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RECENT FIELD STUDIES OF DRY DEPOSITION TO SURFACES IN PLANT CANOPIES*

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ABSTRACT

A variety of field techniques were used to assess the dry deposition of sulfur. In a deciduous forest canopy in eastern Tennessee, inert petri plates and adjacent chestnut oak leaves showed similar $SO_4^{=}$ deposition velocities of about 0.1 cm s^{-1}. In the same forest, statistical analysis of throughfall yielded a deposition velocity of 0.48 cm s^{-1} for total sulfur ($SO_4^{=}$ plus SO_2). The throughfall technique appears useful for scaling individual surface measurements to larger spatial and temporal scales. On a grassy field in Illinois, flat Teflon plates, petri dishes, and dustfall buckets were exposed side by side. Measured sulfate deposition increased with increasing rim height on the collection surface, and deposition velocities ranged from 0.14 to 0.70 cm s^{-1}. Much of the deposition to these surfaces can be attributed to large-particle $SO_4^{=}$. Dry season (summer) deposition velocities of ^{7}Be in California were found to be similar to dry deposition velocities of ^{212}Pb in Tennessee, ranging from 0.18 to 0.35 cm s^{-1}. These natural radionuclides attach to submicron aerosols in the atmosphere and may be useful tracers of submicron $SO_4^{=}$ deposition.

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*(See reference 18 in attached list for place of publication).

Introduction

The need to quantify dry deposition is widely recognized. However, despite considerable research, most commonly used methods have several disadvantages.[1] Development of new methods suitable for use in complex vegetation canopies and compatible with particles in a wide range of sizes is a major research need. We present here some aspects of several recent field studies on particle dry deposition to various surfaces, with emphasis on their application to sulfur deposition. The different methods used have been applied to a number of atmospheric constituents and are described in detail in the literature.[2-6]

Methodologies and Results

Deposition to Leaves and Surrogate Surfaces

Inert (surrogate) surfaces have proved successful for estimating the dry deposition flux of some particles; however, these data must be compared with deposition to foliage because of the difficulty of simulating plant surfaces. The traditionally used dustfall buckets suffer from unpredictable aerodynamic and surface effects and are subject to contamination by local soil dust. We have minimized these problems by using inert surfaces of different design, by deploying the surfaces in the actual plant canopy of interest, and by analyzing adjacent foliage for comparison.

Our surrogate surface research has involved a limited number of short-term comparisons of particle fluxes to rimless Teflon plates (13.3-cm-diam) and polycarbonate petri dishes (9.5-cm-diam) with 1-cm rims situated 1 m above a 30-cm high grass canopy in an open field in Illinois. During September 1981, we completed five comparison experiments during several-day dry periods in which deposition was also collected by others using standard dryfall buckets (26-cm-diam) with 26-cm rims and 1-m^2 flat Teflon sheets.[7] During June 1982, we completed an additional five comparisons concurrent with micrometeorological measurements made by others at the same site.[3]

More extensive experiments were performed in a deciduous forest canopy at the Walker Branch Watershed in eastern Tennessee. During the period 7/81 to 6/83 we completed 24 experiments using the petri dish deposition plates situated in and above a 20-m oak forest canopy (144 plates exposed).[4] For these experiments we designed a sampler that provided automatic protection of the samples from unexpected rainfall (Fig. 1,2).[2]

Research in the forest canopy also included direct comparison of particle fluxes to plates and leaves. We developed a method of foliar extraction that separates external surface from internal leaf pools, allowing an estimate of dry deposition using oak leaves (Quercus prinus) collected sequentially in the forest canopy (Fig. 3).[2] During the summers of 1981 and 1982 we completed eight experiments in which leaves and plates were exposed simultaneously side by side at four locations in and above the canopy.[4] These collections involved 5 to 8 replicate plates and 8 to 32 "replicate" leaves per experiment.

The results of all of these experiments are summarized in Table I as dry deposition rates of $SO_4^{=}$ to the surface of interest. In the first set of experiments in the grass field, the measured deposition rate increased from 26 to 60 to 190 $\mu g\ m^{-2}\ h^{-1}$ as the sampler rim height increased from 0 to 1 to 25 cm. Whether the influence of the rim is to increase turbulent mixing over the sampler (and hence particle transport into still areas of the rimmed collectors) or to decrease the bounce-off or resuspension of deposited particles from the collector surface is unknown. The effect may well be a combination of these factors as well as differences in chemical composition between collectors.[3]

The difference between the rimmed and rimless plates was less pronounced during the second set of experiments in the grass field; the petri dish deposition rates exceeded those to the Teflon plates by 37% compared with 130% during the initial experiments. We feel that this lower number is a more accurate reflection of the true difference because the samplers were exposed within 50 cm of each other during 6/82 (within 5 m during 9/81) and were collected, extracted, and analyzed by the same person during 6/82 but not during 9/81. Still, the influence of the rims is measurable. Airborne particle size distributions measured during 6/82 indicated that much of the $SO_4^{=}$ aerosol mass was associated with particles of 0.5-to 1μm diam [3]. However, using these data as input, a dry deposition model[3] suggests that larger particles may be responsible for most of the mass deposition of $SO_4^{=}$ onto these surfaces. Hence, inertial impaction and sedimentation of supermicron particles were more important than deposition of submicron material in influencing deposition onto plates at this site.

The mean deposition rates measured to petri dishes at the forested site are comparable with rates measured at the Illinois site during the growing season (Table I). However, during the dormant period in the oak forest, deposition rates of $SO_4^{=}$ are approximately two times greater. This may be due to a combination of higher winter wind speeds and an increased proportion of large particles in the overall $SO_4^{=}$ size distribution during this period.

