EFFECTS OF SLUDGE APPLICATIONS ON SOIL WATER AND VEGETATION IN A NORTHERN HARDWOOD FOREST IN NEW ENGLAND

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ABSTRACT

Dewatered and limed sludge obtained from a primary treatment plant was spread at two rates, 25 and 125 wet t/ha, on sandy loam soils in a northern hardwood forest in New Hampshire. Changes in soil water chemistry were assessed using water samples collected at 20 and 45 cm depths with suction lysimeters. Changes in the understory vegetation were monitored using species diversity and frequency of individual species while changes in the overstory were based on the basal area growth of trees > 5 cm in diameter.

The light application of 25 t/ha caused only minor changes in soil water chemistry, but the heavy application of 125 t/ha caused increases in most of the major ions in soil water. At 20 cm, mean Cl concentrations rose to a maximum of 27.5 mg/l from background values of < 0.4 mg/l. SO_4 initially declined 1.0 mg/l below background concentrations averaging 5.0 mg/l, and then rose to a maximum of 13.5 mg/l. Ca, which averaged about 2.0 mg/l before the heavy application, also rose to about 13.0 mg/l. Mg, Na, K, H, and NO_3 showed maximum increases in concentration that averaged 2- to 3-fold greater than background values of < 0.5 mg/l. At 45 cm, the increases in ionic concentrations following the heavy application were less than one half the increase that occurred at 20 cm. The concentration of most ions returned to background values within one year after the sludge was applied.

The background chemistry of soil water at 45 cm was similar to streamwater chemistry in the study area. Thus, changes that occurred in soil water chemistry at this depth after sludge applications may

reflect potential impacts on streams. Despite the temporary enrichment of soil water caused by sludge applications, there were minor changes in the diversity and frequency of herbs and shrubs in the understory. And there was no significant increase in the basal area growth of trees.

INTRODUCTION AND OBJECTIVES

Within the last decade, a considerable effort has been made to evaluate the feasibility of using forest lands as potential sites for recycling nutrients in sludges and effluents (Sopper and Kardos, 1973; Sopper, 1977). However, in New England, where forests occupy almost 80% of the land surface, there is little information on the environmental effects of spreading sewage sludge in forests. This is unfortunate for in addition to their abundance, forests possess other attributes that make them attractive sites for sludge disposal. For example, the diversity in forest phenology results in an extended growing season and period of nutrient uptake. Also, the extensive rooting networks of forest communities suggest optimum occupancy of most soil horizons and maximum opportunity for nutrient uptake. Thus, the application of sludge in forests may provide a means of recycling the nutrients in sludge while possibly increasing forest productivity. And it could provide New England with a better alternative than the present common practice of using landfills as disposal sites where leaching to water supplies may be a serious problem.

Our study was designed to evaluate the environmental impact of sludge applications on a northern hardwood forest in New England. The specific objectives of our study were:

(1) Determine changes in soil water and vegetation during the 2-year period after sludge was applied at two different rates to the floor of a northern hardwood forest.

(2) Use available literature and the data we collected on soil water chemistry to determine factors affecting the fate of nutrients in applied sludge.

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(3) Determine the potential impact of sludge applications on chemistry of stream water.

(4) Develop guidelines and recommendations for the application of sludge to forests.

MATERIALS AND METHODS

The Hubbard Brook Experimental Forest located in the White
Mountain National Forest in New Hampshire was chosen as the study area
(Fig. 1). The 15 years of intensive research on the geology, hydrology,
vegetation, soils and nutrient cycling of this forest (Likens et al.,
1977) provided valuable information in interpreting our study results.
The specific site for our study was on a north-facing slope of 5 to
15 percent.

The soils at the site are a mixture of haplorthods and fragiorthods
and have a sandy loam texture. The haplorthodic soils are Peru and
Colton while the fragiorthodic soil is a Marlow. The soils are moderate
to well drained, at least 50 cm in depth to the fragipan when it occurs,
and are covered with an organic layer up to 20 cm in depth.

The understory vegetation consists of a variety of herbs, shrubs,
and tree seedlings and has been described by Siccama et al. (1970).
Major tree species include sugar maple (Acer saccharum F.), yellow birch
(Betula alleghaniensis Marsh), white ash (Fraxinus americana F.) and
aspen (Populus tremuloides F.). The tree stratum is uneven-aged, 50 to
70 years old, and typical of other low elevation forests described for
Hubbard Brook (Whittaker et al., 1974; Bormann et al., 1970).

