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1987 Annual Group Leader Reports of the Integrated Forest Study

S. E. Lindberg
D. W. Johnson

Environmental Sciences Division
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ENVIRONMENTAL SCIENCES DIVISION

1987 ANNUAL GROUP LEADER REPORTS OF THE INTEGRATED FOREST STUDY

Compiled by Group Discussion Leaders from
Data Presented at the 1987 Annual IFS Review
Pinehurst, NC February 8-12, 1988

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Integrated Forest Study

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CONTENTS

ABSTRACT

The summaries contained in this report describe the trends in atmospheric deposition, canopy interactions, soil solution fluxes, and ecosystem budgets for the major chemical species studied. Preliminary results indicate several interesting trends across sites. The deposition estimates confirm the expected increase in atmospheric deposition at high elevation sites compared to lower elevation forests. Cloudwater interception, enhanced dry deposition due to high wind speeds, and increased rainfall due to this trend. At the other sites dry deposition was important in the input of all ions, particularly of the base cations and particularly at the dryer southeastern sites.

The soil solution fluxes and ecosystem budget data illustrate the difficulties in predicting site status with respect to the degree of sulfate and nitrate leaching. The magnitude of nitrate leaching from each forest is not readily predicted by any single parameter. Similarly, the sulfur input/output budgets do not fit with any expected soil series effects on sulfur retention. However, patterns of aluminum and base cation leaching do appear to match predictions from theoretical soil solution considerations.

INTRODUCTION AND OVERVIEW

The principal objective of the Integrated Forest Study on Effects of Atmospheric Deposition is to determine the effects of atmospheric deposition of sulfur and nitrogen on forest nutrient cycling. The study integrates a field monitoring component, involving quantification of atmospheric deposition and nutrient cycling in a variety of forest sites, and experimental research, including laboratory and field studies to investigate selected atmospheric and soil processes in greater detail. The research is being conducted at forested sites in the northwestern, northeastern, and southeastern United States and in Canada and Norway (Fig. 1). The sites selected for this study represent a range of conditions in climate, air quality, soils, and vegetation, which will facilitate testing hypotheses about the effects of atmospheric sulfur and nitrogen deposition on forest nutrient cycles.

Responsibility for coordination of synthesis activities for the IFS project is shared by nine individuals with expertise in the areas of either atmospheric science or forest ecosystems. For the purpose of comparing data across a range of sites, these individuals have been designated as synthesis group leaders for the major chemical components being studied at each IFS research site: sulfur, nitrogen, acidity, and base cations. Data from the initial year of field research was supplied to these individuals by each IFS site. The summaries contained in this report describe the trends in atmospheric deposition, canopy interactions, soil solution fluxes, and ecosystem budgets for the major chemical species studied.

Preliminary results indicate several interesting trends across sites. The deposition estimates confirm the expected increase in atmospheric deposition at high elevation sites compared to lower elevation forests. However, the magnitude of this effect at the Smoky Mountains site in the Southeast was a surprise. Cloudwater interception, enhanced dry deposition due to high wind speeds, and increased rainfall due to orographic effects all contribute to this trend. At the other sites dry deposition was important in the input of all ions, particularly of the base cations and particularly at the dryer southeastern sites.

The soil solution fluxes and ecosystem budget data illustrate the difficulties in predicting site status with respect to the degree of sulfate and nitrate leaching. The magnitude of nitrate leaching from each forest is not readily predicted by any single parameter. Similarly, the sulfur input/output budgets do not fit with any expected soil series effects on sulfur retention. However, patterns of aluminum and base cation leaching do appear to match predictions from theoretical soil solution considerations. These observations indicate the need to quantify system fluxes for a variety of forests in different environments in order to understand the possible effects of deposition on nutrient cycling processes.

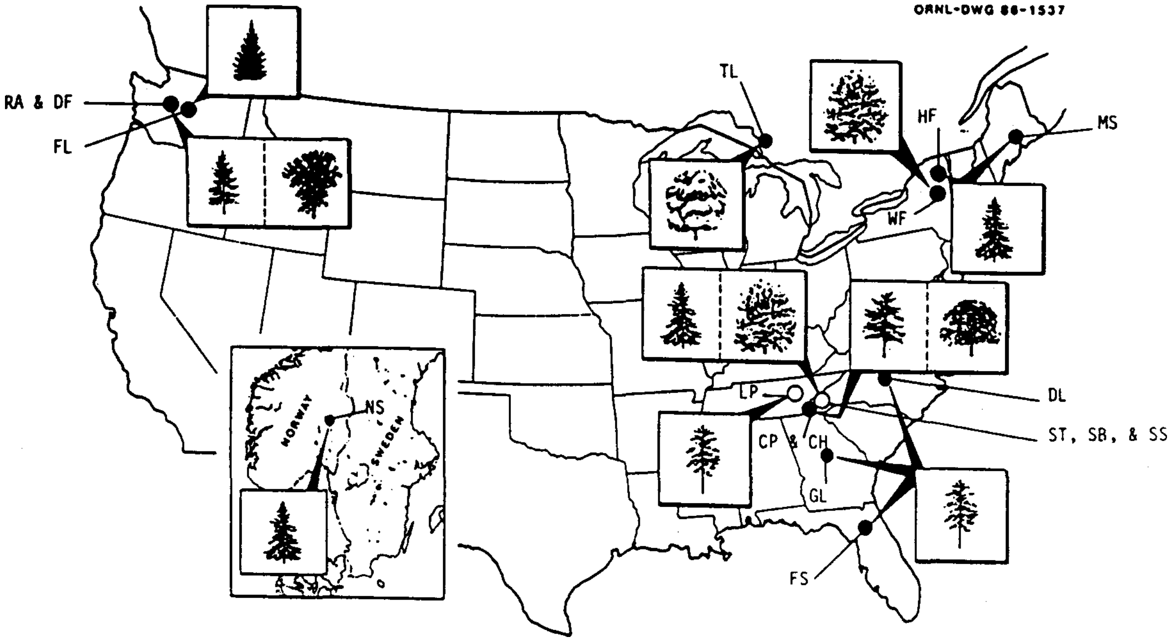

Figure 1. Locations and codes for IFS research sites

ATMOSPHERIC DEPOSITION OF SULFUR TO FORESTS IN THE INTEGRATED FOREST STUDY

Summarized on 2/8/88 by

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Environmental Sciences Division
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Oak Ridge, TN 37831-6038

Atmospheric fluxes of the major forms of airborne sulfur have been determined above and below the forest canopies at eight of the IFS Intensive Deposition Measurement Sites for a one year period: Washington red alder (RA) and Douglas fir (DF), Oak Ridge loblolly pine (LP), Smokies red spruce (ST), Coweeta white pine (CP), Duke loblolly pine (DL), and Huntington Forest mixed hardwood (HF). The sampling years began as early as 4/86 and ended as late as 10/87; hence somewhat different periods were summarized for each site. In addition, data were available from the Turkey Lakes Watershed in Ontario (TL) for 1982 for comparison. Data were also available from the following sites but were not included in this summary: Georgia loblolly and Norway spruce (delayed field startup, chemical analyses incomplete), Whiteface spruce/fir (missing records, some chemical analyses not complete). No data were available for the new Maine spruce and Florida pine sites.

At each site standard IFS deposition sampling protocols were followed for precipitation, throughfall, atmospheric aerosols and vapors, deposited coarse particles, cloud and fog water, and meteorological parameters (Lindberg and Lovett, in preparation). Common protocols were also followed for computation of deposition fluxes. Wet deposition above and below the canopy was determined from the product of seasonal weighted mean concentrations and measured water fluxes. Dry deposition was determined for sampled dry periods using measured air concentrations and modelled dry deposition velocities (Hicks et al., 1987) based on meteorological parameters measured at each site (Lindberg and Lovett, in preparation). Cloudwater flux was determined from weighted mean chemistry data and seasonal estimates of cloudwater deposition amount calculated from hydrologic data. The sites collected from 40 to 100% of the total rainfall (as wet-only events; mean ~80%) and sampled from 25 to 100% of the dry periods (mean ~40%). Detailed uncertainty analyses have been done for the LP and ST sites as discussed below.

Total airborne S at the sites ranged from 1.1 to 9.9 $\mu g/m^3$ and was dominated by SO_2 at most sites. At the more remote sites, SO_2 and particle SO_4^{2-} were comparable. The highest levels occurred in the Southeast, the lowest in the Northwest. These trends were not reflected in the rain concentration data for SO_4^{2-}, with the highest concentrations (~40 μMc/L) at low elevation northeastern and southeastern sites, and the lowest concentrations (~20 μMc/L) at the ST and RA/DF sites.

The estimates of total atmospheric deposition by wet, dry, and cloudwater processes as determined for each site are illustrated in Figure 1. The sites are ranked in order of total deposition and also grouped by region.

Total SO_4^{2-} fluxes ranged over an order of magnitude from ~20 to ~200 mMc m^{-2} y^{-1} (dry deposited SO_2 is assumed to be converted to SO_4^{2-} within the ecosystem for these calculations). The highest flux occurs at the ST mountain site, with fluxes generally decreasing from the Southeast to the Northeast to the Northwest. Approximately half the flux at the ST site is due to cloudwater interception (we estimated a hydrologic flux of ~35 cm cloudwater), with inputs by dry deposition and precipitation approximately equal. Dry deposition was most important at the dryer Southeastern sites (~40-60% of total), while wet deposition dominated at the other sites (70-80% of total), indicating that dry/wet input ratios are site specific. Dry deposition at all sites was dominated by SO_2.

The results for the ST site support the idea that mountain forests are exposed to higher atmospheric loading because of climatologic factors and site characteristics (orographic effects, high winds, cloud immersion, and the gap nature of the high surface area spruce canopy), despite their location distant from major emission sources. Although we expected this trend, the magnitude of the enhancement was a surprise (two to three times higher S fluxes than at the nearby LP and CP sites). The S input data are reflected by the deposition estimates for N, H+, and base cations, as well as the data for soil fluxes of S and N (summarized in the attached reports).

All of these estimates are subject to considerable uncertainty, but particularly those at high elevation sites because of the importance of cloudwater which is difficult to quantify. The magnitude of the uncertainty in S deposition was determined for the ST site based on combined errors due to analytical, sampling, hydrologic, and spatial factors. Using these data and considering physical bounds on fluxes wherever possible (eg. hydrologic fluxes based on independent measurements), it is possible to bound the best estimate of the S flux: minimum - 140, best - 200, maximum - 350 mMc m^{-2} y^{-1}. Given this relatively large range, it is still clear that the S flux to this site exceeds that measured elsewhere.

Comparing the estimates of total deposition to the forest canopies with the measurements of S fluxes below these canopies in stemflow and throughfall (Figure 2) gives an indication of the interactions and fate of atmospheric S in the canopy, and supports the trends described above. The flux of sulfate in throughfall at the ST site exceeds the next highest flux (LP) by a factor of 2.8, and exceeds that at the geographically nearest site (CP) by more than a factor of 6. The throughfall fluxes indicate that the IFS sites fall into two categories of S behavior in the canopy: sites where the the flux below the canopy exceeds that to the canopy (ST, HF, TL, DF, RA), and sites where the opposite is true (LP, DL, CP).

Behavior of the first type indicates a source of S in the canopy (foliar leaching). The second type suggests that the canopy is a sink for deposited S, most likely as dry deposited SO_2. Such behavior is obviously species specific, but the IFS data suggest some generalities which are potentially useful for modelling . All of the "sink" canopies are pine (loblolly and white), while the "source" canopies are hardwoods or spruce/fir. When the leaching flux of sulfate is expressed as a percentage of total throughfall plus stemflow, this term decreases with increasing concentrations of atmospheric S. Thus leaching becomes relatively less important at sites with high S loading where dry deposition wash-off dominates the below canopy flux.

At five of the IFS sites the fluxes are essentially the same above and below the canopy, and at the remaining sites the differences are generally within the range of uncertainty in the deposition estimates. Thus, relative to the total S flux, canopy interactions (uptake and leaching) appear to be small. Figure 3 illustrates the relationship between the estimated total annual wet plus dry deposition of SO_4^{2-} to each IFS site and the measured flux in throughfall plus stemflow (the ninth point in the graph represents data from the previous EPRI study at Walker Branch). The variance in the flux of S in throughfall plus stemflow accounts for 98% of the variance in estimated total S deposition. If this behavior is supported by data from future years at these sites and from the five IFS sites not included in this report, it will result in a much simplified method for estimating the seasonal and annual S loading to forests and forested watersheds.

ACKNOWLEDGEMENTS AND REFERENCES

These data could not have been summarized without the help of the principal investigators at each IFS site. In particular the considerable efforts of the field researchers who collect these data are appreciated: D. Silsbee, J. Owens, D. Schaefer, G. Wolfe, J. Panec, L. Reynolds, J. Shepard, P. Conklin, D. Todd, E. Miller, P. Homann, R. Gonea, and R. Briggs.

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ORNL-DWG 88-2084

TOTAL DEPOSITION OF SO4= AT IFS SITES

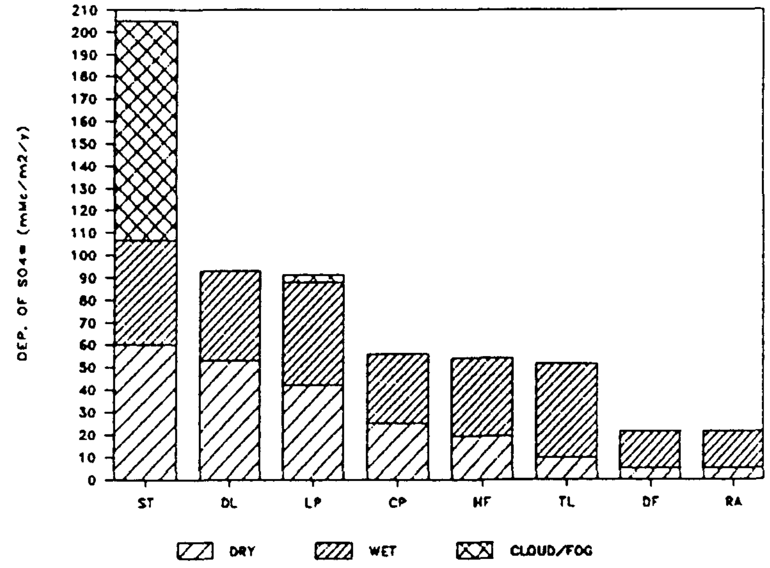

Fig. 1. Total annual atmospheric deposition of sulfur as sulfate (mMc m^{-2} y^{-1}) to IFS sites by wet, dry, and cloud/fog interception processes.

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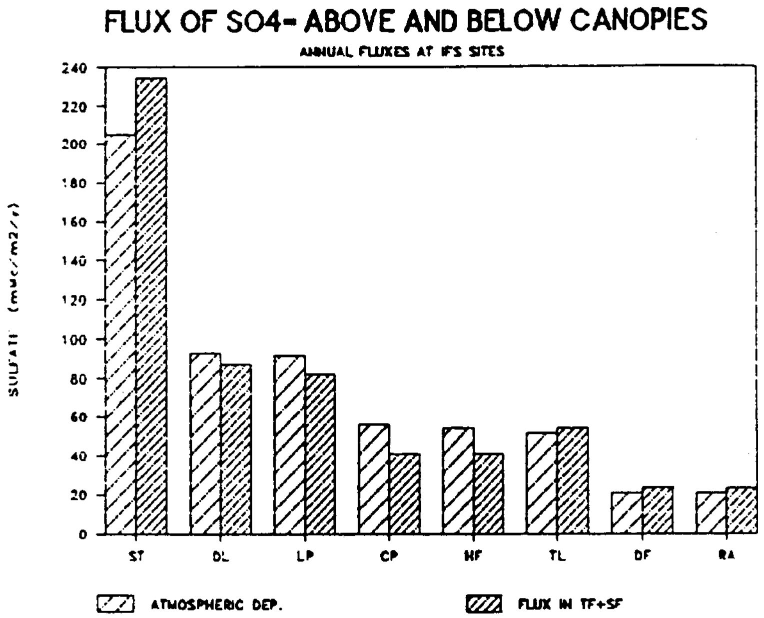

Fig. 2. Annual flux of sulfate (mMc m^{-2} y^{-1}) above and below the forest canopies at IFS sites.

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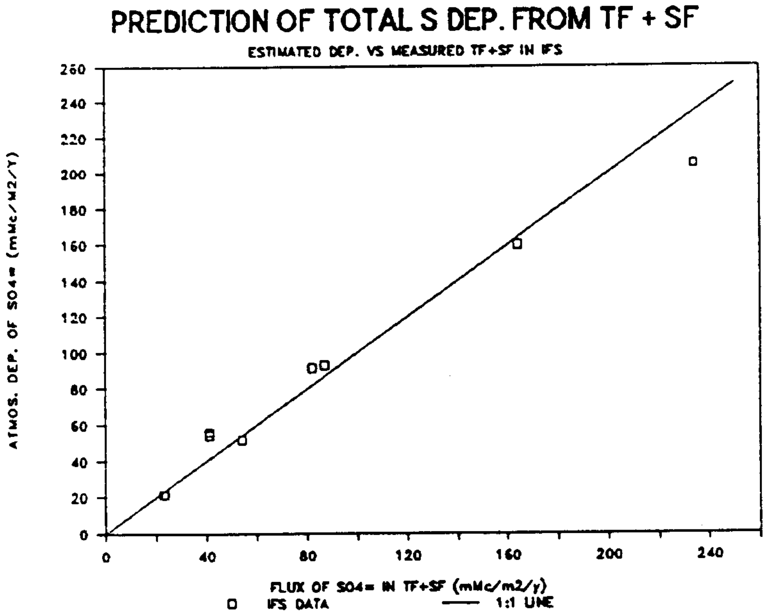

Fig. 3. The relationship between the estimated total annual atmospheric deposition of sulfate to the canopy (mMc m^{-2} y^{-1}) and the measured flux of sulfate in throughfall (TF) plus stemflow (SF) at IFS sites.