The results of the paired leaf/plate experiments in the forest indicated good agreement in average deposition rates between surfaces, despite the complexity of such a comparison. The grand mean dry deposition rate of $SO_4^{=}$ to 53 plates was 33 $\mu g\ m^{-2}\ h^{-1}$, and that determined from extraction of 116 pairs of leaves collected before and after the same periods was 31 $\mu g\ m^{-2}\ h^{-1}$. The individual data from each experiment are plotted in Figure 4 for $SO_4^{=}$ and the other ions analyzed.[4] The degree of scatter in the data is apparent and reflects the uncertainty in the comparison of mean values. Despite the similarity in means, the values for $SO_4^{=}$ lie above a 1:1 line and also above the linear regression fit for all of the ions. As we have discussed,[4] individual deposition rates estimated from leaf extractions are subject to large uncertainty because of considerable variability in concentrations of ions extracted from leaf surfaces. For $SO_4^{=}$ only five of the eight comparison experiments are plotted in Figure 4. The remaining three leaf deposition values were not significantly

different from zero because of large standard errors of the surface concentrations at the beginning and end of each dry deposition period (note also that one of the five points for $SO_4^{=}$ falls outside of the scale in Figure 4).

Several factors contribute to the variability and to the difficulty in interpreting these results: the spatial variability of surface concentrations of all ions on leaves is considerably higher than that for replicate deposition plates; large internal pools of these ions in leaves can cause measurable interference with "surface" extractions because many of these ions are highly mobile in vegetation; biological uptake of dry deposition can result in underestimates of deposition to leaves; non-rain moisture runoff from leaves (e.g., dew, fog) will result in lower estimates of dry deposition of soluble ions compared with rimmed plates from which surface wetness will evaporate without removing soluble material; dry deposition of SO_2 to leaves may contribute $SO_4^{=}$ during leaf surface extractions, but is probably unimportant in the total $SO_4^{=}$ flux to the inert plates; and obvious differences in the surface morphology of leaves and plates will influence the particle capture characteristics of each surface, particularly for small particles.

In considering these effects and our results in detail,[4] we drew several conclusions. Sedimentation of large particles is a major process of transport of particle mass onto both leaf and inert surfaces in this forest canopy; hence, differences in surface morphologies which affect small particle deposition may not be strongly reflected in plate/leaf dry deposition comparisons. Examination of over 1000 particles by electron microscopy indicated similar shapes for deposited and suspended particles and yielded similar mass median diameters for deposited particles on leaves and plates (Fig. 5), but much lower values for suspended particles (3 to 5 μm for deposited particles, 0.5 μm for aerosols). This indicates that the largest particles in the air, those most subject to gravitational sedimentation, transfer mass more efficiently to both surfaces than do smaller particles. The generally higher deposition rates of $SO_4^{=}$ onto leaves relative to inert surfaces are partially attributable to higher deposition rates for SO_2 to leaves, some fraction of which is extractable as "surface bound" $SO_4^{=}$. Because of the complicating influences of vapor dry deposition (e.g., SO_2 and HNO_3), the relationship between particle dry deposition rates to leaves and inert plates is best represented by our data for Ca^{++} (Figure 4), which yield a mean plate/leaf deposition ratio of 1.7 ± 0.3 (SE).

Our foliar extraction method yields deposition rates for $SO_4^{=}$ and other ions which are comparable with values measured by adjacent inert surfaces.[2,4] These methods must be attempted for several ions and with other vegetation types to determine their applicability to general dry deposition studies. In situations where particle sedimentation dominates dry deposition flux, consistent relationships might be expected. These relationships and further investigations of deposition processes in the canopy will be useful in determining the distribution of deposited particles in complex canopies and will assist in the extrapolation of single-surface deposition measurements

to the entire canopy. One promising method for quantifying these relationships involves analysis of rain above and below the vegetation canopy as described in the following section.

Assessing Dry Deposition by Analysis of Throughfall

The water that reaches the ground beneath a forest canopy (including throughfall and stemflow, but primarily throughfall) contains solutes from three sources: (1) the wet deposition incident to the canopy, (2) dry deposition accumulated between rain events and washed off the canopy surfaces by the rainfall, and (3) substances leached from the plants and their associated microflora. Incident wet deposition is easy to measure and can be subtracted from throughfall deposition to calculate net throughfall deposition, which has components of dry deposition and canopy leaching. These two ion sources are difficult to quantify independently and thus difficult to separate.

We approached this problem statistically using measurements of throughfall chemistry collected on a single-event basis. Collection of throughfall after discrete dry period/rain event cycles permits accurate specification of (1) the duration of the dry period, which should be correlated with the amount of accumulated dry deposition, and (2) the duration and amount of rainfall, which should be correlated with the amount of canopy leaching. A multiple regression of net throughfall deposition against dry period duration and rainfall amount (or rain event duration) will then statistically separate the dry deposition and canopy leaching effects.

We performed this analysis on a data set of 64 throughfall collections from the period July 10, 1981, to May 8, 1983. The collections were made using two collectors under each of two tree crowns, a white oak (Quercus alba) and a chestnut oak (Quercus prinus), at the Walker Branch Watershed site. A complete analysis of the methods and results is presented elsewhere[5] and only the results for sulfate will be discussed here.

For all regressions, the intercept terms were not significantly different from zero, indicating that if the rainfall amount and dry period duration approach zero, then the expected net deposition of throughfall $SO_4^{=}$ approaches zero also (Table II). The chestnut oak shows a $SO_4^{=}$ dry deposition rate in the growing season that is higher than the rate in the dormant season. This is expected for deciduous trees, in which the canopy surface area is much reduced during the dormant season. The deposition rates shown for white oak have large standard errors, reflecting the variability introduced by higher rates of biological $SO_4^{=}$ leaching from that species.[5] The dry deposition rates for white oak are not significantly different from zero nor from those for the chestnut oak. The chestnut oak rates, which are less sensitive to the confounding effects of $SO_4^{=}$ leaching, appear to be better estimates of true $SO_4^{=}$ dry deposition rates to this mixed oak forest.[5]

The dry-deposited sulfate that accumulates on canopy surfaces can be deposited as $SO_4^{=}$ particles or as SO_2 gas, which can be oxidized to $SO_4^{=}$ on the leaf. Thus, Table II estimates the sum of these two components.