Six 8 x 12 m plots were established along a contour at the study
site with adjacent plots separated by at least a 2 m buffer strip. Six
tube-type suction lysimeters were placed in each plot; three at a depth
of 20 cm, or just below the major organic horizons, and the other three
at 45 cm, or just above the discontinuous fragipan. Hasan and Harris
(1975) provide recommendations for using this type of lysimeter. In
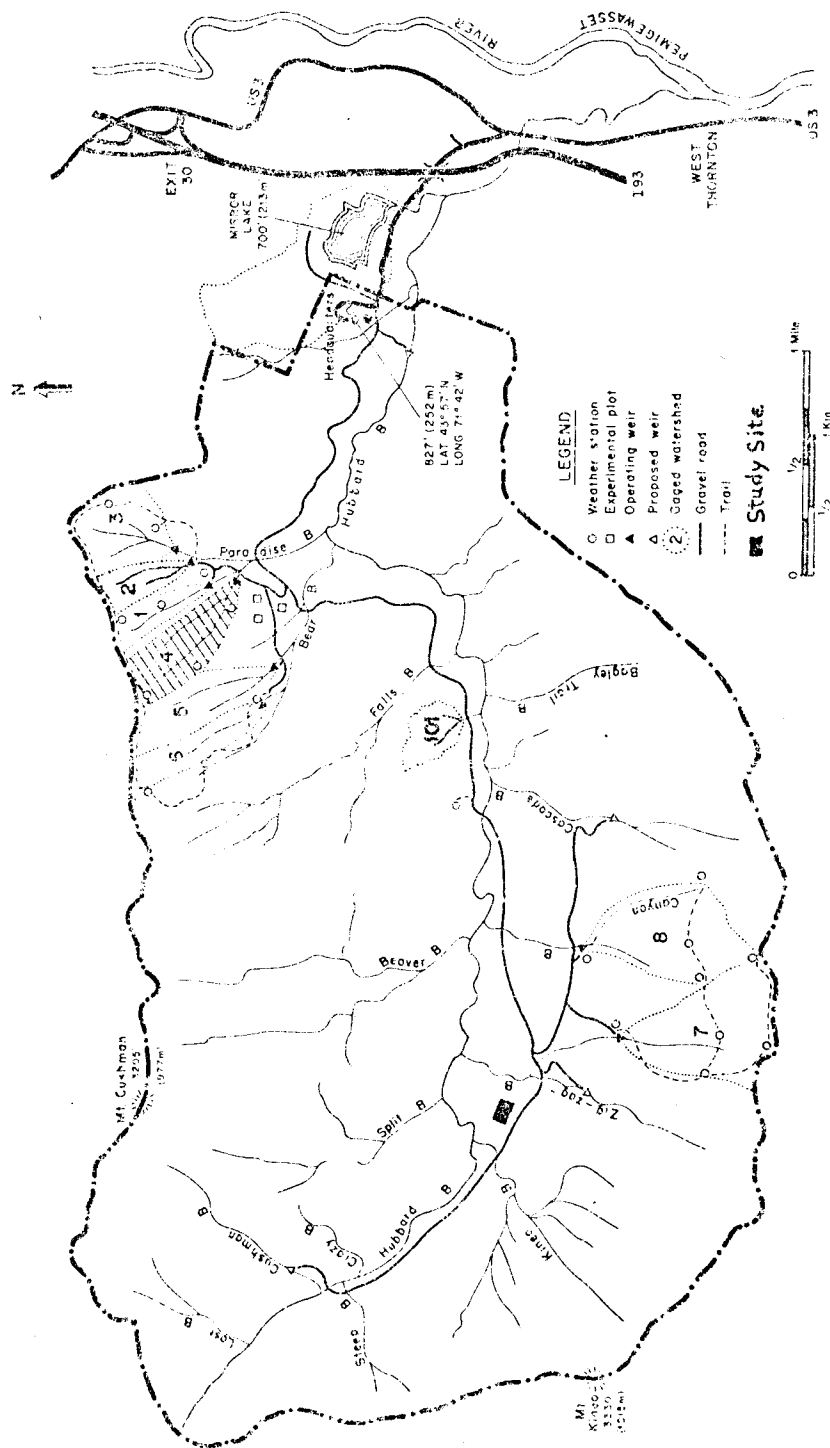
Figure 1.   The Hubbard Brook Experimental Forest and study site.

our study, the lysimeters were installed by removing a soil core to the desired depth. The lower portion of the core was mixed with water to form a slurry which was then poured into the hole. The lysimeter was inserted and slowly rotated to force the slurry up and around the porous sampling cup. The remaining core material was repacked around the lysimeter tube according to the depth and type of soil horizon.

The lysimeters were installed in July, 1974, about one year before the sludge was applied, to allow for a recession of soil disturbance effects associated with lysimeter installation. Soil water samples were collected at 1 to 2 week intervals from July, 1974 to November, 1976, except when air temperatures were below freezing. An initial vacuum of -86 kPa (-86 centibars) was used on each lysimeter. In addition to soil water samples, streamwater samples, precipitation, and air temperature data were taken in instrumented watershed #7 adjacent to the study site (Fig. 1). This baseline data was used to aid in interpreting study results.

Changes in the understory vegetation were monitored by recording the diversity and frequency of plant species occurring on two 1-m^2 subplots located in each of the 6 major plots. Diversity and frequency counts were made at two month intervals during each growing season from 1974 to 1976. Changes in the tree growth were monitored by computing changes in basal area for each plot based on dendrometer band measurements from trees > 5 cm diameter at breast height.

Sludge was applied to four of the six plots in late June, 1975. Using random selection, two plots received 25 wet or 5.8 dry t/ha, two plots received 125 wet or 28.0 dry t/ha, and two plots were kept as controls.

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The sludge was obtained from the primary treatment plant at Plymouth, N. H. (pop. 6,000). At the plant, the sludge is limed for pathogenic control and flocculated with ferric chloride before being dewatered using a vacuum coil filter. The odor-free sludge, in the form of a fibrous, fragmented mat, is about 22% solids by weight.

Sludge applications on selected plots were completed by a six person crew within 3 hours after the sludge was obtained at the Plymouth plant. During this time, the sludge was transported to Hubbard Brook, a distance of 25 km, weighed out in 225 kg portions, and spread easily and uniformly on the forest floor using hand tools and portable scaffolding. Nutrient loadings for the two application rates are given in Table 1 (Data provided by B. V. Salotto, EPA, Cincinnai).

Ca, Mg, Na, and K in water samples were determined by atomic absorption. NH_4, Cl, NO_3, and SO_4 were determined by automated, colorimetric analyses. TOTAL-P was determined by initial persulfate digestion (Menzel and Corwin, 1965), conversion of PO_4 to a phosphomolybdenum-blue complex (Murphy and Riley, 1962), and colorimetric analysis of the complex after it was extracted into butyl acetate. Conductance and soil water pH measurements were made according to standard methods (American Public Health Association, 1971).

The mean values for the two treatments and control were plotted for each ion in soil water for each soil depth on each collection date. The impacts of sludge treatments on soil water chemistry were based on comparisons of this plotted data. No statistical analyses were made because treatment effects on soil water chemistry were either readily apparent or were so small as to be ecologically and environmentally unimportant. Standard errors are given with many of the values presented in this paper as a reference for sample variability.