INTEGRATED FOREST STUDY

SULFUR SYNTHESIS: CYCLING AND BUDGETS

February 1988

Summarized by:
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Syracuse, NY

The quantification of sulfur budgets of forested ecosystems is an important component of the IFS project since sulfur inputs are major contributors to mineral acidity and the flux of sulfate through the soil effects the dynamics of other elements, such as base cations which are important for forest nutrition (Johnson et al., 1982). A diagram of a generalized sulfur budget for a forest ecosystem is given in Figure 1. Major inputs are through dry and wet deposition. Major sulfur pools include the vegetation and both inorganic and organic sulfur fractions in the soil. Fluxes include sulfate uptake by vegetation, litter and root input of sulfur to the forest floor and soil, microbial immobilization-mineralization, abiotic sulfate adsorption-desorption, and sulfate leaching (David et al., 1987). The information presently available on the sulfur content of IFS sites is shown in Figure 2. As has been established for other forest ecosystems, the soil serves as the major sulfur pool (Johnson, 1984). The sulfur content shows little variation across the IFS sites with the exception of high level of sulfur in the Duke Loblolly Pine Site (DL) which may have been fertilized with super-phosphate when the site was in agricultural use (D. Binkley, personal communication).

The flux of sulfate through the strata (total deposition, throughfall plus stemflow, forest floor, and lower B horizon) of IFS sites is given in Figure 3. Among the sites, there is an order of magnitude difference of input from 2.0 to 0.2 kmoles(+or-)/ha/y. The high input sites are red spruce stands (ST, SS) in the high elevations of the Smokey Mountains with the lowest input sites being found in the Cascades of Washington which have low background levels of atmospheric sulfur.

It has often been hypothesized that sulfate adsorption is in steady state for soils with low sulfate adsorption capacity and sulfate outputs should equal inputs for these ecosystems (Rochelle et al., 1987). This hypothesis was tested by calculating net sulfate retention (total deposition - leaching from B horizon). Surprisingly, a wide range of net sulfate retention-loss values were found (Figure 4). For sites with low sulfur inputs, such as the red alder (RA) and Douglas fir (DF) systems in Washington, retention may be explained by nutrient demand for this element. However, other forest systems with higher sulfate inputs also have net retention which may be due sulfate adsorption and/or microbial immobilization (Figure 5). A number of sites, especially those with higher levels of input, appear to be exhibiting net losses of sulfur (Figure 6). These net losses could be due to underestimates of sulfur inputs, overestimates of sulfate losses, sulfate desorption, and/or net sulfur mineralization. Due to the preliminary stage of these data, it cannot be stated with certainty which of these factors are operative. However, these results clearly show that the hypothesis of steady state with respect to sulfate budgets cannot be validated across the IFS sites since only the Huntington Forest (HF) shows no net sulfur retention or loss (Figure 7).

These findings demonstrate the need for independent estimates of inputs and outputs for sulfur budgets of forested ecosystems such as those being carried out in the IFS project. The assumption that dry deposition can be calculated by the difference between sulfate leaching and wet deposition may be incorrect for a wide range of forested ecosystems. The determination of whether sulfur is being retained or lost in forested ecosystems has important implications not only for estimating nutrient status of these sites, but also for predicting how changing atmospheric inputs of sulfur will affect surface water chemistry (Hornberger et al., 1986). These results also show the importance of having accurate hydrological budgets for these forest ecosystems since the flux of sulfate is tightly coupled to that of water (Mitchell and Fuller, 1988). Increased emphasis on estimating site hydrology is being initiated in the IFS project.

Experimental tasks have begun to ascertain how sulfate adsorption-desorption and the potential for organic sulfur immobilization-mineralization vary among sites. For example, data from Rob Harrison have shown that sites with low sulfate retention generally have a higher capacity for sulfate adsorption. However, this retention in not related to inorganic iron and aluminum fractions which should affect adsorption capacities (Fuller et al., 1985). John Fitzgerald has also shown that among the sites there is a high capacity for organic sulfur immobilization. Further work is in progress to evaluate the sulfur budgets in each site and to relate these findings to more detailed analyses of processes affecting soil sulfur dynamics.

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FIGURE 1

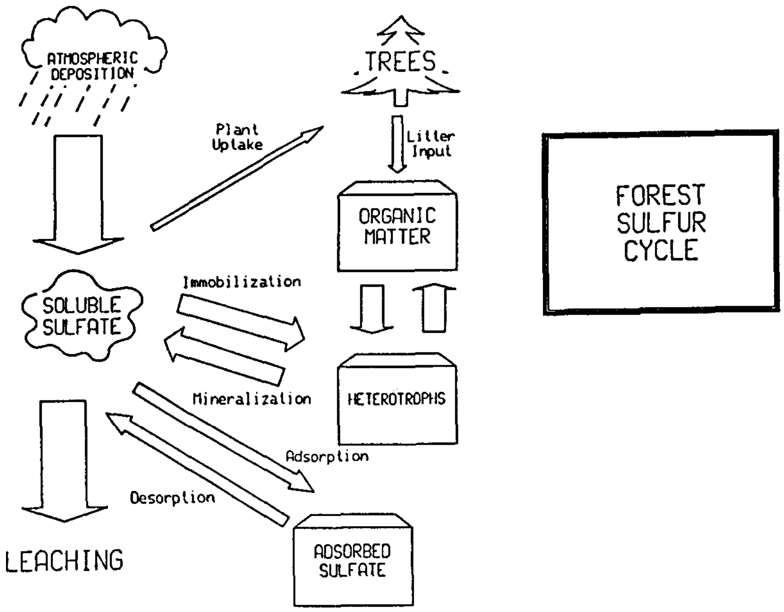


FIGURE 2

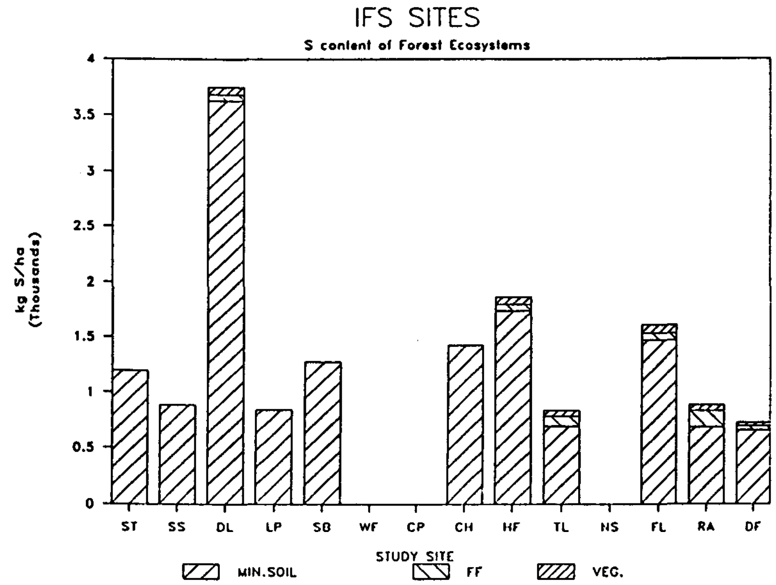


FIGURE 3

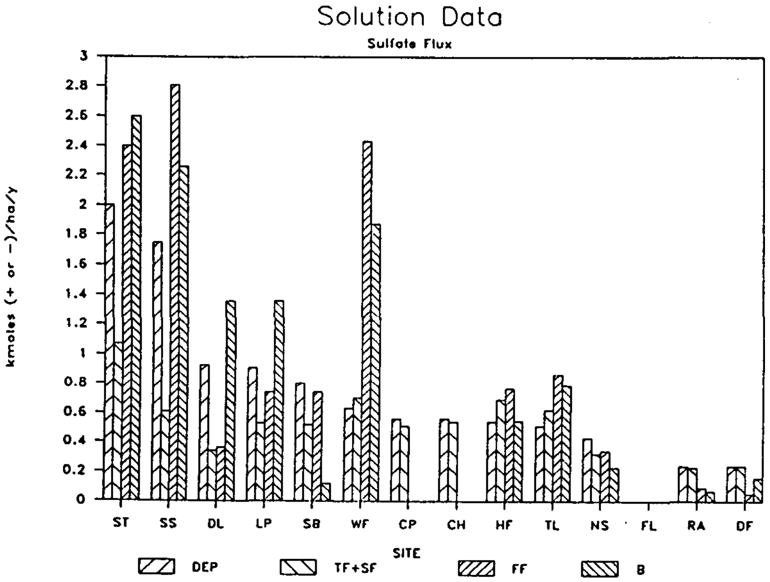


FIGURE 4

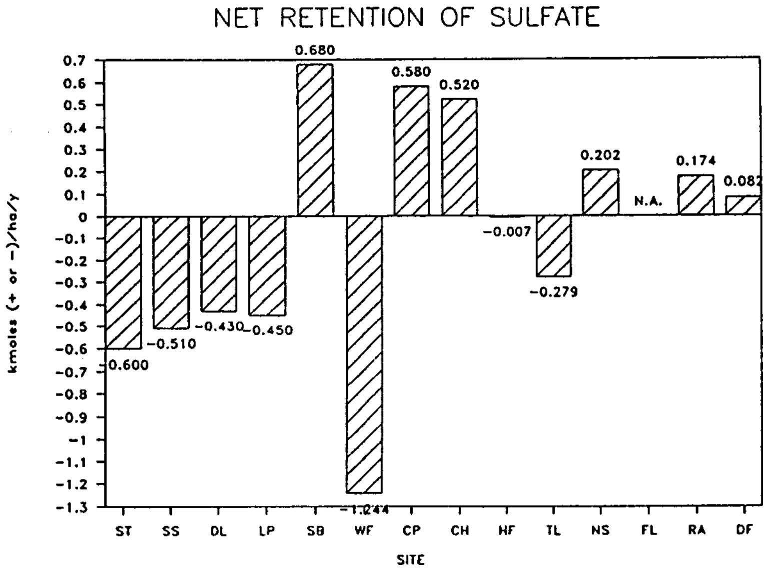
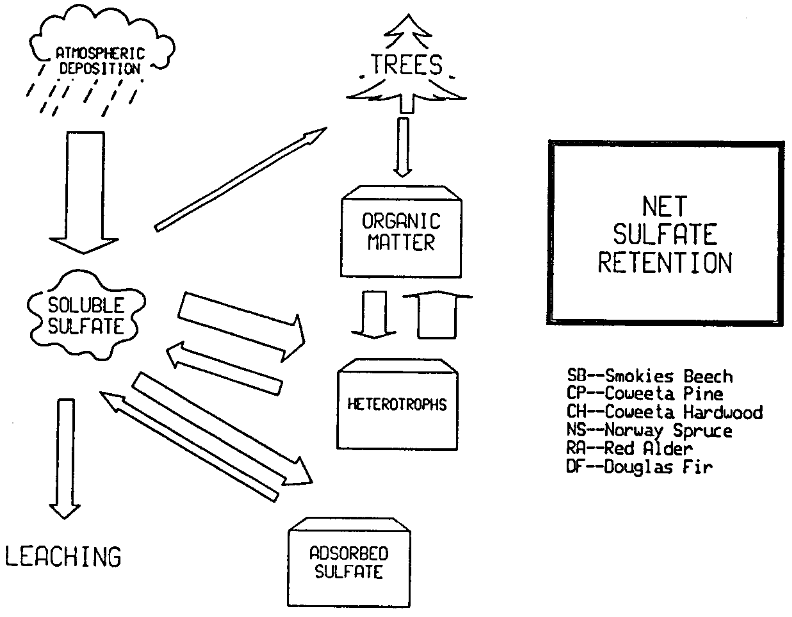


FIGURE 5



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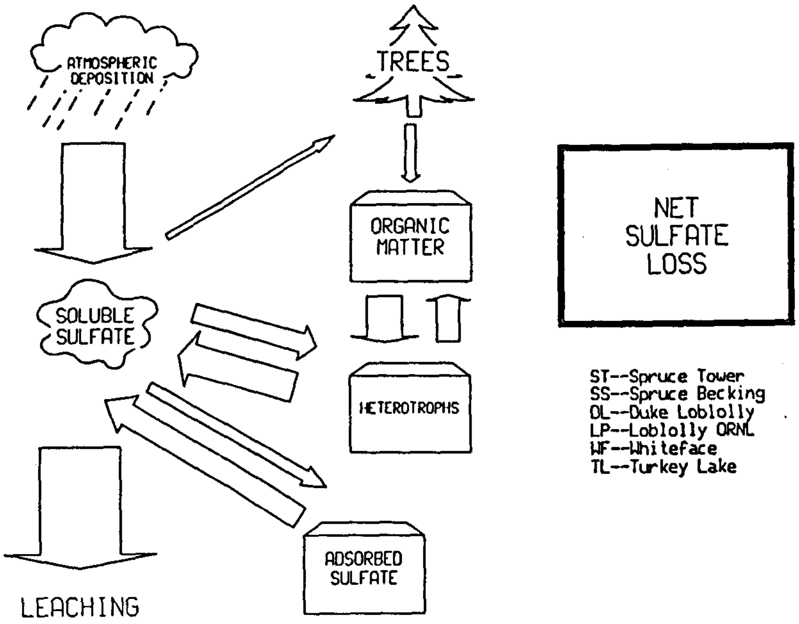
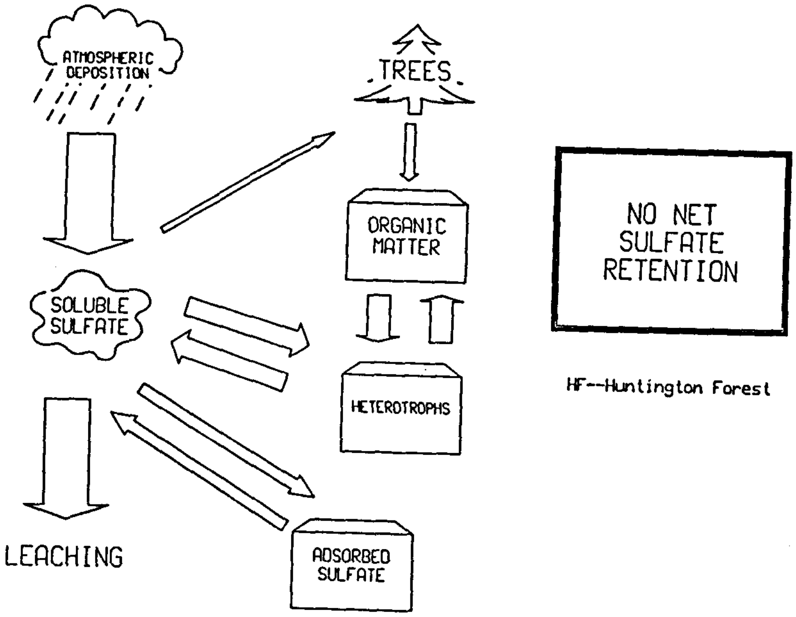


FIGURE 7



DEPOSITION AND CANOPY EXCHANGE
OF NITROGEN AT THE IFS SITES

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Site Codes

ST: Clingman's Dome, NC; red spruce

LP: Oak Ridge, TN; loblolly pine

CP: Coweeta Hydrologic Lab, NC; white pine

DL: Duke Forest, NC; loblolly pine

HF: Huntington Forest, NY; mixed hardwoods

WF: Whiteface Mt., NY; red spruce and balsam fir

DF: Thompson Forest, WA; douglas fir

RA: Thompson Forest, WA; red alder

NS: Nordmonen, Norway; Norway spruce

This report is a summary of data compiled for the Pinehurst, NC meeting of the IFS project, February 8-12, 1988. The data were collected through the diligent efforts of the many cooperators of IFS, using methods described in the IFS protocol documents. The data from the "intensive deposition" sites included wet deposition, atmospheric concentrations, dry deposition, and fluxes of throughfall and stemflow. Cloud water deposition was also measured at the Clingman's Dome (ST), Oak Ridge (LP) and

Whiteface (WF) sites. At the other sites, cloud water deposition was not considered to be a significant input process. At the time these data were summarized, all sites had been in operation for at least a year, and the numbers presented here are annual means and fluxes. At the Whiteface site, collections were not made during the winter, so the data are extrapolated to a full year, using the growing-season measurements. All of the site-to-site comparisons made in this report should be viewed skeptically until several years' data can be averaged for each site.

Atmospheric concentrations of the various N species are shown in Figure 1. Coarse particle concentrations are not shown here because their deposition was measured directly on artificial surfaces without measuring atmospheric concentration. The dominant form of N in the air at these sites is fine-particle NH_4^+, with HNO_3 vapor second, and fine-particle NO_3^- generally unimportant. The NO_x species were measured at some sites but the data were not reported at this meeting.