However, this estimate may be low for several reasons: (1) the canopy may be incompletely washed by some rain events, (2) some portion of the deposited sulfur may be absorbed by the plant, immobilized by microbes, or degassed as reduced sulfur compounds, (3) stemflow, which was not collected, may carry a significant portion (perhaps 10%) of the sulfur to the forest floor. Thus, the deposition rates shown in Table II must be considered minima.

The Use of ^{212}Pb and ^{7}Be to Evaluate Dry Deposition of Submicron Aerosols to Vegetation Canopies

One approach to evaluating the transfer of submicron aerosols to vegetation is to relate vegetation concentrations of selected radionuclides to their ambient air concentrations. If radionuclides are chosen carefully, they can be interpreted as having entered the vegetation system by the deposition of small atmospheric particles. Proper selection of sampling conditions assures that dry deposition is the process being reflected. Because ^{212}Pb (derived from ^{220}Rn) has a finite life, its presence on a vegetation surface represents a finite deposition history. For the average ^{212}Pb atom this is less than a day. Because of the short half-life of ^{212}Pb, the decay constant, λ_r, can be assumed to be equal to the inverse of the mean residence time of the radionuclide on a surface. A biomass-normalized deposition velocity, V_D, can thus be derived:

$$V_D = \lambda_r \, CR_{v,a} \qquad (1)$$

where λ_r is the radionuclide decay constant (1.8×10^{-5} s^{-1}) and $CR_{v,a}$ is the vegetation-to-air concentration ratio derived from C_v, the concentration of the radionuclide on vegetation (pCi kg^{-1}) and C_a, the air concentration (pCi m^{-3}) averaged over at least three half-lives and preferably five. The resulting V_D (m^3 kg^{-1} s^{-1}) represents the effective air volume being depleted of aerosols by 1 kg of vegetation each second. Multiplying V_D by the biomass density (kg m^{-2} ground area) results in the deposition velocity, V_d (in units of m s^{-1}). Such measurements on a variety of vegetation were made in Tennessee in 1983.[6]

The uniformity in ^{7}Be air concentrations expected on a regional basis suggested that sampling of summer vegetation in various locations in California (dry summer climate) could be used to empirically evaluate the transfer of submicron aerosols in all stages of aggregation.[6] The combined processes of dry deposition, growth dilution, radiodecay, etc., would be reflected in the observed vegetation concentrations because the vegetation would have been continuously exposed to a relatively constant ^{7}Be air concentration. For these measurements, ^{7}Be air concentrations at Lawrence Livermore National Laboratory, Livermore, California, were used as generic values.[8]

Unlike the ^{212}Pb measurements, V_D for ^{7}Be can only be estimated given information on λ_e, the effective rate for the decay, growth dilution, and weathering of ^{7}Be from vegetation. This effective loss rate is defined as

$$\lambda_e = \lambda_r + \lambda_w \quad , \qquad (2)$$

where λ_r is the radioactive decay constant for ^{7}Be (1.5×10^{-7} s^{-1}) and λ_w is an effective first-order rate constant representing all other processes affecting the removal of ^{7}Be from vegetation with time. Although λ_w for ^{7}Be has not been determined in a dry climate, literature values of λ_w do not differ substantially for different chemical or physical forms, regardless of climate. Therefore, λ_e was derived from a range of λ_w values summarized in the literature[9] with the median value for growing herbaceous vegetation being 4.7×10^{-7} s^{-1}. This constant is substituted for λ_r in Equation (1).

Natural radioactivity on vegetation and air filters was measured using a high-resolution photon detector and a computer-based data reduction system. Air was sampled using a high-volume system (1.8 m^3 min^{-1}) and 20- x 25-cm glass fiber filters. Vegetation was either measured fresh (^{212}Pb) or after dry ashing at 450°C (^{7}Be).[6]

Because both ^{212}Pb ($t_{1/2}$ 10.64 h) and ^{7}Be ($t_{1/2}$ 53.3 d) decay appreciably over the time periods employed in evaluating deposition, certain assumptions are made in calculating deposition rates. For the results discussed here, it was assumed that the deposition rates did not change on a diurnal basis (^{212}Pb) or over the growing season (^{7}Be). Furthermore, ^{7}Be air concentrations for the individual months of July, August, and September 1983 at Livermore, California, were averaged, after decay-weighting each month's value, to provide a comparable time period for Oak Ridge, Tennessee, data.[6] Similarly, ^{212}Pb air concentrations were calculated assuming that a constant deposition rate to the filter occurred. This is not what actually occurs because of diurnal variations in ground-level concentrations, but as with ^{7}Be, vegetation will be exposed to the same variations as the air filter. What is important, therefore, is that the period of air sampling be representative of the vegetation exposure conditions.