Table 1. Field application rates for various sludge nutrients based on the dry weight of sludge applied. Dry weight ratio of two rates was about 4.8 while wet weight ratio was 5.0 due to variability in solids content of sludge when applied.

| | Application Rate | |
Nutrient	Heavy	Light
	Kg/ha	
Ca	1674	347
$Fe_{III)}$	202	42
Mg	41.9	8.7
Na	14.0	2.9
K	9.1	1.9
$N_{(total\ Kjeldahl)}$	477	98.8
NH_3	57.4	11.9
SO_4	209	43.4
Cl	79.5	16.5
$P_{(total)}$	55.8	11.6
$P_{(H_2O\ Extraction)}$	5.3	1.1

RESULTS AND DISCUSSION

Soil Water Chemistry

Soil disturbances associated with lysimeter installation in July, 1974, caused increases in the concentration of most ions in soil water (Figs. 2-6). These disturbance effects gradually subsided during summer and autumn of 1974, and were completely gone by the spring of 1975. Consequently sludge was applied on June 20, 1975.

Specific Conductance. Only the heavy application caused increases in the specific conductance of soil water (Fig. 2). At the 20 cm depth, the average increase was as high as 5-fold, from background values of 20 $\pm$ 1 to a maximum of 108 $\pm$ 7 μmhos/cm by September, 1975. At 45 cm, the average conductance rose to 55 $\pm$ 4 μmhos/cm, one half the 20 cm maximum. Specific conductance then declined at each soil depth and averaged only 4 to 5 μmhos/cm above control values during the 1976 growing season.

Specific conductance is usually a good indicator of ionized substances in water. Values for the control plots and the 1975 pretreatment period reflect the low levels of cations and anions found in streams flowing from undisturbed forest areas in Hubbard Brook (Likens et al., 1977). The rise and decline in conductance in the treated plots reflect the leaching of sludge nutrients as will be shown by the similarity in the pattern of change for specific conductance and some of the ions discussed below.

Chloride. Before sludge applications, Cl concentrations averaged < 0.4 mg/l (Fig. 3). Cl concentrations began to increase in the treated plots soon after the sludge was applied. The heavy application caused mean Cl concentrations to reach 27.5 $\pm$ 2.9 mg/l, over a 90-fold increase, by

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SPECIFIC CONDUCTANCE

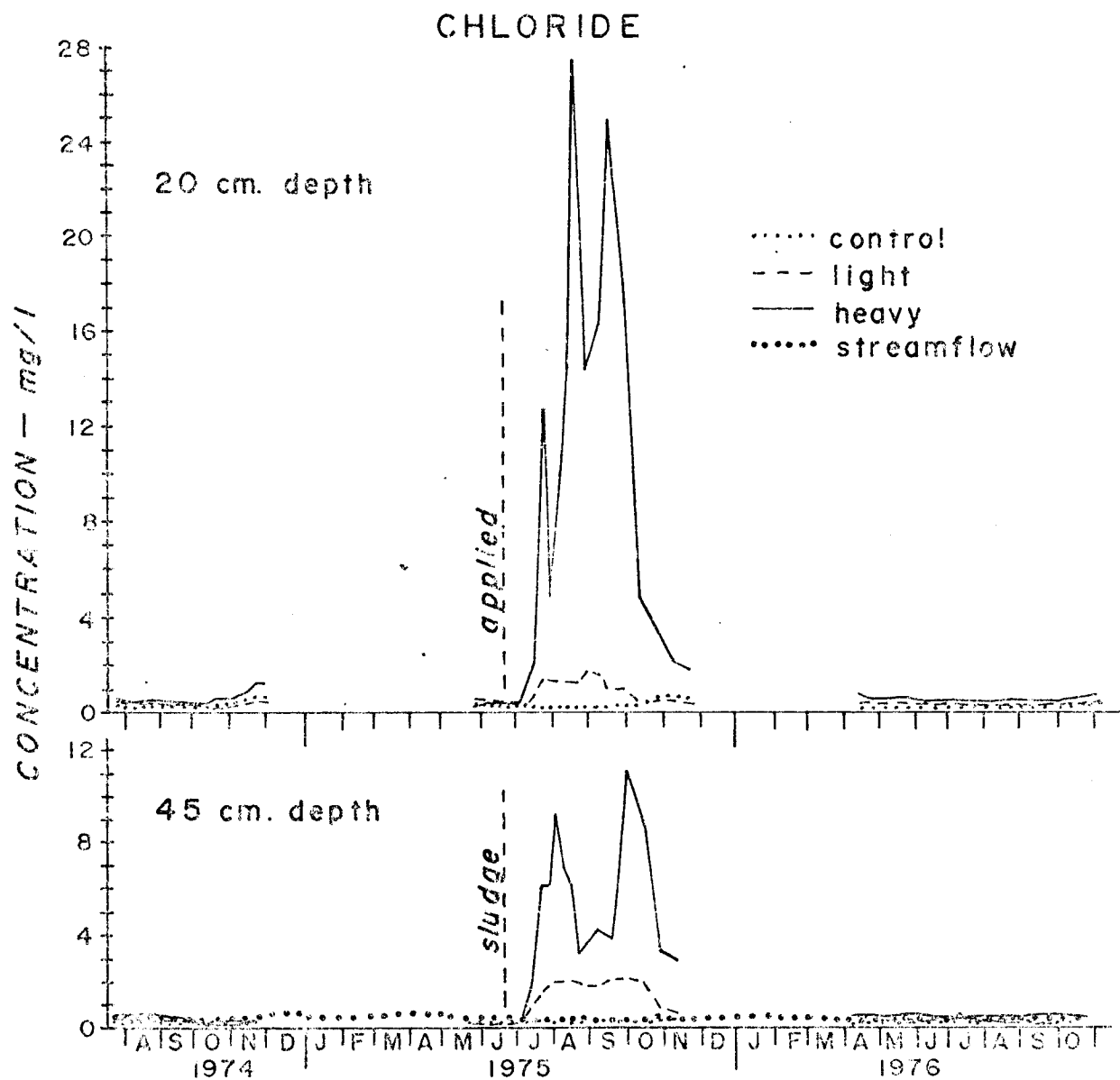Figure 2. Mean conductance in soil water and stream.