Because the reactive gas HNO_3 deposits very efficiently but fine particles do not, NH_4^+ is much less important than HNO_3 in delivering N to these ecosystems. Dry deposition of nitrogen at the sites is shown in Figure 2, expressed as Kg N deposited per hectare per year, and with the contribution of the various atmospheric species distinguished. Nitric acid vapor contributes between 60% and 97% of the total. Deposition rates vary from 0.8 (DF and RA sites) to 14 Kg N ha^{-1} yr^{-1} (ST site). Dry deposition is measured by several methods, none of which is without error. Our analysis suggests an uncertainty of about

± 50% in these dry deposition fluxes. Experimental task A4 (see Technical Highlight #9) is providing valuable information to help evaluate dry deposition rates of HNO_3 and NO_X to forests.

Wet, dry, and cloud water deposition of nitrogen are shown for each site in Figure 3. The sites in the eastern U.S. show a remarkable similarity in wet deposition of N, all receiving between 4-6 Kg N ha^{-1} yr^{-1} from rain and snow. The sites in the western U.S. (DF and RA) receive less, and the Norway site appears to receive more, although this may be misleading because bulk deposition was measured in Norway while wet-only deposition was measured at the other sites. Cloud water deposition delivers a significant amount of N at the ST site, minor amounts at LP and WF, and negligibly small amounts at the other sites. Dry deposition contributes roughly half (49-62%) of the total N deposition at the eastern U.S. sites, regardless of elevation. The importance of measuring dry deposition of N in nutrient cycling studies in the East is clearly indicated by these data. In the West, dry deposition appears to be less important, contributing only 25% of the total at the DF and RA sites. Wet, dry, and cloud water deposition total from 4 to 25 Kg N ha^{-1} yr^{-1} at the IFS sites, with the ST site the highest and DF/RA the lowest. (The RA site actually receives much more N deposition than this, because nitrogen fixation in the root nodules of the red alder adds over 80 Kg N ha^{-1} yr^{-1}.) At the ST site, deposition of NO_3^- and NH_4^+ can contribute a major fraction of the N requirement of the forest.

Whether or not the forest canopy takes up or releases inorganic N depends on how the exchange is measured (Figure 4). If it is calculated as throughfall plus stemflow (TF + SF) minus wet deposition, most of the canopies in the eastern U.S. appear to release N. However, this release can be accounted for by dry and cloud water deposition, so that the calculation TF + SF - total deposition indicates a net uptake of inorganic N by all the canopies. We note that the greatest uptake is shown in the high-elevation ST and WF sites, which have large communities of chemically active epiphytic lichens in the canopy. In general, the amount of N uptake increases with the amount of N deposition.

It is not surprising that N-limited forests take up deposited N. The amount of canopy uptake (<10 Kg N ha^{-1} yr^{-1}) is in general dwarfed by the amount of root uptake by these trees. This restores confidence that we do, indeed, know the function of the different plant organs, despite recent suggestions that the canopy is a major site of N uptake. We do not, in fact, know that the entire canopy uptake shown in Figure 4 is uptake by the trees themselves. Epiphytic macro- and microorganisms may be accumulating N or releasing it as organic N. Organic N in throughfall was measured sporadically at some of the sites, and the data suggest a net release from the canopy of several Kg of N ha^{-1} yr^{-1}, partially compensating the observed uptake of inorganic N.

FIGURE 1.

AEROSOL AND VAPOR N

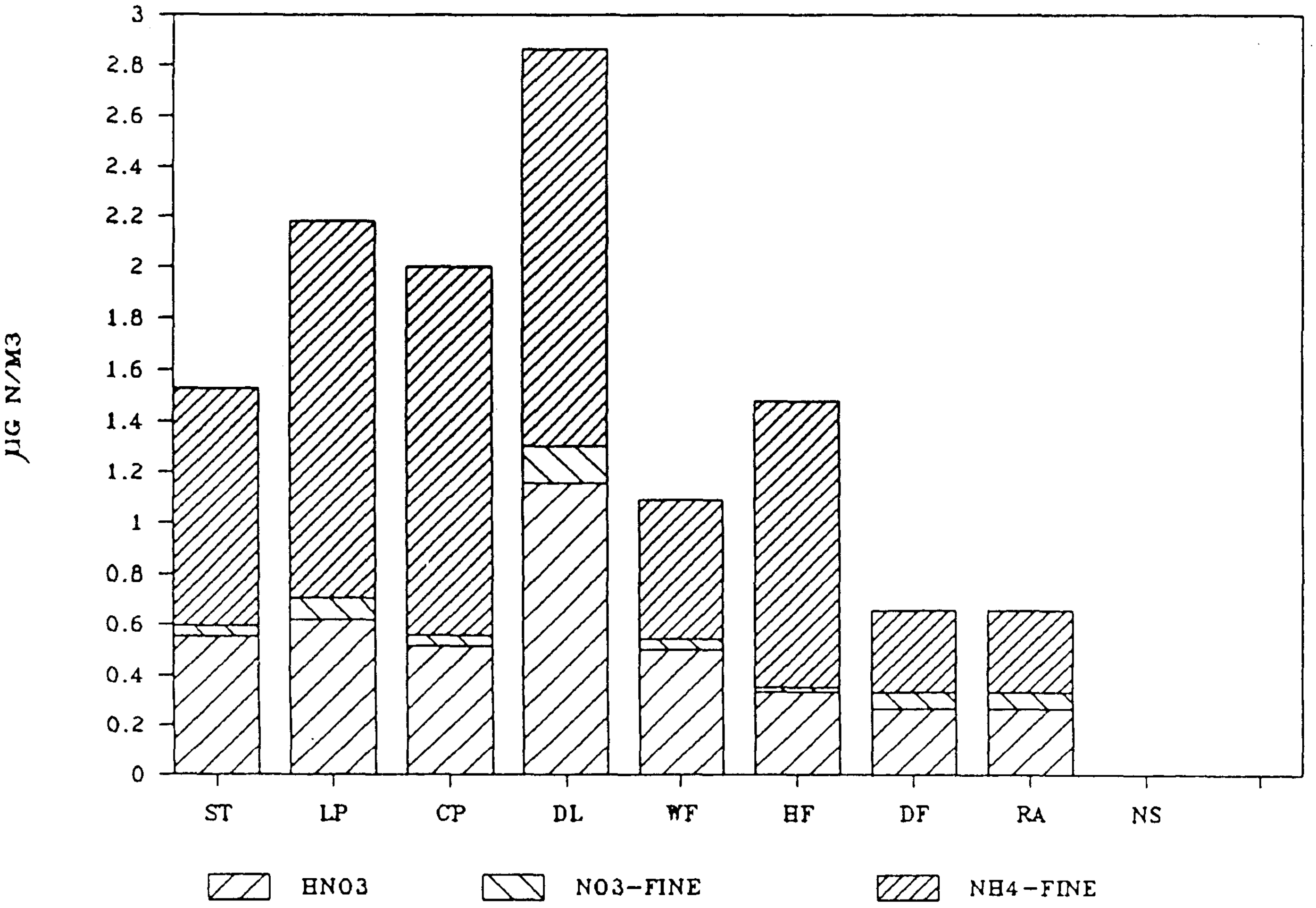


FIGURE 2.

DRY DEPOSITION OF N

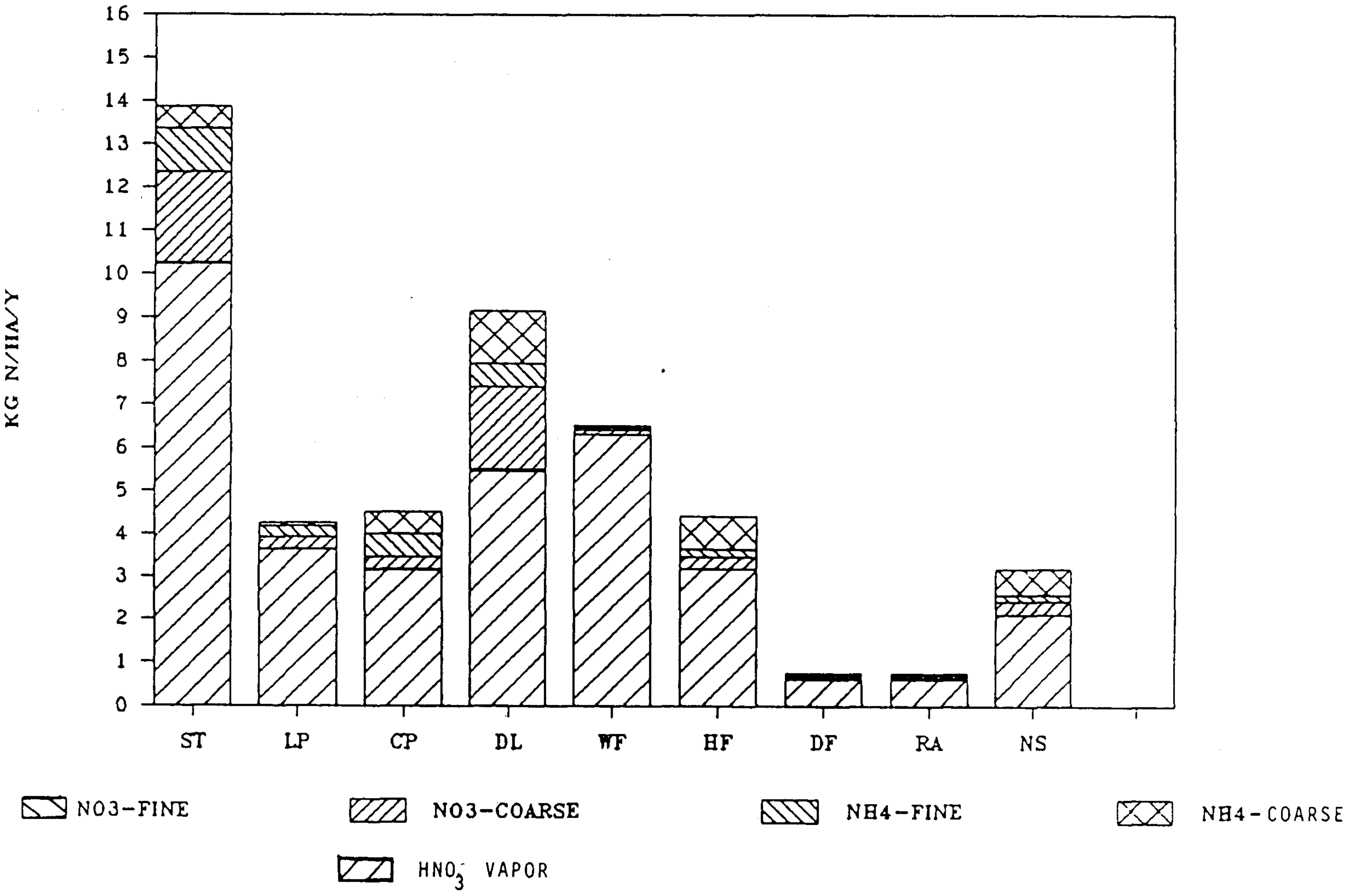


FIGURE 3.

N DEPOSITION

IFS Sites

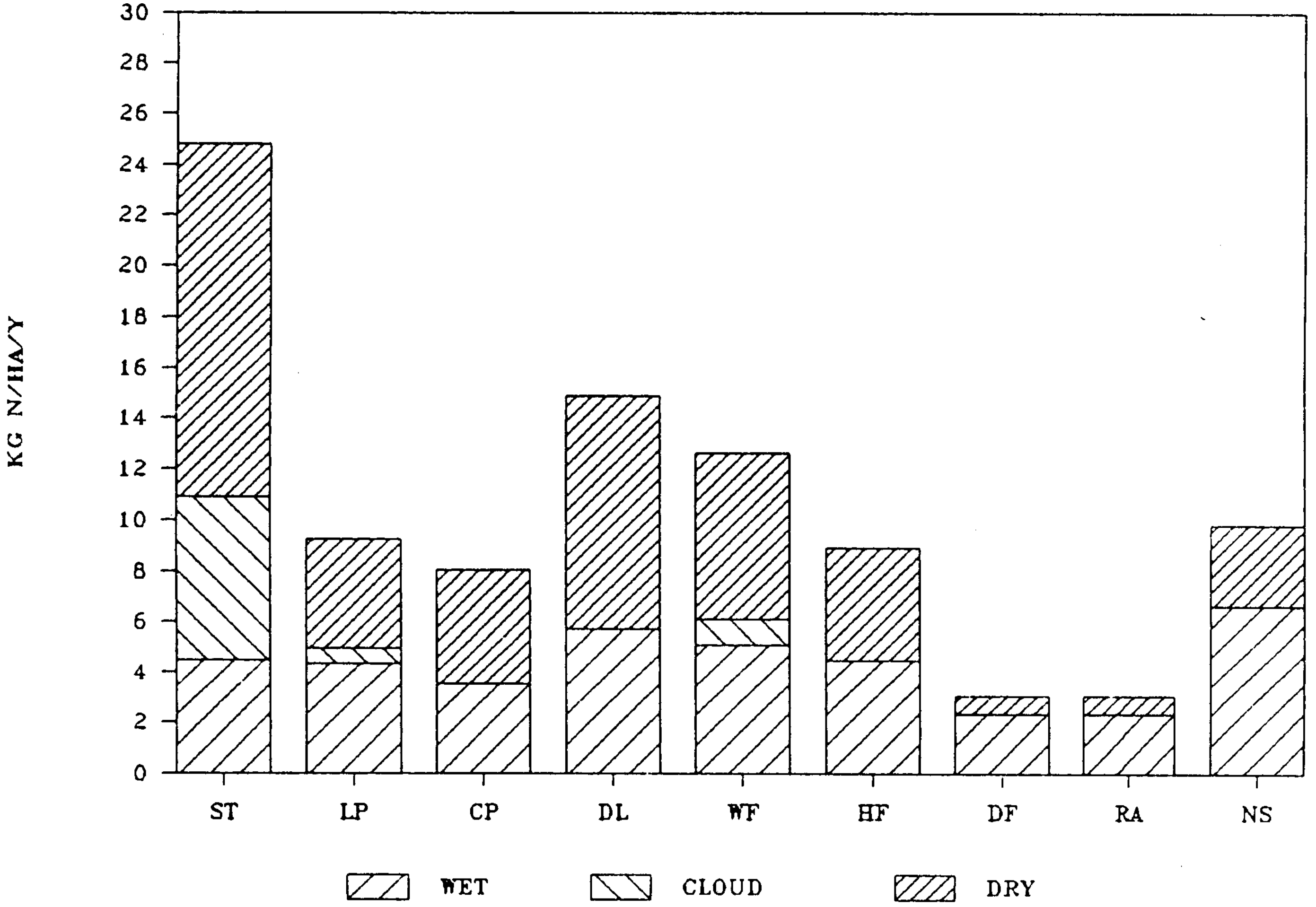
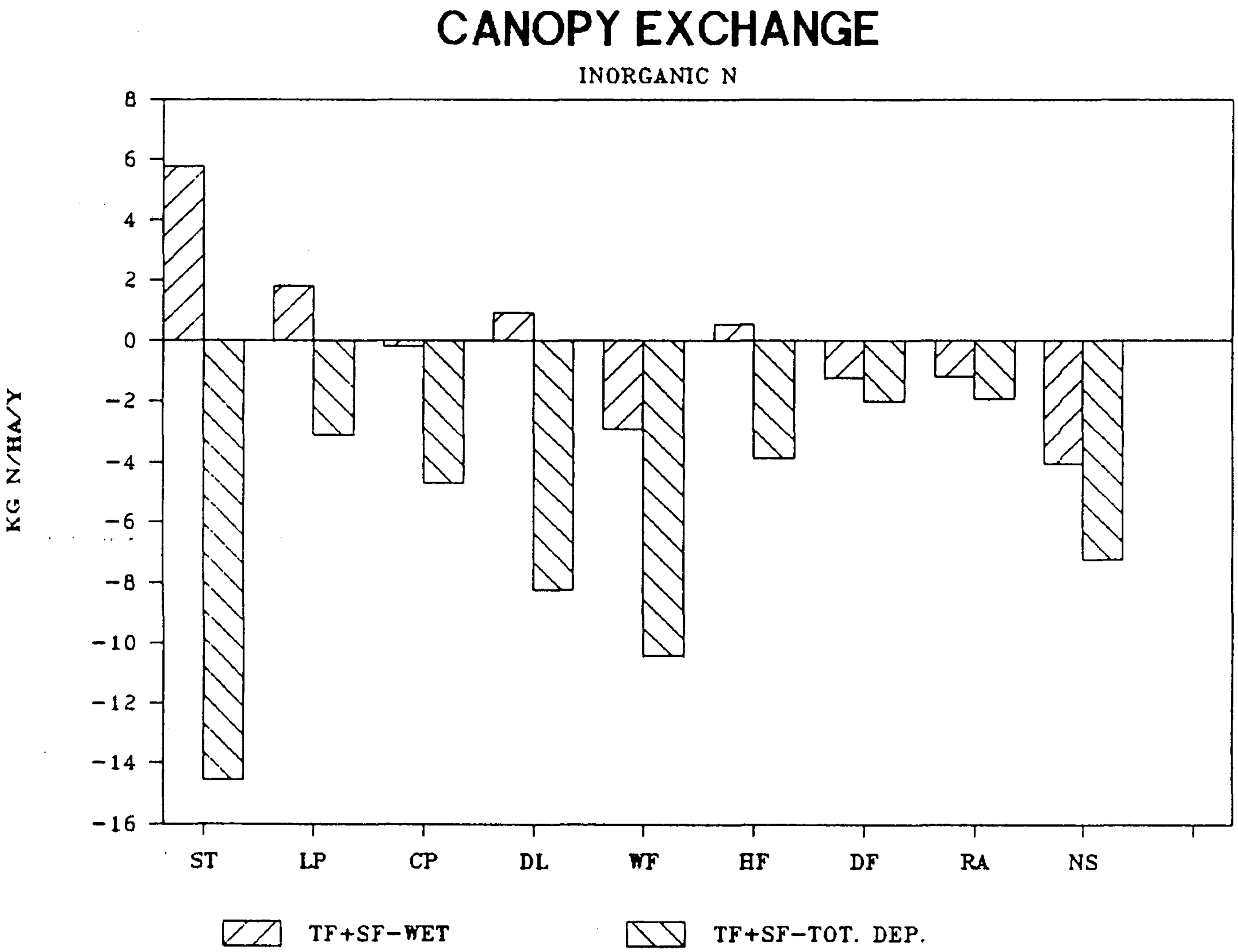


FIGURE 4.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-12
-14
-16
ST
LP
CP
DL
WF
HF
DF
RA
NS
TF+SF-WET
TF+SF-TOT. DEP.