Dry deposition velocities (V_D and V_d) for the California ^{7}Be data and the Tennessee ^{212}Pb measurements are presented in Table III. The ground-area deposition velocities (V_d) for the ^{212}Pb data were calculated from Equation (1) by assuming a biomass density, Y, of 0.4 kg m^{-2} (leaves only a representative value for local deciduous forests but one that would not apply to all measured species; hence,

$$V_d = CR_{v,a} \lambda_r Y \quad . \tag{3}$$

The V_d calculation for the ^{7}Be data is more complicated because of the longer exposure of vegetation to ^{7}Be; thus, the average exposure time, T, as well as the average biomass density, Y (assumed to be 0.4 kg m^{-2} unless measured), over the exposure period must be considered. Therefore:

$$V_d = C_{v,a} Y \lambda_e (1 - e^{-\lambda T}) \quad . \tag{4}$$

Because T is long with respect to the reciprocal value of λ_e, errors in its estimation will not affect the estimate of V_d greatly. These calculations, summarized in Table III and in Equation (1), demonstrate that ^{212}Pb and ^{7}Be are depositing with atmospheric aerosols at very similar rates.

By taking advantage of the fact that radionuclides such as ^{212}Pb and ^{7}Be are subject to the condensation and coagulation forces acting on other atmospheric nuclei, it is possible to estimate deposition rates of natural submicron aerosols to complex vegetation canopies. The results summarized here and elsewhere[6] indicate that deposition velocities on the order of 0.2 cm s^{-1} are probable when vegetation densities of 0.4 kg m^{-2} are considered. Other factors such as wind speed, humidity, and leaf surface structure and size might exert effects that are not identified here. As the measurement technique develops, a further understanding of their importance should emerge.

Discussion

A useful parameter in many dry-deposition field studies and in the modeling of deposition flux is the deposition velocity discussed above. This is the flux to a surface divided by the concentration in the air ($V_d = F\ C^{-1}$, units in cm s^{-1}). This is normally determined from the air concentration at some reference height measured concurrently with the flux. For most of our experiments, this calculation is straightforward: air concentrations of particle-borne $SO_4^{=}$ were measured in Illinois at a 1-m height using standard filtration methods;[3,7] in Tennessee $SO_4^{=}$ air concentrations were determined at 1 m above the forest canopy (20 m above the ground) by similar methods.[4] All of these samples were collected concurrently with the deposition measurements. However, because of long averaging times (throughfall method) or a lack of local air measurements (^{7}Be), the deposition velocities for these methods were determined using mean reference concentrations as discussed earlier for the radionuclides. For the throughfall method, we divided the mean deposition flux by our measured mean aerosol $SO_4^{=}$ plus SO_2 concentration for the growing season (8.1 µg of S m^{-3}).

These data are summarized in Table IV for measurements taken during the growing season at each site. Overall the deposition velocities range over a factor of 7, from 0.10 to 0.70. At Illinois, the values during 9/81 reflect the influence of rim height discussed above, increasing from approximately 0.15 cm s^{-1} for rimless samplers to 0.70 cm s^{-1} for the bucket. The deposition velocities during 9/81 are 20 to 30% lower than during 6/82, a period characterized by lower air concentrations and somewhat higher wind speeds. The petri dish values for the oak canopy in Tennessee are approximately three times lower than for these same surfaces in the grass field in Illinois. This may reflect a somewhat higher transfer efficiency to the samplers near ground level in the field because they are within the soil-particle resuspension layer. A higher proportion of efficiently deposited large-particle $SO_4^{=}$ would be expected at 1 m above the ground in an agricultural area than at 20 m above the ground in a forest.

As discussed above, the data from the throughfall method are difficult to interpret because of SO_2 deposition and the potential biological uptake of dry deposited S in the canopy. Without further data, this deposition velocity (0.48 cm s^{-1}) must be considered as an average value representing the effective transfer of dry-deposited sulfur from the atmosphere to the

forest floor, keeping in mind that some irreversible absorption of sulfur probably occurs in the canopy itself. The ratio of the deposition velocity for the whole canopy (throughfall technique) to that for individual leaves is 4.4, somewhat less than the estimated leaf area index of this canopy (5 to 7). This could indicate that not all surfaces in the canopy are receiving equal sulfur deposition, as would be expected if atmospheric sulfur concentrations decrease within the canopy. This comparison of techniques holds some promise for scaling individual surface measurements to whole canopies.

Although the data are limited, the radionuclide values provide a very interesting comparison with the Teflon plate and petri dish surrogate surface values. Considering all of the data together, the deposition velocities of $SO_4^{=}$ to these individual surrogate surfaces (primarily deposited as large particles) range from 0.10 to 0.35 cm s^{-1} and the values for the deposition of radionuclides to foliage range from 0.18 to 0.35. Extrapolation of any of these values to the full canopy is very difficult, as discussed earlier. However, to the extent that the behavior of ^{7}Be and ^{212}Pb can be considered to reflect the behavior of submicron $SO_4^{=}$ particles, these data suggest deposition velocities for large-particle and submicron $SO_4^{=}$ are of the same order of magnitude. Preliminary reports of deposition velocities measured to the grass canopy specifically for submicron $SO_4^{=}$ by micrometeorological methods during the 6/82 Illinois experiment revealed values also in the range 0.1 to 0.4 cm s^{-1} (results presented at AGU Spring Meeting, Baltimore, May 1983).

