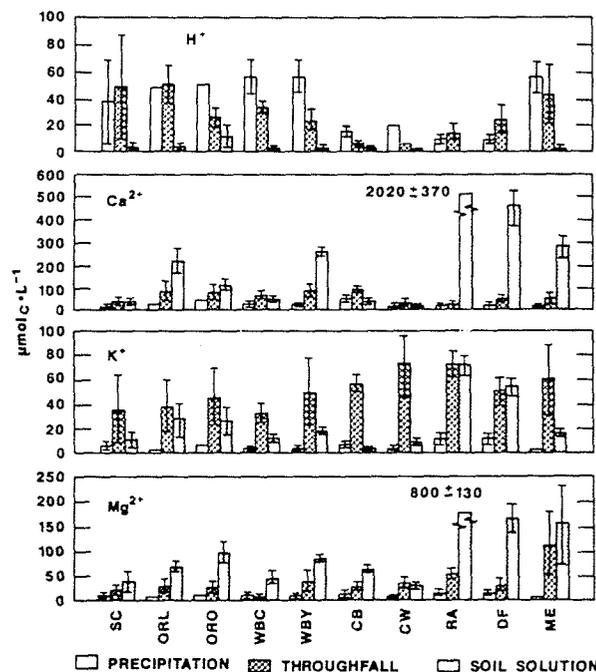
Fig. 3. Concentrations of SO_4^{2-}, NO_3^-, and HCO_3^- in bulk precipitation, throughfall, and soil solution in the various study sites. Standard deviations are given where available. See Fig. 1 for abbreviations.

Fig. 4. Concentrations of H⁺, Ca^{2+}, K⁺, and Mg^{2+} in bulk precipitation, throughfall, and soil solution in the various study sites. Standard deviations are given where available. See Fig. 1 for abbreviations.

and Oak Ridge loblolly sites (Fig. 1 and Table 2). Each of these sites showed a net ecosystem export of SBC even without WTH. Leaching appeared to exceed WTH exports of SBC at the Walker Branch yellow poplar, Oak Ridge mixed oak, and Clemson sites, but not to the extent that the differences in leaching and WTH flux exceeded the uncertainties associated with these estimates. Only at the Camp Branch site did SBC export via WTH significantly exceed SBC export via leaching. The Camp Branch, Coweeta, and Clemson sites showed either net annual gains or virtually no net losses of SBC without WTH. All sites except Camp Branch showed net SBC losses when subjected to WTH (Fig. 1). The higher level of base cation input at Camp Branch was associated primarily with elevated Ca^{2+} and Na^+ concentrations. The Ca^{2+} concentrations, for example, were 38% higher than those at Oak Ridge and Walker Branch sites and 75% higher than values from the other sites (see Fig. 4 for Ca data). This difference is thought to be due primarily to local dust entering bulk precipitation collectors.

Somewhat different patterns emerge when base cations are considered individually. As was the case with SBC, leaching clearly exceeded WTH export of Ca at the red alder, Douglas fir, Oak Ridge loblolly pine, and Maine spruce-fir sites (Fig. 1 and Table 2). However, WTH export of Ca clearly exceeded leaching at the Oak Ridge chestnut oak, Camp Branch, and Coweeta sites (Fig. 1). Without harvesting, the Coweeta and Camp Branch sites showed net Ca gains from atmospheric deposition (bulk deposition flux exceeded soil solution flux), the Walker Branch Chestnut oak, and Clemson sites were either in balance or gaining Ca (bulk deposition flux approximately equaled and throughfall flux exceeded soil solution flux), and only the red alder, Douglas fir, and Maine spruce-fir sites clearly showed net losses (Fig. 1 and Table 2). When WTH is added, all sites except Camp Branch showed net Ca losses. Leaching of K^+ exceeded WTH export of K^+ at the red alder site and possibly also at the Douglas fir and Oak Ridge mixed oak sites. The WTH export of K exceeded leaching of K^+ at the Walker Branch chestnut oak and Camp Branch sites, whereas WTH and leaching were within 50% of one another at the other sites. Only Camp Branch showed a net gain of K without WTH, but the Clemson, Walker Branch Chestnut oak and Coweeta sites were at least in balance for K (Fig. 1). With the exception of the Camp Branch site, all sites show a greater Mg loss from leaching than from WTH. This was especially true at the Pack Forest and Maine sites (Fig. 1). No site showed a clear net gain of Mg without WTH: bulk precipitation inputs were clearly exceeded by soil solution Mg^{2+} fluxes in all but the Camp Branch site, where the two were approximately equal (Fig. 1).

The extent to which the leaching of individual cations can be accounted for by SO_4^{2-}, NO_3^-, and HCO_3^- could be evaluated from a modeling approach (e.g., Reuss and Johnson, 1986), but such an analysis is beyond the scope of this paper. We note, however, that selectivity relationships dictate that the change of Ca^{2+} and Mg^{2+} (M^{2+}) activity is proportional to the square of the change in K^+ activity (Reuss, 1983). Thus, for a doubling of the M^{2+}

leaching rate, the K^+ leaching rate would increase by a factor of 1.41 ($\sqrt{2}$).