Integrated Forest Study Highlight
April, 1988

NITROGEN SYNTHESIS TASK: CYCLING AND BUDGETS

Dale W. Cole, University of Washington
Helga Van Miegroet, Oak Ridge National Laboratory

The primary objective of the N task has been to investigate the role of atmospheric N inputs in causing N saturation and excess NO_3 leaching in forest ecosystems. At the onset of the IFS project, the following study questions were formulated for that purpose:

1. Can atmospheric N input lead to N saturation (i.e., is there a direct correlation between NO_3 leaching and atmospheric N inputs)?
2. Is N saturation also part of a normal process which develops naturally in forests as N pools accumulate, N uptake decreases, and mineralization rates increase?
3. Can N saturated systems recover after N inputs are significantly curtailed? If so, at what rate and to what extent?

Last year's preliminary data summary, based on incomplete information from a limited number of IFS sites had already suggested that the amount of NO_3 leaching from a site was not simply a reflection of atmospheric input rates, but that internal N transformation processes were likely to play a role also. The focus of this year's N synthesis work was precisely to get a better handle on the magnitude and the role of some of these internal fluxes.

More refined geographic and elevational trends in N deposition across the cooperating sites became available, including an assessment of the importance of dry deposition in total N input. In addition, a series of field mineralization experiments were initiated at most sites, which provided a rough but comparative measure of N mineralization input across a wide spectrum of soils, climatic conditions, and vegetation types. From the ongoing mineral cycling work also resulted more realistic estimates of N pool sizes and N uptake, although the degree of certainty on the latter differed by site. Nevertheless, the N input data, coupled to the N pool size, and N flux information brought us another step closer towards understanding the processes regulating N saturation and NO_3 leaching.

This year's input/output data reaffirms the earlier suggestion that no direct correlation exists between total N input from the atmosphere and NO_3 leaching below 40 cm soil depth. Sites either showed N leaching output in excess of atmospheric deposition input or an apparent net N accumulation (Fig. 1). This suggests that additional N sources and sinks within the ecosystem were not accounted for. One logical N input source is N mineralization. The buried bag technique used in this study to quantify mineralization has obvious limitations in that only the upper 10 cm of soil is extracted periodically and N transformations occur under somewhat artificial circumstances. The numbers obtained are nevertheless useful for comparative purposes. The IFS sites represent a wide array of mineralization and nitrification potentials (Fig. 2), and the sites with the highest average monthly NH_4 and NO_3 production rates were generally those with higher NO_3 leaching rates, although the relationship was otherwise not very tight. This relative ranking of mineralization rates was then used in combination with the total soil N pools to roughly estimate the annual N input from the soil via mineralization, thus also incorporating the role of past soil N accumulation in causing N saturation of the soil. In other words, where considerable amounts of N had accumulated over the history of the site annual N input via mineralization could still be substantial even at relatively low measured monthly rates (e.g., spruce sites in the Smokies). Once again, no firm correlation between estimated mineralization input and leaching output could be established (Fig. 3). Finally, the role of N uptake by trees was assessed. In some sites N uptake could be derived directly from the available nutrient cycling data, and where that was not the case some rough estimate was made. No distinct pattern emerged when NO_3 leaching was plotted against N uptake.

Although none of the N fluxes individually could explain the NO3 leaching pattern observed, a much clearer picture emerged when they were combined into the term "excess available N" (Figs. 3 and 4). It was calculated as the sum of N input from the atmosphere and mineralization minus N removal via tree uptake. For the red alder site the input term also included an estimate for N input via symbiotic N-fixation. Figure 4 seems to suggest that NO_3 leaching is regulated by excess N availability, whether this results from high atmospheric N input rates, mineralization of inherently large N pools, rapid organic matter turnover, low N immobilization in tree growth, or a combination thereof. Indeed, most sites are now spread around the 1:1 line and very few fall above it (i.e., NO_3 leaching seldom exceeds the calculated amount of N available for nitrification). A further refinement of the N flux values, including estimates of the (annual) variability will be necessary to firmly establish the importance of excess N to N saturation. If this relationship between NO_3 leaching and excess N availability indeed holds, then any decrease in excess N (whether caused by a decrease in atmospheric input or an increase in tree N uptake; e.g., through replacement of a decadent stand by a younger and more vigorous one) should result in a decline in NO_3 leaching. Conversely, an increase in the excess N availability should stimulate nitrification which, in turn, could result in increased NO_3 leaching, if no further N immobilization or denitrification occurs.

We currently have no good estimate of immobilization rates, nor do we know to what extent denitrification contributes to the N balance. Either one or both processes could be responsible for the relatively low NO_3 leaching output in the alder stand compared to the current estimate of excess available N (Fig. 4). The role of denitrification obviously needs to be investigated further, preferentially in a few target sites with the greatest potential for measurable denitrification (high NO_3 levels in soil and solution, high organic matter content, appropriate water regime) such as the Smokies, Turkey Lakes, and the Washington red alder sites.

As to the question of possible desaturation/recovery, the soil solution data from the forest conversion plots in Washington clearly show that a reduction in N input (in this case through removal of the symbiotic N-fixers) has significantly decreased NO_3 solution concentrations at all collection levels throughout the 3-year observation period, and irrespective of the composition of vegetative cover (Fig. 5). However, solution pH has not changed significantly, and a recovery of the soil properties to pre-saturation characteristics, as exemplified by the Douglas-fir stand, is rather unlikely at this time. The presence of a N-fixer, with the potential for increased N input and N availability, has failed to cause a significant increase in NO_3 leaching at this time (Fig. 5).

In conclusion, the integration of N pool and flux data across the IFS sites indicates that N saturation can occur, either as the result of elevated atmospheric N deposition, naturally when mineralization input of accumulated N exceeds biological immobilization, or through a combination of these external and internal processes. Ecosystems appear to differ substantially in their capacity to store N, which will clearly affect the point at which N saturation is reached and accelerated NO_3 leaching occurs. There are suggestions that denitrification should be incorporated in the N balance calculations at some sites. Experimental data confirmed the hypothesis that a decline in excess available N results in decreased NO_3 leaching. The rate of recovery from N saturation will vary with the inherent characteristics of the ecosystem in question.

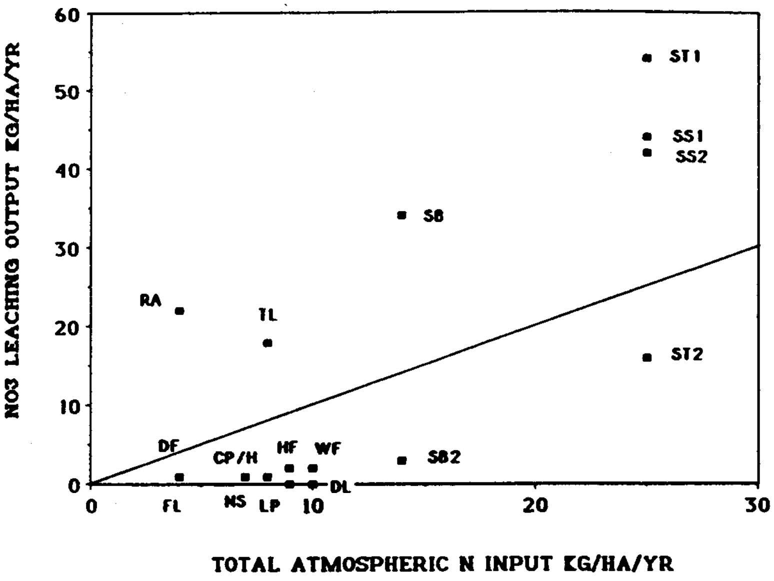


Figure 1. Nitrate leaching vs total atmospheric N input (kg/ha/yr).

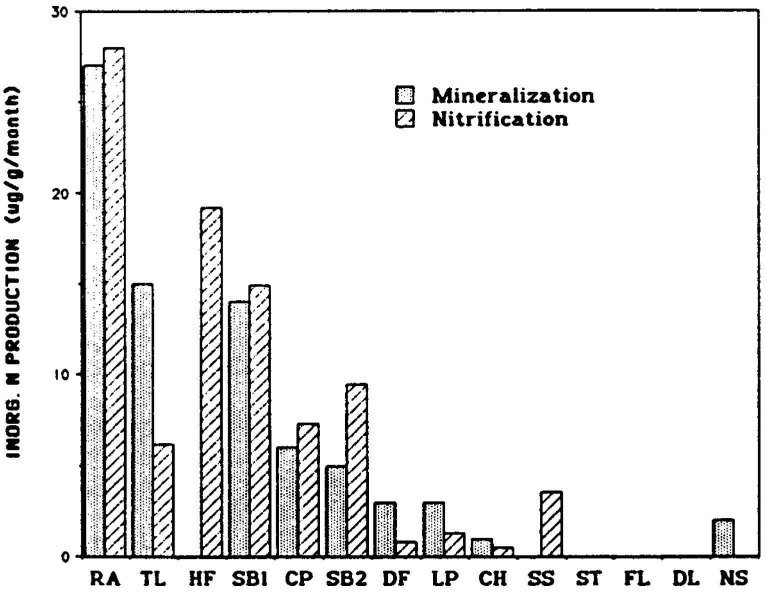


Figure 2. Ranking of IFS sites according to soil mineralization rate.

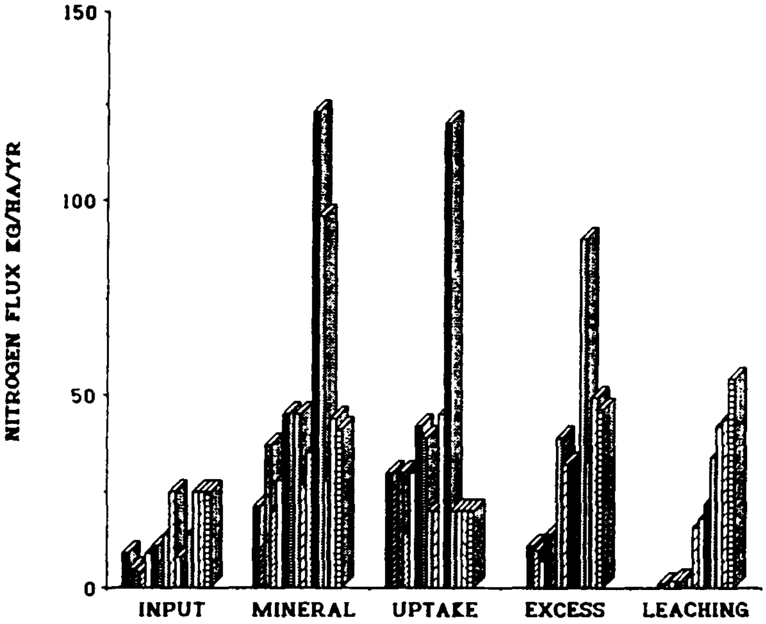


Figure 3. Nitrogen fluxes at the IFS sites [input, mineralization, uptake, excess available, and leaching (kg/ha/yr)].

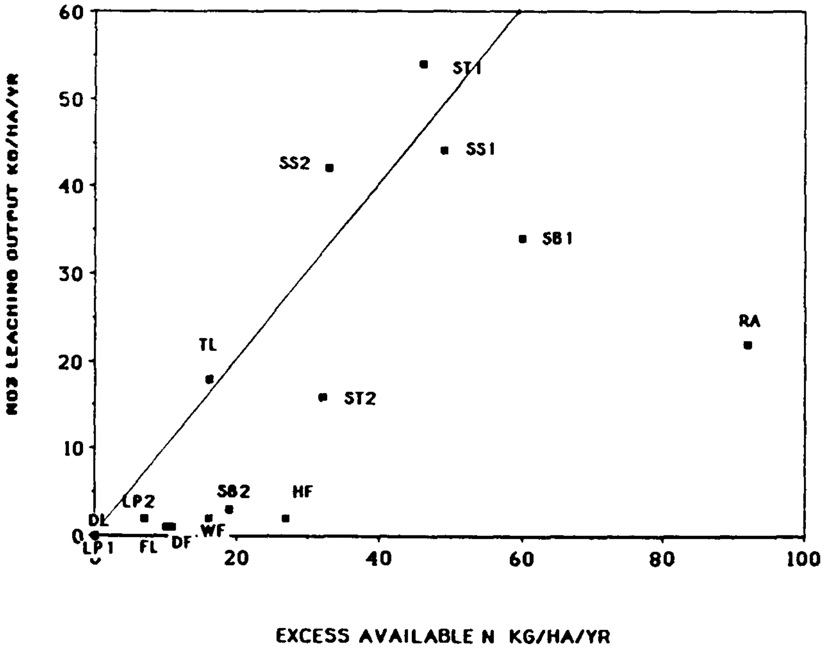


Figure 4. Nitrate leaching vs estimated excess available N (kg/ha/yr).

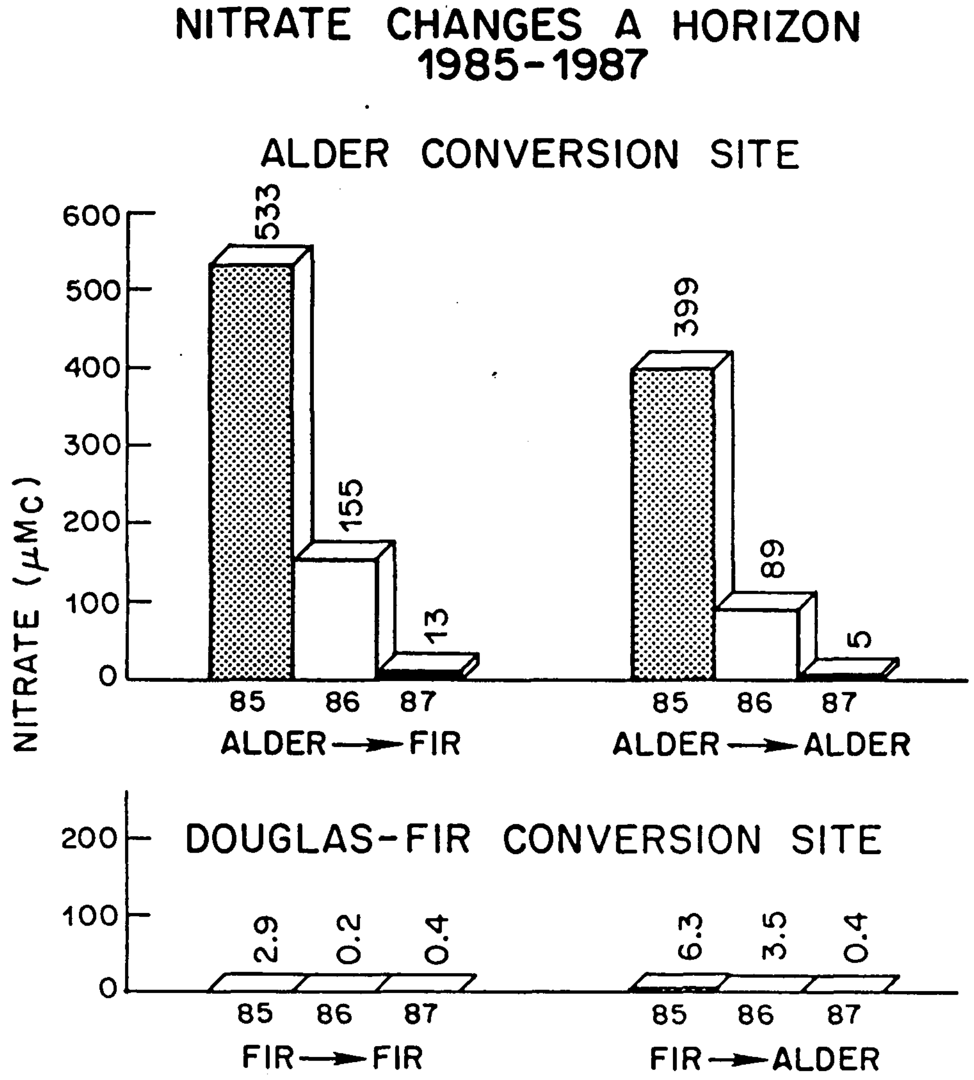


Figure 5. Changes in nitrate concentration of solutions collected in the A horizon of the Washington conversion sites from 1985 through 1987.