Our data indicate that novel approaches to measurement of large-particle, submicron, and total sulfur flux using inert and foliage surface analyses give reasonable results and deserve further attention. Because different sources and transport processes affect large and small particles, one would not necessarily expect deposition velocities inferred from large-particle measurement methods (surrogate surface) and small-particle methods (radionuclide analysis, micrometeorological measurements) to agree. These various methods should be viewed as complementary, contributing data on different size classes as they influence total dry deposition to the surface of interest. The crucial point of focus should thus be on the extent of overlap of the size classes measured by the different methods.[10]

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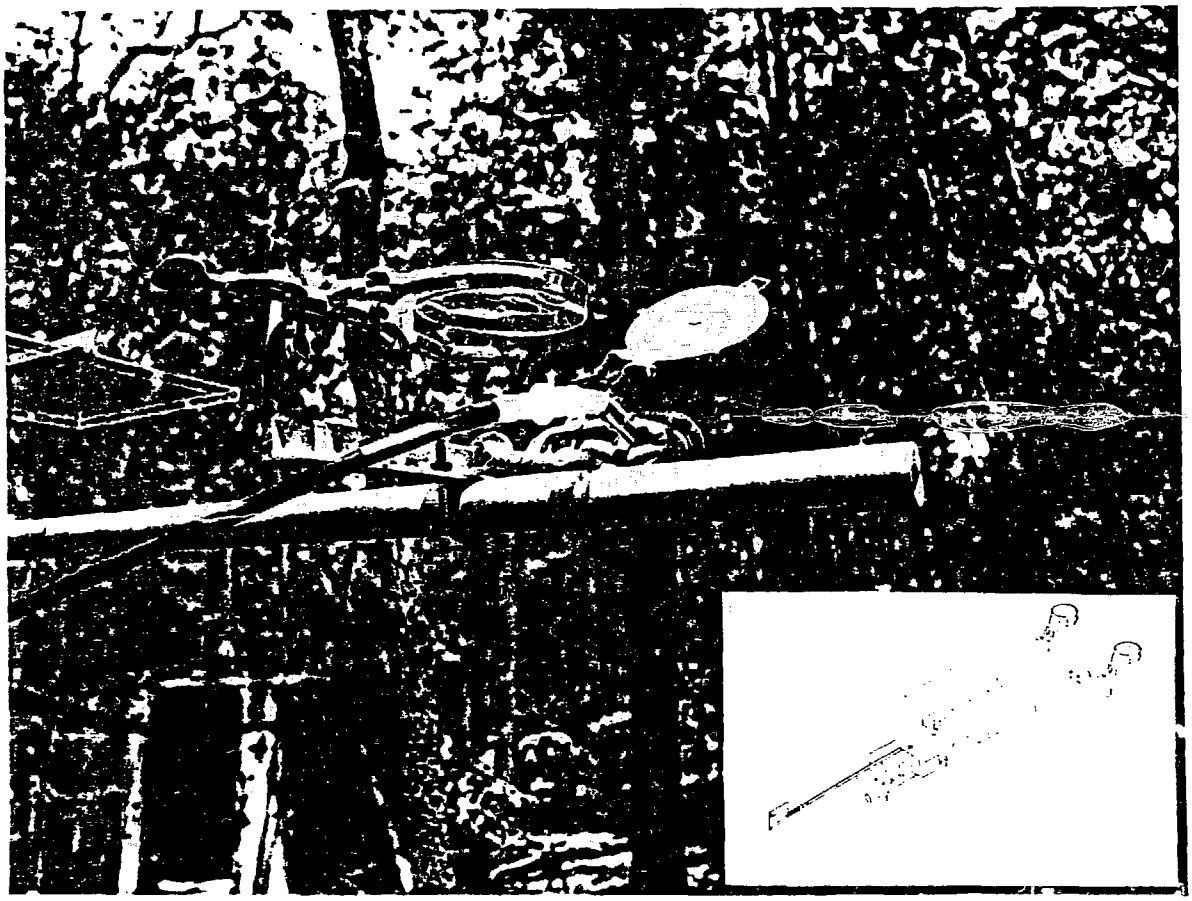

Figure 1. Apparatus for exposure of deposition plates in forest canopy with automatic protection from rain. Three devices are held in the collector, a polycarbonate petri dish, a Whatman Filter, and a total aerosol filter (3 L per min. flow rate).