The relative magnitudes of exchangeable cation pools and estimated leaching rates suggest that soil exchangeable pools of one or more cations could be turned over (i.e., leaching approximately equaled or exceeded soil exchangeable capital) at least once during a rotation due to leaching alone in many cases. It also appears as if tree uptake and accumulation alone (as manifested by WTH export) could turn over soil exchangeable SBC and Ca at the Oak Ridge mixed oak, Walker Branch chestnut oak and yellow poplar, and Coweeta sites; and K at the Clemson and Pack Forest Douglas fir sites (Table 2).

The changes, if any, in soil exchangeable capital due to leaching and tree uptake will depend to a great extent on the weathering rates in these soils. We have no estimates of weathering rates, but we can compare the exports via leaching and WTH with soil total cation capital in many cases to get a general picture of the potential for weathering to replenish exchange sites. Exports of K equaled very small (<1-5%) proportions of total soil K in all sites for which data are available. Exports of Mg equaled similarly small proportions of total soil Mg in all but the red alder site (where leaching equaled over 40% of total soil Mg capital over a rotation; Table 2). Exports of Ca equaled over 30% of total soil Ca in the Oak Ridge mixed oak, Walker Branch chestnut oak and yellow poplar, Coweeta, and Pack Forest Red alder sites, however (Table 2). The WTH was the major factor in the export of Ca in the three oak sites (Oak Ridge, Walker Branch, and Coweeta), WTH and leaching were of approximately equal importance in the Walker Branch yellow poplar site, and leaching was the dominant factor in the red alder site.

SUMMARY AND CONCLUSIONS

Overall, the results from this comparative study support the hypothesis that cation export via WTH is nearly unrelated to soil exchangeable reserves of cations whereas soil leaching exports are strongly affected by both exchangeable cation reserves and the input, production, and mobility of anions. With regard to the latter, it is interesting to note that by far the highest leaching rates were due to natural HCO_3^- production in the relatively unpolluted Pack Forest red alder and Douglas fir sites, a consequence of high base saturation in these soils. It is also noteworthy that NO_3^- concentrations were quite high in the red alder stand as a result of excessive N fixation, as noted previously by Van Miegroet and Cole (1984) for another site in western Washington. Sulfate was either the dominant or a major anion (i.e., accounted for >30% of total anion concentration) in soil solutions from the eastern sites, but total leaching rates from these sites were much lower than at the Pack Forest sites. Some of the southeastern sites with Ultisols showed the expected net retention of SO_4^{2-} retention, presumably by soil SO_4^{2-} adsorption. The sites in the Tennessee Valley near Oak Ridge showed surprisingly little ecosystem SO_4^{2-} retention,

however, perhaps due to lower soil SO_4^{2-} adsorption capacity in these soils than in other Ultisols.

It is clear from this analysis that assessments of the nutritional effects of acid deposition must be oriented toward individual base cations (Ca^{2+}, K^+, and Mg^{2+}) and must consider both biological and chemical mechanisms, which act to conserve those cations when in short supply. It is also clear from both this and other studies that WTH will tend to have a greater effect on the export of limiting or potentially limiting cations than leaching will, whether the latter is increased by acid deposition or not. This is due to the differences in the nature of cation selectivity during plant uptake, where substitutions among individual cations cannot be made to meet plant nutritional needs, vs. cation selectivity during cation exchange and leaching from soils, which are dictated by purely chemical selectivity coefficients. A modeling approach (e.g., Reuss, 1983) to analyze the potential depletion of individual cations by leaching vs. WTH under various scenarios of atmospheric input, natural acid production, and plant uptake would be both useful and interesting.

ACKNOWLEDGMENTS

REFERENCES

Alban, D.H. 1982. Effects of nutrient accumulation by aspen, spruce, and pine on soil properties. Soil Sci. Soc. Am. J. 6:853–861.

Biggar, C.M., and D.W. Cole. 1983. Effects of harvesting intensity on nutrient losses and future productivity in high and low productivity red alder and Douglas-fir stands. p. 167–178. In R. Ballard and S.P. Gessel (ed.) I.U.F.R.O. Symp. on Forest Site and Continuous Productivity. USDA For. Serv. PNW-163. USDA-FS, Portland, OR.

Boring, L.R., C.D. Monk, and W.T. Swank. 1981. Early regeneration of a clear-cut southern appalachian forest. Ecology 62(5):1244–1253.

Boyle, J.R., J.J. Phillips, and A.R. Ek. 1973. "Whole-tree" harvesting: Nutrient budget evaluation. J. For. 71:760–762.