ANNUAL H^+ ION DEPOSITION AND CANOPY EXCHANGE - IFS SITES 1986/87

by

Kenneth R. Knoerr - Duke University

Annual H^+ ion precipitation wet deposition ranged from a low value of about 15 meq/m^2/yr at the pacific northwest DF/RA sites to nearly 60 meq/m^2/yr at the eastern high elevation WF and ST sites (Fig. 1). Some of the sites had similar H^+ volume weighted precipitation concentrations: ST,CP$\approx$30 uM$^\pm$/L; WF,LP,DL$\approx$45uM$^\pm$/L. Within each of these concentration groups the total H^+ precipitation deposition was linearly related to the annual amount of precipitation (Fig. 1).

For a number of the sites the precipitation ratio of $H^+/(SO_4^= + NO_3^-)$ was close to 1, indicating a near balance between the acidity and the sulfate and nitrate ions (Fig. 2). However, for some of the sites, particularly HF and DF/RA, this ratio was considerably less than 1. In these cases the precipitation acidity was partially neutralized by base aerosols or some of the sulfate and nitrate ions were from non-acidic sources. For all of the sites the sulfate source was about two to three times the nitrate source of H^+ ions (Fig 2).

Total H^+ deposition was about equally divided between wet deposition and dry deposition (Fig. 3). Exceptions were the DF/RA and WF sites where dry deposition was somewhat less than wet deposition. While total precipitation deposition was similar at the two high elevation WF and ST sites (Fig. 1), total wet deposition was considerably greater at the ST site (Fig. 3). This was due to the considerably greater cloud water deposition at the ST site.

Canopy exchange generally depleted H^+ ions. Thus, the precipitation reaching the forest floor as throughfall and stemflow had less H^+ ions than the total deposition (Fig. 3). The exceptions were the DF and ST sites where there was no net canopy exchange. The canopy exchange (depletion) of H^+ ions was not related to either total wet H^+ deposition or the H^+ concentration of precipitation.

Fig 1.

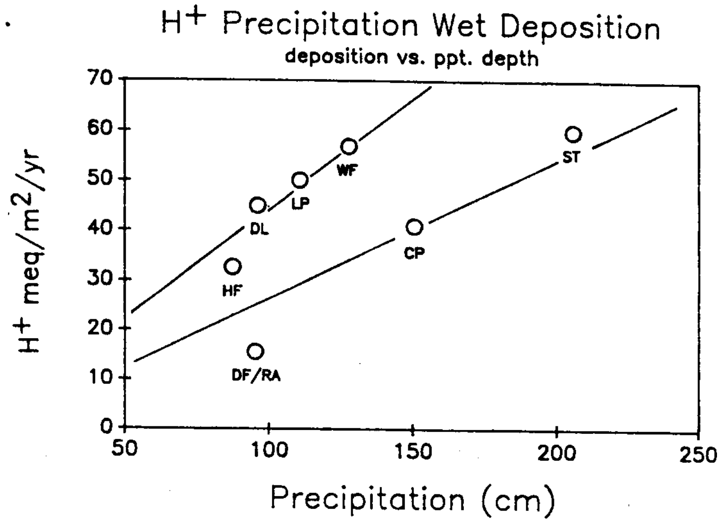


Fig 2.

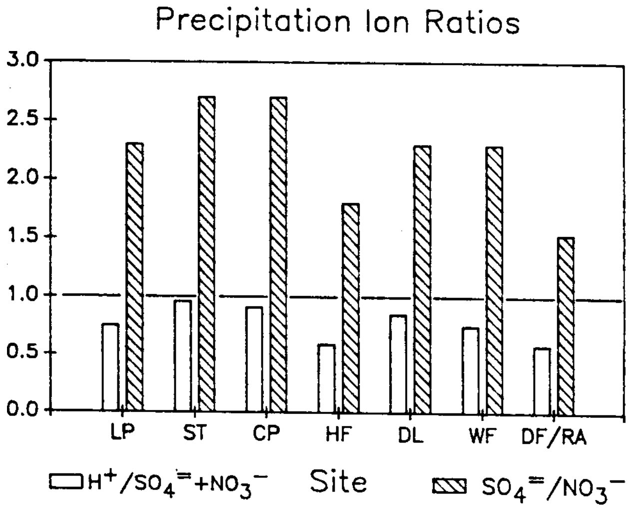
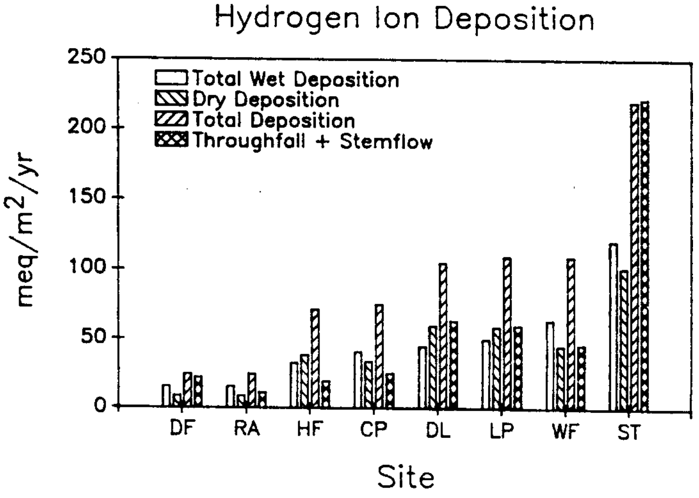


Fig 3.



Synthesis of H+ Budgets for IFS sites--February 1988

Dan Binkley, Dept. of Forest Science, Colorado State University, Ft. Collins, CO 80523

Forests generate and consume vast quantities of H+ as a natural component of nutrient cycles. The potential effects of acidic deposition need to be evaluated in the context of these natural fluxes. The net production or consumption of H+ are controlled by the balance among:

H^+ deposition from the atmosphere
Accumulation or depletion of cations from the ecosystem
Transfers of cations among pools within the ecosystem
Accumulation or depletion of anions from the ecosystem.

On an annual timescale, the net increase or decrease in H+ in the ecosystem drives changes in soil pH primarily by changing the distribution of cations on the exchange complex.

Among the IFS sites with sufficient data, the two mountain sites (Smokies Tower and Whiteface) have the greatest net increase in H^+ (Figure 1). At 400 mol_c/m^2 annually (4 $kmol_c/ha$), the net flux of H^+ at the Smokies Tower site is among the highest reported in the world. The Duke Loblolly site is also fairly high (about 170 mol_c/m^2 annually), due to a large net loss of sulfate. This site has an unusually high amount of total S in the soil (about 3,000 kg/ha, or about 10 mol S/m^2), probably resulting from use of superphosphate fertilizers (which contain slightly more S than P) in agricultural management. Many forests in the Southeast probably received similar treatments before abandonment of agriculture, and they may also show net losses of sulfate. The site with the next-highest net production of H^+ is the Washington red alder site, where net loss of nitrate leads to high H^+ production. Across all sites, the H^+ deposited from the atmosphere account for about 42% of the net H^+ production (or input) in the ecosystems (Figure 2). The major outlier is the red alder site, where deposition is small relative to H^+ production associated with the net loss of nitrate.

At this point, we cannot apportion cation leaching into the relative contributions from mineral weathering and stripping from the exchange complex. This is an important distinction, because the alkalinity generated in mineral weathering (in the form of silicate anions) consumes acidity within the soil (by producing silicic acid which then leaches). The alkalinity associated with the removal of "base" cations from the exchange complex (such as $KHCO_3$) transfers alkalinity to downstream ecosystems at the expense of the forest soil. Hopefully, next year's synthesis can examine these processes individually.

Of the 9 sites with complete information, 5 appear to lose sulfate, and 4 appear to accumulate sulfate (Figure 3). In the Eastern U.S., only the Coweeta sites retain sulfate. Therefore, H^+ production associated with net losses of sulfate contribute a large portion of the total in several ecosystems. In contrast, only the Smokies Tower site and the red alder sites lose more nitrate than they receive in deposition (Figure 4). In the other ecosystems, the nitrate retention represents generation of alkalinity that consumes acidity. The production of bicarbonate is an important source of H^+ in 6 of the 9 ecosystems, but the pH is too low for carbonic acid to dissociate in the other 3 (Figure 5).

In summary, the H^+ dynamics differ greatly among sites (Figure 6). The Smokies Tower site shows the greatest net H^+ flux, due to the combination of atmospheric deposition and net nitrate loss. The Duke loblolly pine site has a high flux due to net loss of sulfur left over from agricultural fertilization. The red alder site's flux is dominated by net nitrate loss. In contrast, the Douglas-fir site is unexciting in all categories!

What are the implications for soil acidity? This question requires more synthesis among the tasks, including computer simulation. The answer will vary substantially among sites, due to differences in soil buffering capacities. In a non-IFS, old-field pine site in South Carolina, we (C. Wells, D. Valentine, and U. Valentine) found that net H^+ production of about 200 to 300 mol_c/m^2 annually changed soil pH by 0.3 to 0.8 units (varying among horizons) in just 20 years, and depleted from 20 to 80% of the "base" cations. This soil was probably buffered more poorly than most IFS sites, but it does indicate that the rates of net H^+ production found in the IFS sites may substantially alter soil chemistry on a time scale of decades.

Figure 1. Net H^+ balance for IFS sites. 100 $mmol_c/m^2$ = 0.1 $kmol_c/ha$.

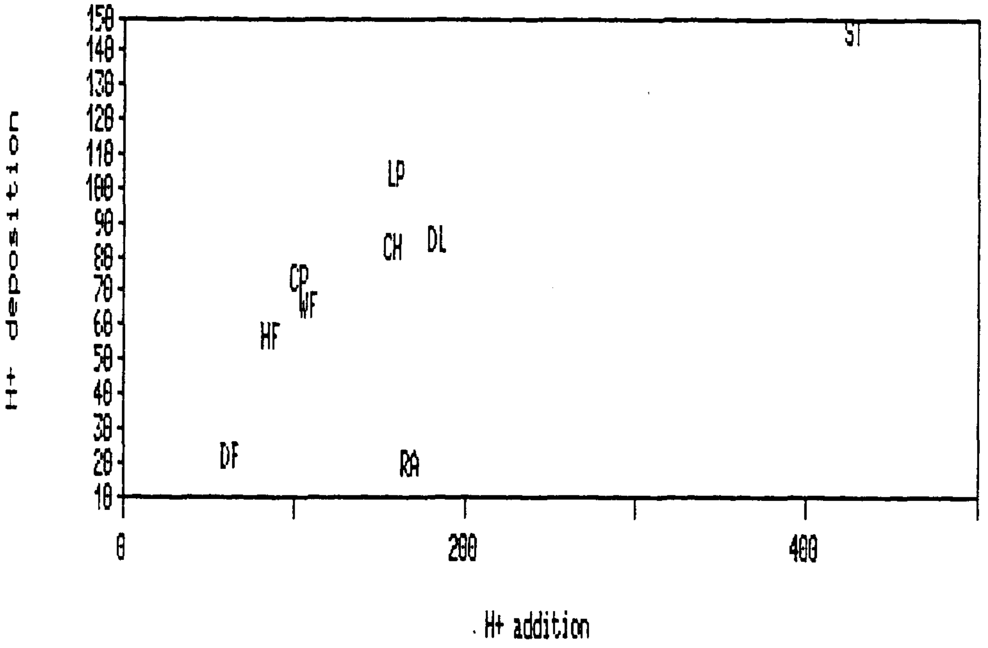


Figure 2. H^+ deposition as a function of net H^+ balance. Slope is 0.42.

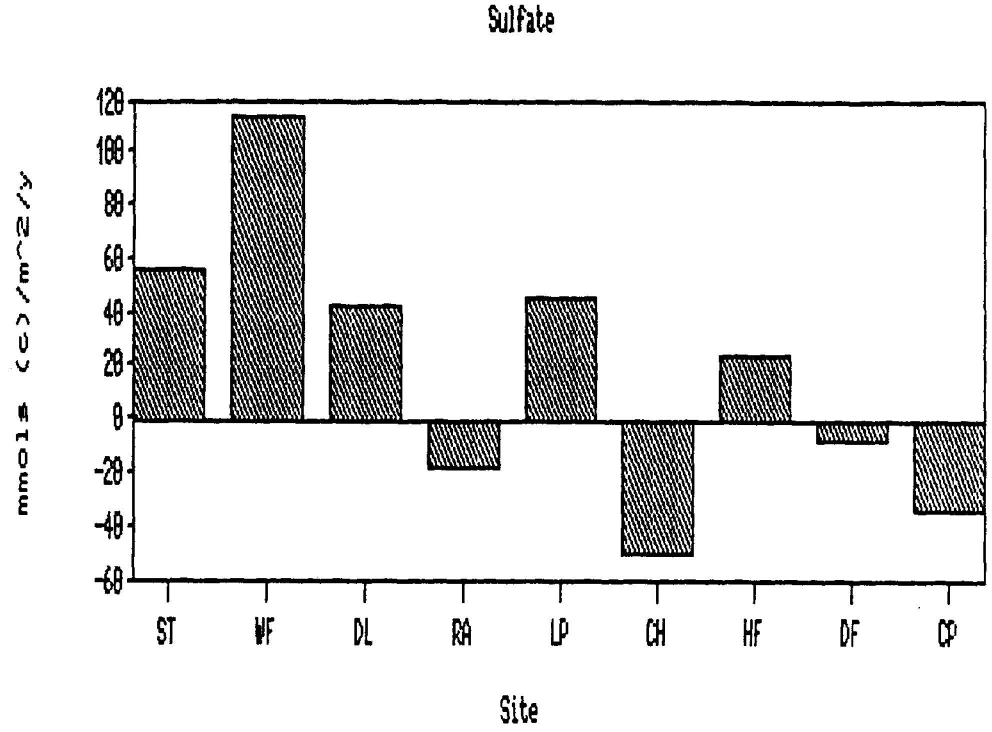


Figure 3. Net sulfate balance for IFS sites. Above 0 means net S loss, and net H^+ production.

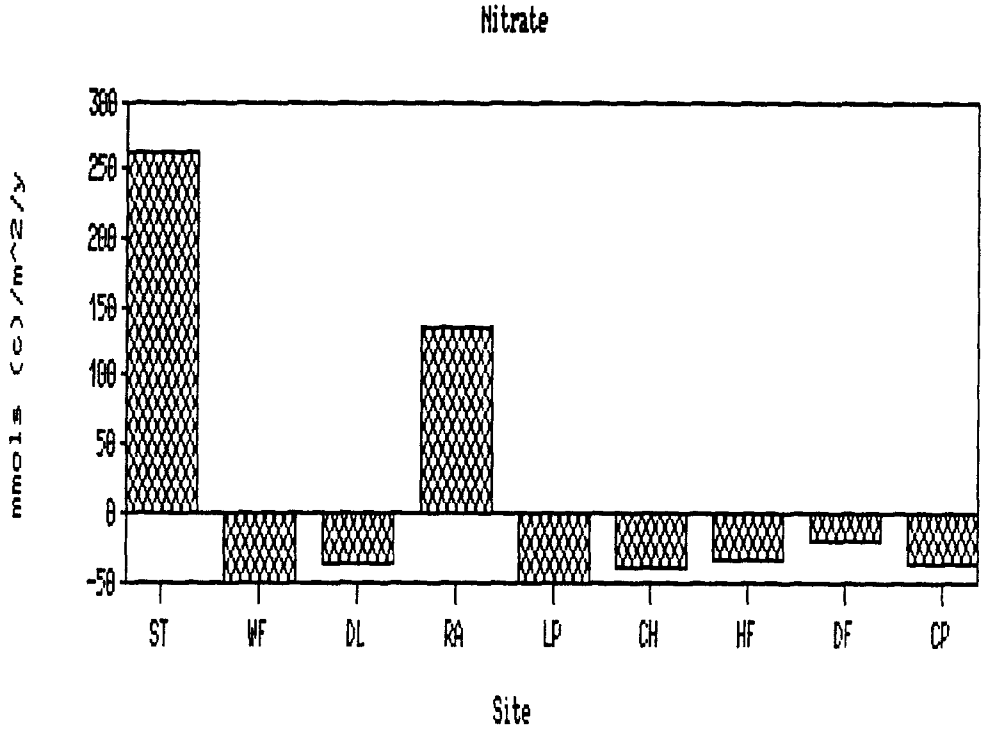


Figure 4. Net nitrate balance for IFS sites. Above 0 means net N loss, and net H^+ production.