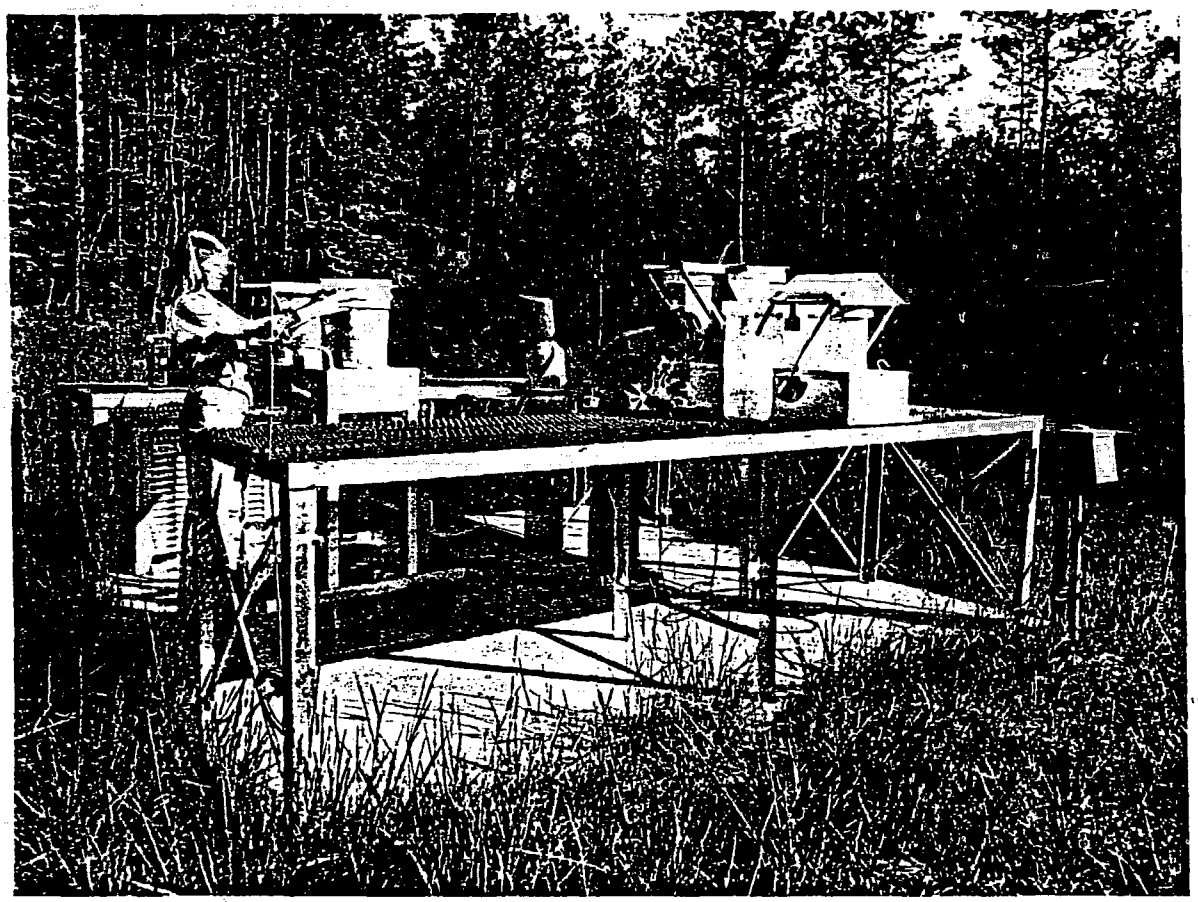

Figure 2. Apparatus for collection of precipitation as wetfall-only, shown for comparison with dry collectors. These devices are situated in forest clearings, as illustrated, and beneath different canopy types. Wetness sensing grids are used to automatically expose sealed collectors to wet deposition.

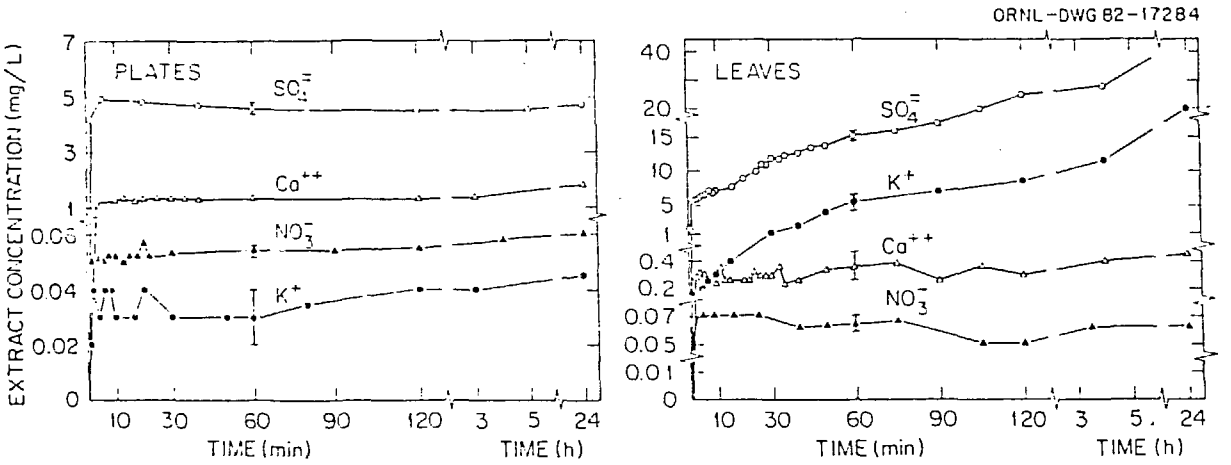

Figure 3. Dissolution kinetics of several ions from particles deposited on inert plates and on leaves. In practice, all leaves are extracted for 3 minutes to maximize removal of external deposits and minimize removal of internal material. Inert plates are extracted for 30 minutes, although 5 minutes would be sufficient to remove essentially all soluble ions.