Cole, D.W., and M. Rapp. 1981. Elemental cycling in forest ecosystems. p. 341–409. In D.E. Reichle (ed.) Dynamic properties of forests ecosystems. Cambridge Univ. Press, London.

Drever, J.I., and D.R. Holcomb. 1986. Neutralization of atmospheric acidity by chemical weathering in an alpine drainage basin in the North Cascade Mountains. Geology 14:221–224.

Johnson, D.W., J.W. Hornbeck, J.M. Kelly, W.T. Swank, and D.E. Todd. 1980. Regional patterns of soil sulfate accumulation: Relevance to ecosystem sulfur budgets. p. 507–520. In D.S. Shriner et al. (ed.) Atmospheric sulfur deposition: Environmental impact and health effects. Ann Arbor Science, Ann Arbor, MI.

Johnson, D.W., J.M. Kelly, W.T. Swank, D.W. Cole, J.W. Horn-

beck, R.S. Pierce, and D. Van Lear. 1985a. A comparative evaluation of three effects of acid deposition, natural acid production and harvesting on cation removal from forests. Final rep. to EPA. ARNL/TM-9706. Oak Ridge National Lab., Oak Ridge, TN.

Johnson, D.W., D.D. Richter, G.M. Lovett, and S.E. Lindberg. 1985b. 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., and D.E. Todd. 1983. Relationships among iron, aluminum, carbon, and sulfate in a variety of forest soils. Soil Sci. Soc. Am. J. 47:792–800.

Johnson, D.W., and D.E. Toss. 1987. Nutrient export by leaching and whole-tree harvesting in a loblolly pine and mixed oak forest. Plant Soil 102:99–109.

Johnson, D.W., D.C. West, D.E. Todd, and L.K. Mann. 1982. Effects of sawlog vs whole-tree harvesting on the nitrogen, phosphorus, potassium, and calcium budgets of an upland mixed oak forest. Soil Sci. Soc. Am. J. 46:1304–1309.

Kelly, J.M. 1984. Power plant influences on bulk precipitation, throughfall, and stemflow nutrient inputs. J. Environ. Qual. 13:405–409.

Lindberg, S.E., G.M. Lovett, D.D. Richter, and D.W. Johnson. 1986. Atmospheric deposition and canopy interactions of major ions in a forest. Science (Washington, D.C.) 231:141–145.

Lovett, G.M., S.E. Lindberg, D.D. Richter, and D.W. Johnson. 1986. The effects of acid deposition on cation leaching from a deciduous forest canopy. Can. J. For. Res. 15:1055–1060.

Mann, L.K., D.W. Johnson, D.C. West, D.W. Cole, J.W. Hornbeck, C.W. Martin, H. Riekerk, C.T. Smith, W.T. Swank, L.M. Tritton, and D.H. Van Lear. 1988. Effects of whole-tree and stem-only clearcutting on postharvest hydrologic losses, nutrient capital, and site productivity. Forest. Sci. (in press).

Marion, G.K. 1979. Biomass and nutrient removal in long-rotation stands. p. 98–110. In A.L. Leaf (ed.) Impact of intensive harvesting on forest nutrient cycling. State Univ. of New York, Syracuse.

Reuss, J.O. 1983. Implication of the calcium-aluminum exchange system for the effect of acid precipitation on soils. J. Environ. Qual. 12:591–595.

Reuss, J.O., and D.W. Johnson. 1986. Acid deposition and the acidification of soils and waters. Ecological studies 59. Springer-Verlag, New York.

Richter, D.D., D.W. Johnson, and D.E. Todd. 1983. Atmospheric sulfur deposition, neutralization, and ion leaching in two deciduous forest ecosystems. J. Environ. Qual. 12:263–270.

Smith, C.T., M.L. McCormack, J.M. Hornbeck, and C.W. Martin. 1986. Nutrient and biomass removals from a red spruce-balsam fir whole-tree harvest. Can. J. For. Res. 16:381–388.

Swank, W.T. 1984. Atmospheric contributions to forest nutrient cycling. Water Resour. Bull. 20(3):313–321.

Ulrich, B., R. Mayer, and P.K. Khanna. 1980. Chemical changes due to acid precipitation in a lossess-derived soil in central Europe. Soil Sic. 130:193–199.

Van Lear, D.H., W.T. Swank, J.E. Douglass, and J.B. Waide. 1983. Forest management practices and the nutrient status of a loblolly pine plantation. p. 252–258. In R. Ballard and S.P. Gessel (ed.) I.U.F.R.O. Symp. of Forest Site and Continuous Productivity. USDA For. Serv. PNW-163. USDA-FS, Portland, OR.

Van Miegroet, H., and D.W. Cole. 1984. The impact of nitrification on soil acidification and cation leaching in red alder (Alnus rubra Bong.) ecosystem. J. Environ. Qual. 13:586–590.

Weetman, G.L., and B. Webber. 1972. The influence of wood harvesting on the nutrient status of two spruce stands. Can. J. For. Res. 2:351–369.