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Figure 3. Mean concentrations for Cl in soil water and stream.

August, 1975. At 45 cm, Cl rose to a mean maximum of 11.0 $\pm$ 1.0 mg/l or about one-half the maximum increase of 20 cm. The light application of sludge caused less than 6-fold increases in the average concentration of this ion at either depth. Within 9 months after the sludge was applied, Cl values in the treated and control plots were again similar.

Juang and Johnson (1967) have shown that Cl is not readily incorporated in the nutrient cycle of a northern hardwood forest. Thus, the changes in the concentration of this ion in the treated plots probably result from seasonal changes in the magnitude and movement of soil moisture. For example, the major increase in Cl in the treated plots corresponds to a period when soil moisture declines, particularly in the upper soil horizons, as a result of high evapotranspiration. This seasonal decrease in soil moisture could have resulted in the concentration of Cl in the upper soil horizons during the summer and early autumn of 1975. The major decline in Cl that occurred later in the 1975 growing season corresponds to the cessation of transpiration resulting in soil moisture recharge and possible dilution and leaching of Cl from the treated plots.

Dilution also probably accounts for the decline in Cl concentrations between soil depths. Consistently higher volumes of soil water were collected from the lysimeters at 45 cm suggesting that higher soil moisture in the lower soil horizons provided greater opportunities for dilution. Later it will be shown that seasonal changes in soil moisture may also be influential in controlling the concentration and leaching of some of the other sludge nutrients.

Sulfate. Background concentrations of SO_4 averaged about 4.5 $\pm$ 1.0 mg/l making it the dominant anion in soil water. Light applications of sludge caused only minor changes in SO_4, while the heavy application resulted in

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an initial decline and later rise in the concentration of this ion (Fig. 4). SO_4 declined to about 3.5 $\pm$ 0.3 mg/l at both soil depths within a few months after the heavy application. But in August of 1975, SO_4 at 20 cm began to rise and by November, 1975, reached a maximum of 13.5 $\pm$ 2.0 mg/l, or about three times the background values. During the 1976 growing season, SO_4 averaged 2 to 3 mg/l above control values at 20 cm. At 45 cm, SO_4 did not increase much above background by November, 1975, but it was similar in concentration to SO_4 at 20 cm during the summer and autumn of 1976.

Unlike Cl, the treatment effects on SO_4 do not appear to be entirely due to seasonal fluctuations in soil moisture. It may be that sludge applications caused changes in the biological and chemical processes governing the cycling of this nutrient (Likens et al., 1977). Increased microbial populations in the treated plots in the summer and autumn of 1975 could have temporarily immobilized SO_4 and thus reduced its concentration in soil water. Within a month after the sludge was applied, large mats of fungal hyphae covered the sludge. And although there was no significant increase in the number of fecal coliforms, the natural populations of soil coliforms also increased in the treated plots. This increase was greatest in plots receiving the heavy application and amounted to over a 100-fold rise in usual background counts of 100-1000/gm of soil. Microbial populations declined rapidly in the treated plots and had almost returned to background values during late autumn, 1975. This may have resulted in a decline in immobilization of SO_4 and could explain why the concentration increased in soil water during the late autumn of 1975 and remained elevated during the 1976 growing season. The continued occurrence of elevated SO_4 concentrations during the 1976 growing season indicates
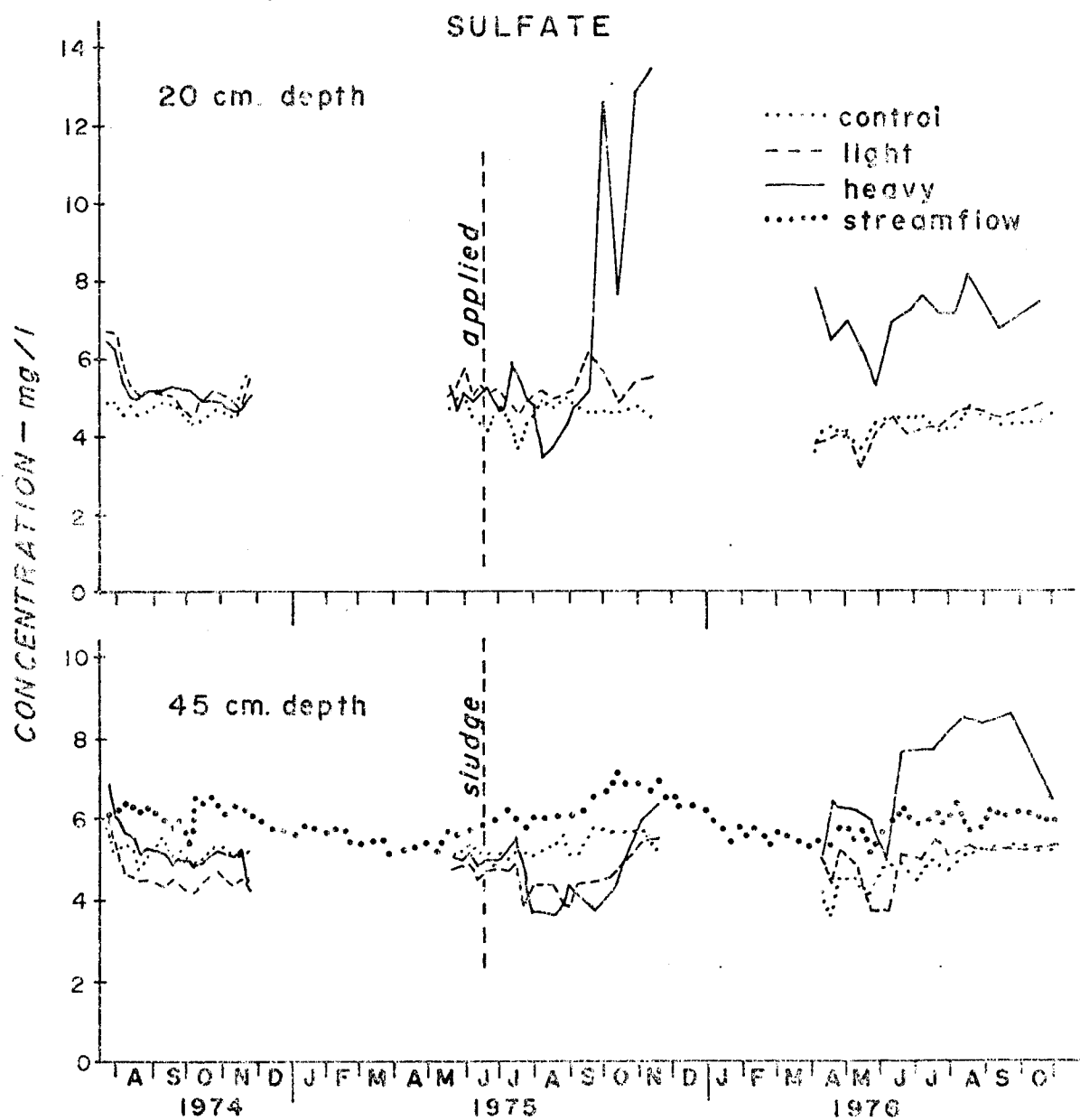
Figure 4. Mean concentrations for SO_4 in soil water and stream.