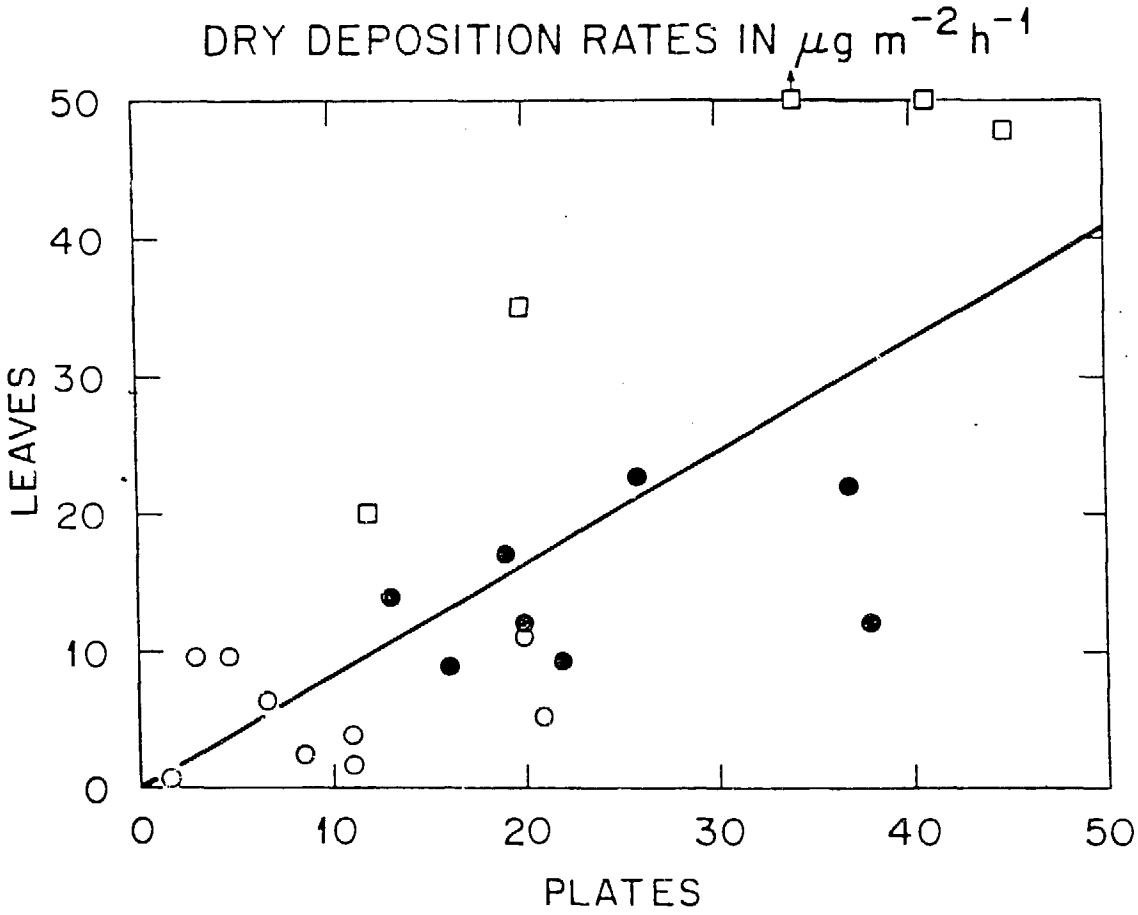

Figure 4. Comparison of dry deposition rates measured in a forest by foliage extraction (Y-axis) and petri dish deposition plates (X-axis). Squares represent data for $SO_4^{=}$, solid circles for Ca^{++}, and open circles for K^{+} and NO_3^{-}. The line is a linear regression fit to all data (n = 22) for which the $R^2 = 0.42$ ($P<0.01$) The point for $SO_4^{=}$ that falls off scale is (88,33) and has been included in the analysis.

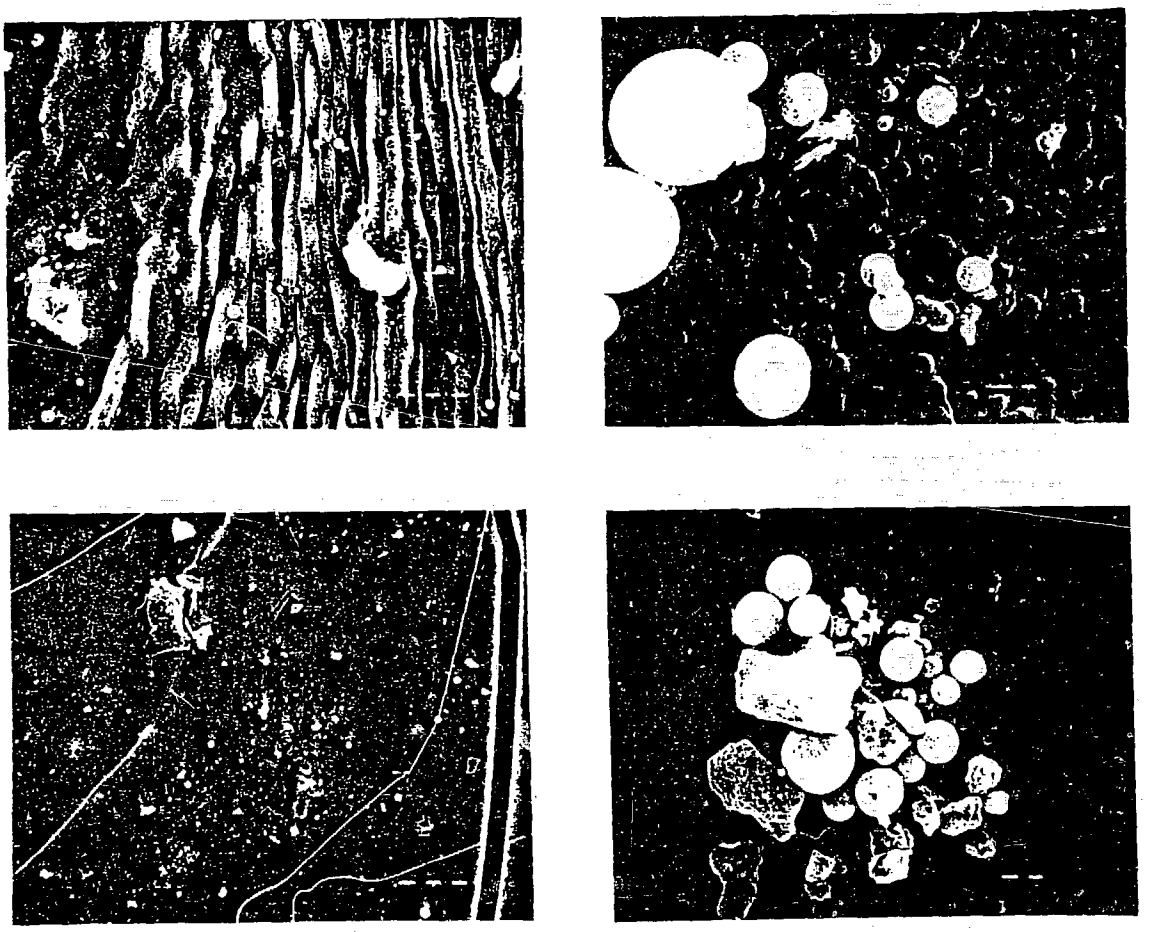

Figure 5. Scanning electron photomicrographs of particles deposited on chestnut oak (Quercus prinus) leaf (upper two photographs) and inert deposition plate (lower two photographs) exposed in the upper canopy. Scale line furthest to the left in the lower right-hand corner of each picture is as follows: 10 μm on both left-hand pictures and 1.0 μm on both right-hand pictures.