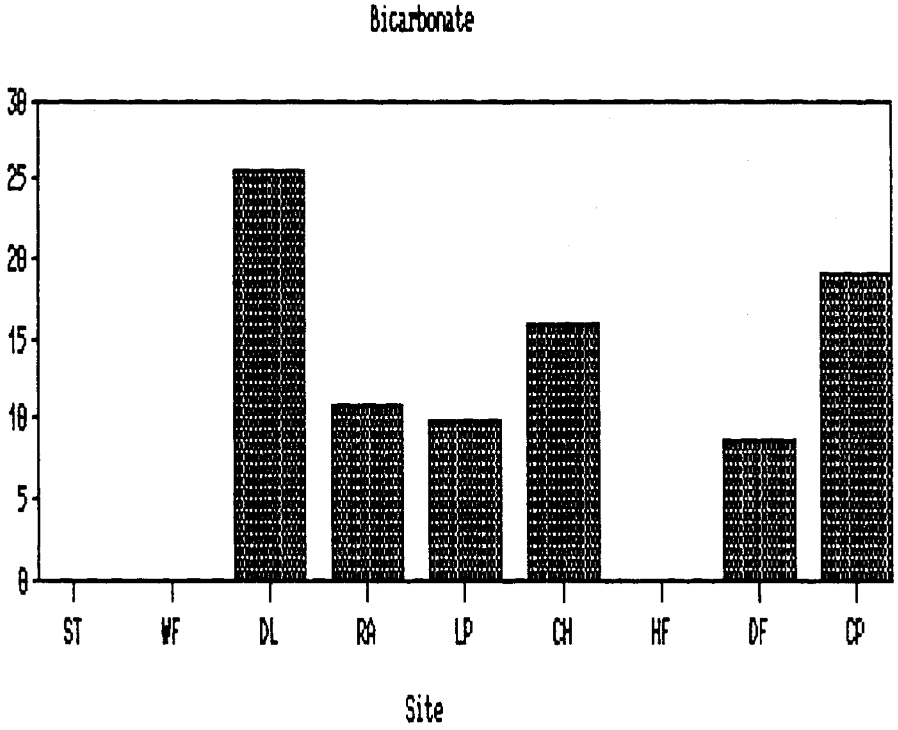


Figure 5. Net bicarbonate balance for IFS sites. Above 0 means net bicarbonate loss, and net H^+ production.

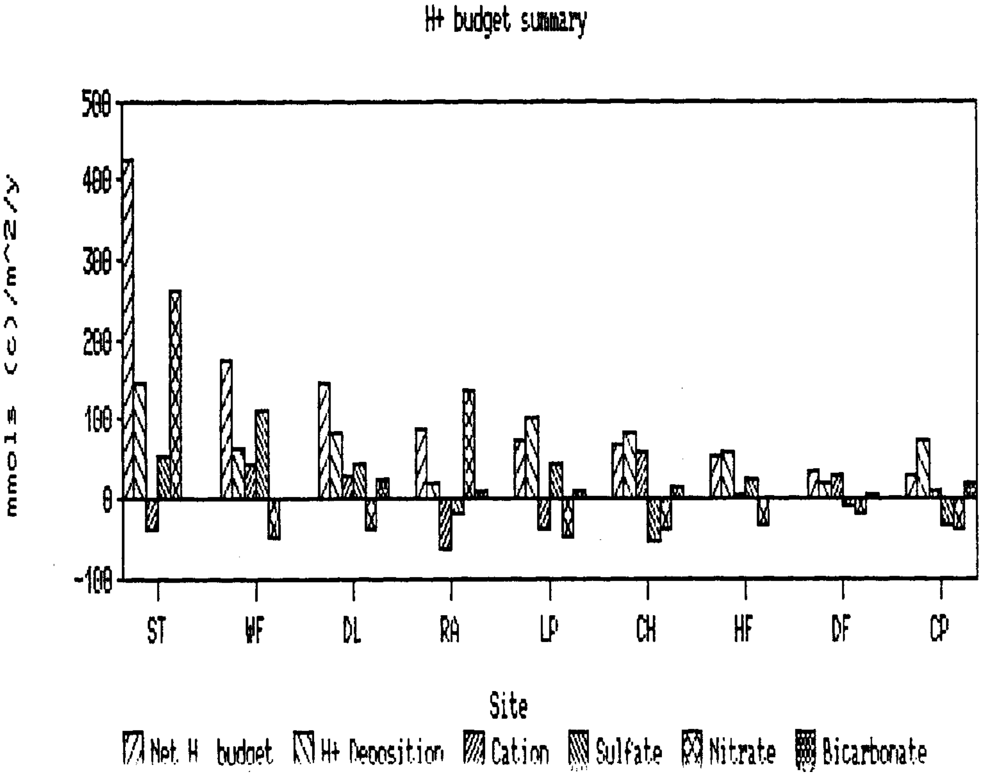


Figure 6. Net H^+ budget (first bar), and the components of the net budget. Above 0 means net H^+ production; below 0 means net H^+ consumption.

FIGURE 1

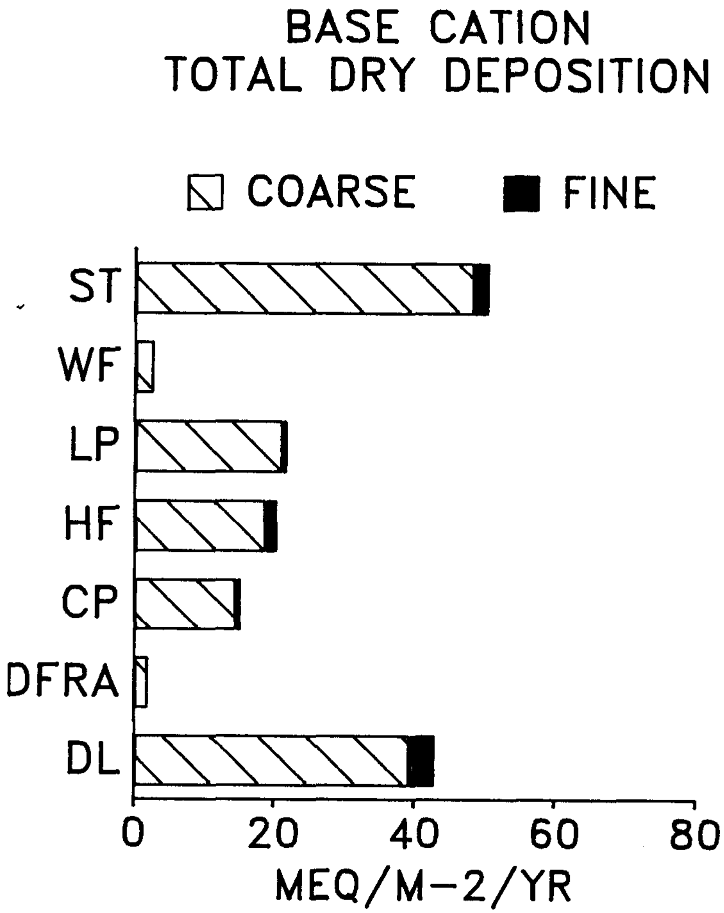


FIGURE 2

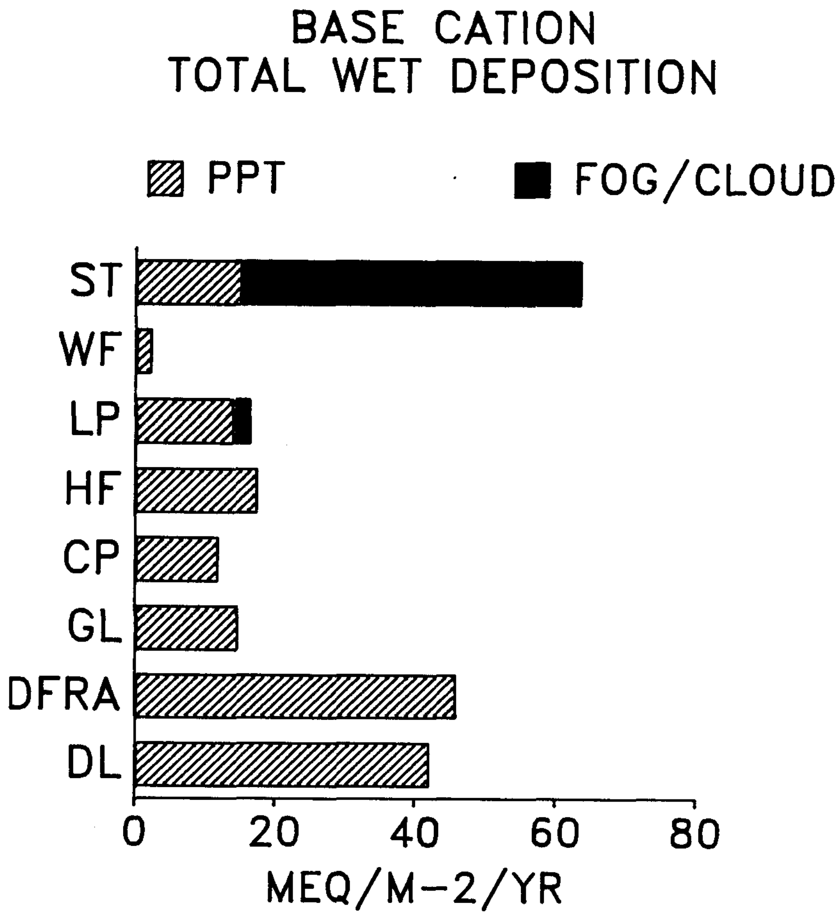


FIGURE 3

BASE CATION
THRUFALL+STEMFLOW
DEPOSITION TO FOREST FLOOR

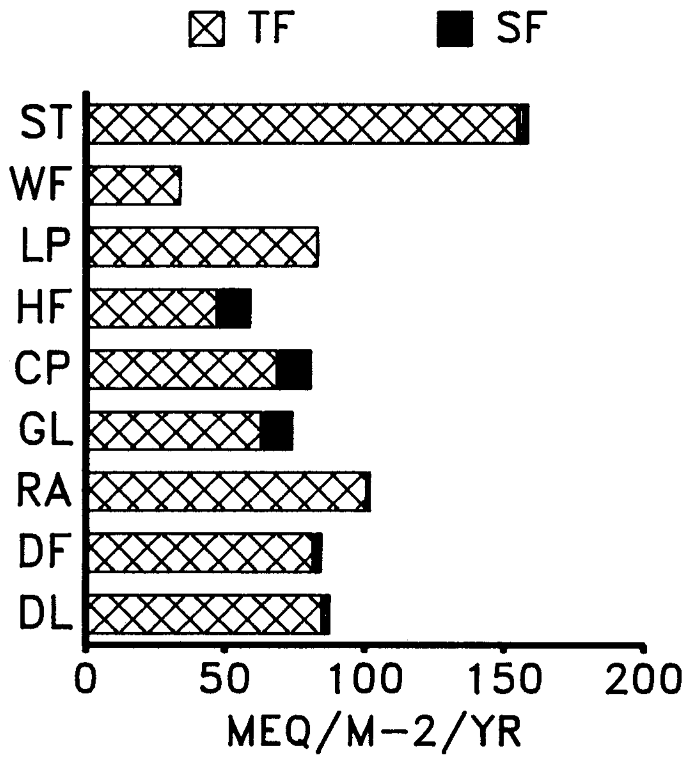


FIGURE 4

BASE CATION
NET CANOPY FLUX

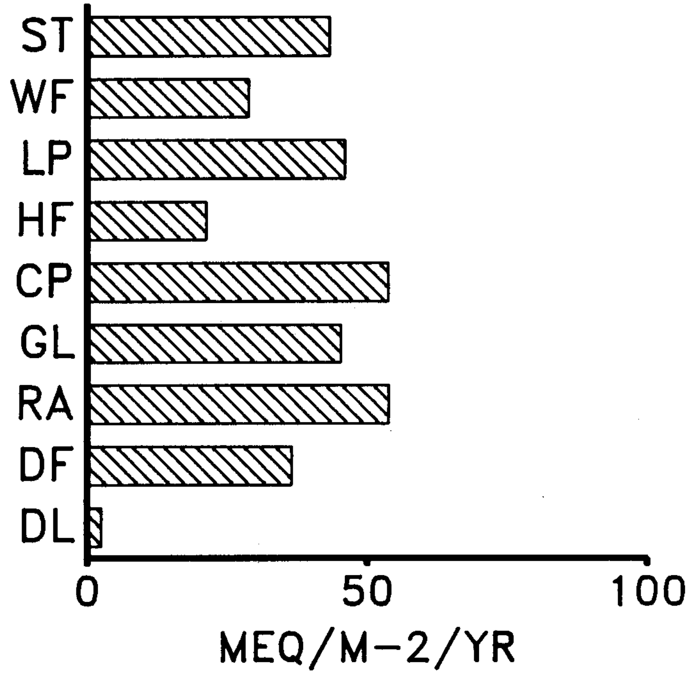


FIGURE 5.'86-87 COARSE DRY FLUX

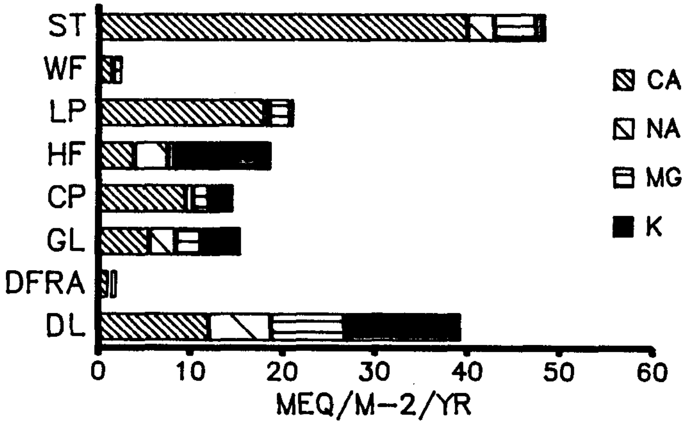


FIGURE 6. '86-87 PPT FLUX

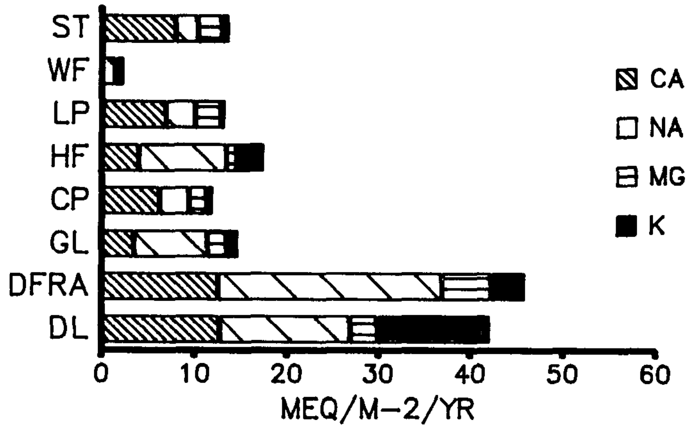


FIGURE 7.'86-87 TF+SF DEPOSITION

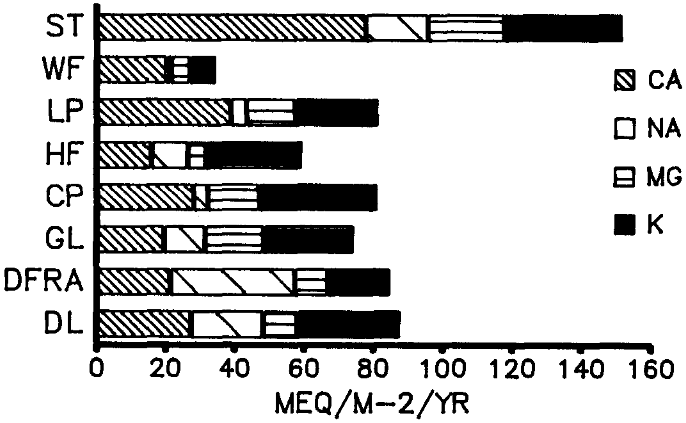
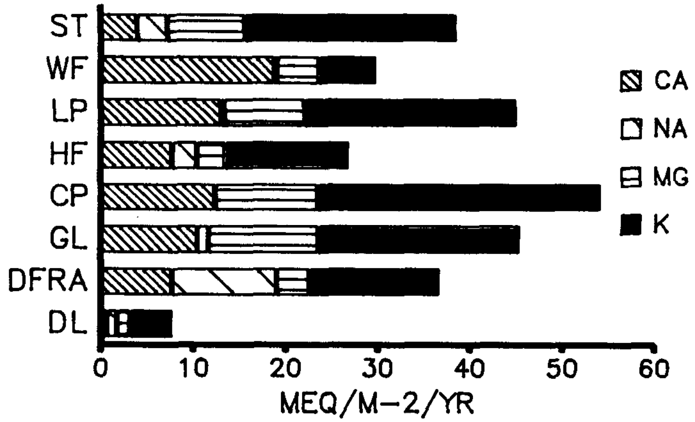


FIGURE 8.'86-87 NET CANOPY FLUX



INTEGRATED FOREST STUDY
CATIONS SYNTHESIS: CYCLING AND BUDGETS

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The ultimate goal of the cations synthesis task within the Integrated Forest Study is to assess the current cation nutrient status of the various forest ecosystems within the project, assess the role that atmospheric deposition is affecting the cation nutrient status of these systems, and to forecast, with the aid of simulation modeling, the potential changes in this nutrient status that may occur both naturally and in response to atmospheric inputs. This analysis will require a detailed examination of the cation nutrient cycles of each site and integration of this information with that obtained from the cation exchange and weathering experimental tasks. At the present time, we have detailed cation nutrient cycling data from only a few sites, and the cation exchange and weathering experimental tasks are not completed yet, so the full analysis described above remains in the future. At this stage, however, we can conduct a capacity-intensity type analysis on the soils and soil solutions from many of the IFS sites, and we can examine the behavior of some of the key equations that will be incorporated into the simulation model that is contemplated as one of the outcomes of this study.