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that leaching of SO_4 and some of the other ions will persist at low concentrations for at least two or more growing seasons after sludge application.

The pattern of decline and rise in SO_4 may also be the result of changes in the adsorption capacity of this ion in the forest soil. The addition of large amounts of iron and calcium in the sludge (Table 1) may have enhanced SO_4 adsorption during the dry summer months in 1975 (Barrow, 1972; Chao et al., 1963, 1964) and resulted in a decline of SO_4 in soil water. Later the adsorbed sulfate could have been slowly released with a simultaneous decline in the adsorption capacity of the soil due to increased movement of water through the soil profile during autumn recharge (Barrow, 1968; Harward and Reisenhauer, 1966).

Phosphorus. This nutrient is of special interest because of its low availability in northern hardwood forests (Gosz et al., 1976). The amount of TOTAL-P in soil water was low, averaging about $0.8 \pm 0.1 \mu g/l$ in the plots before sludge applications. There was no detectable change in the average concentration of this nutrient after sludge applications. There are several potential reasons for low leaching of P including the insolubility of P in the sludge (Hunter, 1974; Hsu, 1973), increased microbial activity in the sludge treated plots, and, in particular, the high fixation capacity of Hubbard Brook soils for this nutrient (Gosz et al., 1976).

Nitrogen. Sludge applications also had relatively little effect on NH_4 or NO_3, the two major forms of N in soil water. NH_4 rarely exceeded 0.02 mg/l, the detection limit, in the control or treated plots despite initial inputs of up to 60 kg/ha of NH_3 (Table 1). NH_3 volatilization may have reduced the leaching of N as NH_4. The initial sludge pH, air temperatures,

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and soil moisture were conducive to rapid volatilization (Adriaro et al., 1974; King and Morris, 1974; Chao and Kroontje, 1964). Biological immobilization by vegetation and increased microbial activity in the treated plots may also have prevented leaching of ammonium.

Background NO_3 values in soil water ranged between 0.2 ± 0.1 and 1.0 ± 0.2 mg/l with the lowest values corresponding to periods of maximum vegetative and microbial uptake (Likens et al., 1977). After heavy applications, NO_3 concentrations tended to rise during the summer and autumn of 1975, but the maximum concentrations averaged $\leq 1.5 \pm 0.3$ mg/l implying at most a 1 to 1.5 mg/l increase in NO_3 (Fig. 5). The increase in NO_3 at 20 cm during the spring of 1976 was solely due to high concentrations found in one lysimeter that may have been disturbed during the winter of 1975-1976. Since the amount of NO_3 initially present in the sludge was negligible and no increase in ammonium was detected in the treated plots, it would appear that the small increases in NO_3 in soil water were a result of nitrification.