Table I. Measured dry deposition rates for $SO_4^{=}$ to inert and foliar surfaces in grass and forest canopies

Canopy Type	Period	Surface Type	Number of Experiments[a]	Dry Deposition Rate ($\mu g\ m^{-2}\ h^{-1}$)	
				Mean	Standard Error
Mixed grass	9/81	Teflon plate[b]	5	26	6.0
	9/81	Teflon sheet[b]	5	27	6.0
	9/81	Petri dish[b]	5	60	7.2
	9/81	Dustfall bucket[b]	5	190	29
	6/82	Teflon plate[c]	5	27	9.0
	6/82	Petri dish[c]	5	37	7.7
Oak forest	GS 1981-1983[f]	Petri dish[d]	24	44	2.7
	DS 1981-1983[f]	Petri dish[e]	8	74	7.3
	GS 1981-1982	Petri dish[d]	8	33	2.1
	GS 1981-1982	Oak leaves[d]	8	31	8.0

[a] Surrogate surface experiments involved 2 to 4 replicates each, leaf experiments generally 16.

[b] Reference 7.

[c] Reference 3.

[d] Reference 4.

[e] Unpublished data.

[f] GS = forest growing season of approximately 4/1 to 10/31. DS = dormant season of approximately 11/1 to 3/31.

Table II. Results of regression analysis of net throughfall deposition of $SO_4^{=}$

Canopy Type	Period[a]	N[b]	Dry Deposition Rate ($\mu g\ m^{-2}\ h^{-1}$)[c]	SE[d]	(R^2)[e]
Chestnut oak	GS	21	210	56	0.80
	DS	18	54	24	0.64
White oak	GS	12	120	130	0.86
	DS	13	120	210	0.63

[a]GS = forest growing season of approximately 4/1 to 10/31. DS = dormant season of approximately 11/1 to 3/31.

[b]N = number of storm event samples analyzed.

[c]The dry deposition rate is the regression coefficient for the "dry period duration" term.

[d]The SE is the standard error of the regression coefficient.

[e]The R^2 is the coefficient of determination of the multiple regression.

Table III. Estimation of biomass-normalized (V_D) and traditional (V_d) dry deposition velocities from measured values of ^{7}Be and ^{212}Pb vegetation-to-air concentration ratios ($CR_{v,a}$) and estimated values of λ_e, Y, and T (defined in the text)

Vegetation Type	$CR_{v,a}$ ($m^3\ kg^{-1}$)	λe (d^{-1})	$\overline{Y}$ ($kg\ m^{-2}$)	$\overline{T}$ (d)	$V_D(x10^3)$ ($m^3\ kg^{-1}\ s^{-1}$)	V_d ($cm\ s^{-1}$)
California (^{7}Be)						
Sycamore	9000	0.054	0.400	90	5.7	0.23
Maple	4600	0.054	0.400	90	2.9	0.12
Fescue	6300	0.054	0.384[a]	45	4.3	0.16
Fescue	7700	0.054	0.848[a]	45	5.2	0.44
Tennessee (^{212}Pb)[b]						
Sycamore	272	1.56[c]	400	--	4.9	0.20
Red oak	205	1.56	400	--	3.7	0.14
Yellow poplar	272	1.56	400	--	4.9	0.20
Black locust	516	1.56	400	--	9.3	0.37
Cottonwood	278	1.56	400	--	5.0	0.20
Honey locust	422	1.56	400	--	7.6	0.30
Black willow	261	1.56	400	--	4.7	0.19
Weeping willow	194	1.56	400	--	3.5	0.14
Fescue	489	1.56	400	--	8.8	0.35
White pine	150	1.56[c]	400	--	2.7	0.11
Loblolly pine	250	1.56	400	--	4.5	0.18
Hemlock	278	1.56	400	--	5.0	0.20
Red cedar	417	1.56	400	--	7.5	0.30

[a] Measured.

[b] Average of all measurements of species.[6]

[c] λ_r.

Table IV. Dry deposition velocities (V_d) for sulfur and small particles determined from several field experiments using different methods[a]

Canopy	Method	Period	Deposition Surface	Atmospheric Species	V_d (cm s^{-1}) N	Mean	SE
Grass	Inert surface[3,7]	9/81	Teflon sheet	$SO_4^=$	5	0.16	0.05
			Teflon plate	$SO_4^=$	5	0.14	0.04
			Petri dish	$SO_4^=$	5	0.30	0.06
			Dustfall bucket	$SO_4^=$	5	0.70	0.18
		6/82	Teflon plate	$SO_4^=$	5	0.22	0.05
			Petri dish	$SO_4^=$	5	0.35	0.07
Forest	Inert surface[4]	GS[b]	Petri dish	$SO_4^=$	8	0.10	0.03
	Leaf extraction[4]	GS	Oak leaves	$SO_4^=$	8	0.11	0.06
Forest	Throughfall[5]	GS	Chestnut oak canopy	total S	21	0.48	0.13
Trees	Radionuclide[6]	7/83-9/83	Deciduous foliage	^{212}Pb	8	0.22	0.03
			Coniferous foliage	^{212}Pb	4	0.20	0.04
Grass	Radionuclide[6]	7/83-9/83	Deciduous foliage	^{212}Pb	8	0.22	0.03
			Foliage	^{212}Pb	1	0.35	----
Trees	Radionuclide[6]	7/83-9/83	Deciduous foliage	^{7}Be	2	0.18	----
Grass	Radionuclide[6]	7/83-9/83	Foliage	^{7}Be	2	0.30	----

[a]Values are given as measured to the surface listed (see text); no attempt has been made to extrapolate all of the values to full vegetation canopies.

[b]GS = 1981 to 1982 forest growing season.

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