For this particular analysis, we define capacity- vs intensity-type effects as per Reuss and Johnson (1986): a capacity-type effect as one that causes a change in the soil (such as a decrease in base saturation) and an intensity-type effect as one that causes a change in soil solution (such as an increase in soil solution Al^{3+}). The factors that can cause an intensity-type effect can be seen from the Gapon equation (one of the three equations commonly used to describe cation exchange in soils, the others being the Gaines-Thomas and Vaneslow); solving for soil solution Al^{3+} activity, or (Al^{3+}), we see that (Al^{3+}) increases as the 3/2 power of (Ca^{2+}) (Fig. 1). By the same principle, (Al^{3+}) increases as the third power of K^+, or Na^+ (Reuss and Johnson 1986). Thus, an increase in the total ionic strength of soil solution, for instance by increasing the concentration of mineral acid anions NO_3^- and SO_4^{2-}, causes the ratio of $(Al^{3+})/(Ca^{2+})$ or (Al^{3+}) to other base cations to increase. This effect can perhaps be seen more clearly in the schematic diagram in Fig. 2; the introduction of a mineral acid anion (depicted as A^- in Fig. 2) to an extremely acid soil will cause the mobilization of Al^{3+} from the exchange sites of that soil into the soil solution, and this effect will occur whether A^- is added as a salt or as an acid (as long as the total amount of salt or acid added remains small relative to the cation exchange capacity, or the total pool of exchangeable Al^{3+} and base cations).

Within the IFS project, we have a spectrum of soils in terms of base saturations and soil solution mineral acid anion concentrations that allow us to test whether the behavior of cation exchange equations such as the Gapon qualitatively describe the behavior of soil solution Al^{3+} and base cations. From Fig. 3, we see that the Smokies Tower, Smokies Becking, Whiteface and Findley Lake sites all have extremely acid soils (%base saturation less than 10%), especially in subsurface horizons. From Fig. 4, we also see that soil solutions from these four sites have very different mineral acid anion concentrations: there is a clear gradient from the relatively pristine Findley Lake site to the more SO_4^{2-}-rich Whiteface site to the SO_4^{2-}- and NO_3^--rich Smokies sites. From Fig. 5, we see that, true to predictions, the ratio of soil solution Al^{3+} to M^{2+} ($M^{2+} = Ca^{2+} + Mg^{2+}$) increases steadily from the Findley Lake to the Whiteface to the Smokies sites.

The other aspect of this analysis, the capacity-type analysis, is far more complicated. In order to forecast potential changes in soil cation pools, we need to know the rates of atmospheric input, leaching, plant uptake, and, most critically, weathering. It is the latter that will be the most difficult to quantify; indeed, we will likely be able only to give semi-quantitative or perhaps just qualitative estimates of weathering rates, based upon the size and relative weatherability of the soil mineral pools (see the report by Newton and April).

There are two possible scenarios in which we would predict little or no additional acidification of soils within the next few decades, even if weathering rates remain unknown, however. The first would be in those sites with extremely acid soils, where we expected to find relatively low base cation leaching (but relatively high Al^{3+} leaching, for the reasons discussed above). Under these conditions, it is quite possible that atmospheric cation inputs exceed cation leaching, making any further soil acidification very unlikely. This scenario is depicted schematically in Fig. 6. The other scenario is one in which the size of the exchangeable cation pools are so large compared to base cation leaching rates that changes in the former are unlikely to occur in less than many decades to centuries.

Figure 7 shows the total base cation depositions and estimated leaching rates for several of the IFS sites. Surprisingly, there are apparently considerable net base cation exports from the sites with extremely acid soils (Whiteface, Huntington Forest, Smokies), and only the Douglas-fir forest in Washington state shows any evidence of a net cation gain. Looking at Ca^{2+}, K^+, and Mg^{2+} inputs vs leaching individually and comparing the net fluxes (leaching minus input) of these cations with the exchangeable pools gives us the somewhat surprising picture that the most acid systems (Whiteface, Huntington Forest, Smokies, Turkey Lakes) show the highest rather than the lowest fraction of soil exchangeable base cations lost by leaching (i.e., leaching minus atmospheric deposition; Fig. 8). The loblolly pine sites at Duke and ORNL and the Douglas-fir site in Washington show little potential for change based upon this analysis. The primary reason for these patterns is the very large differences in soil exchangeable cation capital among these sites, the two loblolly pine sites being by far the highest (Fig. 9). Data on soil total cation pools is much more limited, and only a few sites can be compared. In this case, the Smokies Becking site shows the largest fraction of total Ca lost by leaching, but this fraction was quite small (<0.008; Fig. 10), and the

fraction was either smaller or positive (indicating a net cation gain) in all other cases for which data was available.

In summary, we find at this stage that the high elevation and northern sites in the east (Whiteface, Huntington Forest, Smokies, Turkey Lakes) have the highest soil solution Al^{3+} concentrations, and that this is due to a combination of low base saturation in the soils of these sites combined with a relatively high mineral acid anion concentration in soil solution. We note that low base saturation is a necessary but not a sufficient condition for the presence of Al^{3+} in soil solution, as evidenced by the low soil solution Al^{3+} in the Findley Lake site. We note with some surprise that the most acid soils are the ones that show the greatest fraction of their exchangeable base cation capitals lost annually to leaching, but hastily add that neither weathering nor deep rooting are included in this assessment, and that it must not yet be considered as one in which we predict additional soil acidification. We will pursue the simulation modeling approach to forecasting potential changes in these (and other) ecosystems in the near future, being careful at all times to "keep our feet on the ground" with respect to the uncertainties in estimating some of these fluxes, especially weathering.

REFERENCE

Reuss, J. O. and D. W. Johnson. 1986. Acid Deposition and the Acidification of Soils and Waters. Springer Verlag, New York.

GAPON

$$\frac{E_{Al}\,(Ca^{2+})^{1/2}}{E_{Ca}\,(Al^{3+})^{1/3}} = Kgp$$

$$(Al^{3+}) = \frac{E_{Al}^{3}\,(Ca^{2+})^{3/2}}{E_{Ca}^{3}\ Kgp^{3}}$$

(Al^{3+}) increases as the 3/2 power of (Ca^{2+})

(Al^{3+}) decreases as the 3rd power of E_{Ca}

(Al^{3+}) increases as the 3rd power of E_{Al}

E = Exchange phase fraction
( ) = Solution phase activity

Fig. 1. Gapon equation for cation exchange in soils.

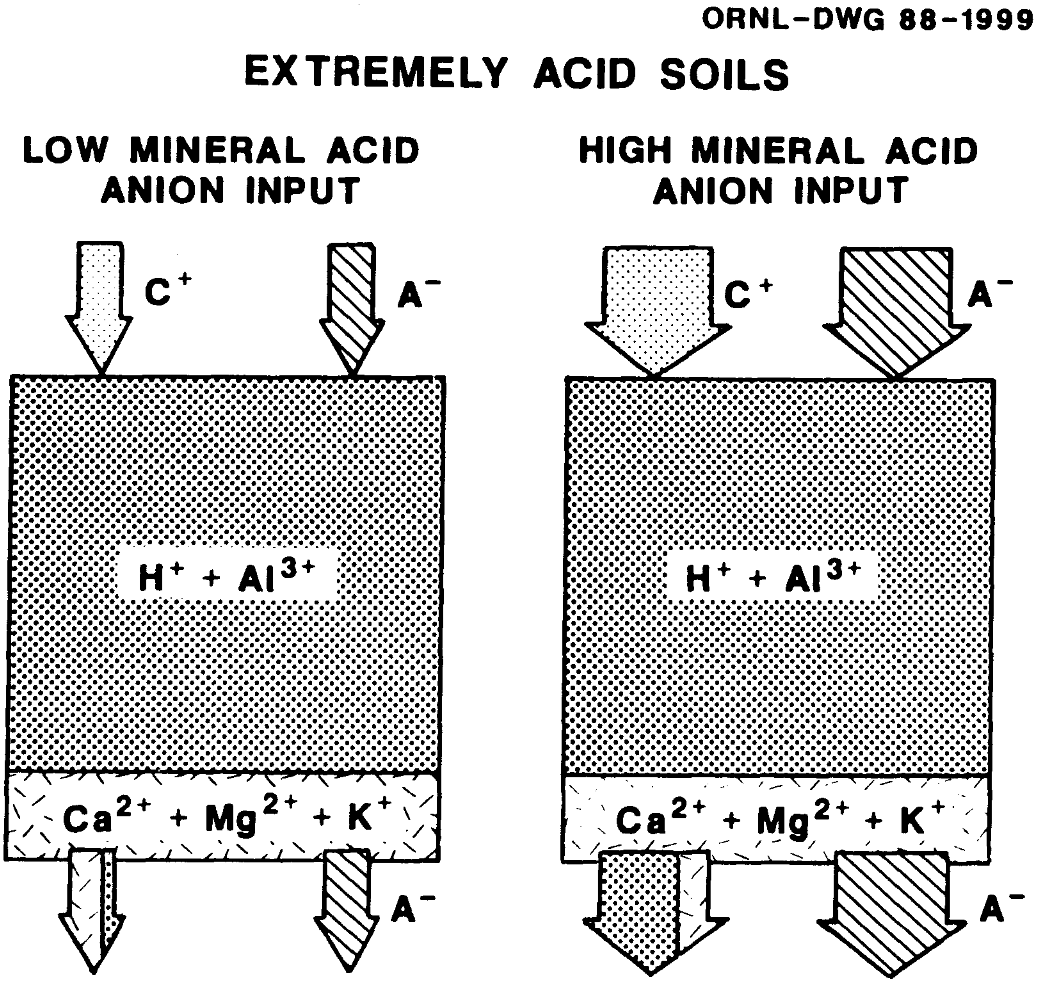

Fig. 2. Schematic diagram of the response of soil solution to low vs high mineral acid anion inputs to extremely acid soils.

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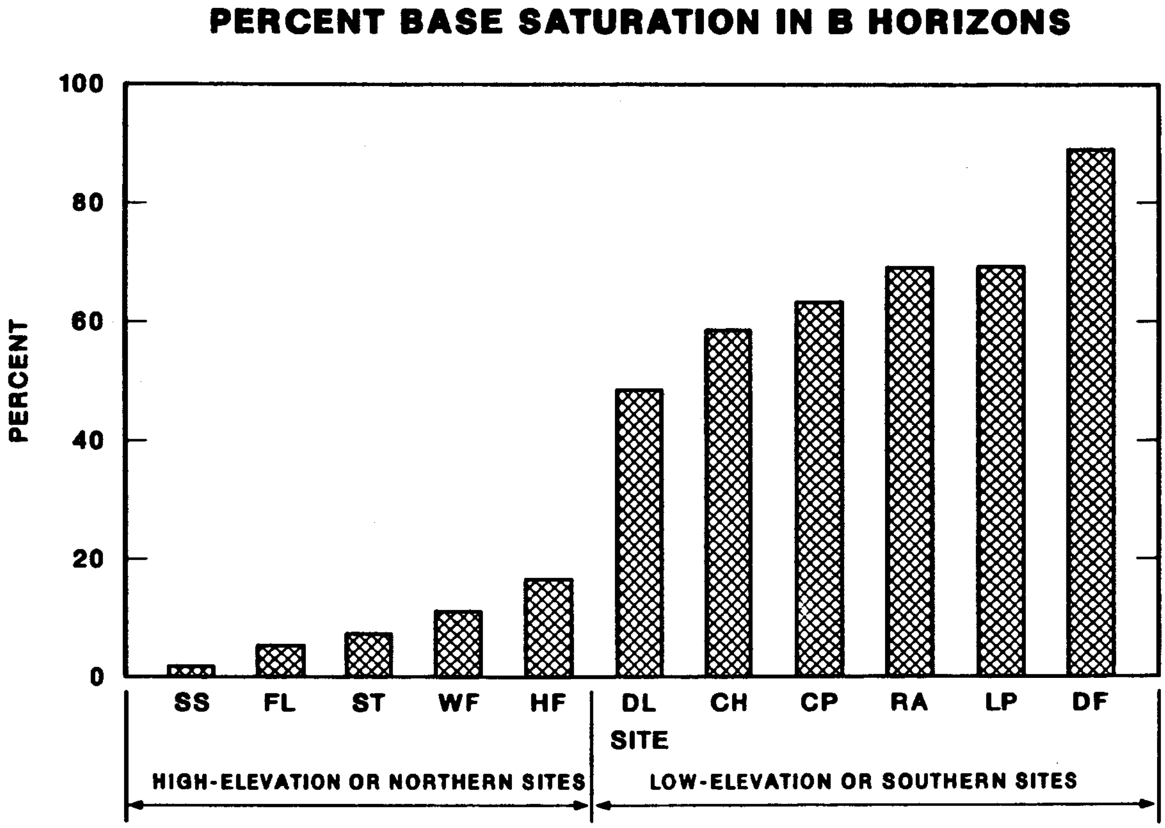

Fig. 3. Percent base saturation in soils from the IFS site (see legend at back for site abbreviations).

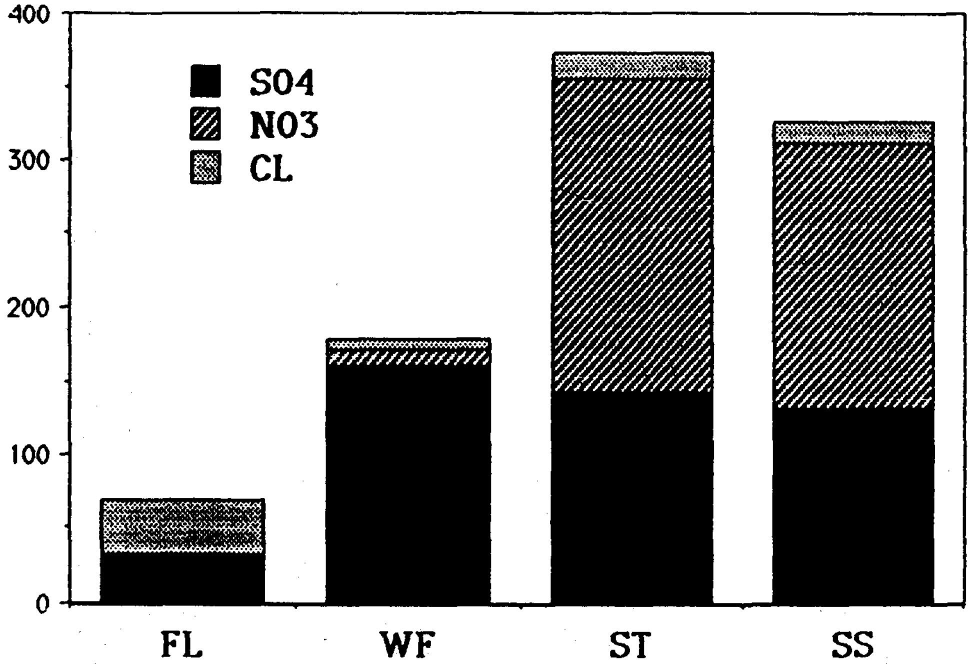

Fig. 4. SO_4^{2-}, NO_3^-, and Cl^- in soil solutions from the Findley Lake (FL), Whiteface (WF), Smokies Tower (ST), and Smokies Beckin (SB) sites.

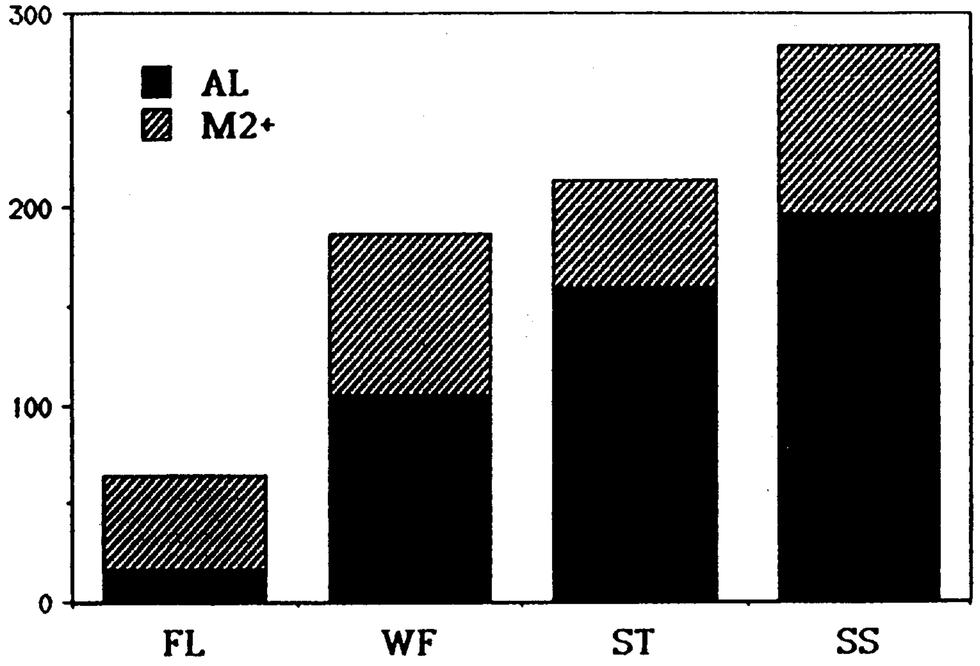

Fig. 5. Al^{3+} and the sum of Ca^{2+} plus Mg^{2+} (M^{2+}) in soil solutions from the Findley Lake (FL), Whiteface (WF), Smokies Tower (ST), and Smokies Becking (SB) sites.