Nitrogen losses in soil water appeared minimal in the treated plots despite the application of up to 477 kgs total Kjeldhal N/ha (Table 1). Most of the applied N is probably retained in the upper soil horizons or sludge (King and Morris, 1972b, 1974). Investigators have shown that N in sludge is not easily mineralized (Molina et al., 1971; Premi and Cornfield, 1971; Beauchamp and Moyer, 1974). Since mineralization rates are naturally low at Hubbard Brook with the most rapid rate occurring during maximum biological uptake (Mellilo, 1977), it would appear that N in the sludge would undergo slow mineralization with the most rapid release occurring at times of greatest demand.
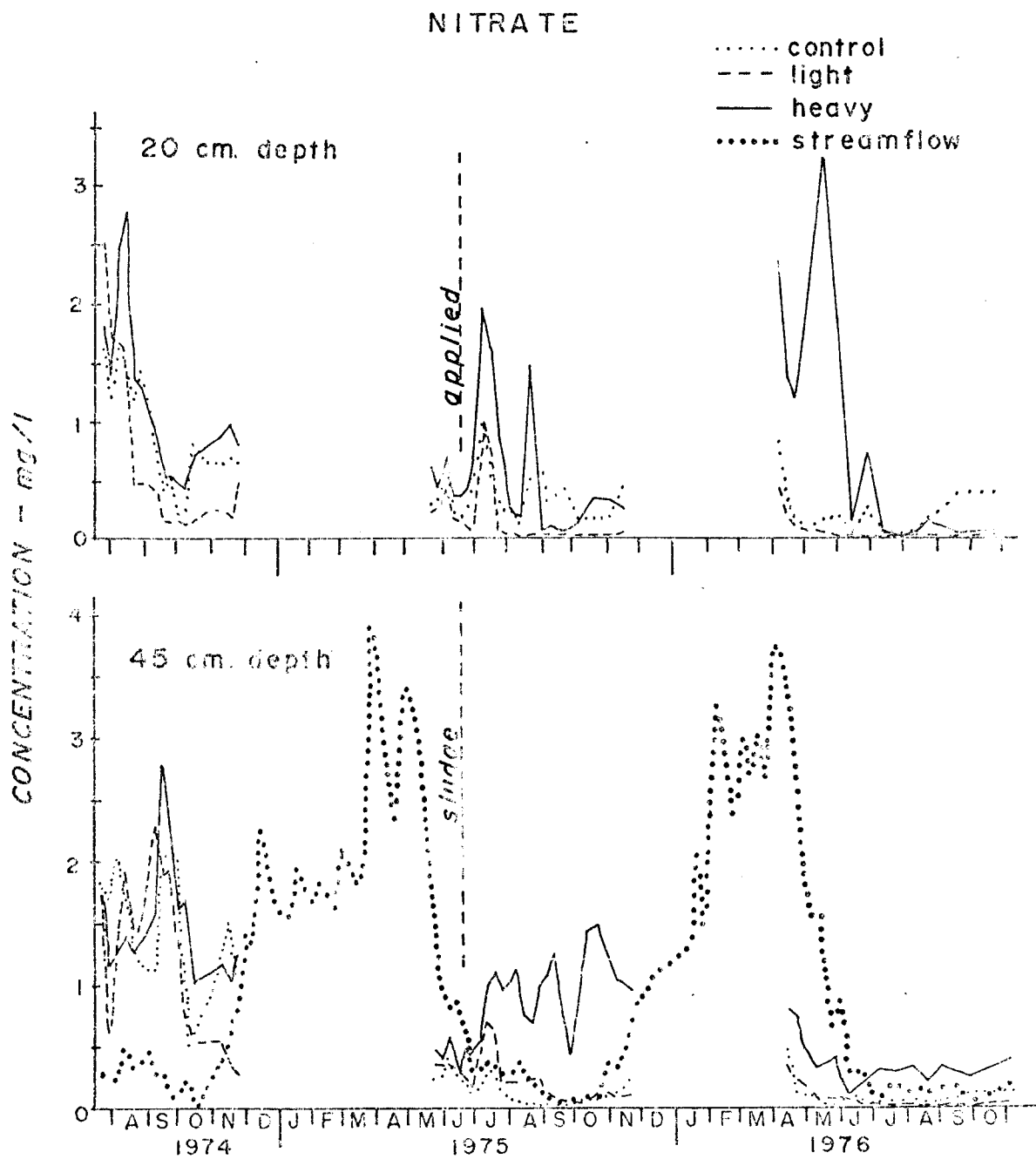
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Figure 5. Mean concentrations for NO_3 in soil water and stream.

Calcium. Ca is the dominant cation in soil water with background concen-
trations averaging about 2.5 ± 0.2 mg/l. The light application caused
little change in the concentration of this ion while the heavy applica-
tion resulted in Ca increases at both soil depths (Fig. 6). At 20 cm,
Ca rose to 12.8 ± 1.0 mg/l or over 5-fold above background values by
September, 1975. At 45 cm, the average concentration rose to a maximum
of only 6.5 ± 0.4 mg/l, or about one half the increase observed at 20 cm.
During most of the 1976 growing season, Ca values averaged about 0.5 mg/l
higher in the plots receiving the heavy application.

The pattern of response for Ca soon after heavy application was
similar to Cl suggesting that fluctuations in soil moisture influenced
initial Ca concentrations. However, Ca remained above background values
for a considerably longer period of time than Cl. The continued leaching
of Ca during late 1975 and in the 1976 growing season was probably governed
more by the rate of sludge decomposition and subsequent release of Ca than
by fluctuations in soil moisture. Overall, the increases in Ca in soil
water were less than half those for Cl, even though much more Ca was
applied in the sludge (Table 1). This suggests a large amount of the
applied Ca is retained in the sludge or soil horizons.

Magnesium. Background concentrations for Mg averaged about 0.5 ± 0.1 mg/l.
Sludge applications caused concentrations of this ion to change in a pattern
similar to that of Ca although the magnitude of change was much less.
Maximum values under the heavy application were 2.0 ± 0.2 mg/l and 1.0 ±
0.1 mg/l at 20 and 45 cms. The similarity of response for Mg and Ca
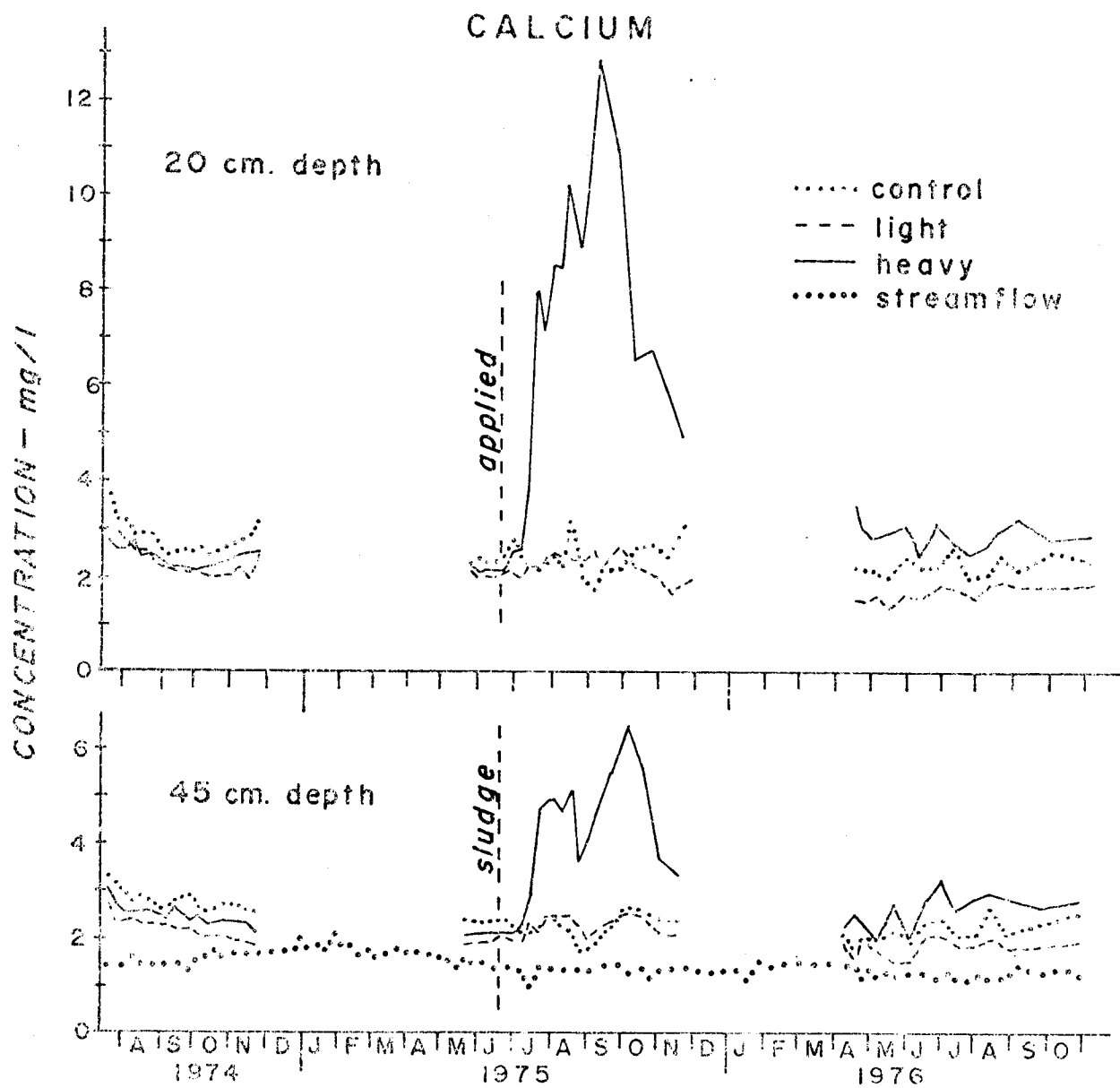suggests the same mechanisms influence treatment impacts.

Figure 6. Mean concentrations for Ca in soil water and stream.

Sodium, Potassium, and Hydrogen. Background concentrations of Na, K, and H averaged about 0.5 ± 0.1 mg/l, 0.10 ± 0.03 mg/l and 2.0 ± 0.5 μg/l respectively. Light applications had no detectable effect on the concentrations of these ions, whereas heavy applications caused two-fold increases in the average concentrations of Na and H and a three-fold increase in the average K concentration at 20 cm. Increases at 45 cm were about one half those occurring at 20 cm. The increase in hydrogen occurred despite an initial pH > 11 (Table 1), and may be the result of increased nitrification and sulfur oxidation in the soil horizons not immediately in contact with the sludge (King and Morris, 1972a). K in the treated plots remained either above or about equal to control values throughout the period of this study. This suggests that K was not limiting to plant growth despite the fact that the sludge may be low in available K (Harter, 1975).

Stream Water Chemistry

The changes in soil water chemistry caused by sludge applications may indicate potential changes in stream water chemistry. The concentrations of most ions at 45 cm in control plots were similar to those in the stream draining a nearby forested watershed (Figs. 2 thru 6). Thus the changes in soil water caused by sludge applications at 45 cm may be extrapolated as an approximation of impacts on streams. On this basis, the light application would cause little change in stream chemistry. The heavy application could cause changes in the concentrations of most of the major nutrients found in streams. However, based on the soil water data, the changes would be short-lived and relatively small in magnitude.

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Vegetation

In general, there was no major response of vegetation to sludge applications. Before treatment, basal area for trees $>$ 5 cm diameter at breast height was uniform among the six plots and averaged 26.6 $\pm$ 1.4 m^2/ha. The total increases in basal area over the two growing seasons after sludge applications was 0.9 m^2/ha for the control plots, 1.3 m^2/ha for plots receiving the light application, and 0.8 m^2/ha for plots receiving the heavy application. The differences between plots are not statistically significant indicating that diameter growth was not affected by sludge applications.

Data from the 1 m^2 subplots showed 31 different plant species occurred in the forest understory before sludge applications. Aside from the germination of a few vegetable seeds that came mixed with the sludge, species diversity was not affected by sludge applications. In terms of specie frequency, the heavy application caused only three changes that were detectable. Sugar maple seedlings declined from an average of 22/m^2 before application to 17/m^2 after, mostly as a result of physical damage during spreading of the sludge. Hobblebush (Viburnum alnifolium Marsh), a shrub commonly associated with northern hardwoods, increased from an average of 5/m^2 before application to 11/m^2 after. And a native herb, Viola pallens Brainerd, increased to 21/m^2 from the pretreatment average of 15/m^2.

CONCLUSIONS AND RECOMMENDATIONS

Our study indicates than when sludge is spread on the floor of a northern hardwood forest most nutrients in the sludge become incorporated into nutrient cycles, leaching losses are not greatly increased, and there is little change in vegetation. The changes that occur in soil water chemistry are primarily dependent on the amount of sludge applied, but the pattern of change in the concentration of most ions also depends on one or more of the following factors:

1. <u>Seasonal fluctuations in soil moisture</u>. During summer and early autumn of 1975, the decrease in the magnitude and movement of soil moisture because of evapotranspiration probably caused increases in Cl and most cations in plots receiving sludge applications. Dilution effects associated with higher soil moisture at 45 cm could account for the fact that the increases at this depth were only about one-half those occurring at 20 cm. Soil moisture recharge beginning in autumn of 1975 may have caused dilution and leaching and explain the rapid decline in treatment effects during this period.

2. <u>Changes in microbial populations in sludge-treated plots</u>. The increase in fungi and indigenous soil bacteria may have provided a temporary sink for some nutrients, especially N, P, and SO_4. At the same time, decomposition of sludge by these microbes could have aided in mobilization and eventual leaching of cations. The eventual return of microbial populations to background levels in autumn, 1975, could have resulted in a release of nutrients, particularly SO_4.