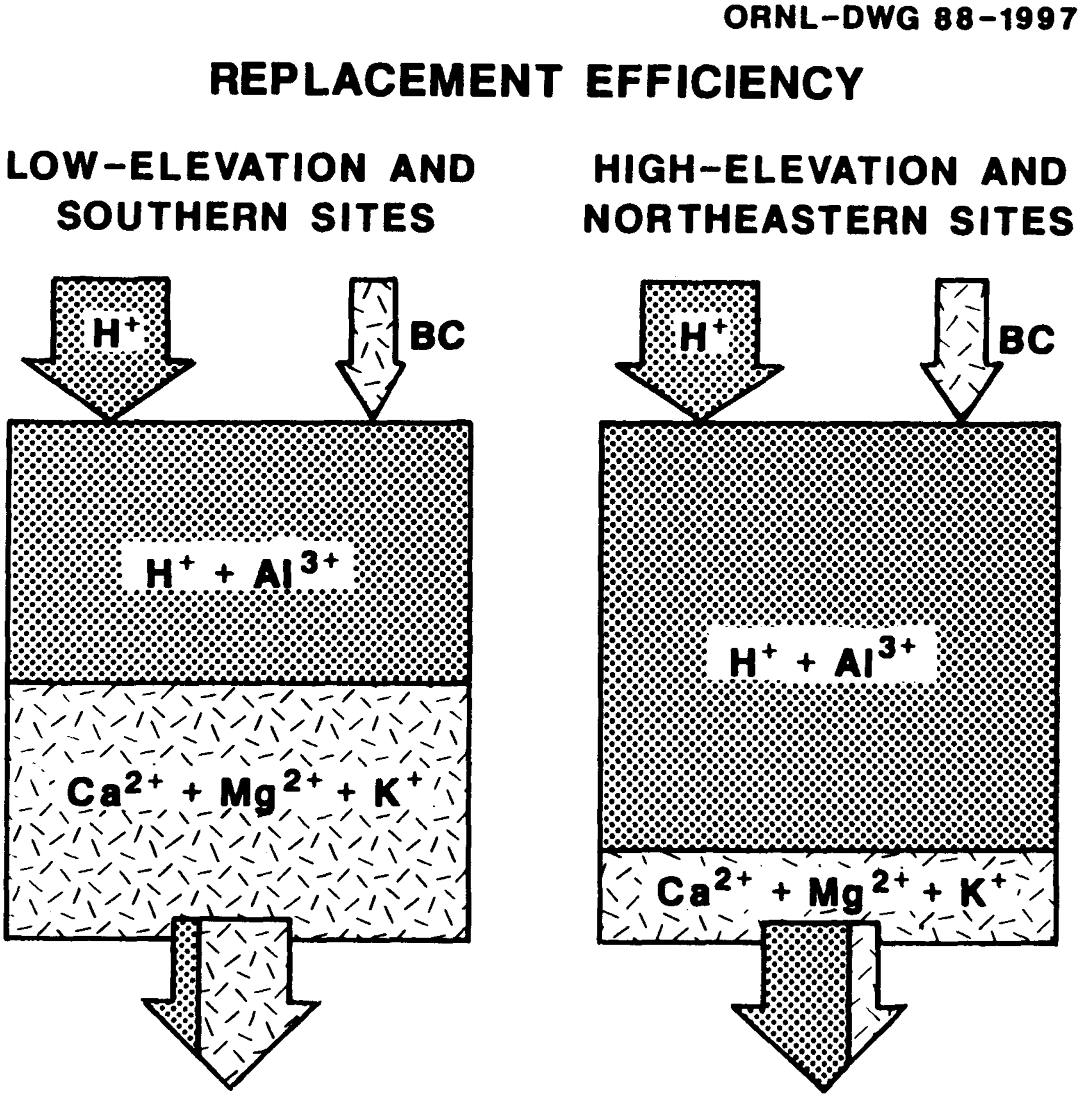

Fig. 6. Schematic diagram of the replacement efficiency of H$^+$ for base cations vs Al^{3+} in slightly acid (left) vs extremely acid soils.

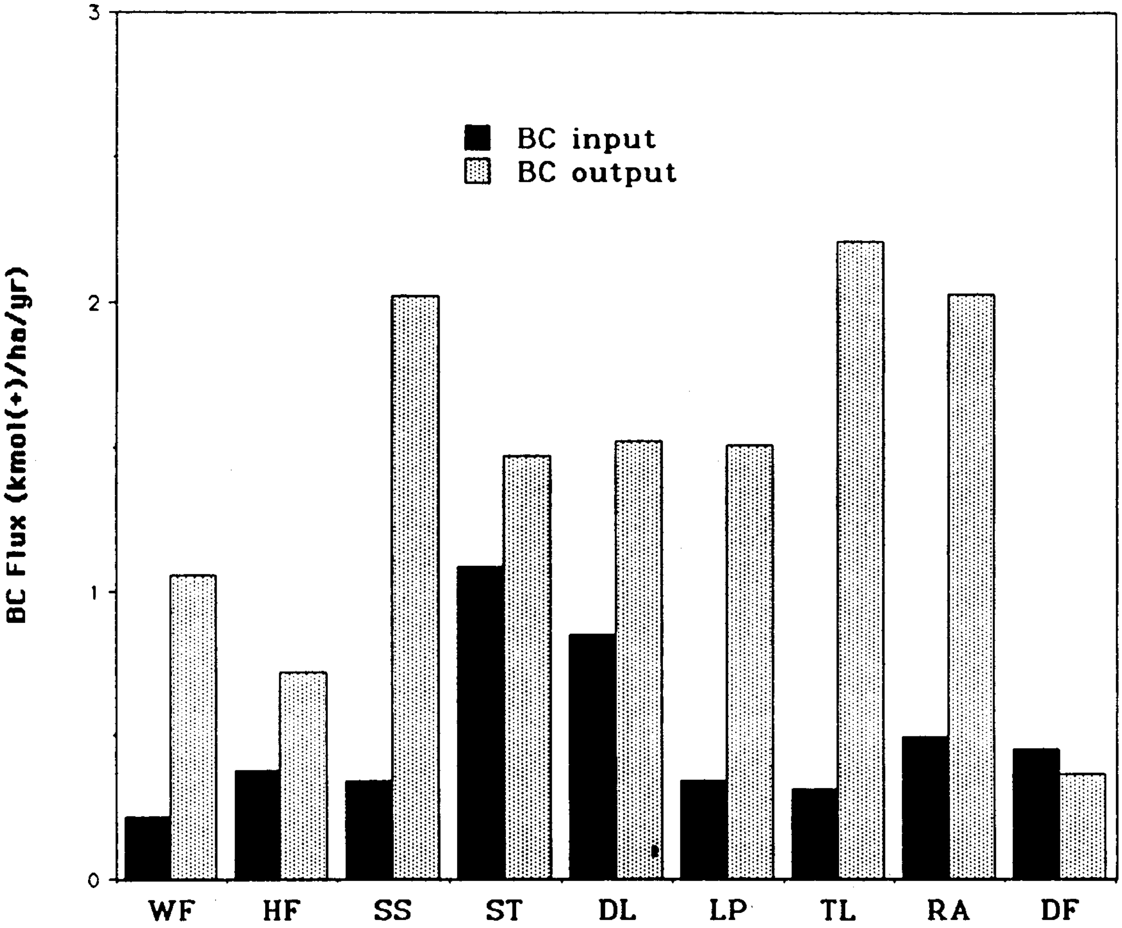

Fig. 7. Base cation inputs and leaching outputs in selected IFS sites.

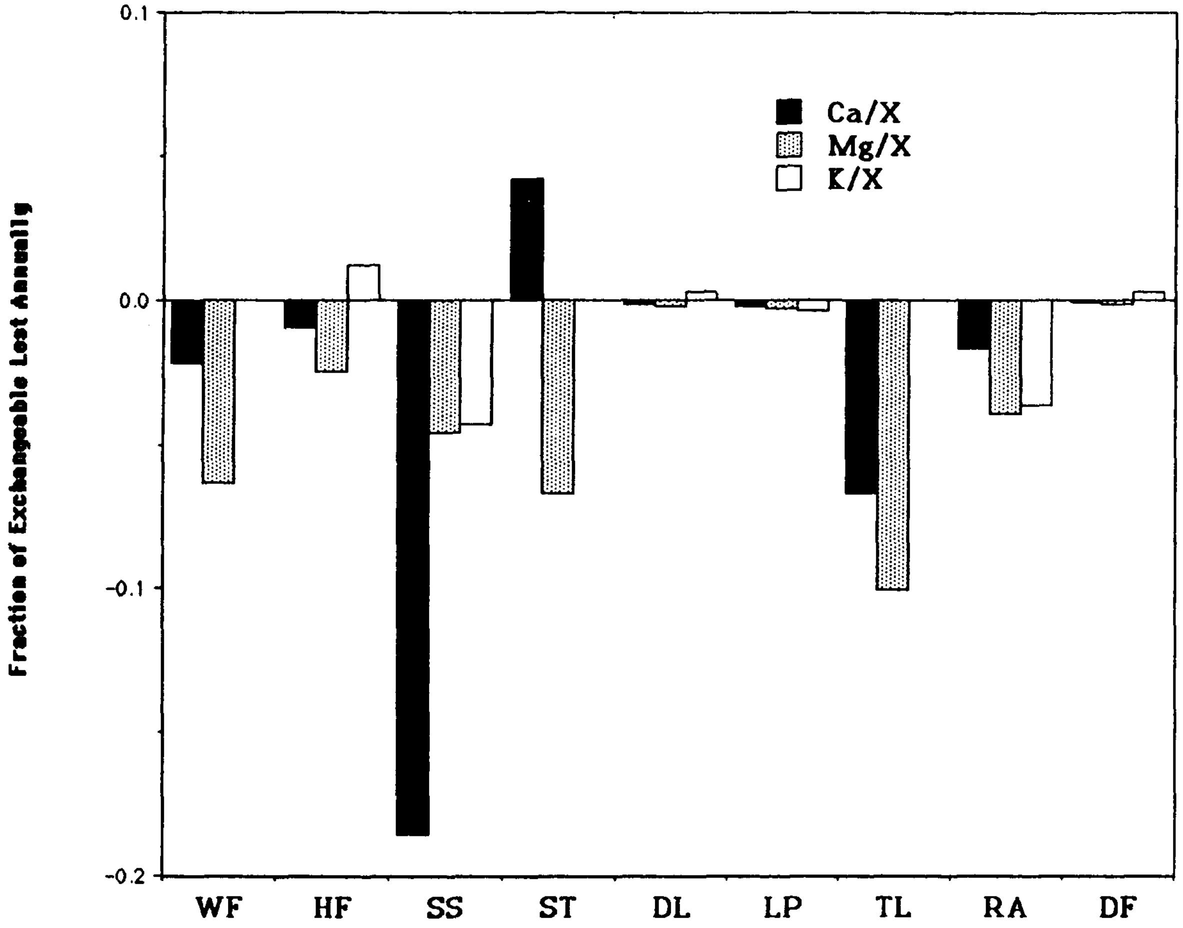

Fig. 8. Net export (leaching output minus atmospheric deposition) of Ca, Mg, and K as a fraction of exchangeable Ca, Mg, and K at selected IFS sites.

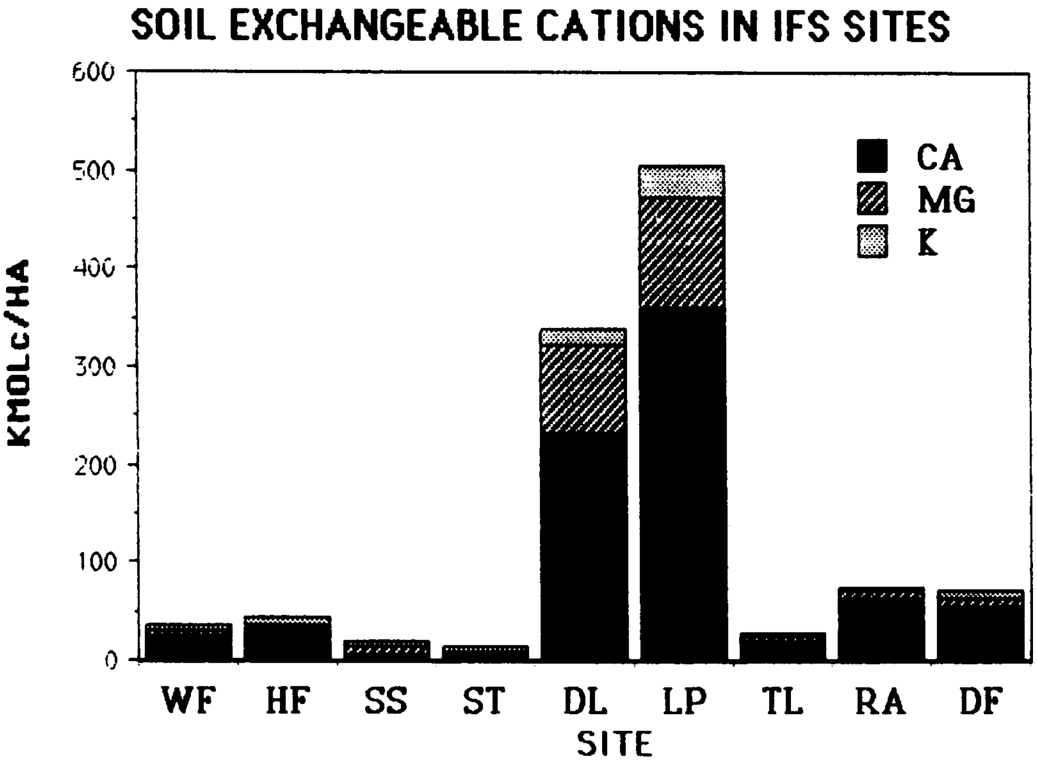

Fig. 9. Soil exchangeable cations in selected IFS sites.

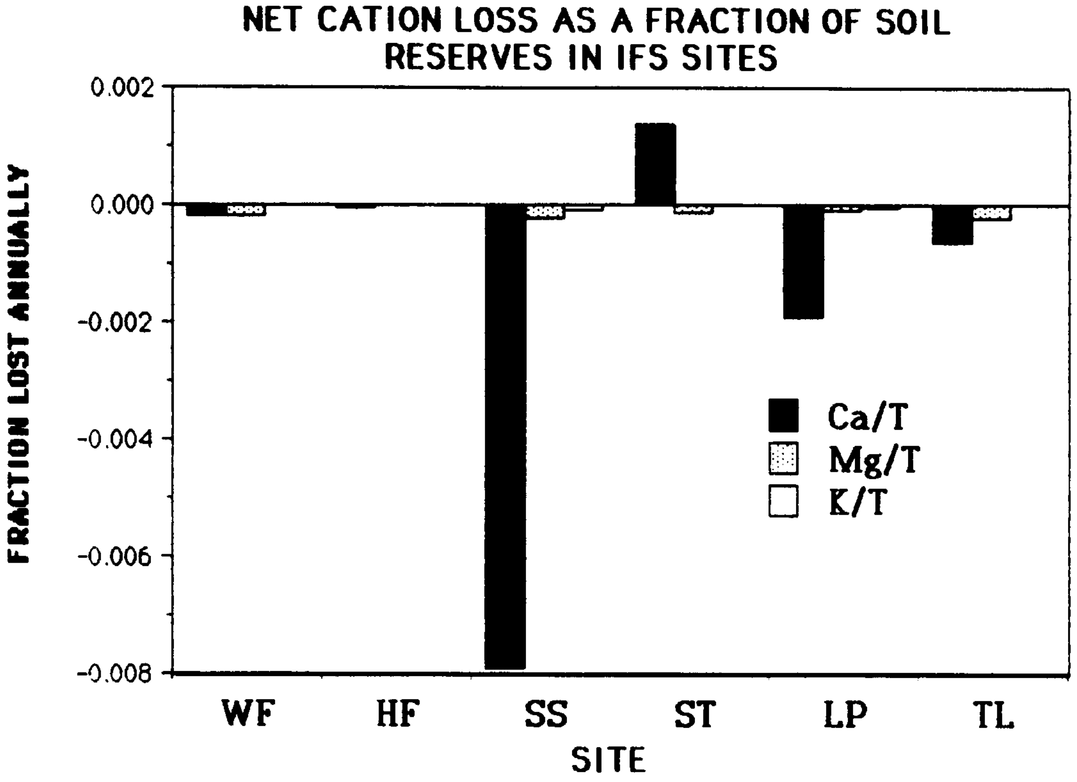

Fig. 10. Net export (leaching output minus atmospheric deposition) of Ca, Mg and K as a fraction of total Ca, Mg, and K at selected IFS sites.

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