3. <u>Chemical and biological retention of nutrients by the soil and plant communities</u>. Sufficient evidence can be found in the literature to suggest that some nutrients, for example, P, N, and to some extent Ca, Mg, and K,

could have been immobilized by chemical retention in the sludge or soil and by vegetative uptake (Koterba, 1977).

4. Volatilization. N leaching may have been reduced by the volatilization of ammonia.

Our study indicates that heavy applications of sludge could cause changes in streamwater chemistry. In order to reduce the likelihood of any changes, the factors listed above have been incorporated in designing the following guidelines for applying sludge to the forest floor:

1. Application rates. Sludge applied at a rate intermediate between the two tested in this study should have little impact on stream and would fall close to the rates recommended by Cunningham (1971) for forests and Harter (1975) for land areas in New Hampshire. Some dilution of any treatment effects on streams could be achieved by maintaining a buffer strip at least 50 m in width between the treated area and stream channel, and by applying sludge only to portions of a watershed.

2. Timing of applications. In our study, sludge was applied during the growing season, the optimum time for vegetative and microbial uptake of nutrients. Also, soil moisture levels are reduced by evapotranspiration which minimizes leaching. The results of our study indicate a high efficiency in maintaining nutrients on site if sludge is applied during this period.

During winter months, November to about April, biological inactivity could mean reduced efficiency in nutrient retention. Readily available nutrients could be leached from the site during periods of temporary thawing or spring melting of the snowpack. These nutrients would probably be leached in relatively low concentrations during major runoff periods, but the loss of nutrients from the site is undesirable. Another problem

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associated with winter disposal is the often limited access to forest sites. Until more information is available on nutrient losses, we hesitate to recommend winter applications of sludge.

Although winter conditions may restrict forest applications, there are alternatives that could complement forest disposal during the growing season. For example, sludge can be composted at relatively low cost during the winter months (Anon., 1976). This composted material can be used for a variety of local uses. However, a major problem confronting the use of composted material on an annual basis is that production often exceeds demands. Thus winter composting alternated with forest disposal of uncomposted material during the growing season may be the answer to year-round sludge disposal.

3. Site characteristics. Soils at our study site were sandy loams, having moderate to high infiltration, and were about 50 cm in depth. Soils with poor drainage or shallower depth may not be as satisfactory in preventing nutrient leaching. Vegetation at the study site was characteristic of a northern hardwood forest. Sites having only sparse vegetative cover, for example, a recent clearcut, may be inadequate in preventing erosion or leaching losses of sludge nutrients. This is particularly important if forest regrowth in the area is less than two or three years old, since natural nutrient losses may already be quite high (Pierce et al., 1972).

4. Sludge characteristics. Our study used sludge that was primarily of domestic origin. The chemical composition of the sludge varied little from week to week (Koterba, 1977). Though some plant to plant variation exists, most primary, domestic sludges are probably suitable for recycling. However, there are other factors which must be considered. For example, the sludge in this study had a high solids content (22 percent by weight).

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A sludge with a high solids content, usually obtainable using a vacuum coil filter, may be less susceptible to rapid leaching of nutrients than sludge in more liquefied states.

The sludge used in this study was also relatively low in heavy metals. Since little information is available on the effects of heavy metals in forests, it may be best to limit forest applications to sludges that do not greatly exceed the heavy metal content of the area of application (Harter, 1975). And finally, the sludge used in this study was limed to a pH of 11-12. Liming is an effective deodorizer and means of destroying pathogenic organisms (Dean and Smith, 1973; Farrel et al., 1974; Counts et al., 1974). The lime may also improve the fertility of acidic soils and reduce leaching and erosion losses by increasing the aggregate stability of both sludge and soil. Thus sludges that have been limed are most suitable for land application.

5. Monitoring systems. It is recommended that any land-recycling system be monitored for changes in water quality or vegetation. Then if any problems arise, they can be corrected before site quality is severely altered. In this study, it was not difficult to maintain a monitoring system capable of detecting changes in vegetation and soil water chemistry. The lysimeters are relatively easy to install and operate. It is important to allow sufficient time for subsidence of soil disturbance effects associated with lysimeter installation. Our study indicates that after 6 to 8 months, soil water samples appear to accurately reflect natural conditions and allow for quick detection of changes in water chemistry at almost any soil depth. Analyses of the samples could probably be obtained through a government agency or private analytical laboratory.

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Changes in vegetation can also be easily monitored through the use of surveys similar to those used in this study. They require little time or expense.

The above recommendations are no more than common sense evaluations that should be made before a community adopts any type of land spreading of sewage sludge. Communities which are able to utilize this approach may find it an attractive alternative for sludge disposal. The amount of forest land needed for sludge disposal is not large. For example, the Plymouth treatment plant serves a population of 5,000-7,000 and produces an average of 3.5 metric tons of wet sludge per week. At an application rate of 75 wet tons/ha, Plymouth would require about 1.3 ha of forest land to spread the sludge produced during the growing season. For eco-systems such as we studied which seem capable of absorbing most of the applied nutrients, sludge could probably be reapplied to the same area every third or fourth year. This means that a total of 5-6 ha of suit-able forest land could probably take care of Plymouth's sludge output during the growing season for 10 to 20 years. If further research in-dicates it is possible to spread sludge throughout the year, only 10 to 12 ha of suitable forest land would be required to take care of Plymouth's sludge for several decades.

The initial cost of this forest land might seem expensive but recent evidence suggests that land spreading of sludge may be the least costly, yet environmentally safe, means of sludge disposal. For treatment plants processing less than 10 million gallons/day, which includes all New Hampshire communities, the cost of sludge disposal in ¢/1000 gallons treated wastewater is smallest for land spreading compared to incineration and land filling (Shea and Stockton, 1975). Thus, under good management

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and with proper site conditions, many communities may find sludge

applications to forests a suitable alternative to the common and

apparently more expensive practice of disposing of sludge at land-fill

sites.

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